Western North American Naturalist 65(1), © 2005, pp. 91–102

EFFECTS OF GRAZING EXCLUSION ON RANGELAND VEGETATION AND SOILS, EAST CENTRAL IDAHO

Jeffrey J. Yeo[1]

ABSTRACT.—Nineteen exclosures on sagebrush steppe and shadscale rangelands, varying in age from 18 to 38 years, were sampled for plant species richness, plant composition, indicators of soil erosion, ground cover, vegetative cover, and herb–low shrub layer screening cover. Features within the exclosures were compared with adjacent sites of the same size that were open to grazing by livestock and wildlife. Species richness typically was slightly greater inside exclosures than in adjacent grazed sites (median = 2 more species inside exclosures), but the difference was not significant ($P = 0.16$). Similarity of plant community composition between exclosures and adjacent grazed sites ranged from 45% to 82%. Evidences of soil movement, soil pedestals, and soil flow patterns were all more pronounced outside exclosures than inside ($P \leq 0.02$), even though many sites were on flat to mild slopes (median slope = 12%). Meta-analysis of the 19 exclosure sites indicated that grazing exclusion resulted in less bare ground cover compared with adjacent grazed sites ($P \leq 0.05$). The effect of grazing exclusion on visible soil surface cryptogams was significant ($P \leq 0.05$), with generally greater cover inside exclosures. Cryptogam cover differences between grazed sites and exclosures tended to increase with the number of years of grazing exclusion ($r = 0.64$, $P = 0.046$). *Pseudoroegneria spicata,* a principal livestock forage, averaged greater basal cover inside exclosures than outside on 4 of 10 sites where it occurred, although no exclosure sites had greater *P. spicata* cover on grazed sites. Meta-analysis of the 10 sites indicated that grazing exclusion resulted in greater *P. spicata* cover compared with adjacent grazed areas ($P \leq 0.05$). *Poa secunda,* a short-growing grass that initiates growth early in the spring and is not important livestock forage, averaged greater basal cover outside exclosures on 5 of 15 sites where it occurred. Meta-analysis of the 15 sites indicated a significant treatment effect ($P \leq 0.05$), with greater *Poa secunda* basal cover outside exclosures. Grazing exclusion resulted in greater screening cover in the herb–low shrub layer (0–0.5 m height; $P \leq 0.05$). These results indicate that despite improved livestock grazing management over the past half century, livestock grazing still can limit the potential of native plant communities in sagebrush steppe ecosystems, and that the health of semiarid ecosystems can improve with livestock exclusion in the absence of other disturbances. A few exclosure sites were similar for the measured parameters, suggesting that these sites were ecologically stable and that exclusion of livestock grazing was not sufficient to move succession toward more pristine conditions, at least within the time periods studied. Managed disturbance such as fire or mechanical brush treatments may be necessary to restore herb productivity on these ecologically stable sites.

Key words: grazing exclosures, long-term vegetation change, erosion, vegetative cover, screening cover, cryptogams, wildlife.

An issue central to rangeland management concerns the ability of plant communities to respond to changing livestock management practices. Range managers recognize the historic impacts of overgrazing prior to passage of the Taylor Grazing Act in 1934 but assume that recent practices of grazing management (e.g., rest-rotation, deferred rotation, short-duration grazing) are adequate to protect rangeland resources given appropriate stocking rates (Laycock 1994). Removal of livestock from western arid and semiarid rangelands has been advocated because of widespread evidence of overgrazing and the impacts on biodiversity (Fleischner 1994, Noss 1994, Donahue 1999). Yet, reduction in numbers or removal of livestock may not result in rangeland improvement, at least in arid and semiarid ecosystems such as sagebrush steppe (West et al. 1984, Bork et al. 1998, West 2000).

Traditional models of plant community succession (Clements 1916, Dyksterhuis 1949, Huschle and Hironaka 1980) postulate generally linear pathways of succession to a predictable climax state following disturbances such as fire and grazing. More recent community succession models suggest that vegetation can exist in multiple quasi-stable states depending on the history of the site, and that transition between states requires some biotic or abiotic force to move the community beyond a threshold (Noy-Meir 1975, Hanley 1979,

[1]The Nature Conservancy, 116 First Avenue North, Hailey, ID 83333.

Westoby et al. 1989, Tausch et al. 1993). The first 10–15 years of heavy grazing by livestock may have the greatest impact on vegetation (Hull 1976). Once vegetation degrades to some threshold, cessation of grazing may not halt continued decline or at least may not allow the community to improve (Whitford 1995). Cycles of drought and soil changes may preclude recovery of plant communities to pregrazing conditions. The availability of native plant seed, availability of nutrients, and invasion of exotic plants may alter the succession pathways available. Reduction of fine fuels because of livestock grazing and fire-suppression policies has resulted in a change in the frequency and intensity of natural fire, which historically played an important role in rangeland dynamics.

Ecological advantages reported for livestock grazing include enhanced seeding/germination by trampling seed into the ground, fertilization from feces and urine, and plant growth stimulation from grazing (see reviews by Briske and Richards 1994, Pieper 1994). However, overgrazing can result in the reduction or extinction of forage preferred by livestock, degradation of biodiversity, introduction or dominance of annuals and exotic plants, reduction or elimination of cryptogamic soil crusts that can reduce nitrogen available for plant growth, increased soil erosion, and soil compaction with impeded water infiltration (Fleischner 1994, Belsky and Blumenthal 1997). Factors other than livestock grazing can also affect vegetation dynamics. For example, cycles of drought can have pronounced effects on the composition and structure of sagebrush steppe vegetation (Anderson and Inouye 2001).

Grazing exclosures have been constructed in the Bureau of Land Management (BLM) Challis Resource Area and surroundings since the 1st half of the 20th century (Idaho Department of Fish and Game records, Salmon, ID). Yet, few of these exclosures apparently were sampled when constructed or the records have been lost, and even fewer have been evaluated in the intervening years. This research addresses the question, What are the effects on vegetation and soils of exclusion of livestock grazing after a long history of grazing (and often overgrazing)? I assumed in this study that sites were in similar condition inside and outside the exclosures at the time of construction, and that paired sites experienced similar environmental conditions since fencing so that any observed differences were a result of continued grazing or its absence. My expectation was that sites inside exclosures would have greater cover of palatable grasses, perennial forbs, cryptogams, and screening cover, and less bare ground than adjacent sites outside exclosures. I also expected that evidence of soil erosion would be less pronounced within exclosures than outside.

