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Measuring Plant Diversity in the Tall Threetip Sagebrush Steppe: Influence of Previous Grazing Management Practices

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ABSTRACT / In July 2000, a 490-ha wildfire burned a portion of a long-term grazing study that had been established in 1924 at the US Sheep Experiment Station north of Dubois, Idaho, USA. Earlier vegetation measurements in this tall threetip sagebrush (*Artemisia tripartita* spp. *tripartita*) bunchgrass plant community documented significant changes in vegetation due to grazing and the timing of grazing by sheep. A study was initiated in May 2001 using 12 multiscale modified Whittaker plots to determine the consequences of previous grazing practices on postfire vegetation composition. Because there was only one wildfire and it did not burn all of the original plots, the treatments are not replicated in time or space. We reduce the potential effects of psuedoreplication by confining our discussion to the sample area only. There were a total of 84 species in the sampled areas with 69 in the spring-grazed area and 70 each in the fall- and ungrazed areas. Vegetation within plots was equally rich and even with similar numbers of abundant species. The spring-grazed plots, however, had half as much plant cover as the fall- and ungrazed plots and the spring-grazed plots had the largest proportion of plant cover composed of introduced (27%) and annual (34%) plants. The fall-grazed plots had the highest proportion of native perennial grasses (43%) and the lowest proportion of native annual forbs (1%). The ungrazed plots had the lowest proportion of introduced plants (4%) and the highest proportion of native perennial forbs (66%). The vegetation of spring-grazed plots is in a degraded condition for the environment and further degradation may continue, with or without continued grazing or some other disturbance. If ecosystem condition was based solely on plant diversity and only a count of species numbers was used to determine plant diversity, this research would have falsely concluded that grazing and timing of grazing did not impact the condition of the ecosystem.

In the summer of 2000, wildfire burned a portion of a long-term grazing study. The grazing study had been established in 1924 at the US Sheep Experiment Station (USSES) north of Dubois, Idaho, USA (Craddock and Forsling 1938) in a tall threetip sagebrush (*Artemisia tripatita* spp. *tripatita*) bunchgrass community. Earlier vegetation studies (Mueggler 1950, Laycock 1967, Bork and others 1998) at this location documented significant changes in vegetation due to sheep grazing and the timing of grazing. These studies concluded that spring grazing reduced perennial herbs and increased annual herbs and sagebrush cover compared to fall and no grazing. In addition, fall grazing decreased sagebrush cover compared to spring grazing and no grazing. Given its well-documented vegetation and grazing intensity history (over 70 years of studies) and because tall threetip sagebrush will occasionally resprout after fire (Blaisdell and others 1982), this site lent itself well to a study of post-fire vegetation recovery in the sagebrush steppe.

KEY WORDS: Biodiversity; Seasonal grazing; Wildfire; Modified Whittaker plots; Sagebrush steppe; Tall threetip sagebrush

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Although wildfire is a natural part of the sagebrush steppe ecosystem (Blaisdell and others 1982), landowners and federal agencies have actively suppressed it while encouraging prescribed fire to increase grazable forage. The resultant changes in fuel loads, combined with fragmentation of the native vegetation by roads, farms, and urban areas, utilization of the remaining range by livestock, and invasion by introduced plants, has impacted the overall function of this ecosystem in ways largely unknown.

Some research has been conducted to determine successional changes following fire in the sagebrush steppe. In west-central Utah, with a Clementsian model, Barney and Frischknecht (1974), using data collected from sites that had burned at different times, concluded that an annual weedy stage was followed by a perennial grass/forb stage 3 or 4 years after a fire.

These sites were next dominated by big sagebrush (*Artemisia tridentata*) about 35 years after fire, Utah juniper (*Juniperus osteosperma*) about 70 years after fire, and reached a climax juniper woodland 85–90 years after fire. The development time of these stages varied considerably depending on the size and timing of the burn, weather, grazing, and seed dissemination. The weedy annual phase might not occur if the area had a healthy component of perennial grasses and forbs prior to the burn. In a study following a controlled burn in mountain big sagebrush (*A. tridentata* spp. *vaseyana*) steppe in southeastern Idaho, the annual weedy stage was skipped, where over 33% of the prefire biomass was perennial grasses and forbs (Harniss and Murray 1973). Sagebrush levels began to increase after 12 years and reached prefire levels 18 years later, 30 years after the fire. Wambolt and others (2001) compared 13 prescribed burn areas with adjacent unburned areas in southwestern Montana. Sagebrush recovery was irregular with some areas reestablishing readily, while others had very little sagebrush cover 15 years after fire. Herbaceous response was highly variable despite the removal of the competing shrub layer. In all of the above research, plant diversity before or after the fire was not measured.

Biodiversity as a measure of ecosystem condition has been readily adopted by the public (Redford and Sanderson 1992). This concern for biodiversity has resulted in various types of legislation, such as the National Forest Management Act of 1976 (NFMA) and the Surface Mining Control and Reclamation Act of 1977 (SMCRA). Biodiversity has often focused on species numbers because it is easy to measure and explain, but there are more useful ways to describe and measure biodiversity (West 1993). Among ecologists, biodiversity is often expressed as the number of species in a community and the relative abundances of these species (Peet 1974). Several diversity indices have been developed for characterizing plant communities, but all have limitations due to incorrect assumptions and the confounding of community structure variables (Ludwig and Reynolds 1988). As rangeland management decisions will continue to be based on improving ecosystems and increasing biodiversity, it is important to determine if changes in plant diversity can be adequately measured in the sagebrush steppe.

With a hypothesis that prefire vegetation and the resultant seed banks would influence regeneration after fire (West and Hassan 1985), we believed the wildfire would provide us with an excellent opportunity to measure plant diversity in the sagebrush steppe and to determine the consequences of previous grazing practices on post-fire plant diversity. The objectives of our research were to: (1) determine differences in reestablishing vegetation cover following fire based on grazing history, and (2) determine the usefulness of diversity indices in measuring vegetation in the sagebrush steppe.

