Journal of Arid Environments 132 (2016) 49–59

Contents lists available at ScienceDirect

Journal of Arid Environments

journal homepage: www.elsevier.com/locate/jaridenv

Postfire grazing management effects on mesic sagebrush-steppe vegetation: Spring grazing

Patrick E. Clark*, C. Jason Williams, Fredrick B. Pierson, Stuart P. Hardegree

USDA-ARS Northwest Watershed Research Center, 800 E. Park Boulevard, Suite 105, Boise, ID 83712, USA

ARTICLE INFO

Article history:
Received 22 September 2014
Received in revised form
29 October 2015
Accepted 30 October 2015
Available online 4 January 2016

Keywords:
Burning
Cattle
Fire-grazing interactions
Livestock
Prescribed fire
Rangeland improvement

ABSTRACT

The influence of fire-grazing interactions on ecological pattern and process has been fairly well studied in some rangeland ecosystems (e.g., tallgrass prairie) but is only poorly understood in others. On sagebrush-steppe rangelands of the western US, there has been a long-standing concern that fire followed by grazing can cause substantial mortality in sensitive plant species. Vegetation responses to fire-grazing interactions, however, have never been studied in the higher elevation, more mesic portions of the sagebrush-steppe. We investigated whether graminoid, forb, and litter cover; bare ground; and species density and frequency responses differed among burned areas which were grazed at a very light stocking rate (33 ha AUM^{-1}) during spring (May) without postfire deferment, burned areas where 1–2 growing seasons of grazing deferment were applied, and burned areas completely excluded from postfire grazing. Fire-grazing interactions had very few effects on vegetation but did reduce litter cover and bare ground compared to burning alone. This was a case study; consequently, caution should be taken in applying these results beyond their limited scope of inference. In some situations, however, postfire grazing can likely be employed without deferment or after deferring for only one growing season, and not cause substantial adverse impacts on vegetation.

1. Introduction

Fire is a disturbance which plays a number of critical roles in most rangeland ecosystems. Fire can promote successional cycling and changes in plant species composition (Christensen, 1985; Keane et al., 2004; Spasojevic et al., 2010; Turner et al., 1997; Vermeire et al., 2011), affect nutrient cycling (Boerner, 1982; Ojima et al., 1994), enhance the diversity and productivity of habitats for wildlife and domestic animals (Hobbs and Spowart, 1984; Peek et al., 1979), and increase forage quality, palatability, and availability for grazers (Van Dyke and Darragh, 2007; Willms et al., 1980). Grazing is also a disturbance agent on rangelands that can alter vegetation structure and composition (McNaughton, 1984; Manier and Hobbs, 2007; Milchunas et al., 1988; Milchunas and Lauenroth, 1993), promote enhanced forage quality and palatability (Clark et al., 1998, 2000; Ganskopp et al., 2004; Ganskopp and Rose, 1992) and the effects of fire and grazing can be interactive (Allred et al., 2011a; Hobbs et al., 1991; Knapp et al., 1999; Turner et al., 1994; Zimmerman and Neuenschwander, 1984).

The influence of fire-grazing interactions on ecological pattern and process has been fairly well studied in some rangeland ecosystems but is very poorly understood in others. In native tallgrass prairie, the combination of fire and ungulate grazing can promote and sustain heterogeneity in vegetation cover, affect productivity, and alter species composition (Collins and Smith, 2006; Fuhlendorf et al., 2008, 2009); enhance wildlife habitat (Fuhlendorf et al., 2006); and ultimately feedback to influence the fire regime (Kerby et al., 2007). Outside of the prairie grasslands, some studies have investigated the effects of fire and grazing separately (Underwood and Christian, 2009; Valone and Kelt, 1999) but few have set out to explicitly examine fire-grazing interactions (e.g., Noy-Meir, 1995; Bates et al., 2009).

On sagebrush-steppe rangelands of the western US, there has been a long-standing concern that the combined effects of fire and postfire grazing can cause substantial mortality in sensitive plant species. Citing postfire vigor-recovery rates for perennial bunchgrass species, Wright and Bailey (1982) suggested at least 2 years of rest from livestock grazing may be required before burned

Mention of a proprietary product does not constitute a guarantee or warranty of the product by USDA or the authors and does not imply its approval to the exclusion of the other products that may also be suitable.

* Corresponding author.
E-mail address: pat.clark@ars.usda.gov (P.E. Clark).

http://dx.doi.org/10.1016/j.jaridenv.2015.10.022
0140-1963/Published by Elsevier Ltd.

perennial grasses have recovered sufficient vigor to tolerate postfire herbivory. Based largely on this suggestion, public land-management agencies throughout the sagebrush-steppe ecotype have developed a postfire grazing management guideline which stipulates that burned areas should be rested from livestock grazing for at least 2 growing seasons postfire. Although this guideline has been applied for many years, its scientific foundation remains quite limited, and its legitimacy and efficacy have been and continue to be questioned (Bates et al., 2009; Bruce et al., 2007; Bunting et al., 1998; Sander, 2000).

Early research to evaluate the interactive effects of fire and livestock grazing on sagebrush-steppe vegetation provided some useful information towards resolving this postfire grazing controversy but these studies had some substantial limitations. Rather than applying fire and postfire livestock grazing treatments at the plant-community scale or broader, these early studies used mechanical means to burn and subsequently clip individual bunchgrass plants to evaluate their vigor and mortality responses to simulated fire-grazing interactions (Jirik and Bunting, 1994; Bunting et al., 1998). These burn-and-clip studies intentionally used a very short, uniform stubble height (i.e., 2 cm) to simulate the effects of a high postfire utilization rate on bunchgrass species. Uniform clipping heights, however, may not accurately simulate the defoliation effects of actual cattle grazing on vegetation responses (Wallace, 1990). Furthermore, high forage utilization rates within burned areas are not inevitable. Under a moderate stocking rate, cattle grazing may in fact produce a fairly short, uniform stubble height on bunchgrass during the first few postfire years. Under conservative or very light stocking rates, however, the level of bunchgrass utilization can potentially be much more variable and the mean stubble height somewhat higher than under moderate stocking. Recent work in the short-grass steppe indicates conservative stocking rates are an effective means of minimizing any adverse effects of postfire cattle grazing (Augustine et al., 2010). The effects of conservative postfire cattle stocking rates on responses of sagebrush-steppe vegetation have never been rigorously evaluated, particularly, when applied in combination with differing levels of postfire rest from grazing.

Because of their nature, the burn-and-clip studies also did not evaluate the interactive effects of fire and livestock grazing on plant community-level responses such as cover by growth form or plant species density and frequency by functional group. More recent studies on Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young) or basin big sagebrush (*A. tridentata* Nutt. ssp. *tridentata*)/black sagebrush (*Artemisia nova* A. Nelson) rangelands have, however, directly evaluated plant-community responses to prescribed fire and postfire cattle grazing (Bates et al., 2009; Bruce et al., 2007). Generally, these studies found, even under moderate stocking rates and moderate levels of utilization, that one growing season of rest from postfire cattle grazing allowed herbaceous plant recovery to proceed. These studies, however, were conducted in some of the drier areas and community types of the sagebrush-steppe ecotype. In higher elevation, mesic sagebrush-steppe vegetation; where mountain big sagebrush (*A. tridentata* Nutt. ssp. *vaseyanna* Beetle) is generally dominant or co-dominant, plant community-level responses to the interaction of fire and postfire cattle grazing have never been studied.