STUDY AREA

The exclosures were situated on high-elevation (1585–2315 m), federally managed public rangelands in east central Idaho (Table 1). Vegetation varied from lower-elevation xeric communities of *Atriplex confertifolia* and *Atriplex spinosa* commonly growing on alkaline soils, or *Artemisia tridentata* spp. *wyomingensis* on well-drained soils, to more mesic communities of *Artemisia tripartita* or *Artemisia tridentata* spp. *vaseyana* sometimes mixed with *Purshia tridentata* bordering *Pseudotsuga menziesii* at higher elevations. Slope gradients ranged from nearly flat (1%) to 20%, and aspect of the exclosure sites included most exposures. Exclosure sizes varied from 0.04 ha to 0.8 ha, and the time during which grazing was excluded spanned 18 to 38 years.

Climate at the exclosure sites is semiarid and cool with average annual precipitation ranging from 128 mm to 559 mm (based on soil descriptions, Natural Resources Conservation Service 2002a). Average summer temperature at Challis is 20°C, and winters average –8°C. Climate averages can be deceiving for this area because of the substantial variability due to rugged topography and localized storms. The frost-free period may range from 60 to 100 days (Bureau of Land Management 1998). A consistent annual weather record for Challis began about 1932 (Fig. 1). Drought was apparent in the 1930s and broke in 1940. Drought returned in the late 1940s and continued through about 1963. The late 1980s through the early 1990s also experienced drought. In contrast, the years 1993, 1995, and 1998 were exceptionally wet.

Precipitation in 1999, when the exclosures were sampled, was generally less than the long-term average. This was particularly true during mid- to late summer when sampling occurred. Plant growth in 1999 was initially slowed by a cool June, followed by rapid growth in July that

TABLE 1. Exclosure site characteristics and topography. Predicted range of annual precipitation is based on soil characteristics (Natural Resources Conservation Service 2002a). Plant association: ACHY = *Achnatherum hymenoides*, ARAR = *Artemisia arbuscula*, ARFR = *Artemisia frigida*, ARTR4 = *Artemisia tripartita*, ARTRV = *Artemisia tridentata vaseyana*, ARTRW = *Artemisia tridentata*, ATCO = *Atriplex confertifolia*, FEID = *Festuca idahoensis*, HECO = *Hesperostipa comata*, POSE = *Poa secunda*, PSSP = *Pseudoroegneria spicata*, SPAR = *Sphaeromeria argentea*, and SPCR = *Sporobolus cryptandrus*. Community similarity index is described in Methods.

Exclosure	Predicted precipitation (mm)	Plant association	Elev. (m)	Aspect	Slope (%)	Size (ha)	Years enclosed	Similarity index (%)
Eagle Rock	128–205	ATCO/ACHY-HECO	1585	NE	8	0.81	22	70
Meadow Creek	178–254	ATCO/SPCR	1804	N	3	0.4	38	80
Leaton Gulch	179–256	ATCO/SPCR	1670	W	8	0.4	38	82
Antelope Flats	203–279	SPAR-ARFR/POSE-HECO	1938	N	1	0.04	28	75
McGowan Creek	203–305	ARTRW/HECO	1987	W	12	0.08	18	73
Centennial Flats	203–305	ARTRW/PSSP	1938	E	16	0.36	18	78
Sage Creek No. 2	203–305	ARTRW/PSSP	1951	NE	1	0.08	18	60
Sage Creek No. 1	203–305	ARTRW/PSSP	1975	NE	1	0.12	18	70
East Fork Fan	203–305	ARTRW/PSSP	1743	NE	12	0.36	18	82
Boneyard Gulch	203–305	ARTRW/PSSP	1792	E	12	0.45	29	45
Bradshaw Basin	203–305	ARTRW/PSSP	2219	NW	12	0.04	28	69
Jeff Flats No. 1	279–330	ARTRW/PSSP	1853	NE	12	0.04	28	61
Donkey Hills	203–305	ARAR/PSSP	2146	NE	2	0.4	38	78
Jeff Flats No. 2	305–406	ARTR4/FEID	1926	N	12	0.32	19	80
Broken Wagon	330–406	ARTR4/FEID	2085	N	12	0.08	18	54
Second Spring	305–406	ARTR4/FEID	2292	S	12	0.81	21	77
Martin Creek	330–406	ARTRV/FEID-PSSP	2012	SE	20	0.4	31	72
Third Spring	406–559	ARTRV/FEID	2134	S	20	0.36	22	75
Horse Heaven	406–559	ARTRV/FEID	2316	N	5	0.45	29	54

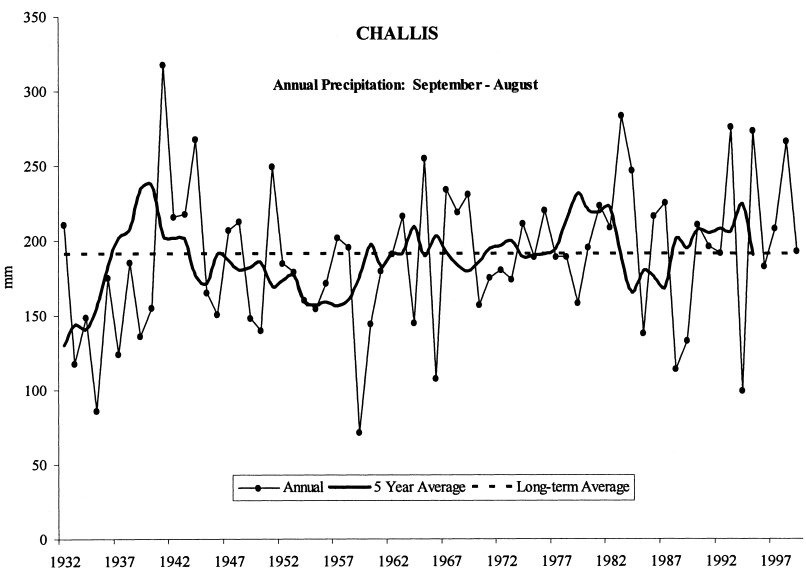

Fig. 1. Long-term annual precipitation records for Challis, Idaho. Median distance to the 19 exclosures from Challis was 23.2 km (range = 8.4–52.3 km).

terminated in early August with desiccation, particularly among forbs.

Livestock grazing of the region began in the 1860s, with overgrazing noted by the 1880s and probably continuing for 40–50 years (Shoup 1935). Domestic sheep numbers began increasing in the late 1880s, reaching >50,000 sheep grazing Bureau of Land Management lands in the Challis area by the 1920s (Bureau of Land Management records, Salmon, ID). After the 1950s cattle became the principal livestock grazing the area. In the 1970s planned grazing systems were implemented.