Materials and Methods

Research was conducted during June and July of 2001 (one year after burn) at the US Sheep Experiment Station, approximately 10 km north of Dubois, Idaho, in the Upper Snake River plain (44°14'44″ N latitude, 112°12'47″ W. Longitude). The site is at an elevation of 1650 m and is in the northeastern part of the sagebrush steppe region (West 1983). The vegetation is a mix of tall threetip sagebrush, bluebunch wheatgrass (*Pseudoroegneria spicata* ssp. *spicata*), and arrowleaf balsamroot (*Balsamorhiza sagittata*). Soils are fine-loamy, mixed, frigid Calcic Argixerolls derived from wind-blown loess, residuum, or alluvium on slopes ranging from 0 to 12% (Natural Resources Conservation Service 1995). Climate is semiarid with cold winters and warm summers. Annual precipitation for the last 69 years averaged 325 mm (Anonymous 1999). At a rain gauge less than 2 km from the study, precipitation was 253 mm in the 12 months preceding the fire and was 213 mm in the 12 months following the fire.

The research site is contained within a long-term sheep grazing study area (65 ha). The study area was established in 1924 to measure the effects of grazing seasonality by comparing spring and fall grazing to fall only grazing (Pechanec and Stewart 1949). Stocking rates and timing of grazing varied early in the study, and ungrazed control pastures were not added until 1941 and 1950. For the last 50 years the fall-grazed plots have received 150 sheep days ha of use and the spring-grazed plots have been stocked with 100 sheep days ha (Bork and others 1998). On 31 July 2000, a 490-ha wildfire burned through a portion of the US Sheep Experiment Station, including a portion of the long-term grazing study (Figure 1). The ungrazed control plots and most of a spring-grazed and a fall-grazed plot were burned. To sample plant diversity and vegetation recovery, 12 modified Whittaker plots (Stohlgren and others 1998), four in each grazing treatment, were established in the spring of 2001 (Figure 1). Placement of plots was random in areas that had a similar soil type (Stohlgren and others 1999) and did not contain rock outcrops or unburned vegetation. Our reasons for the exclusions of rock and unburned areas were that rock outcrops contain a different vegetation component and occupy small areas of the site, and unburned areas were not the focus of this study. The modified Whittaker plot (Figure 2) was chosen as it samples a large area (0.1 ha) and is useful for detecting less abundant plant species (Stohlgren and others 1998).

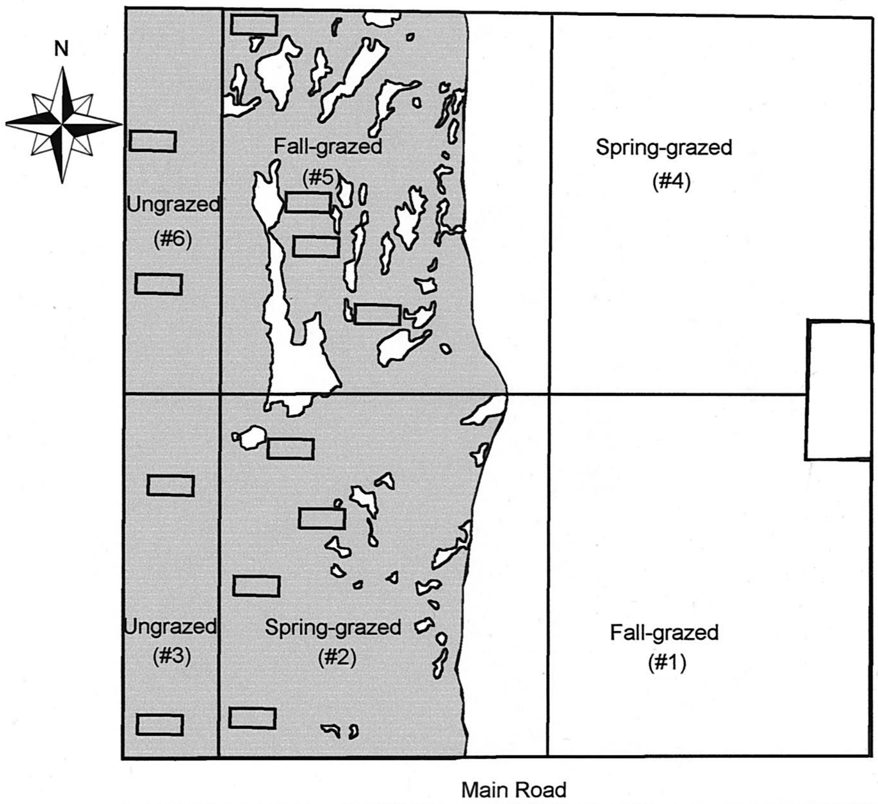

Figure 1. Burned area (gray) of the seasonal grazing paddocks at the UDSA/ARS Sheep Experiment Station, Dubois, Idaho. Pastures 2 and 4 are spring-grazed, pastures 1 and 5 are fall-grazed, and pastures 3 and 6 are ungrazed. Rectangles are the locations of sample plots.

At peak biomass (30 May–15 June) the year following the fire, the plots were sampled according to the protocol established by Stohlgren and others (1998). Each 20 × 50-m plot (1000 m^2) had nested in it one 5 × 20-m plot (100 m^2), two 2 × 5-m plots (10 m^2) and ten 0.5 × 2-m plots (1 m^2; Figure 2). In the 1-m^2 plots, all plant species were identified, and the percentage cover of each species (at maximum crown diameter), bare ground, and other nonplant components (rock, litter, feces and wood) were estimated. Other plant species found in the 10-m^2, 100-m^2, and 1000-m^2 areas were added to the list of species found in the 1-m^2 plots. This list was used to generate species area curves (species richness = $mx + b$, where m is the slope of the line, x is logArea, and b is the y intercept) (Stohlgren and others 1995). All plot corners were located using a global positioning system (± 1 m) to facilitate future resampling.