Our intent with this current study was to address two questions. First, in mesic sagebrush steppe, do plant community-level responses differ among different durations of postfire rest from cattle grazing when a very light stocking rate is used? Second, do vegetation responses to postfire grazing-rest treatment differ among sagebrush-steppe community or vegetation types? We hypothesized mesic sagebrush-steppe vegetation will have similar amounts of perennial grass, forb, and litter cover; bare ground; and species density and frequency levels whether these burned areas received no postfire grazing at all; were grazed at a very light stocking rate during the month of May within the first postfire growing season (i.e., no rest from grazing); or were subjected to different levels of postfire rest from grazing. Further, we expected these vegetation responses would be similar whether the postfire grazing-rest treatment was applied in a vegetation type dominated by mountain big sagebrush and mountain snowberry (*Symphoricarpos oreophilus* A. Gray) or in a type dominated by antelope bitterbrush (*Purshia tridentata* [Pursh] DC) and mountain big sagebrush. We tested these hypotheses in a case study following a landscape-scale, fall prescribed fire applied to mesic sagebrush steppe rangelands of southwestern Idaho, USA.

2. Methods

2.1. Study area

The research was conducted in the Whiskey Hill prescribed-fire study area (43° 9′ 49″ N, 116° 47′ 51″ W) located on private lands within the Reynolds Creek Experimental Watershed (RCEW) in the Owyhee Mountains about 80 km south of Boise in southwestern Idaho. Climate is continental. Winters are cold and wet. Long-term (1962–2009) mean annual precipitation at the Whiskey Hill gauges (095 and 095b) was 453 mm (NWRC, 2014). Typically about 34% of this precipitation occurs as snow (Hanson, 2001). Summers are warm and dry. The growing season is about 100 days but frost can occur during any month of the year. Long-term (1967–2010) mean daily maximum, minimum and mean air temperatures at the nearby Lower Sheep Creek weather station (127x07) were 12.7, 3.8, and 8.3 °C, respectively (Hanson et al., 2001; NWRC, 2014).

The Whiskey Hill study area (324 ha) is a fenced rangeland pasture which spans a north-south ridgeline and includes the adjoining west and east-facing hillslopes. Elevation at Whiskey Hill ranges from 1523 to 1878 m. Slopes range from flat to very steep (176.8% or 60.5° maximum) with aspects in all four cardinal directions well represented. Soils are primarily derived from granitic parent materials and composed of a complex of Takeuchi (coarse, loamy, mixed, frigid Typic Haploxerolls) and Kanlee (fine, loamy, mixed, frigid Typic Argixerolls) soil series (Seyfried et al., 2001).

Vegetation at Whiskey Hill is dominated by three cover types: i) mountain big sagebrush – mountain snowberry, ii) antelope bitterbrush – mountain big sagebrush, and iii) native bunchgrass. Besides the 2 dominant species, the mountain big sagebrush-mountain snowberry type includes western juniper, yellow rabbitbrush (*Chrysothamnus viscidiflorus* [Hook.] Nutt.), Saskatoon serviceberry (*Amelanchier alnifolia* [Nutt.] Nutt. ex M. Roem. *alnifolia*), bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] A. Löve), squirreltail (*Elymus elymoides* [Raf.] Swezey), Idaho fescue (*Festuca idahoensis* Elmer), Sandberg bluegrass (*Poa secunda* J. Presl.), silvery lupine (*Lupinus argenteus* Pursh), tapertip hawksbeard (*Crepis acuminata* Nutt.), and western stoneseed (*Lithospermum ruderale* Douglas ex Lehm.) (See online Supplemental Materials for a full species list). Other components of the antelope bitterbrush-mountain big sagebrush type include western juniper, native bunchgrasses and forbs. Of these two, shrub-dominated cover types, the mountain big sagebrush-mountain snowberry type generally had the most herbaceous cover, both in the interspaces and under the shrub canopy (Clark unpublished data). The native bunchgrass cover type is dominated by bluebunch wheatgrass, squirreltail, Idaho fescue, and Sandberg bluegrass. Cheatgrass (*Bromus tectorum* L.), an exotic annual grass, exhibits a minor to common presence within all three of these dominant vegetation types. A curl-leaf mountain mahogany (*Cercocarpus* *ledifolius* Nutt.) woodland with very sparse understory occurs as a fourth, less common vegetation type and is primarily confined to the granite outcrops on ridge-crowns and adjacent slopes.

2.2. Prescribed fire

On 27 September 2004, the Whiskey Hill prescribed fire was applied to 131 ha within the study area. This vegetation treatment was intended to decrease shrub cover, improve forage availability, and kill encroaching western juniper trees while avoiding any substantial adverse impacts on ecosystem health. The fire was conducted by the USDI Bureau of Land Management (BLM) using their standard procedures for prescribed burning in juniper-encroached, sagebrush steppe rangeland (BLM, 2003). Immediately after the fire was extinguished, fire-severity polygons were digitized using a dual-channel GPS unit (Trimble Pro XRS, Trimble Navigation, Inc., Sunnyvale, California). The prescribed fire produced a mosaic of lightly (68 ha), moderately (40 ha), and severely burned areas (23 ha) as well as some unburned areas (33 ha) within the fire perimeter. Generally, stands of the antelope bitterbrush-mountain big sagebrush cover type tended to have received moderate severity while bunchgrass grasslands and mountain big sagebrush-snowberry stands received light fire severity. In all burned areas, however, prescribed fire generally killed or greatly suppressed mountain big sagebrush and bitterbrush shrubs and juvenile juniper trees.

Postfire recovery of herbaceous vegetation on burned areas proceeded rapidly. Perennial forbs increased in cover on formerly shrub-dominated areas during the first postfire year (2005). A large increase in perennial grasses on these same areas occurred during the second year postfire (Clark unpublished data). The postfire landscape, during the remainder of the study (2005–2008) can generally be described as perennial grassland with burned shrub and tree skeletons and occasional inclusions of unburned woody vegetation.