METHODS

I sampled upland grazing exclosures on public lands in which at least half the area inside the exclosure appeared undisturbed since its construction. Visually disturbed areas either inside or outside exclosures (i.e., soil disturbance from range improvements or experimental seeding) were not sampled. I sampled inside each exclosure and at a nearby (<30 m distant) site outside the exclosure exhibiting similar size, vegetation, aspect, slope, and elevation as the exclosure. Therefore, sampling site selection emphasized similarities between the exclosure and outside.

Sample points, 5–15 m apart (depending on the size of the exclosure, 10–25 samples were collected per treatment), were systematically located on a randomly positioned grid to achieve good dispersion of points throughout the exclosure and adjacent grazed site. A 5-m buffer zone inside and outside the exclosure fence was not sampled to avoid possible effects of the fence (e.g., livestock trailing and trampling, fertilization from birds perched on the fence or other animals concentrating along the fence). At each sampling point I measured horizontal vegetation cover, ground cover, and screening cover (vertical vegetation cover). Soil surface condition was evaluated for each treatment as a whole.

I measured canopy cover of shrubs and forbs, basal cover of graminoids, and ground cover of bare ground, litter, and cryptogams using a 50 × 100-cm point-interception frame (Floyd and Anderson 1982) placed at each sampling point. Each point-interception frame contained 36 intersection points (created by 2 superimposed grids, 15 cm apart, of 10 × 10-cm squares of string) at which cover was recorded. The point-interception frame was leveled with a bubble level to maintain a consistent vertical projection on the ground. All plant species beneath each point were recorded based on "hits" of live vegetation of forbs and shrubs and basal areas of graminoids. Points were visually projected through the overstory so that a single point could include more than 1 species but not ground cover. Points intercepting bare ground, litter, or cryptogams were recorded only if no vascular plants occurred above the point.

At 3 exclosure sites I used a 20 × 50-cm plot frame to measure cover. I did this either because vehicle access was >1.6 km from the exclosure and the combination of equipment was too cumbersome to transport to the sites, or because shrubs were too tall and dense to use the point-interception frame effectively. Cover was estimated using Daubenmire's (1959) cover classes (in percentage): 0, 1–5, 6–25, 26–50, 51–75, 76–95, and 96–100.

Screening cover is important to rangeland wildlife such as nesting Greater Sage-Grouse (*Centrocercus urophasianus*) and pronghorn (*Antilocapra americana*) fawns, both for security from predators and for thermal cover (Autenreith 1978, Connelly et al. 2000). Screening cover was estimated at each sampling point using a 1.5-m-tall cover pole (Griffith and Youtie 1988). The pole was divided into three 0.5-m sections, with each section divided into five 10-cm segments. The 3 sections were designated as herb–low shrub layer (0–0.5 m height), medium shrub layer (0.5–1.0 m height), and tall shrub layer (1.0–1.5 m height). The number of 10-cm segments at least 50% covered by live vegetation was recorded separately for each of the three 0.5-m sections. Four readings at a distance of 5 m from each of the cardinal directions were taken at each sampling point. The 2 lower layers were read from a kneeling position to reduce parallax error. Analyses of screening cover comparisons are presented only for the herb–low shrub layer because many fewer sites contained screening cover in the 2 taller layers.

I evaluated soil erosion using 7 indicators of soil surface condition: evidence of soil movement, surface litter, surface rock, soil pedestals, evidence of flow patterns, rills, and gullies (Bureau of Land Management 1973). Soil surface condition factors were evaluated for each exclosure site sampled (i.e., separately for the entire area within the exclosure and for the

sampled area outside the exclosure). Each erosion indicator was evaluated on an ordinal scale with 1 indicating little or no evidence of erosion and 5 indicating severe erosion.

Species richness was compared between the exclosure and the adjacent grazed area. Sorensen's community coefficient (similarity index), weighted by % cover (Barbour et al. 1980), was calculated for each exclosure site. Depending on the distribution of the data for each parameter estimated, I used an unpaired *t* test or its nonparametric equivalent (Mann-Whitney rank sum test) to compare the effects of grazing and grazing exclusion. Correlations between the duration of grazing exclusion and measured parameters were analyzed using Pearson's correlation index. Syntheses across exclosure sites of treatment effects for each of the cover parameters estimated were analyzed using meta-analytic methods (Rosenberg et al. 2000). The treatment effect size (with the area open to grazing representing the control and the area excluding grazing representing the experimental treatment) for each exclosure site was standardized as the natural log of the response ratio (the ratio of the mean estimate within the exclosure to that of the area open to grazing). This measure of effect size estimates the change resulting from grazing exclusion. A random-effects model was used because the vegetation communities differed among exclosures such that the response to grazing exclusion would not be expected to be consistent among exclosure sites. Confidence intervals about the mean effect size were calculated using bias-corrected bootstrapping (Rosenberg et al. 2000). Indicators of soil erosion were compared across all exclosures using a Wilcoxon signed rank test. Significance level was set at $P \leq 0.05$. Plant names follow the National PLANTS database (Natural Resources Conservation Service 2002b).

RESULTS

I encountered 20 species of graminoids, 60 species of forbs, 20 species of shrubs, and 1 tree species (*Pseudotsuga menziesii*) at the 19 exclosure sites. Nonnative plants were sparse (only 6 species found) and, except for *Agropyron cristatum* seeded at 3 sites, had only trace amounts of cover at any exclosure site.

Species richness observed at each of the exclosure sites ranged from 8 to 31 species, with differences between treatments ranging from 0 to 9 species. Twelve of 19 sites had greater species richness within exclosures than outside (median difference = 2 more species within exclosures than on adjacent grazed sites), but this difference was not significant ($P = 0.16$). Community similarity ranged from 45% to 82% (median = 73%), with Boneyard Gulch having the lowest community similarity (45%) of 19 exclosure sites studied (Table 1). The number of years of grazing exclusion was not related either to differences in species richness between treatments or to community similarity ($r \leq 0.13$, $P \geq 0.60$).

Slope steepness at the exclosure sites ranged from 1% to 20% (median = 12%; Table 1). Exclusion of livestock grazing resulted in a consistent pattern of improved soil surface conditions compared with areas open to grazing. Evidence of soil movement, soil pedestals, and soil flow patterns were all more pronounced outside exclosures than inside ($n = 19$; $P \leq 0.02$).