Jaccard's coefficient (Krebs 1989) was used to compare species overlap between: (1) spring-grazed and fall-grazed plots; (2) spring- and ungrazed plots; and (3) fall- and ungrazed plots with data from the 1000-m^2 plot. Jaccard's coefficient (J), which gives equal weight to all plant species, is derived from (equation 1):

$$J = A/(A + B + C), \qquad (1)$$

where A is the number of species found in both sites, B is the number of species found only in the first site, and C is the number of species found only in the second site (Krebs 1989). Jaccard's coefficient varies from 1 (complete overlap) to 0 (no overlap) and is a good similarity measurement (Stohlgren and others 1997). It is often expressed in percent as $J \times 100$.

Species diversity in the study was compared using Shannon's index and Simpson's index (Ludwig and Reynolds 1988). Shannon's index (H') for a sample is defined as (equation 2):

$$\check{H}' = \sum_{i=1}^{S} \left[\left(\frac{n_i}{n}\right) \ln\left(\frac{n_i}{n}\right) \right] \qquad (2)$$

where n_i is the cover of the ith species of S species in the sample and n is the total cover of all species in the sample. Simpson's index (λ) for a sample was defined as (equation 3):

$$\lambda = \sum_{i=1}^{S} \frac{n_i(n_i - 1)}{n(n - 1)} \qquad (3)$$

The values from these indices were then combined in a method recommended by Ludwig and Reynolds (1988), which differ from the similarity analysis of the

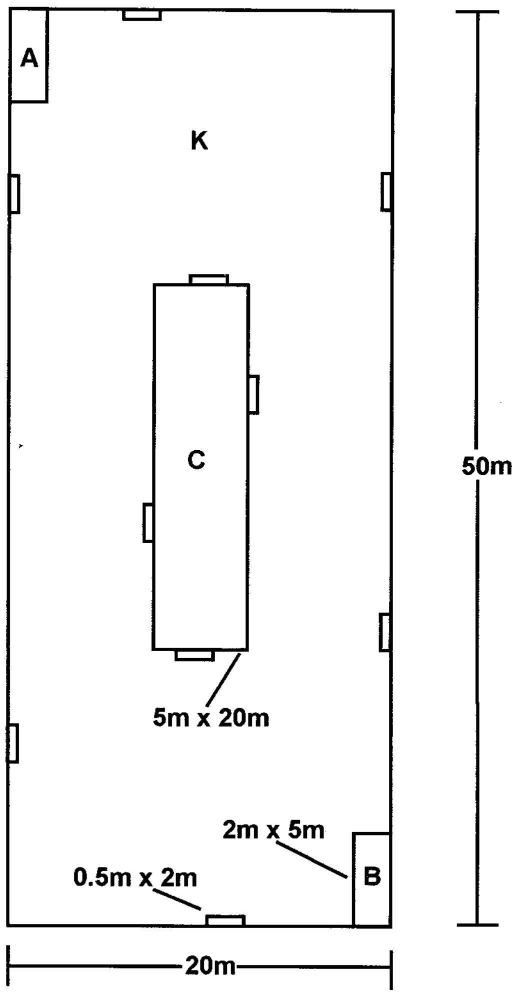

Figure 2. Sampling layout for the modified Whittaker plots with ten 1-m^2 plots, two 10-m^2 plots (A and B), one 100-m^2 plot (C), and one 1000-m^2 plot (K).

Jaccard's coefficient by weighting the abundant (N1) and very abundant species (N2). N1 was calculated as

$$N1 = e^{H'}$$

and N2 was calculated as

$$N2 = 1/\lambda.$$

With the values from the above equations, a modified Hill's ratio can then be determined as a measure of evenness (E5) (Ludwig and Reynolds 1988). $E5$ was calculated as (equation 4):

$$E5 = \frac{(1/\lambda) - 1}{e^{H'} - 1} = \frac{N2 - 1}{N1 - 1} \qquad (4)$$

As $E5$ approaches zero one species becomes more dominant in the total cover component. Higher values of $E5$ indicate a more even division of cover among the species in the sample area.

Differences in plant cover were determined using data collected in the 1-m^2 plots (40 per grazing treatment). To satisfy assumptions of normality, all data were $\log_{10}$-transformed after adding 1. Because of the inherent site variability and the scarcity of plants growing after the fire, probably magnified by the drought, cover analysis by individual plant species was not possible. Therefore, all plants were grouped into the following categories for percentage cover analysis: introduced annual forbs, introduced annual grasses, introduced perennial forbs, introduced perennial grasses, native annual forbs, native perennial forbs, native perennial grasses, shrubs, and total based on Hitchcock and Cronquist (1981). There were no native annual grasses at this location. Additional cover analyses were conducted to determine differences in the following broader categories: introduced, natives, forbs, grasses, annuals, and perennials, as these can be important for land management decisions.

An analysis of variance (ANOVA) was used to determine differences among spring-, fall-, and ungrazed pastures for these groups of plants using a randomized complete block design. A Tukey's test was used to compare means when the F test was significant ($P < 0.05$). All statistical analyses were conducted with SAS (version 8, SAS Institute, Cary, North Carolina, USA). The same ANOVA procedures were used to compare differences in species composition, diversity, and evenness.

Results

As a measure of species richness, there were a total of 84 species in the sampled areas (Appendix). At the 1-m^2 scale there were 55 species in the ungrazed plots, 56 in the fall-grazed plots, and 52 in the spring-grazed plots. At the 1000-m^2 scale, the similarity was greater with 70 species in the ungrazed and fall-grazed plots and 69 in the spring-grazed plots (Table 1). Generally as the sample area increased, the number of species found increased with virtually no differences among grazing treatments. Introduced species ranged from 13% to 19%, grass species ranged from 13% to 17% and annuals species were about 25% of the total species in the 1- and 1000-m^2 plot areas, respectively (Table 1). The species area curve for the combined modified Whittaker plots at this site was $y = 5.41 \times \mathrm{Ln}(x) + 12.56$, where y is the total number of species, with an R^2 of 0.91. There were no differences in the slopes of the curves among the three grazing treatments (data not shown).