2.3. Postfire grazing-rest treatments

Cattle grazing on the study area had traditionally occurred, with moderate stocking, for about 30 days during the month of May when forage plants were actively growing but prior to peak production which generally occurred in June-early July. The phenology of bunchgrass forages like bluebunch wheatgrass generally progressed from the new leaf-growth stage, to the boot stage, and into the inflorescence-emergence stage during the month of May. After the prescribed fire, the study area continued to receive cattle grazing for 30 days during May thus the pasture was never rested from grazing postfire. For two growing seasons prior to the prescribed fire (2003 and 2004) and for 3 postfire years of the study (2005–2007), the study area pasture was stocked with 10 beef cow–calf pairs and 2 bulls for the first 15 days of the grazing trial. Cattle used during these first 15 days were GPS collared as part of study evaluating cattle resource-selection responses to prescribed fire (Clark et al., 2014). For the remaining 15 days of the 30-day trial, an additional 12 uncollared, cow–calf pairs were added to the herd grazing the study area. Assuming each cow and her calf represented 1.15 metabolic Animal Unit Equivalents (AUEs) and each bull represented 1.5 metabolic AUEs, stocking rate in the pasture during the first 15 days was about 0.045 AUEs ha-1 or 44.7 ha AUM^{-1} and was about 0.087 AUEs ha^{-1} or 22.9 ha AUM^{-1} during the remaining 15 days of the trial. Average stocking rate for the entire 30-day trial was thus 0.066 AUE ha^{-1} or 33 ha AUM^{-1}. Typically, this average stocking rate would be considered a very light rate for mesic sagebrush steppe rangelands which are commonly stocked by the USDI Bureau of Land Management (BLM) at 4–7 ha AUM^{-1}. Our intent with this research, however, was to provide both private and public land managers with information useful in developing post-fire cattle grazing management strategies. While private lands, such as this study area, are often grazed during the first 2 years following a prescribed fire, resource managers on federal agency lands in the USA typically follow a guideline of excluding livestock grazing entirely from burned pastures for at least 2 years postfire. This postfire grazing-rest guideline is intended to allow burned vegetation to recover vigor before being grazed (Wright and Bailey, 1982). This guideline is just that, a guideline and not a strict rule or law. Consequently, rather than apply a moderate stocking rate as has typically been done in postfire grazing studies (e.g., Bates et al., 2009), our intent here was to use very conservative stocking rates such as those a public lands resource manager would likely use during the first 2 postfire years if he/she chose not to strictly follow the agency postfire-rest guidelines. Findings on short-grass steppe (Augustine et al., 2010) lend support to the concept of using conservative stocking rates as an alternative postfire grazing strategy to deferment or rest from livestock grazing postfire.

The postfire grazing-rest treatment levels in this study were applied using sets of livestock exclosures. Six sets of livestock exclosures were established during the postfire winter 2004-05 within areas that had received light or moderate fire severity. Three exclosure sets were randomly located on sites formerly occupied by mountain big sagebrush-snowberry stands which had received light fire severity. The remaining 3 exclosure sets were located at random on sites previously occupied by antelope bitterbrush-mountain big sagebrush stands which had received moderate fire severity. Sites representing moderately and severely burned mountain big sagebrush-snowberry stands and lightly or severely burned antelope bitterbrush-mountain big sagebrush stands were excluded from selection as these sites were atypical and represented only small minorities of the available stands from these two vegetation cover types on our study area.

Each exclosure set consisted of 3 exclosures and one nearby, unexclosed area, all of which were square in shape and 529 m^2 in size. Exclosures were constructed with steel fence posts and 4 strands of barbed wire. The exclosures and unexclosed area of each set represented 4 levels of a postfire grazing-rest treatment. The unexclosed area represented the No-Rest postfire grazing treatment level and was intended to be grazed by cattle during the month of May for postfire years 2005–2007. One exclosure, representing the 1-Year Rest treatment level, of each set was to be dismantled and removed just prior to the second postfire grazing season (2006) thus exposing the vegetation to cattle grazing during May for study years 2006–2007. The second exclosure of each set, which represented the 2-Year Rest treatment level, was grazed by cattle during May of the third postfire year (2007). The remaining exclosure of each set was maintained as a Control thus the vegetation inside had been burned in the prescribed fire but did not receive any postfire cattle grazing cattle during the study (2005–2008). Wild herbivores (e.g., mule deer [*Odocoileus hemionus* Rafinesque], elk [*Cervus elaphus* Linnaeus], and rabbits [*Lepus* spp.]), however, had access to this and all exclosures of each set throughout the study.

2.4. Vegetation sampling

Forage utilization was assessed during study years 2005–2007 using the caged paired-plot method described by Cook and Stubbendieck (1986). Rather than simply assessing cattle use at the end of the 30-day grazing trials, utilization assessments were made at 15-day intervals within each trial to minimize issues associated with differential plant growth (Cook and Stoddart, 1953) and differing stocking rates. Utilization within a specific study year was only sampled in areas within the experimental layout which were to be exposed to grazing during that year. Immediately prior to the grazing trial, three 1-m^2 plots were randomly located within each treatment area to be sampled. Each of these 3 plots was paired, based on similarity in vegetation composition and productivity, with a plot location nearby. One plot from each of the 3 plot-pairs was randomly selected to be caged to prevent cattle herbivory and trampling and the remaining plot was left exposed to grazing. After 15 trial days, all herbaceous vegetation in each of the 6 plots within each sampled treatment area were clipped to a 2.54 cm stubble height, bagged by plot, and oven-dried at 60 °C until a constant dry weight was reached. Percentage utilization for the first 15 days of the trial was calculated based on the average difference in dry herbage weight between the caged and uncaged plots. Six, new paired plots were established and maintained within each sampled treatment area for the last 15 days of the grazing trial. Utilization for this second 15-day period was determined in the same fashion as for the first period. Utilization level for the entire 30-day trial for each sampled treatment area of each study year was determined as the average of the utilization values from the two 15-day periods per trial. Treatment areas not yet exposed to cattle grazing were visually-assessed periodically to determine if any substantial wildlife herbivory or trampling damage was occurring in these areas. Only occasional trace amounts of wildlife utilization were detected inside these exclosures.

Basal cover by growth form, litter cover, and bare ground percentage were assessed using point-quadrat sampling. Four 1-m^2 quadrats were randomly located within each exclosure or unexclosed area. Point sampling was conducted using a 20-pin frame applied at 5 stations per 1-m^2 quadrat thus totaling 100 pins per 1-m^2 area. Pin-point contacts of the bases of rooted, live vegetation were recorded as graminoid, forb, shrub, or tree. Basal contacts with standing litter and contacts with downed litter were recorded as litter. Contacts with soil or rock were used to determine bare ground percentage. Sampling campaigns were conducted immediately prior to the 30-day grazing trial of 2005 (pre-treatment assessments), during treatment application years 2006 and 2007, and during the post-treatment assessment year (2008).

Plant species density was measured in each 529-m^2 exclosure or unexclosed area by conducting an exhaustive census of all macroscopic plant species within the exclosure or areal boundaries. Species was classified into function groups based on growth form (i.e., graminoid, forb, and shrub) and duration (i.e., annual or perennial). Species counts within each functional group for each exclosed area were used to calculate species density (# species 529-m treated area^{-1}) for each group (Magurran, 1988).

Species frequency was sampled at each of the random locations used for the basal-cover sampling (see above). A nested frequency approach (Bonham, 2013) was applied using 5 different-sized, nested plot sizes ranging from 1 m^2 to 0.0008 m^2. All macroscopic plant species rooted in the nested plots were recorded. From this larger set, several graminoid, forb, and shrub species were selected for frequency analysis. Selection of these target species was based on perceived importance as livestock forage; as wildlife forage, browse, and cover; or as having special status such as being an invasive exotic (e.g., cheatgrass). The 20–80 percent rule (Despain et al., 1991) was used for choosing which plot size to use in the frequency calculation. Simply put, the proper plot size for analysis is the one where a target species occurs in 20–80 percent of plots sampled. In most cases, frequency was calculated using presence data from the 1-m^2 plot size but data from the 0.016-m^2 plot, the second smallest size, was used for cheatgrass and for silvery lupine, a perennial forb.