Cover of bare ground ranged from 3% to 51%, with differences between treatments ranging from 0% to 39%. The amount of bare ground was greater outside exclosures at 9 of 19 sites ($P \leq 0.017$), and there was no apparent relationship between the duration of grazing exclusion and bare ground differences between grazed and exclosure treatments (Fig. 2). Although the correlation between slope steepness and the difference between excluded and grazed sites was not significant ($P = 0.07$), there was a tendency for flatter sites to have greater differences between treatments than steeper sites ($r = 0.42$). Meta-analysis of the 19 exclosure sites indicated that the amount of bare ground exposed was reduced when grazing was excluded ($P \leq 0.05$).

Differences between treatments for ground cover of litter were variable but were significant at 5 exclosure sites. Two sites had greater litter cover outside the exclosures and 3 had greater litter cover inside. There was no correlation between litter cover differences between treatments and the duration of grazing exclusion ($r = 0.02$, $P = 0.92$). In addition, there was no consistent effect of grazing exclusion when all exclosure sites were analyzed together.

Soil cryptogams, found at 10 sites, had significantly greater cover inside exclosures at 6 sites compared with adjacent grazed areas ($P \leq 0.015$; Fig. 3). Cover ranged from 1% to 36%,

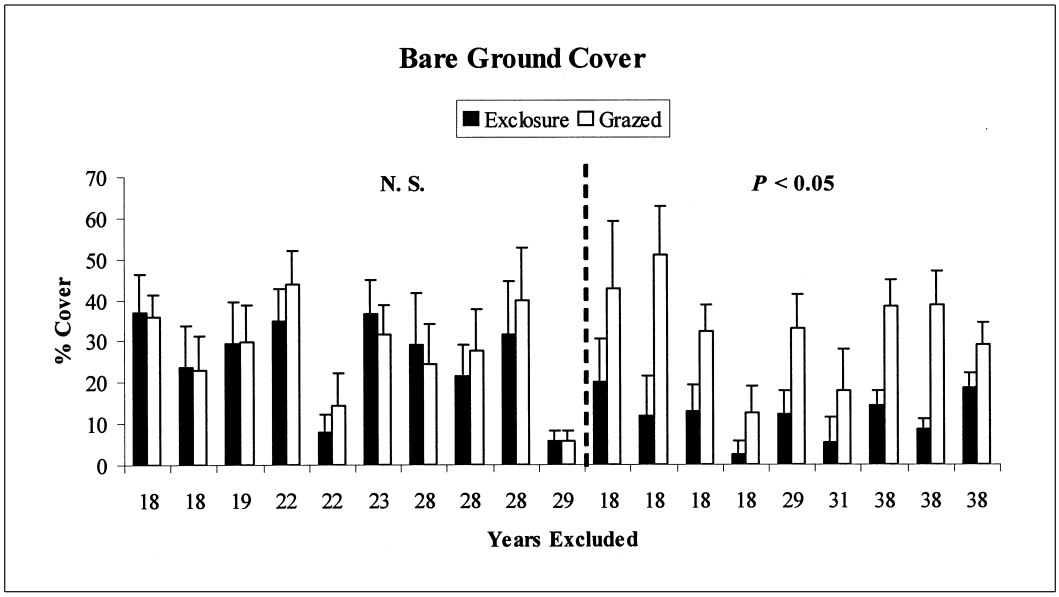

Fig. 2. Comparison between exclosure sites and adjacent grazed sites for bare ground cover, arranged by number of years of grazing exclusion within the exclosure. **N. S.** groups those sites where the difference between exclosure and grazed sites was not statistically significant; $P < 0.05$ groups those sites where treatment differences were statistically significant. Error bars denote 95% confidence intervals.

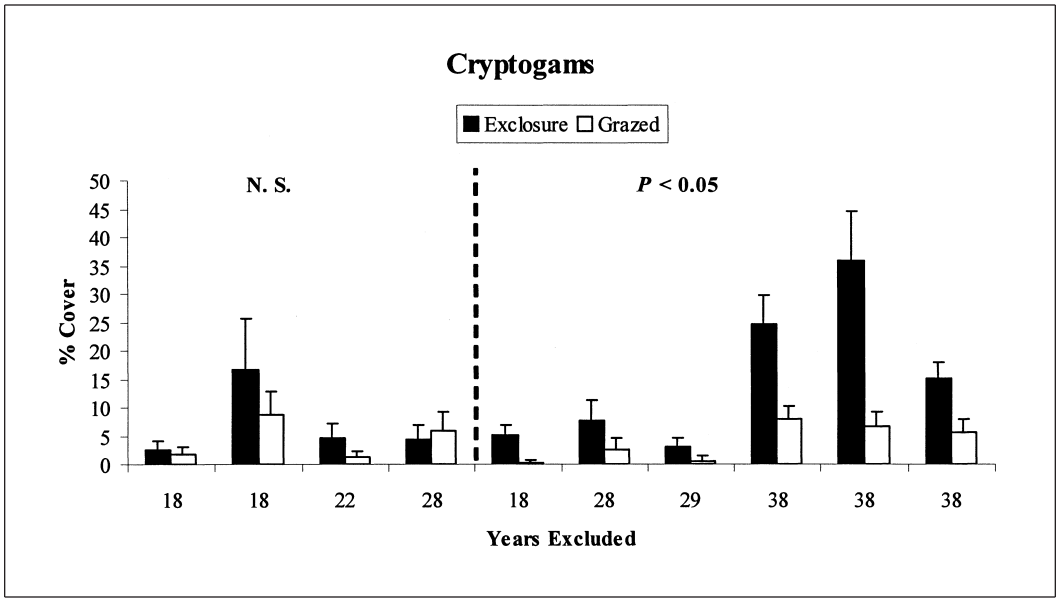

Fig. 3. Comparison between exclosure sites and adjacent grazed sites for cover of crytogams, arranged by number of years of grazing exclusion within the exclosure. **N. S.** groups those sites where the difference between exclosure and grazed sites was not statistically significant; $P < 0.05$ groups those sites where treatment differences were statistically significant. Error bars denote 95% confidence intervals.