Based on Jaccard's coefficient, on average, there was 65% overlap in species composition between spring-grazed versus fall-grazed plots, spring-grazed versus un-

Table 1. Species richness as a function of plot size and plant category of plant populations recovering from fire and previous grazing practices

	1-m² plots				1000-m² plots			
	Ungrazed	Fall-grazed	Spring-grazed	Total	Ungrazed	Fall-grazed	Spring-grazed	Total
Introduced annual forbs	4	7	5	9	6	8	8	10
Introduced annual grasses	1	1	1	1	1	1	1	1
Introduced perennial forbs	1	1	2	2	2	1	2	3
Introduced perennial grasses	1	0	0	1	1	1	2	2
Native annual forbs	5	7	8	9	9	8	9	11
Native perennial forbs	31	29	25	35	37	37	31	41
Native perennial grasses	7	7	6	7	7	7	9	9
Native shrubs	5	4	5	6	7	7	7	7
Introduced plants[a]	7	9	8	13	10	11	13	16
Native plants	48	47	44	57	60	59	56	68
Forbs	41	44	40	55	54	54	50	65
Grasses	9	8	7	9	9	9	12	12
Annuals	10	15	14	19	16	17	18	22
Perennials	45	41	38	51	54	53	51	62
Total	55	56	52	70	70	70	69	84

[a]Numbers in the introduced plants, native plants, forbs, grasses, annuals and, perennials categories are summations of species from the first eight categories.

Table 2. Measures of species composition (Shannon's index and Simpson's index), diversity (*N*1 and *N*2), and evenness (*E*5) from percent cover estimates of 1-m² plots in spring-grazed, fall-grazed, and ungrazed plots

	Shannon's index	Simpson's index	*N*1	*N*2	*E*5
Spring-grazed	2.1	0.18 a[a]	8.2	6.4	0.73
Fall-grazed	1.9	0.22 ab	7.3	5.6	0.72
Ungrazed	1.9	0.23 b	7.1	5.3	0.68
	(0.18)[b]	(0.05)	(1.2)	(1.1)	(0.06)

[a]Means within a column followed by the same letter are not different ($P > 0.05$).
[b]Minimum significant difference.

grazed plots and fall-grazed versus ungrazed plots, with 60–62 species in common and 8–10 species in one treatment but not in another. When the plant species from each of the four 1000-m² plots in a grazing treatment were combined, there was 78% overlap in species composition between the spring-grazed versus the fall-grazed areas, 76% for the spring-grazed versus the ungrazed areas, and 79% for the fall-grazed versus the ungrazed areas.

There were no differences in Shannon's index, diversity (N1, N2) or evenness (E5) among grazing treatments (Table 2). A small difference was detected between the Simpson's index values of the spring-grazed and ungrazed treatments ($P = 0.046$). There were significant grazing treatment by plot interactions for all the measures except evenness. Coefficient of variation values ranged from 15.9 to 46.3, and R^2 values were 0.1 to 0.3 in all the measures. The coefficient of variation is a relative measure of variation, and the values obtained in this study would indicate a high level of variation among the plots. The R^2 indicates how well the data are described by the equation used in the analysis, and the values obtained in this study indicate low predictability. The condition of a high level of variability and correspondingly low predictability is a good description of the vegetation in the sagebrush steppe where many plants occur in discrete patches.

There were differences among plant categories when analyzed by percentage cover (Table 3). Normalized plant cover (percentage cover was normalized by dividing the percentage cover by the total percentage cover of the grazing treatment) could not be statistically analyzed, but is listed in Table 3. Overall, the vegetation in the study area was 60% forbs and 34% grasses. The ungrazed treatment had the highest forb cover and the fall-grazed plots had the highest grass cover (Table 3). Percentage cover by annual species was highest and cover by perennials was lowest in the spring-grazed treatment (Table 3). The ungrazed treatment had less introduced annual forb cover than the spring- and fall-grazed treatments. The only introduced annual grass was downy brome (*Bromus tectorum*), and the percentage cover of this species was highest in the spring-grazed treatment and lowest in the ungrazed treatment.

Table 3. Percent cover (actual and normalized) of plant categories of plant populations recovering from fire and previous grazing practices

	Actual percent cover			Normalized percent cover		
	Ungrazed	Fall-grazed	Spring-grazed	Ungrazed	Fall-grazed	Spring-grazed
Introduced annual forbs	0.5 b[a]	0.9 a	1.5 a	2	5	14
Introduced annual grasses	0.3 c	0.5 b	1.1 a	1	3	10
Introduced perennial forbs	0.0 b	0.0 b	0.3 a	0	0	3
Introduced perennial grasses	0.05	0.0	0.0	0	0	0
Native annual forbs	1.1 a	0.2 b	1.1 a	6	1	10
Native perennial forbs	13.0 a	7.3 b	3.6 c	66	40	33
Native perennial grasses	3.9 b	8.0 a	2.8 b	20	43	26
Native shrubs	0.9	1.3	0.4	4	7	3
Introduced plants[b]	0.8 c	1.5 b	2.9 a	4	8	27
Native plants	18.9 a	16.9 a	7.9 b	96	92	73
Forbs	14.6 a	8.5 b	6.5 b	74	46	60
Grasses	4.2 b	8.5 a	3.9 b	22	46	36
Annuals	1.8 b	1.7 b	3.7 a	9	9	34
Perennials	17.9 a	16.7 a	7.1 b	91	91	66
Total	19.7 a	18.4 a	10.8 b	100	100	100

[a]Means with the same letter in a row are not different ($P > 0.05$).
[b]Numbers in the introduced plants, native plants, forbs, grasses, annuals, and perennials categories are summations of species from the first eight categories.