2.5. Statistical analysis

A split –plot experimental design was used where, the 3 exclosure sets within each of the two vegetation cover types were whole plots and exclosures representing 4 treatment levels within exclosure sets were sub-plots. While locations of the exclosure sets within vegetation types were completely randomized, treatment levels among the exclosures within each set were systematically-rather than randomly-assigned, as dictated by logistical constraints. Consequently, although our statistical analyses were conducted as if treatment assignment was randomized, it is possible our results were biased by underlying environmental patterns that by chance coincided with our systematic treatment assignment layout. Given the randomization at the exclosure-set level, the potential for this biasing is probably quite low but the reader should be aware of this design limitation.

A mixed-model analysis of covariance (ANCOVA) approach was used to examine basal cover, plant species density, and frequency responses to postfire grazing-rest treatment levels. Vegetation type and treatment were fixed effects, nesting of sets within vegetation types was a random effect, and pre-treatment measurements (i.e., basal cover, species density, or frequency measured in 2005 prior to treatment application) was the continuous covariant in the ANCOVA. Analyses proceeded in two steps. First, the assumption of independence between treatment and covariate was tested by contrasting the treatment vs. covariate regression slopes for each treatment level (Doncaster and Davey, 2007). Differing regression slopes among treatment levels would indicate the independence assumption was violated and ANCOVA was not an appropriate analysis approach. Second, where independence was confirmed, an ANCOVA model was fitted using the Mixed procedure in SAS (SAS Institute Inc., 2011) and the results reported at the 0.05 alpha level. The Kenward-Roger method was used to adjust denominator degrees of freedom for all F-tests on fixed effects (Kenward and Roger, 1997). Where differences were detected, multiple comparisons of least-squared means were conducted using a studentized maximum modulus (SMM) adjustment for p-values and confidences intervals. Square-root transformations were applied to the grass, forb, and litter basal cover data to obtain normality of ANCOVA model residuals. Results from these analyses were back transformed for presentation.

Application of the different treatment levels within the experimental design was executed in stages where, the No-Rest treatment level was fully executed at the end of the first postfire year (2005), the 1-Year Rest treatment level at the end of the second postfire year (2006), and so on. Consequently, the ANCOVA described above were applied during each year of treatment application (2006 and 2007) and during the final study year (2008) after all treatment levels had been fully implemented. Our intent for the 3 separate analyses was to detect potentially interesting, short-term responses occurring in the earlier stages of treatment application which would not have been detected if ANCOVA were only conducted for the final study year (2008), after all treatments had been fully implemented. Sampling of the vegetation responses during 2006, however, was only conducted in the No-Rest and 1-Year Rest treatment level areas. Consequently, ANCOVA for 2006 data involve only these 2 treatment levels. In all subsequent study years, sampling was conducted in all areas of the experimental design whether the respective treatment levels had yet been applied or not.

Although live shrubs were present in the exclosures postfire, as evidenced by the species density and frequency data; basal cover of live shrubs was generally very near zero both prior to and after

Table 1
Back-transformed least-squares mean estimates and 95% confidence limits for percentage basal cover responses within 4 categories to 4 different levels of rest from postfire cattle grazing as measured in 2008, 4 years after application of the Whiskey Hill prescribed fire in the Owyhee Mountains of southwestern Idaho.

		Basal cover		
Category	Treatment	Estimate	Lower CL	Upper CL
		———————%———————		
Graminoid	No-Rest	4.12[a†]	2.27	6.51
	1-Yr Rest	NA*	NA	NA
	2-Yr Rest	5.25[a]	3.12	7.92
	Control	6.07[a]	3.75	8.93
Forb	No-Rest	2.87[a]	1.82	4.16
	1-Yr Rest	2.73[a]	1.71	3.98
	2-Yr Rest	2.41[a]	1.45	3.62
	Control	1.98[a]	1.12	3.08
Litter	No-Rest	12.5[a]	9.21	16.4
	1-Yr Rest	12.9[a]	9.60	16.7
	2-Yr Rest	17.1[ab]	13.2	21.5
	Control	19.6[b]	15.4	24.2
Bare	No-Rest	77.0[a]	71.5	82.5
Ground	1-Yr Rest	75.4[a]	70.0	80.9
	2-Yr Rest	69.8[ab]	64.3	75.3
	Control	65.0[b]	59.5	70.5

†Within each category, least-squares mean estimates labeled with differing letter codes were significantly different among treatment levels at the 0.05 alpha level based on SMM-adjusted multiple comparisons. In cases where the confidence limits derived from the ANCOVA model seem to contradict the multiple comparisons, the SMM-adjusted multiple comparison findings were given precedent.
*The 1-Yr Rest treatment level was dropped from the ANCOVA model to satisfy treatment-covariate independence assumptions.

application of the postfire grazing rest treatments. Consequently, no statistical analyses were conducted for shrub basal cover responses. Annual graminoid species density was extremely limited; consequently, statistical analyses were not applied for this species group. With rare exceptions, cheatgrass was generally the only annual graminoid recorded in the treated areas during all study years. Only one tree species, western juniper, occurred within the experimental layout and then only very rarely. Consequently, statistical analyses were not conducted for the tree growth form.

3. Results

3.1. Utilization levels

Utilization of herbaceous plants was relatively light and quite similar among experimental areas exposed to grazing during each year of treatment application. Under our very light stocking rates, however, there was a great deal of variability observed in the field in terms of the amount material consumed from individual bunchgrass plants. Some plants were uniformly grazed to a 3–4 cm stubble while others exhibited no evidence of herbivory at all. Still other bunchgrass plants were grazed but retained some ungrazed tillers which continued to develop into reproductive tillers as the season progressed. Data regarding this response were only collected as informal remarks on the field data forms but, these remarks suggest partial herbivory tended to occur more frequently for squirreltail than for other dominant bunchgrass species.

In 2005, when only areas assigned to No-Rest treatment level were exposed to postfire cattle grazing, mean utilization was 23.2 ± 1.34% SD. Mean utilization rates were 27.1 ± 2.67% SD in the No-Rest level and 22.9 ± 2.11% SD in the 1-Yr Rest level during 2006. In 2007, mean utilization rates were 25.4 ± 4.98% SD, 21.8 ± 3.01% SD, and 19.9 ± 1.52% SD for the No-Rest, 1-Yr Rest, and 2-Yr Rest treatment levels, respectively. In 2008, sampling of other vegetation responses was conducted before any cattle grazing commenced that year, consequently, utilization was not formally assessed. An informal scan survey, however, confirmed very little wildlife herbivory had occurred within the treatment areas during the 2008 season before response sampling was conducted. We should also note that; based on data collected for another, as yet unpublished study, using random plots distributed across the entire pasture, utilization ranged from about 3 to 18 percentage points higher in or near the postfire-rest treatment sites than in the burned area in general (Clark et al., unpublished data). This discrepancy suggests there was some attractiveness of the specific study sites or the exclosure facilities to cattle. We can only speculate to the cause of this attractiveness but as a consequence, this study was conducted under higher utilization levels than might be expected if a postfire grazing treatment or strategy had been applied to the entire pasture.