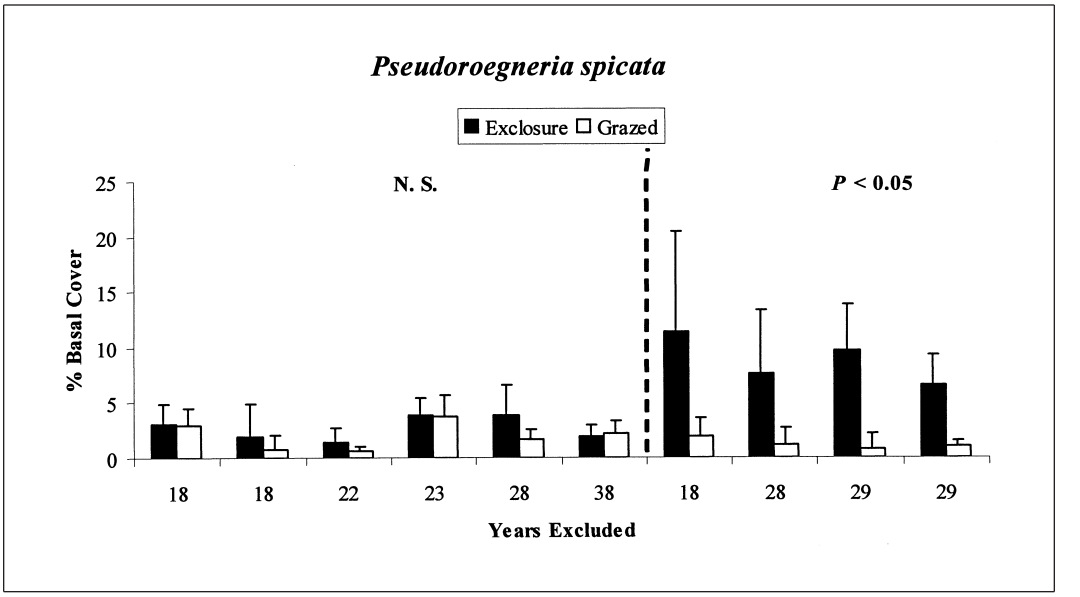

Fig. 4. Comparison between exclosure sites and adjacent grazed sites for cover of *Pseudoroegneria spicata*, arranged by the number of years of grazing exclusion within the exclosure. **N. S.** groups those sites where the difference between the exclosure and grazed sites was not statistically significant; ***P* < 0.05** groups those sites where treatment differences were statistically significant. Error bars denote 95% confidence intervals.

with cover differences between treatments ranging from 1% to 29%. Cryptogam cover differences between exclosures and adjacent grazed areas tended to increase with more years of grazing exclusion ($r = 0.64$, $P = 0.046$). Meta-analysis of all 10 sites indicated a significant treatment effect with increased cryptogam cover corresponding to exclusion of grazing ($P \leq 0.05$).

Graminoid basal cover ranged from 5% to 26%. The 2 most consistently encountered grasses, and generally with the largest cover, were *Pseudoroegneria spicata* and *Poa secunda.* Other grasses common (≥5% basal cover) on some sites included *Festuca idahoensis* (2 sites), *Sporobolus cryptandrus* (1 site), and *Agropyron cristatum* (1 site).

Pseudoroegneria spicata, probably the most important livestock forage species in the region (Yeo 1981), occurred at 10 exclosure sites. Basal cover of *P. spicata* was significantly greater within the exclosures at 4 sites ($P \leq 0.037$), with cover differences ranging from 6% to 10% (Fig. 4). No sites had significantly greater cover of *P. spicata* outside the exclosures. There was no apparent relationship between the duration of grazing exclusion and basal cover differences between treatments ($r = -0.01$, $P = 0.96$). Meta-analysis of all 10 sites showed a significant treatment effect with increased basal cover with grazing exclusion ($P \leq 0.05$).

Poa secunda, an early growing, small, fine-leaved grass of limited forage value, averaged greater basal cover outside the exclosure at 5 of 15 sites at which it occurred ($P \leq 0.05$; Fig. 5). *Poa secunda* had greater basal cover within the Sage Creek No. 2 exclosure than outside ($P = 0.031$). There was not an apparent relationship between duration of exclusion and amount of *Poa secunda* basal cover differences between grazed and excluded sites ($r = -0.11$, $P = 0.64$). *Poa secunda* basal cover decreased with grazing exclusion ($P \leq 0.05$), based on meta-analysis of the 15 sites.

Average cover of perennial forbs ranged from trace amounts to 24% at the 19 exclosure sites. Only at 1 site (Boneyard Gulch) did cover of perennial forbs differ significantly between treatments, with about a threefold greater forb cover outside the exclosure ($P \leq 0.001$). *Phlox* spp. and *Lomatium* spp. were the principal contributors to greater cover outside the Boneyard Gulch exclosure. There was no apparent

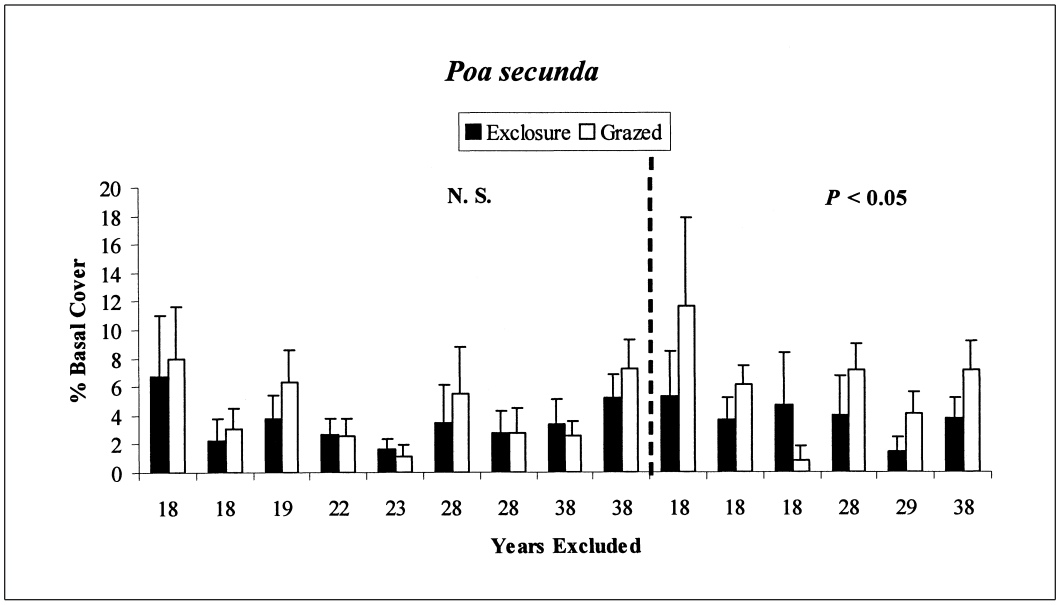

Fig. 5. Comparison between exclosure sites and adjacent grazed sites for cover of *Poa secunda*, arranged by number of years of grazing exclusion within the exclosure. **N. S.** groups those sites where the difference between exclosure and grazed sites was not statistically significant; $P < 0.05$ groups those sites where treatment differences were statistically significant. Error bars denote 95% confidence intervals.