The spring-grazed treatment also had higher introduced perennial forb cover than the fall- and ungrazed treatments. Crested wheatgrass (*Agropyron cristatum*) was the only introduced perennial grass in the 1-m^2 plots and only occurred in the ungrazed plots. Crested wheatgrass, which had been planted in pastures near the experimental site, is a widely planted species in the sagebrush steppe ecosystem and was found at the 10-m^2 scale in the fall-grazed plots and at the 100 m^2 scale in the spring-grazed plots. Although not planted in this area, another introduced perennial grass (Kentucky bluegrass, *Poa pratensis*) was identified at the 1000-m^2 scale in the spring-grazed plots. The spring- and ungrazed treatments had a greater percentage cover of native annual forbs than did the fall-grazed treatment. The fall-grazed treatment, however, had greater cover of native perennial grass than the spring- and ungrazed treatments. Native perennial forb cover was greatest in the ungrazed treatment and least in the spring-grazed treatment. There was an abundance of native perennial forb species, with the most common being arrowleaf balsamroot, tapertip hawksbeard (*Crepis acuminata*), lesser rushy milk vetch (*Astragalus convallarius*), and pale bastard toadflax (*Comandra umbellata* ssp. *pallida*). There were no differences in the percentage cover of native shrubs among treatments (Table 3). Cover of the native plants group was higher in the fall- and ungrazed treatments compared to the spring-grazed treatment. The spring-grazed treatment had the highest percentage cover of introduced plants while the ungrazed treatment had the lowest (Table 3).

Postfire vegetation composition varied considerably among grazing treatments. Over a third of the vegetation cover in the spring-grazed plots was composed of annual plants, whereas the fall- and ungrazed plots had less than 10% cover composed of annual plants (Table 3). These annuals in the spring-grazed plots were evenly distributed between introduced annual forbs (14%), native annual forbs (10%), and introduced annual grasses (10%). The most common introduced annual forbs were desert alyssum (*Alyssum desertorum*), littlepod falseflax (*Camelina microcarpa*), hornseed buttercup (*Ceratocephata testiculata*), and yellow salsify (*Tragopogon dubius*), none of which have been planted in this area. The most common native annual forbs were small flower blue-eyed Mary (*Collinsia parviflora*), western stickseed (*Lappula occidentalis* var. *occidentalis*), slimleaf goosefoot (*Chenopodium leptophyllum*), and dwarf groundsmoke (*Gayophytum humile*). In the fall-grazed and ungrazed plots the percentage cover of these annuals was considerably less (Table 3). Native annual forbs cover was a very small component in the fall-grazed plots (0.2%), but had a higher cover in the ungrazed and spring-grazed plots (1.1%). As a proportion of the total vegetation cover, however, native annual forbs comprised 10% in the spring-grazed plots, 6% in the ungrazed plots, and 1% in the fall-grazed plots (Table 3).

Over 25% of the vegetative cover in the spring-grazed plots was composed of introduced species, compared to 8% in the fall-grazed plots and only 4% in the ungrazed plots. About half the introduced cover was annual forbs and half annual grasses, with a small percentage of perennial forbs (Table 3). The most common of the introduced perennial forbs was common dandelion (*Taraxicum officinale*). In the spring-grazed plots there was some yellow sweetclover (*Melilotus officinalis*), and in the ungrazed plots there was some alfalfa (*Medicago sativa*), both of which have been planted within 10 km of the site. In all plots, the only introduced annual grass was downy brome and the only introduced perennial grass was crested wheatgrass.

As expected, shrub biomass declined considerably after the fire from over 20% (Bork and others 1998) to about 1%, composed mostly of gray horsebrush (*Tetradymia canescens*), tall threetip sagebrush, and green rabbitbrush (*Chrysothamnus viscidiflorus*) with no differences among the treatments. Most of the gray horsebrush was in the ungrazed plots, and most of the tall threetip sagebrush was in the fall-grazed plots. All the above shrubs, including some of the tall threetip sagebrush, will sprout after a fire. As a percentage of the total vegetative cover, shrubs occupied 3% of the spring-grazed plots, 7% of the fall-grazed plots, and 4% of the ungrazed plots.

Expressed as actual percentage cover, there was less grass in the ungrazed (4.2%) and spring-grazed (3.9%) plots compared to the fall-grazed (8.5%). The same is true when downy brome is removed from the grass analysis leaving only native perennial grasses (Table 3). The most common native perennial grass species were bluebunch wheatgrass, Sandberg bluegrass (*Poa secunda*), thickspike wheatgrass (*Elymus lanceolatus* ssp. *lanceolatus*), needle-and-thread grass (*Hesperostipa comata*), and prairie junegrass (*Koeleria macrantha*).

Discussion and Conclusion

The primary and untested assumption in this study was that the initial conditions of all sites were similar prior to the initiation of grazing treatments in 1924. We believe this to be generally true for soils and vegetation based on the report of Craddock and Forsling (1938). A second issue is one of psuedoreplication (Hurlbert 1984), because the treatments were not replicated in space or time, we reduce the potential effects of psuedoreplication by confining our discussion to this sample area only. We view this as an important case study of well measured, but perhaps site-specific results.

Diversity has been defined in the ecological literature as: (1) the total number of species in a community and (2) a dual concept that combines number of species with the relative abundances of the species (Peet 1974). Whittaker (1972) described three levels of diversity of interest to ecologists: (1) alpha or within-habitat diversity; (2) beta or between habitat diversity (i.e., changes along ecological gradients), and (3) gamma or large-scale landscape diversity. In this study the dual concept is used at the alpha level of diversity. Seasonality of sheep grazing has had an effect on plant composition in the long-term grazing study at the US Sheep Experiment Station (Mueggler 1950, Laycock 1967, Bork and others 1998). Continuous spring sheep grazing shifted the plant community towards higher annual plant cover and may have allowed establishment of a sizable population of some introduced species. Continuous fall grazing increased the cover of native perennial grasses but reduced native annual forbs. Exclusion from grazing may have resulted in a loss of grass cover. The wildfire may have exacerbated the effects of the grazing treatments.

The species mix in the three grazing treatments was remarkably comparable, with even numbers of similar and different species. When the various plant species were grouped into categories, there were no differences in number of species among the three grazing treatments (Table 1). The three grazing treatments had seven to eight abundant species and five to six very abundant species in the 1-m^2 plots, indicating similar degrees of diversity. The evenness index (E) was very high at 0.7 for the three treatments, again indicating that no one species dominated and there were no significant differences among the grazing treatments (Table 2).