3.2. Basal cover and bare ground

Basal cover of graminoids was not affected by the postfire grazing-rest treatment during the 2006 ($P = 0.3985$), 2007 ($P = 0.4719$), and 2008 ($P = 0.2541$; Table 1) study years. Similar to graminoids, no treatment differences were detected in forb basal cover during the 2006 ($P = 0.3112$), 2007 ($P = 0.0769$), or 2008 ($P = 0.5805$; Table 1) study years. Vegetation type ($P = 0.0228$) was the only significant factor in the ANCOVA model for forb basal cover in 2008. Treated areas in the mountain big sagebrush/snowberry vegetation type had greater forb basal cover (3.77 ± 0.0198% SE) four years postfire than in the antelope bitterbrush/mountain big sagebrush type (1.47 ± 0.0198% SE).

Treatment-related differences in litter cover were not detected in 2006 ($P = 0.0840$). In 2007, data from the Control treatment level was dropped from the full ANCOVA model to satisfy the treatment-covariate independence assumption. In 2007, litter cover in the Control exclosures tended to decrease with increasing pre-treatment cover ($P = 0.0463$). A reduced ANCOVA model, excluding the Control level data and the treatment by covariate interaction term, indicated the postfire grazing-rest treatment ($P = 0.0018$) and vegetation type ($P = 0.0196$) affected litter cover in 2007. Areas receiving the No-Rest treatment level (7.37 ± 0.213% SE) had less litter cover than areas receiving the 1-Yr Rest or 2-Yr Rest treatment levels (13.1 ± 0.209% SE or 15.9 ± 0.210% SE, respectively). Note that areas assigned to the 2-Yr Rest level had not yet received postfire grazing at the time of sampling in 2007. Litter cover in 2007 was greater in the mountain big sagebrush-snowberry type (25.0 ± 0.346% SE) than in the bitterbrush-

Table 2
Least-squares mean estimates and 95% confidence limits for species density responses within 4 functional groups to 4 different levels of rest from postfire cattle grazing as measured in 2008, 4 years after application of the Whiskey Hill prescribed fire in the Owyhee Mountains of southwestern Idaho.

Functional group	Treatment	Species density		
		Estimate[†]	Lower CL	Upper CL
		——# of Species Treated-Area^{-1}——		
Perennial[‡] Graminoids	No-Rest	4.07[a]	3.00	5.14
	1-Yr Rest	4.80[a]	3.74	5.86
	2-Yr Rest	5.13[a]	4.07	6.19
	Control	4.50[a]	3.45	5.55
Perennial Forbs	No-Rest	11.5[a]	9.31	13.7
	1-Yr Rest	11.0[a]	8.75	13.2
	2-Yr Rest	13.2[a]	11.0	15.4
	Control	14.1[a]	11.9	16.4
Annual Forbs	No-Rest	6.63[a]	5.27	7.98
	1-Yr Rest	6.74[a]	5.42	8.05
	2-Yr Rest	5.04[a]	3.68	6.39
	Control	5.60[a]	4.29	6.91
Shrubs	No-Rest	4.87[a]	3.96	5.78
	1-Yr Rest	5.79[b]	4.89	6.70
	2-Yr Rest	5.79[b]	4.89	6.70
	Control	6.04[b]	5.13	6.95

[†]Within each functional group, least-squares mean estimates labeled with differing letter codes were significantly different among treatment levels at the 0.05 alpha level based on SMM-adjusted multiple comparisons. In cases where the confidence limits derived from the ANCOVA model seem to contradict the multiple comparisons, the SMM-adjusted multiple comparison findings were given precedent.
[‡] Annual graminoid species density was extremely limited; consequently, statistical analyses were not applied for this species group and least-squared mean estimates were not determined.

mountain big sagebrush type (3.54 ± 0.346% SE). In 2008, the postfire grazing-rest treatment ($P = 0.0019$) and the treatment by vegetation type interaction ($P = 0.0437$) were the only significant factors in the ANCOVA model. Litter cover values in the No-Rest and 1-Yr Rest treatments were 7.1 and 6.7 percentage points less, respectively, than for the Control (Table 1). In the mountain big sagebrush/snowberry vegetation type, litter cover was greater in the Control (23.6 ± 0.106% SE) than in the No-Rest treatment level (11.8 ± 0.114% SE) but litter cover values were similar between these treatment levels (15.9 ± 0.106% SE and 13.3 ± 0.105% SE, respectively) in the bitterbrush/mountain big sagebrush type.

In 2006, there was more bare ground in areas receiving the No-Rest treatment level (73.7% ± 3.54% SE) than in areas receiving the 1-Yr Rest level (59.7 ± 3.54% SE). During 2007, more bare ground occurred in areas assigned to the No-Rest treatment level (67.3 ± 5.25%) than in areas receiving the Control level (52.6 ± 5.27%). While at 4 years postfire (2008), areas receiving the No-Rest and 1-Yr Rest treatments had greater amounts of bare ground than the Control (Table 1).

3.3. Species density by functional group

The postfire grazing-rest treatment did not affect species density in any functional group evaluated during any of the study years except for shrubs during 2008 ($P = 0.0047$; Table 2) when less shrub species occurred in the No-Rest treatment level than in any other level. The treatment by vegetation type interaction term also influenced for the shrub species density response during 2008 ($P = 0.0133$). While shrub species density within the mountain big sagebrush-snowberry type was greater in areas receiving the 1-Yr Rest (5.55 ± 0.525 species treatment-area^{-1}) or the Control (6.22 ± 0.525 species treatment-area^{-1}) levels than in the No-Rest treatment level (3.90 ± 0.552 species treatment-area^{-1}), species density values for all these treatment levels were similar within the bitterbrush-mountain big sagebrush type. None of the factors in the 2008 species density ANCOVA models for perennial graminoids, perennial forbs and annual forbs were significant.

A summary of species counts by year and functional group, across all postfire treatments, is provided in Table 3. A full species list for this study is provided in the online Supplemental Materials associated with this paper.

3.4. Species frequency

The frequencies of 4 perennial graminoid species, which are typically dominant in mesic sagebrush steppe, were evaluated (Table 4). In 2006, bluebunch wheatgrass which differed in frequency between the No-Rest and 1-Year Rest treatment levels ($P = 0.0542$) was the only one of these species exhibiting treatment differences. The frequencies of these species did not differ among treatments during 2007. In 2008, a treatment by covariate interaction ($P = 0.0172$) was detected for Sandberg bluegrass frequency. Post-treatment frequency of Sandberg bluegrass in areas receiving the Control treatment level tended to decline with increased pre-

Table 3
A summary of plant species counts by study year and functional group across all postfire grazing treatments applied following the Whiskey Hill prescribed fire in the Owyhee Mountains of southwestern Idaho.

Years	Graminoids		Forbs		Shrubs
	Perennial	Annual	Perennial	Annual	
2005	8	3(2)[a]	34	20	7
2006	12	2(2)	36	22	9
2007	7	1(1)	31	16	9
2008	8	3(3)	35	18	9

[a] Numbers in parentheses indicate the number of species which were exotics.

Table 4
Least-squares mean estimates and standard errors for the frequency responses of 4 perennial bunchgrass species and one exotic, annual grass species during 2 treatment application years (2006–2007) and the post-treatment year (2008) under 2or 4 levels of postfire grazing-rest treatment at the Whiskey Hill prescribed fire study area within the Reynolds Creek Experimental Watershed in the Owyhee mountains of southwestern Idaho.