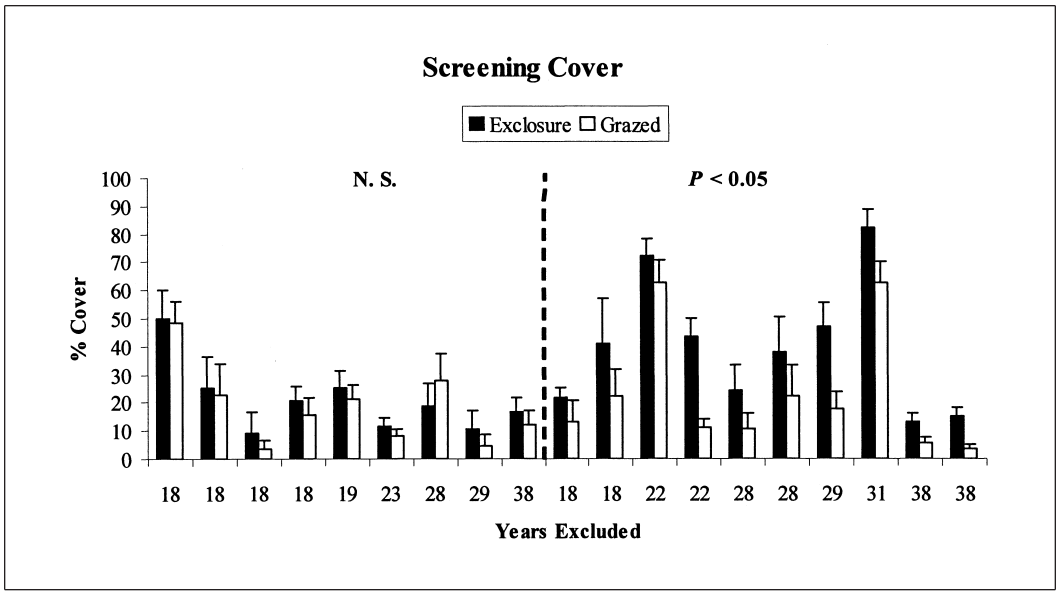

Fig. 6. Comparison between exclosure sites and adjacent grazed sites for screening in the herb–low shrub layer, arranged by number of years of grazing exclusion within the exclosure. **N. S.** groups those sites where the difference between exclosure and grazed sites was not statistically significant; $P < 0.05$ groups those sites where treatment differences were statistically significant. Error bars denote 95% confidence intervals.

treatment effect on perennial forb cover analyzed across all exclosure sites by meta-analysis ($P \geq 0.05$).

Screening cover in the herb–low shrub layer was significantly greater inside exclosures at 10 of 19 sites ($P \leq 0.05$; Fig. 6). Jeff Flats No. 1 was the only exclosure site where screening cover averaged greater outside the exclosure than inside. The relationship between years of grazing exclusion and differences between treatments of herb–low shrub layer screening cover was not significant ($r = 0.10$, $P = 0.70$). Meta-analysis indicated that grazing exclusion resulted in increased screening cover in the herb–low shrub layer ($P \leq 0.05$).

Artemisia tridentata occurred at 14 of 19 sites. Canopy cover was similar between exclosures and adjacent areas open to grazing. Meta-analysis indicated no effect on *A. tridentata* cover due to grazing exclusion. The lack of treatment effect remained true when the 2 subspecies of *A. tridentata* encountered, *A. t.* ssp. *wyomingensis* and *vaseyana*, were analyzed separately.

DISCUSSION

Each exclosure site represented a case study of the effects of grazing exclusion. Exclosures varied in size (which affected sample sizes), vegetation types (which affected types of treatment responses possible), and number of years of exclusion (which could limit the magnitude of potential treatment responses). Because of these differences, variability of ecological effects and effect sizes should be expected, and meta-analyses are recommended to identify treatment effects for these situations of variable results commonly encountered in field studies (Johnson 2002).

Many differences were evident at most Challis exclosure sites that can be attributed to the exclusion of livestock grazing. These include reduction of bare ground cover and reduction of evidence of soil erosion, increased principal forage cover, increased cover of cryptogams, and increased screening cover. These differences are consistent with those reported in reviews of the ecological effects of livestock grazing (e.g., Fleischner 1994, Belsky and Blumenthal 1997). At least some of these differences were evident at most, but not all, exclosure sites, indicating that environmental complexity and site history are integral to understanding community responses to livestock grazing or its exclusion.

Despite changes in grazing management, continued livestock grazing may hinder the rate or magnitude of vegetation response compared to livestock exclusion. For example, the Idaho National Engineering and Environmental Laboratory (INEEL), a 2315-km^2 reserve about 65 km south of the study area, has been protected from livestock grazing for 45 years. Shrub cover increased (during the first 25 years after livestock exclusion), followed by increased grass abundance, increased average species richness, and increased vegetation heterogeneity over the next 20 years (Anderson and Inouye 2001). Vegetation outside the INEEL, which was available for livestock grazing, showed a similar although less pronounced pattern of vegetation change.

West et al. (1984), studying *Artemisia tridentata* ssp. *tridentata* shrub steppe (annual precipitation = 280–347 mm), reported that exclusion of livestock would not necessarily improve native perennial grass biomass. They stated that disturbance was "mandatory" to return these semiarid communities to dominance by perennial grasses. However, in the absence of obvious disturbance, many of the plant communities inside exclosures reported here showed improvements based on indicators of rangeland health (National Research Council 1994, Pellant et al. 2000). Valone et al. (2001), working on arid grassland sites in Arizona (annual precipitation = 222–376 mm), suggested that there might be time lags of 20 years or more before perennial grasses respond to livestock grazing removal. For the exclosures reported here, some exclosures in place for >30 years showed no difference for principal grass basal cover, while some exclosures <20 years old had greater grass cover inside exclosures. The lack of correlation between period of grazing exclusion and vegetation response suggests that site history and site potential may be important factors determining rates of vegetation recovery.

Pseudoroegneria spicata is a principal livestock forage species on these rangelands comprising as much as 80% of cattle forage use in *Artemisia tridentata wyomingensis* communities (Yeo 1981). Results presented here suggest that cattle preference for *P. spicata* was

suppressing its recovery, a situation in which *Poa secunda* was apparently capitalizing. *Poa secunda* provides less forage and lower nutritive quality than *Pseudoroegneria spicata* (Willms and McLean 1978) and, because of its low growth form, affords less cover for wildlife.