Reinforcing the similarity between the grazing treatments, the sampled area (4000-m^2) in the spring-grazed paddock had 69 plant species, whereas the fall-grazed and ungrazed paddocks had 70 each. In the entire sampled area there were 84 plant species (Table 2 and Appendix), a number that is considerably larger than past estimates (Craddock and Forsling 1938, Mueggler 1950, Laycock 1967, Bork and others (1998). Bork and others (1998) identified 58 species in 1995 and 57 in 1996. An argument could be made that the fire caused an increase in species numbers with the reasoning being that the fire changed the environment to one favorable for seeds of previously unrepresented species to germinate. An explanation for the increase in species numbers may be the use of an improved sampling technique (Stohlgren and others 1998). In this study, there were from 52 to 56 total plant species in the 40 1-m^2 plots that were used to measure cover components for each grazing treatment with a total of 70 species in 120 1-m^2 plots in the entire study (Figure 1), which corresponds well with the 58 and 57 species found in 1995 and 1996 on 8 and 30 1.75-m^2 plots that were sampled by Bork and others (1998). A narrow definition of plant diversity that was satisfied with a tally of species (species richness) would conclude that sheep grazing in the sagebrush steppe has had no impact on diversity, and therefore, ecosystem health has not been altered. This narrow definition, however, does not reflect the substantial ecological shifts that have taken place in response to grazing seasonality.

Previous research at this site has determined spring grazing by sheep results in more annual grasses and shrubs compared to fall- and ungrazed plots. In addition fall-grazing results in more perennial forbs and less shrub cover, than spring-grazed, but similar to ungrazed (Craddock and Forsling 1938, Mueggler 1950, Laycock 1967, Bork and others 1998). In 1995–1996, total plant cover was about 50% in each of the three grazing treatments. One year after the fire in 2001, live vegetative cover was 19.7% in the ungrazed, 18.4% in the fall-grazed, and 10.8% in the spring-grazed plots (Table 3). Recovery of total vegetative cover in the spring-grazed plots was reduced compared to the fall and ungrazed plots. The fire did burn more vegetation in the spring-grazed area (95%) than the fall-grazed area (81%; Figure 1). This was probably not the main cause of the reduced plant cover, however, as plots were located in areas where all vegetation was burned. The ungrazed area, where vegetation recovery was the greatest, was more completely burned (almost 100%) than the spring-grazed area (Figure 1). Perhaps this difference in post-fire vegetation recovery is due more to the type of vegetation that was in these plots before the fire (Harniss and Murray 1973).

There was a large difference in the cover of annual plants, with over 33% of the total vegetative cover in the spring-grazed plots composed of annual species compared to less than 10% in the fall- and ungrazed treatments. There was also a large difference in the cover of introduced species, with over 25% of the total vegetative cover in the spring-grazed plots composed of introduced species compared to 8% in the fall-grazed plots and 4% in the ungrazed plots. These differences were not picked up by a diversity index, evenness index, or species count.

Except for downy brome, no other mention of introduced species has been made for these plots in past research, although there are seven listed in the 1995–1996 species list (Edward Bork, personal communication), which is considerably fewer than the 16, representing 19% of the species, identified in this present study. Downy brome is a problematic weedy winter annual grass in the sagebrush steppe that is capable of increasing fire frequency and converting the plant community into a near monoculture of downy brome (Pellant and Reichart 1984, Pellant 1990, Whisenant 1990, Peters and Bunting 1994). In 1924, there was no record of downy brome on these plots, however, the first appearance of downy brome in the USSES herbarium was in 1923. In 1949, 0.3% of the herbage dry weight in the fall-grazed plots and 0.9% of the herbage dry weight in the spring-grazed plots was downy brome (Mueggler 1950). By 1964 the percentage of downy brome in the herbage dry weight had increased to 7% in the spring-grazed plots, 1.6% in the fall-grazed plots, and 1.3% in the ungrazed plots (Laycock 1967). These numbers had changed only slightly by 1996, when downy brome plant cover as a proportion of the total plant cover was 10% in the spring-grazed plots, 1.6% in the fall-grazed plots, and 3% in the ungrazed plots (Bork and others 1998). A year after the fire, downy brome, as a percent of the total vegetative cover, did not change from the 1964 and 1996 measurements, with 10% in the spring-grazed plots, 3% in the fall-grazed plots, and 1% in the ungrazed plots (Table 3). Although percentage cover of downy brome did not increase, it was very successful at producing seed in the first year after the fire compared to other plant species (personal observation) and careful attention will be paid to its population dynamics in subsequent years.

The fall-grazed treatment had twice the grass coverage as the spring-and ungrazed treatments (Table 3). In past studies of this site there were differences in the amount of grasses in the ungrazed, spring-grazed, and fall-grazed plots (Craddock and Forsling 1938, Mueggler 1950, Laycock 1967, Bork and others 1998). Craddock and Forsling(1938) measured a decline in grasses from 11% to 3% cover in six years (1924–1930) due to heavy spring and fall grazing, whereas heavy fall grazing alone did not change grass cover (11% to 13%). With a lighter grazing pressure, Mueggler (1950) still measured a reduced grass component in the spring + fall-grazed plots (123 kg/ha) compared to fall-grazed plots (171 kg/ha). Unfortunately, there was not an ungrazed control paddock in the earliest study, and Mueggler (1950) made no mention of the control plot, which was established in 1941. By 1964, when grazing intensity and timing were standardized and an ungrazed treatment had been added, Laycock (1967) reported more perennial grass in the fall-grazed (276 kg/ha) compared to the ungrazed (242 kg/ha) and the spring-grazed (194 kg/ha) treatments. This relationship was still measurable in 1996 when native perennial grass cover in fall-grazed plots was 16.2% compared to 13.3% in the ungrazed and 10.6% in the spring-grazed treatments (Bork and others 1998). It is possible that the increased sagebrush density resulted in a fire that burned more completely and possibly more intensely in the spring- and ungrazed plots. Although we have no measure of fire intensity, an increase in fire intensity may result in more damage to root crowns of perennial species, which could have reduced grass recovery (Blaisdell and others 1982). Sandberg bluegrass, which was abundant in all plots before the fire (Bork and others 1998), seemed particularly sensitive to fire, with over an 80% decline in cover in the spring- and ungrazed plots compared to the fall-grazed plots. This sensitivity to fire may be due to its growth form as at this site it is usually in a small bunch that might be more easily damaged by fire.