		Graminoid species				
Year	Treatment	PSSPS*	ELEL5	FEID	POSE	BRTE†
		—%—				
2006	No-Rest	39.6 ± 4.66[a‡]	29.2 ± 8.43[a]	6.41 ± 11.7[a]	34.7 ± 11.9[a]	33.3 ± 8.84[a]
	1-Yr Rest	22.9 ± 4.66[b]	33.3 ± 8.43[a]	18.6 ± 11.7[a]	48.7 ± 11.9[a]	16.7 ± 8.84[b]
2007	No-Rest	33.6 ± 7.75[a]	26.2 ± 8.94[a]	38.6 ± 12.2[a]	5.72 ± 8.16[a]	58.3 ± 10.2[a]
	1-Yr Rest	14.0 ± 9.00[a]	38.7 ± 8.94[a]	67.4 ± 12.6[a]	16.1 ± 7.77[a]	58.3 ± 10.2[a]
	2-Yr Rest	47.9 ± 8.08[a]	45.8 ± 8.86[a]	42.8 ± 12.2[a]	12.0 ± 8.53[a]	37.5 ± 10.2[a]
	Control	33.6 ± 7.75[a]	47.7 ± 9.19[a]	51.1 ± 12.2[a]	28.6 ± 7.77[a]	54.2 ± 10.2[a]
2008	No-Rest	24.0 ± 8.81[a]	44.2 ± 10.4[a]	0.779 ± 9.27[a]	24.1 ± 8.32[a]	62.5 ± 8.07[a]
	1-Yr Rest	32.3 ± 11.3[a]	69.2 ± 10.4[a]	35.2 ± 9.65[a]	59.2 ± 7.81[b]	66.7 ± 8.07[a]
	2-Yr Rest	32.8 ± 11.0[a]	62.5 ± 10.4[a]	13.3 ± 9.27[a]	33.4 ± 8.62[ab]	75.0 ± 8.07[a]

*Symbols for graminoid species where PSSPS = bluebunch wheatgrass, ELEL5 = squirreltail, FEID = Idaho fescue, POSE = Sandberg bluegrass, and BRTE = Cheatgrass.
†Cheatgrass is an exotic, annual grass while the other species listed are perennial bunchgrasses.
‡Frequency values with differing letter codes within year and species combinations were significantly different between or among treatment levels at the 0.05 alpha level based on SMM-adjusted multiple comparison.
§The Control level was dropped from the ANCOVA model to satisfy treatment-covariate independence assumptions.

Table 5
Least-squares mean estimates and standard errors for the frequency responses of 7 perennial forb species during 2 treatment application years (2006–2007) and the post-treatment year (2008) under 2or 4 levels of postfire grazing-rest treatment at the Whiskey Hill prescribed fire study area within the Reynolds Creek Experimental Watershed in the Owyhee mountains of southwestern Idaho.

		Perennial forb species						
Year	Treatment	CRAC2[a]	LIRU4	LUAR3	ACMI2	COUM	ERHE	PHLO2
		—%—						
2006	No-Rest	41.7 ± 10.6[b]	30.4 ± 5.15	48.3 ± 20.1	59.4 ± 12.4	23.9 ± 6.68	20.8 ± 11.0	51.5 ± 12.0
	1-Yr Rest	58.3 ± 10.6	32.1 ± 5.15	35.1 ± 20.1	53.1 ± 12.4	30.3 ± 6.68	16.7 ± 11.0	40.2 ± 12.0
2007	No-Rest	44.4 ± 8.86	39.9 ± 6.38	43.7 ± 8.65	70.7 ± 9.23	28.3 ± 9.19	25.0 ± 10.8	51.6 ± 10.6
	1-Yr Rest	51.4 ± 8.86	24.9 ± 6.45	45.6 ± 9.15	64.1 ± 9.42	36.5 ± 9.12	20.8 ± 10.8	28.4 ± 11.3
	2-Yr Rest	38.1 ± 8.86	31.6 ± 6.38	17.1 ± 8.30	75.1 ± 9.23	36.2 ± 9.25	16.7 ± 10.8	22.5 ± 10.6
	Control	62.0 ± 8.86	32.8 ± 6.33	22.8 ± 8.65	65.1 ± 9.42	36.5 ± 9.12	20.8 ± 10.8	47.5 ± 10.6
2008	No-Rest	38.6 ± 7.67	35.5 ± 7.73	41.2 ± 13.5	74.6 ± 9.43	17.8 ± 9.03	10.2 ± 1.59	54.9 ± 10.8
	1-Yr Rest	48.9 ± 7.67	37.7 ± 7.80	36.5 ± 14.4	60.3 ± 9.63	32.7 ± 8.96	18.4 ± 1.59	39.6 ± 11.2
	2-Yr Rest	51.1 ± 7.31	31.4 ± 7.73	52.0 ± 12.9	83.7 ± 9.43	33.4 ± 9.09	8.09 ± 1.59	38.2 ± 10.8
	Control	53.1 ± 7.31	33.0 ± 7.68	41.2 ± 13.5	60.5 ± 9.63	24.4 ± 8.96	4.05 ± 1.59	42.4 ± 10.8

[a] Symbols for forb species where CRAC2 = tapertip hawksbeard, LIRU4 = western stoneseed, LUAR3 = silvery lupine, ACMI2 = common yarrow, COUM = bastard toadflax, ERHE = parsnipflower buckwheat, and PHLO2 = longleaf phlox.
[b] All frequency values within year and species combinations were similar between or among treatment levels at the 0.05 alpha level.

treatment frequency (P = 0.0179). Dropping the Control data from the model removed this dependency between the treatment levels and the covariate (P = 0.3240). The final ANCOVA model was then created by excluding the Control data and the treatment by covariate interaction term. The postfire-grazing rest treatment (P = 0.0254) was the only significant factor in this final mixed model. The frequency of Sandberg bluegrass in areas receiving the 1-Yr postfire grazing rest treatment level was greater than in the No-Rest treatment level. Sandberg bluegrass frequency in the 2-Yr Rest treatment level, however, was similar to both the No-Rest and 1-Yr Rest levels. No treatment effects were detected in 2008 for the remaining 3 species.

Cheatgrass, an exotic annual grass was nearly absent in the pretreatment data, consequently, an ANOVA rather an ANCOVA was applied. During 2006, cheatgrass frequency was higher in the No-Rest treatment level than in the 1-Yr Rest level (P = 0.0161; Table 4). Cheatgrass frequency was not affected by treatment levels during 2007 (P = 0.4262) but vegetation type was a significant term in the ANOVA model (P = 0.0390). Cheatgrass occurred in 68.8 ± 7.78% SE sample plots in bitterbrush/mountain big sagebrush type but only about half that often (35.4 ± 7.78% SE) in the mountain big sagebrush/snowberry type. Similar to 2007, the only significant factor in the 2008 ANOVA model was vegetation type (P = 0.0008). Four years postfire, this species occurred much more frequently in the bitterbrush/mountain big sagebrush vegetation type (83.3 ± 5.71% SE) than in the mountain big sagebrush/snowberry type (50.0 ± 5.71% SE). No other annual grasses occurred in the study area at sufficient amounts to warrant statistical analysis.