Independent evidence for the Challis area also suggests that livestock grazing practices have not allowed *P. spicata* to recover from past overgrazing. Range trend monitoring transects were implemented in the BLM's Challis Resource Area in the 1950s. Reexamination of those that could be found in 2001 (n = 11 transect clusters) suggested that between the 1950s and 1970, vegetation cover remained sparse (Yeo 2001). Even though average vegetation cover was 26% greater in 2001 than in the 1950s and 1970, *P. spicata,* typically confined to protection within shrub canopies in the 1950s, showed little change in cover by 2001 and was still confined to shrub canopies. However, periods of rest between livestock grazing may result in improvements of *P. spicata* cover in the Challis area, at least within the drier vegetation types. Within 10 years of implementation of a rest-rotation grazing system and with rest periods of ≥3 years, *P. spicata* cover increased 42% in *Artemisia tridentata* ssp. *wyomingensis*/*Pseudoroegneria spicata* communities (Yeo et al. 1990).

Cryptogams decline under livestock foraging and trampling (Rice and Westoby 1978, Anderson et al. 1982). Anderson et al. (1982) reported that cryptogams could recover within 20 years if protected from livestock grazing and trampling. This study's results indicate that cryptogams are impacted by livestock grazing and that recovery is time related. Cryptogams are an important component of xeric landscapes of the West, contribute to soil stabilization and soil moisture retention, influence nitrogen cycling, and may aid seedling establishment (West 1990, Belnap 2000).

Grazing exclusion resulted in greater screening cover in the herb–low shrub layer that has implications for wildlife. Guidelines for habitat management for Greater Sage-Grouse (Connelly et al. 2000), pronghorn (Autenreith 1978, Allen et al. 1984), and sagebrush-dependent songbirds (Paige and Ritter 1999) recommend mosaics of native sagebrush communities with productive herbaceous understories. These communities afford thermal and security cover for nests and young fawns, as well as forage for pronghorn and Greater Sage-Grouse and productive communities of insects as food for Greater Sage-Grouse and songbirds.

These differences indicate that despite improved livestock grazing management in the past half century, continued livestock grazing has limited the potential of some of these native rangeland communities, or at least slowed their recovery relative to grazing exclusion. Greater productivity (particularly of the herbaceous understory), greater species richness, greater extent of cryptogamic soil crusts with less bare ground and less evidence of soil erosion all are signs of better rangeland health (National Research Council 1994, Pellant et al. 2000) and benefit not only wildlife with increased forage and cover but also livestock with increased forage availability. These differences were not evident at all sites, and range managers should be cautious in their expectations of ecosystem responses to changes in grazing management. Plant species respond individually in a nonlinear fashion and at different rates to disturbance or ecosystem stressors, such as drought (Anderson and Inouye 2001), so land managers should expect unpredictable variability to management actions at a local level. The results reported here, coupled with other evidence reported in the literature, clearly indicate the need for monitoring to guide land management and the worth of contrastive experiments such as exclosures to act as controls or references as part of that monitoring (Ford 2000).

ACKNOWLEDGMENTS

The research was funded through a Bureau of Land Management Challenge–Cost Share agreement with the Challis Field Office. J. Gregson, W. Diage, W. Osborne, and E. Williams of the BLM Challis Field Office provided help throughout the study. K. Rodgers, R. Camper, and K. Gallogly, Salmon-Challis National Forest, found records and soils information for the 2 exclosures on U.S. National Forest Service lands. M. Olson, Natural Resources Conservation Service, Challis Office, and M. Scott, Idaho Department of Fish and Game, Salmon Office, gave access to their exclosure records. I thank J. Gregson, W. Diage, and 2 anonymous reviewers for many helpful comments on earlier drafts.

LITERATURE CITED

ALLEN, A.W., J.G. COOK, AND M.J. ARMBRUSTER. 1984. Habitat suitability index models: pronghorn. U.S. Fish and Wildlife Service, FWS/0BS-82/10.65.

ANDERSON, D.C., K.T. HARPER, AND S.R. RUSHFORTH. 1982. Recovery of cryptogamic soil crusts from grazing on Utah winter ranges. Journal of Range Management 35:355–359.

ANDERSON, J.E., AND R.S. INOUYE. 2001. Landscape-scale changes in plant species abundance and biodiversity of a sagebrush steppe over 45 years. Ecological Monographs 71:531–556.

AUTENRIETH, R., EDITOR. 1978. Guidelines for the management of pronghorn antelope. Pages 473–526 *in* Proceedings of the Eighth Biennial Pronghorn Antelope Workshop, 2–4 May 1978, Jasper, Alberta.

BARBOUR, M.G., J.H. BURK, AND W.D. PITTS. 1980. Terrestrial plant ecology. Benjamin/Cummings Publishing Company, Inc., Menlo Park, CA.

BELNAP, J. 2000. Structure and function of biological soil crusts. Pages 55–62 *in* P.G. Entwistle, A.M. DeBolt, J.H. Kaltenecker, and K. Steenhof, compilers, Proceedings: Sagebrush Steppe Ecosystems Symposium. Bureau of Land Management Publication BLM/ID/PT-001001+1150, Boise, ID.

BELSKY, A.J., AND D.M. BLUMENTHAL. 1997. Effects of livestock grazing on stand dynamics and soils in upland forests of the interior west. Conservation Biology 11: 315–327.

BORK, E.W., N.E. WEST, AND J.W. WALKER. 1998. Threetip sagebrush steppe responses to long-term seasonal sheep grazing. Journal of Range Management 51: 293–300.

BRISKE, D.D., AND J.H. RICHARDS. 1994. Physiological responses of individual plants to grazing: current status and ecological significance. Pages 147–176 *in* M. Vavra, W.A. Laycock, and R.D. Pieper, editors, Ecological implications of livestock herbivory in the west. Society of Range Management, Denver, CO.

BUREAU OF LAND MANAGEMENT. 1973. Determination of erosion condition class, Form 7310-12. U.S. Department of the Interior, Washington, DC.

_____. 1998. Challis Resource Area. Proposed Resource Management Plan and Environmental Impact Statement. U.S. Department of Interior Bureau of Land Management, Salmon, ID.

CLEMENTS, F.E. 1916. Plant succession: an analysis of the development of vegetation. Carnegie Institution of Washington, Publication 242, Washington, DC.