Historically the vegetation component most influenced by spring and fall grazing in these pastures was the native perennial forbs. All previous studies at this site have measured immediate and severe reductions in native perennial forbs due to spring grazing by sheep (Craddock and Forsling 1938, Mueggler 1950, Laycock 1967, Bork and others 1998). In 1995–1996, before the fire, native perennial forbs accounted for 8.5% of the cover of the fall-grazed, 8.4% of the ungrazed, and 2.3% of the spring-grazed plots (Bork and others 1998). After the fire, native perennial forbs rebounded in the ungrazed plots to 13% cover, representing 66% of the total vegetative cover. In the fall- and spring-grazed plots the native perennial forbs remained about the same at 7.3% and 3.6%, respectively, representing 40% of the total vegetative cover in the fall-grazed plots and 33% in the spring-grazed plots. The positive response of the native perennial forbs in the ungrazed plots is surprising, given the completeness of the burn and the severity of the drought. This increase could be due to a reduction of competition with the shrub component. In the ungrazed plots arrowleaf balsamroot cover was fivefold more abundant than in the fall-grazed plots, which in turn was 25-fold more abundant than in the spring-grazed plots. Visually there was a remarkable difference when arrowleaf balsamroot was in flower.

The impact of the fire varied depending on the history of grazing. Before the fire there were fewer shrubs in the fall-grazed treatment (Bork and others 1998). Taking into account the areas that did not burn; the fall-grazed treatments had a much greater shrub component, especially tall threetip sagebrush, than the spring- and ungrazed treatments (Figure 2). Because of seed production from these unburned plants, the fall-grazed treatment could have a much quicker recovery of sagebrush than the other two grazing treatments. Native forbs flourished in the ungrazed plots to levels higher than before the fire. The spring-grazed plots had larger annual and introduced plant components after the fire. The condition of the spring-grazed plots after burn reflects the weedy annual phase predicted by Barney and Frischknecht (1974) for a site in west-central Utah. The quantity of introduced species in the spring-grazed treatment may make a transition back to a more native species-dominated state difficult (Laycock 1991). Probably because of the drought, very few seedling perennials were present, and except for downy brome, very few plant species, annual or perennial, produced large amounts of seed. In mid-October 2001, just before snow began to accumulate, seedling downy brome plants could be found in the burned areas. Differences in species numbers have not changed, but differences in vegetation components, especially as a percentage of total vegetative cover, have increased possibly due to the fire (Table 3).

In making environmental management decisions, a focus on species numbers and diversity indices would lead to the conclusion that the three grazing systems and the wildfire at this location have had no impact on plant diversity and that the ecosystem is healthy and intact. A closer look, however, at the vegetation in the systems would reveal that the spring-grazed plots had half as much plant cover as the fall- and ungrazed plots, and the spring-grazed plots had a larger proportion of plant cover composed of introduced (27%) and annual (34%) plants (Table 3). This closer look would determine that the spring-grazed plots are in an unnatural condition for the environment and further degradation may continue, with or without continued grazing. The spring-grazed plots, with two- and fourfold as much actual percent coverage in introduced and annual plants as the fall and ungrazed plots, seem to be much more susceptible to invasion.

Introduced species represented 8% of the vegetative cover of the fall-grazed and 4% of the cover in the ungrazed plots (Table 3), and each of these treatments have at least 10 introduced species present. One might argue that these introduced species in the fall- and ungrazed treatments could be the advent of a greater infestation and that this greater infestation may occur in a few more years or perhaps after another wildfire or a year with a poor grazing management practice. Because of the history of research on these plots near Dubois, Idaho, it will be possible to document what changes occur and, with controlled experiments, attempt to determine causality.

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Appendix 1. Species list, species type, and smallest plot in which the species occurred of plants identified in the study