The frequencies of the following 7 perennial forb species were evaluated: tapertip hawksbeard, western stoneseed, silvery lupine, common yarrow (*Achillea millefolium* L.), bastard toadflax (*Comandra umbellata* [L.] Nutt.), parsnipflower buckwheat (*Eriogonum heracleoides* Nutt.), and longleaf phlox (*Phlox longifolia* Nutt.). These species were selected based on their typical dominance in mesic sagebrush-steppe rangelands and, in some cases, their prominence in wildlife diets (e.g., sage-grouse [*Centrocercus urophasianus* Bonaparte]) (Pyle and Crawford, 1996; Drut et al., 1994). The postfire grazing-rest treatment did not affect the frequency of any of the perennial forb species during any of the study years, although, the treatment P-value for silvery lupine was border-line (P = 0.0562) in 2007 (Table 5). In 2008, the pre-treatment covariate was the only significant factor in the ANCOVA models for the frequencies of tapertip hawksbeard (P = 0.0427), western stoneseed (P = 0.0002), common yarrow (P = 0.0210), and bastard toadflax (P = 0.0153), thus indicating the post-treatment variability in the frequencies of these 4 species was related the variability in pretreatment frequency values. The pre-treatment vs. post-treatment regression slopes, however, were similar among treatment levels demonstrating the ANCOVA treatment-covariate independence assumption was valid for each of these 4 species. Parsnipflower buckwheat did not occur in the pretreatment sample data, consequently, only an ANOVA was applied. The only significant factor in the 2008 ANOVA model for the frequency of parsnipflower buckwheat was vegetation type (P = 0.0180). This species occurred much more frequently in the mountain big sagebrush/snowberry vegetation type (22.5 ± 0.795% SE) than in the bitterbrush/mountain big sagebrush type (2.02 ± 0.795% SE). None of the factors in the 2008 ANCOVA models for the remaining two perennial forb species, longleaf phlox and silvery lupine, were significant in 2008. Although some annual forb species were present in the treated areas (see online Supplemental Materials for a full species list), the frequencies of these species were not investigated.

The frequencies of mountain big sagebrush, mountain snowberry, and yellow rabbitbrush were evaluated. Too few plants from other shrub species (e.g., bitterbrush or rubber rabbitbrush (*Ericameria nauseosa* [Pall. ex Pursh] G.L. Nesom & Baird) occurred within the treatment areas postfire to justify analysis. In 2006, of the 3 shrub species analyzed, the postfire grazing-rest treatment only affected the frequency of mountain big sagebrush (P = 0.0003) which was more frequent in areas receiving the 1-Yr Rest treatment level than in the No-Rest level (Table 6). The frequencies of these 3 shrub species were not affected by the postfire grazing-rest treatment in 2007 or 2008. The frequency of mountain big sagebrush was, however, influenced by vegetation type (P = 0.0100) and by the treatment by vegetation type interaction (P = 0.0401) in 2008. Mountain big sagebrush occurred more frequently in the mountain big sagebrush/snowberry type (19.6 ± 2.87%) than in the bitterbrush/mountain big sagebrush type (7.44 ± 2.87%) in the final study year. This difference in mountain big sagebrush frequency between vegetation types, however, was limited only to the 1-Yr Rest treatment level. None of the other treatment levels exhibited a vegetation-type effect.

4. Discussion

4.1. Relation to previous work

The potential for heavy livestock use in burned areas is

Table 6
Least-squares mean estimates and standard errors for the frequency responses of 3 shrub species during 2 treatment application years (2006–2007) and the post-treatment year (2008) under 2or 4 levels of postfire grazing-rest treatment at the Whiskey Hill prescribed fire study area within the Reynolds Creek Experimental Watershed in the Owyhee mountains of southwestern Idaho.

Year	Treatment	Shrub species		
		ARTRV*	CHVI8	SYOR
			%	
2006	No-Rest	4.17 ± 1.82[a‡]	52.0 ± 5.42[a]	16.7 ± 9.89[a]
	1-Yr Rest	20.8 ± 1.82[b]	35.5 ± 5.42[a]	16.7 ± 9.89[a]
2007	No-Rest	6.98 ± 5.22[a]	54.7 ± 8.90[a]	8.33 ± 4.78[a]
	1-Yr Rest	27.8 ± 5.22[a]	40.2 ± 8.94[a]	12.5 ± 4.78[a]
	2-Yr Rest	13.4 ± 5.35[a]	46.3 ± 8.97[a]	3.65 ± 4.93[a]
	Control	18.5 ± 5.15[a]	42.2 ± 8.90[a]	4.69 ± 4.93[a]
2008	No-Rest	7.14 ± 4.02[a]	20.7 ± 8.99[a]	4.17 ± 3.56[a]
	1-Yr Rest	19.6 ± 4.02[a]	28.5 ± 9.05[a]	12.5 ± 3.56[a]
	2-Yr Rest	13.1 ± 4.09[a]	17.6 ± 9.11[a]	9.64 ± 3.67[a]
	Control	14.3 ± 3.99[a]	12.4 ± 8.99[a]	2.86 ± 3.67[a]

*Symbols for shrub species where ARTRV = mountain big sagebrush, CHVI8 = yellow rabbitbrush, and SYOR = mountain snowberry.
‡Frequency values with differing letter codes within year and species combinations were significantly different between or among treatment levels at the 0.05 alpha level based on SMM-adjusted multiple comparison.

invariably a primary topic of concern when postfire livestock-grazing management strategies are discussed among natural resource managers, livestock producers, range researchers, and other interested public. Cattle selection or preference for burned over unburned areas has been demonstrated in a wide variety of rangeland ecosystems (Allred et al., 2011a, b; Augustine and Derner, 2014; Clark et al., 2014, 2015; Vermeire et al., 2004). This concern is particularly acute for ecosystems which did not evolve under heavy grazing pressures, such as those in the sagebrush steppe (Mack and Thompson, 1982; but see also Burkhardt, 1996). Some previous work has suggested heavy, poorly-timed postfire grazing can cause substantial reductions in plant vigor and high mortality rates for perennial bunchgrasses and other sensitive species common to the sagebrush steppe (Wright and Bailey, 1982; Bunting et al., 1998). More recent studies on sagebrush-dominated rangelands, however, have shown cattle grazing can be applied to burned areas after only one growing season of rest without substantial adverse effects to plant species abundance or diversity (Bates et al., 2009; Bruce et al., 2007). These more recent findings tend to support those of our case study in mesic sagebrush-steppe rangeland which indicate cattle grazing can even be applied in spring (May) during the first postfire growing season without affecting graminoid or forb basal cover and with only very limited effects on species density or frequency relative to those responses detected under prescribing burning alone (i.e., our Control) or under one or two growing seasons of postfire rest from grazing. Initial treatment differences in bluebunch wheatgrass, cheatgrass, and mountain big sagebrush frequencies detected in 2006 did not extend to the later study years. Treatment differences in Sandberg bluegrass frequencies during the final study year (2008) are difficult to interpret. Since Sandberg bluegrass frequencies in the No-Rest and 2-Yr Rest levels were similar while those in No-Rest and 1-Yr Rest levels differed, we cannot simply conclude deferment from postfire grazing alone produced the observed treatment effect. The vegetation-type differences in the frequencies cheatgrass (2007 and 2008) and parsnipflower buckwheat (2008) are interesting but difficult to explain since these differences did not occur during earlier study years and thus cannot be easily attributed to differences in prefire composition or fire severity between these vegetation types. The treatment by vegetation type interaction for mountain big sagebrush during 2008 was limited to only the 1-Yr Rest treatment level. Taken as a whole, rather than illustrating a clear postfire-grazing effect, our results generally indicate postfire grazing under very light stocking, regardless of the length of deferment, had little or no effect on sagebrush-steppe vegetation beyond that of prescribed burning alone.