CONNELLY, J.W., M.A. SCHROEDER, A.R. SANDS, AND C.E. BRAUN. 2000. Guidelines to manage Sage Grouse populations and their habitats. Wildlife Society Bulletin 28:967–985.

DAUBENMIRE, R.F. 1959. Canopy coverage method of vegetation analysis. Northwest Science 33:43–64.

DONAHUE, D.L. 1999. The western range revisited: removing livestock from public lands to conserve native biodiversity. University of Oklahoma Press, Norman.

DYKSTERHUIS, E.J. 1949. Condition and management of range land based on quantitative ecology. Journal of Range Management 2:104–115.

FLEISCHNER, T.L. 1994. Ecological costs of livestock grazing in western North America. Conservation Biology 8:629–644.

FLOYD, D.A., AND J.E. ANDERSON. 1982. A new point interception frame for estimating cover of vegetation. Vegetatio 50:185–186.

FORD, E.D. 2000. Scientific method for ecological research. Cambridge University Press, Cambridge, UK.

GRIFFITH, D.B., AND B.A. YOUTIE. 1988. Two devices for estimating foliage density and deer hiding cover. Wildlife Society Bulletin 16:206–210.

HANLEY, T.A. 1979. Application of an herbivore-plant model to rest-rotation grazing management on shrub-steppe vegetation. Journal of Range Management 32:115–118.

HULL, A.C. 1976. Rangeland use and management in the Mormon West. *In:* Symposium on Agriculture, Food and Man—Century of Progress. Brigham Young University, Provo, UT.

HUSCHLE, G., AND M. HIRONAKA. 1980. Classification and ordination of seral plant communities. Journal of Range Management 33:179–182.

JOHNSON, D.H. 2002. The importance of replication in wildlife research. Journal of Wildlife Management 66:919–932.

LAYCOCK, W.A. 1994. Implications of grazing vs. no grazing on today's rangelands. Pages 250–280 *in* M. Vavra, W.A. Laycock, and R.D. Pieper, editors, Ecological implications of livestock herbivory in the West. Society of Range Management, Denver, CO.

NATIONAL RESEARCH COUNCIL. 1994. Rangeland health: new methods to classify, inventory, and monitor rangelands. National Academy Press, Washington, DC.

NATURAL RESOURCES CONSERVATION SERVICE. 2002a. Custer-Lemhi Counties, Idaho, soil survey. USDA Natural Resources Conservation Service, Boise, ID.

_____. 2002b. The PLANTS database. Version 3.5 (http://plants.usda.gov). U.S. Department of Agriculture National Plant Data Center, Baton Rouge, LA.

NOSS, R.F. 1994. Cows and conservation biology. Conservation Biology 8:613–616.

NOY-MEIR, I. 1975. Stability of grazing systems: an application of predator-prey graphs. Journal of Ecology 63:459–481.

PAIGE, C., AND S.A. RITTER. 1999. Birds in a sagebrush sea: managing sagebrush habitats for bird communities. Partners in Flight Western Working Group, Boise, ID.

PELLANT, M., P. SHAVER, D.A. PYKE, AND J.E. HERRICK. 2000. Interpreting indicators of rangeland health. Version 3. USDI Bureau of Land Management Technical Reference 1734-6, Publication BLM/WO/ST-00/001+1734.

PIEPER, R.D. 1994. Ecological implications of livestock grazing. Pages 177–211 *in* M. Vavra, W.A. Laycock, and R.D. Pieper, editors, Ecological implications of livestock herbivory in the west. Society of Range Management, Denver, CO.

RICE, B., AND M. WESTOBY. 1978. Vegetative responses of some Great Basin shrub communities protected against jackrabbits or domestic stock. Journal of Range Management 31:28–34.

ROSENBERG, M.S., D.C. ADAMS, AND J. BUREVITCH. 2000. MetaWin: statistical software for meta-analysis. Version 2. Sinauer Associates, Sunderland, MA.

SHOUP, G.E. 1935. Ranges of the Salmon River. Unpublished manuscript, Bureau of Land Management records, Salmon, ID.

TAUSCH, R.J., P.E. WIGAND, AND J.W. BURKHARDT. 1993. Plant community thresholds, multiple steady states, and multiple successional pathways: legacy of the

Quaternary? Journal of Range Management 46: 439–447.

VALONE, T.J., M. MEYER, J.H. BROWN, AND R.M. CHEW. 2001. Timescale of perennial grass recovery in desertified arid grasslands following livestock removal. Conservation Biology 16:995–1002.

WEST, N.E. 1990. Structure and function of soil microphytic crusts in wildland ecosystems of arid and semiarid regions. Advances in Ecological Research 20:179–223.

_____. 2000. Synecology and disturbance regimes of sagebrush steppe ecosystems. Pages 15–26 *in* P.G. Entwistle, A.M. DeBolt, J.H. Kaltenecker, and K. Steenhof, compilers, Proceedings: Sagebrush Steppe Ecosystems Symposium. Bureau of Land Management Publication BLM/ID/PT-001001+1150, Boise, ID.

WEST, N.E., F.D. PROVENZA, P.S. JOHNSON, AND M.K. OWENS. 1984. Vegetation change after 13 years of livestock exclusion on sagebrush semidesert in west central Utah. Journal of Range Management 37: 262–264.

WESTOBY, M., B. WALKER, AND I. NOY-MEIR. 1989. Opportunistic management of rangelands not at equilibrium. Journal of Range Management 42:266–274.

WHITFORD, W.G. 1995. Desertification: implications and limitations of the ecosystem health metaphor. *In:* D.J. Rapport, C.L. Gaudet, and P. Calow, editors, Evaluating and monitoring the health of large-scale ecosystems. NATO ASI Series, Volume 28. Springer-Verlag, Berlin.

WILLMS, W., AND A. MCLEAN. 1978. Spring forage selection by tame mule deer on big sagebrush range, British Columbia. Journal of Range Management 31: 192–199.

YEO, J.J. 1981. The effects of rest-rotation grazing on mule deer and elk populations inhabiting the Herd Creek allotment, East Fork Salmon River, Idaho. Master's thesis, University of Idaho, Moscow.

_____. 2001. Long-term vegetation trend in the BLM Challis Resource Area, east central Idaho. Unpublished report to the Challis Field Office, Bureau of Land Management, Challis, ID.

YEO, J.J., W.T. WITTINGER, AND J.M. PEEK. 1990. Vegetation changes on a rest-rotation grazing system. Rangelands 12:220–225.

Received 25 March 2003
Accepted 8 March 2004