Latin name	Common name	Ungrazed	Spring	Fall
Introduced annual forbs				
Alyssum desertorum Stapf.	desert alyssum	1[a]	1	1
Camelina microcarpa DC.	littlepod falseflax	1	1	1
Ceratocephala testiculata(Cratz.) Bess	hornseed buttercup	10	10	1
Chenopodium album L.	lambsquarter	—	1000	1
Chorispora tenella (Pallas) DC.	blue mustard	—	—	1
Kochia scoparia (L.) Schrad	burning bush	10	—	1
Lactuca serriola L.	prickly lettuce	1	1	—
Salsola tragus L.	Russian thistle	—	100	100
Sisymbrium altissimum L.	Jim Hill tumblemustard	—	1	—
Tragopogon dubius Scop.	yellow salsify	1	1	1
Introduced annual grass				
Bromus tectorum L.	downy brome	1	1	1
Introduced perennial forbs				
Medicago sativa L.	alfalfa	10	—	—
Melilotus officinalis (L.) Larn.	yellow sweetclover	—	1	—
Taraxacum officinale G.H. Weber ex Wiggers	common dandelion	1	1	1
Introduced perenntal grass				
Agropyron cristatum (L.) Gaertn.	crested wheatgrass	1	100	10
Poa pratensis L.	Kentucky bluegrass	—	100	—
Native annual forbs				
Amsinckia menziesii (Lehm.) A. Nels. & J.F. Macvr. var. *menziesii*	Menzies' fiddleneck	10	—	—
Chenopodium leptophyllum (Moq.) Nutt. ex S. Wats	slimleaf goosefoot	1	1	1
Collinsia parviflora Lindl.	small flower blue-eyed Mary	1	1	1
Cordylanthus ramosus Nutt. ex Benth	bushy bird's beak	10	10	1
Descurania incana (Bernh. Ex Fisch & C.A. Mey.) ssp. *incana*	mountain tansymustard	1	1	1
Gayophytum humile Juss.	dwarf groundsmoke	10	1	100
Gymnosteris parvula Heller	small-flowered gymnosteris	10	—	—
Lappula occidentalis (S. Wats.) Greene var. *occidentalis*	western stickseed	1	1	1
Lepidium densiflorum Schrad.	common pepperweed	—	1	—
Plantago patagonica Jacq.	wooly Indian-wheat	—	1	1
Polygonum douglassi Greene	Douglas' knotweed	1	1	1
Native perennial forbs				
Achillea millefolium L.	western yarrow	—	—	100
Agoserus glauca (Pursh) Raf.	pale agoseris	1	1	1
Allium acuminatum Hook.	tapertip onion	1	1	1
Allium textile A. Nels. & J.F. Macbr.	textile onion	1	1	1
Antennaria dimorpha (Nutt.) Torr. & Gray	low pussytoes	1	—	—
Antennaria rosea Greene	rosy pussytoes	1	1	1
Arabis holboelii Hornem.	Holboell's rockcress	1	100	—
Arnica fulgens Pursh	orange arnica	1	—	—
Astragalus convallarius Greene	lesser rushy milkvetch	1	1	1
Astragalus filipes Torr. ex Gray	basalt milkvetch	1	—	10
Astragalus lentigenosus Dougl. ex Hook.	specklepod milkvetch	1	1	10
Astragalus miser Dougl.	timber milkvetch	1	1	1
Astragalus purshii Dougl. ex Hook.	wooly pod milkvetch	1	1	1
Balsamorhiza sagittata (Pursh) Nutt.	arrowleaf balsamroot	1	1	1
Calochortus macrocarpus Dougl.	sagebrush marisopa lily	1	—	1
Castilleja angustifolia (Nutt.) G. Don	northwest Indian paintbrush	1	1	1
Commandra umbellata (L.) Nutt. ssp. pallida (A. DC.) Piehl	pale bastard toadflax	1	1	1
Crepis acuminata Nutt.	tapertip hawksbeard	1	1	1
Erigeron corymbosus Nutt.	purple daisy fleabane	1	1	1
Erigeron filifolius (Hook.) Nutt.	thread-leaf fleabane	10	1	1
Eriogonum caespitosum Nutt.	matted buckwheat	—	—	100

Appendix 1. Continued.

Latin name	Common name	Ungrazed	Spring	Fall
Native perennial forbs				
Eriogonum heracleoides Nutt.	Wyeth's buckwheat	10	100	100
Eriogonum ovalifolium Nutt.	cushion buckwheat	1	—	1
Fritillaria atropurpurea Nutt.	purplespot fritillary	1	1	1
Hackelia patens (Nutt.) I.M. Johnston	spreading stickseed	—	—	100
Lithospermum ruderale Dougl. ex Lehm.	western stoneseed	1	100	1
Lomatium foeniculaceum (Nutt.) Coult. & Rose	desert biscuitroot	1	1	1
Lomatium triternatum (Pursh) Coult. Rose	nineleaf biscuitroot	1	1	1
Lonactis alpina (Nutt.) Greene	crag aster	100	1	1
Lupinus caudatus Kellogg	tailcup lupine	100	—	—
Penstemon aridus Rydb.	stiffleafed penstemon	—	—	1
Penstemon radicosus A. Nels.	matroot penstemon	1	10	1
Phlox hoodii Richards	Hood's phlox	1	1	1
Phlox longifolia Nutt.	long-leafed phlox	1	1	1
Schoenocrambe linifolium (Nutt.) Greene	flaxleaf plainsmustard	1	1	1
Senecio integerrimus Nutt.	lambstongue groundsel	1	10	1
Sphaeralcea munroana (Dougl. ex Lindl.) Spach ex Gray	Munro's globemallow	10	1	100
Stenotus acaulis (Nutt.) Nutt. var. *acaulis*	stemless mock goldenweed	100	100	1000
Viola beckwithii Torr. & Gray	Beckwith's violet	1	1	1
Viola muttallii Pursh	Nuttall's violet	1	1	1
Zigadenus paniculatus (Nutt.) S. Wats.	foothill deathcamas	1	1	1
Native perennial grass and grasslike				
Achnatherum hymenoides (Roemer & J.A. Schultes) Barkworth	Indian ricegrass	1	1	1
Carex filifolia Nutt.	threadleaf sedge	—	10	—
Elymus lanceolatus (Scribn. & J.G. Sm.) Gould ssp. *lacneolatus*	thickspike wheatgrass	1	1	1
Elymus cinereus (Scribn. & Merr.) A. Love	basin wildrye	—	10	—
Elymus elymoides (Raf.) Swezey	bottlebrush squirreltail	1	1	1
Hesperostipa comata (Trin. & Rupt.) Barkworth	needle and thread	1	1	1
Koeleria macrantha (Ledeb.) J.A. Schultes	prairie junegrass	1	1	1
Poa secunda J. Presl	Sandberg bluegrass	1	1	1
Pseudoroegneria spicata (Pursh) A. Love ssp. spicata	bluebunch wheatgrass	1	1	1
Native perennial shrubs				
Artemisia tripartita Nutt. spp. *tripartita*	tall threetip sagebrush	1	1	1
Chrysothamnus viscidiflorus (Hook.) Nutt.	green rabbitbrush	1	1	1
Eriogonum microthecum Nutt.	shrubby buckwheat	1	1	1
Leptodactylon pungens (Torr.) Torr. ex Nutt.	prickly phlox	1	100	100
Opuntia polyacantha Haw.	plains pricklypear	10	1	1
Purshia tridentata (Pursh) DC.	antelope bitterbrush	10	1000	1
Tetradymia canescens DC.	horsebrush	1	1	1

[a]Plot area where plant was identified: 1, 10, 100, or 1000 m2.