Combined, our results and those of Bates et al. (2009) and Bruce et al. (2007) lend support to the argument that rather than strictly adhering to a one-size-fits-all guideline of deferring livestock for at least the first 2 postfire growing seasons, postfire grazing management in the sagebrush steppe should be conservative but also flexible and adaptive to the conditions presented in each individual case (Sander, 2000). Addressing the postfire grazing issue, however, is more complex than simply dealing with species conservation concerns. In our study, we also found applying postfire grazing without rest or with only 1 growing season of postfire deferment from grazing can reduce litter cover and increase bare ground amounts relative to responses obtained under our burned-only Control. This result is not surprising as even a small amount of grazing over time will tend to decrease standing litter and its recruitment into surface litter compared to complete exclosure. These treatment-related reductions in litter and increased soil surface exposure may, however, have ecological and hydrological consequences which should be discussed.

4.2. Possible ecological and hydrologic consequences

Litter deposited on the soil surface serves a number of ecological functions. Litter acts as a mulch to help reduce evaporation, retain near-surface soil moisture and thus decreasing potential drought stress in rangeland plants (Facelli and Pickett, 1991; Villegas et al., 2010). In grasslands or shrub-grasslands where the herb layer is typically dominated by perennial bunchgrasses, this litter mulch layer, if thick enough, may suppress the growth of annual forbs and graminoids which otherwise would compete with the perennials for moisture and soil nutrients. Surface litter also has important hydrologic functions including reduction of evaporative losses, as mentioned above, and shielding the soil surface from rain drop impact and associated splash erosion (Pierson et al., 2010, 2014). Litter can also help retain precipitated moisture where it falls or at least delay or slow its downslope movement such that the opportunity for infiltration is increased (Pierson et al., 2007, 2013; Williams et al., 2014a). Litter forms debris dams on the soil surface which can serve to reduce run-off volume, velocity, and associated erosive energy (Pannkuk and Robichaud, 2003; Cerdà and Doerr, 2008). Furthermore, these dams can also divert or cause branching of run-off rivulets from otherwise straighter, more erosive flow paths. All these effects on runoff can help in preventing or lessening the severity of rill erosion (Pierson et al., 2007, 2009). Consequently, on postfire landscapes, timely recruitment of litter cover can substantially reduce the potential for significant run-off and erosion events that may otherwise occur if precipitation fell on bare soils (Pierson et al., 2002; Moody et al., 2013; Williams et al., 2014b).

Findings from our case study, suggest postfire grazing without deferment can slow the recovery of litter cover and increase soil surface exposure compared to burning alone. Treatment-related reductions in litter likely occurred because, even under very light cattle grazing in May, herbivory reduced leaf litter availability to some degree and, perhaps more importantly, delayed phenological development of these bunchgrasses and thereby decreases the amount of reproductive culms produced (Anderson and Scherzinger, 1975; Clark et al., 1998, 2000; Westenskow-Wall et al., 1994). Simply put, spring grazing can reduce the amount of standing bunchgrass litter available for recruitment into the surface litter layer. Alternatively, it has been suggested if bunchgrasses are allowed to accumulate some standing litter before postfire grazing is applied, the movement and trampling action of foraging cattle would aid in rapidly recruiting surface litter from these standing litter supplies. As such, the logical next question would be, is resting burned areas for a full growing season necessary or can rapid litter recruitment be accomplished by simply deferring grazing ? Under late-season conditions, selective grazing (Ganskopp et al., 1992, 1993) would tend to ensure that substantial ungrazed culms remained after cattle were removed and thus would be available for recruitment to surface litter.

4.3. Questions about fire-grazing interactions

Counterpoints to the arguments for routine postfire rest or deferment from grazing are generally centered around disruptions of agency rotational-grazing management schedules and difficulties for the livestock producer in securing alternate pasture or forage sources while grazing is suspended or deferred following fire. What is rarely discussed, however, is whether there are potential ecological benefits if postfire grazing is applied without rest or deferment. If recent research in the sagebrush steppe suggests postfire cattle grazing can be managed such that postfire rest or deferment from grazing is not a strict necessity, perhaps we should then question what the ecological consequences of routinely excluding large herbivores (e.g., cattle) from burned sagebrush steppe may be? Does postfire rest from grazing short-circuit a desirable fire-grazing interaction? Fire can both, increase or decrease heterogeneity on rangelands. Large, hot wildfires or poorly-conducted prescribed burns can reduce heterogeneity, converting a shrub steppe with widely-varying structure and composition into a simpler grassland, perhaps, even one dominated by exotic annual graminoids. Cooler, patchy burns, where fire intensities and resultant severities vary greatly in space, can enhance heterogeneity thus promoting increased diversity in habitat and foraging opportunities for wildlife and livestock (Clark et al., 2014, 2015). A number of studies in the tallgrass prairie, shortgrass steppe, and other rangeland ecosystems indicates that postfire herbivory can prolong and/or augment heterogeneity increases created by fire (Allred et al., 2011a, b; Archibald et al., 2005; Archibald and Bond, 2004; Augustine and Derner, 2014; Fuhlendorf and Engle, 2001; Fuhlendorf et al., 2009). Furthermore, postfire grazing can affect fuel loading and hence, feed back to influence the frequency and behavior of subsequent fires (Fuhlendorf and Engle, 2004; Kerby et al., 2007). An argument can be made that these fire-grazing interactions are important in prairie and steppe ecosystems which evolved under an intense herbivory but these mechanisms may not apply in the sagebrush-steppe with its evolutionary history of more moderate levels of herbivory. However, despite differences between prairie grasslands and sagebrush steppe in terms of the grazing tolerance and/or avoidance adaptations of their dominant forage species, the question of whether fire-grazing interactions play somewhat similar and important roles on both these different types of rangeland is still quite open to debate. Fire-grazing interactions may indeed have a yet unappreciated importance in the sagebrush steppe.

4.4. Conclusions

Our results are similar to those in other sagebrush-dominated rangeland types (Bates et al., 2009; Bruce et al., 2007; West and Yorks, 2002). Combined, these findings suggest vegetation of the sagebrush steppe or other sagebrush rangelands do not strictly require at least 2 full growing seasons of rest from grazing after fire. In some cases, perhaps many, carefully-managed postfire grazing can likely be applied after resting for only one growing season. Under very light stocking conditions, no deferment of grazing may be necessary to avoid or minimize adverse impacts on perennial graminoid and forb cover and plant species density and frequency. Indeed, Augustine et al. (2010) reported conservative stocking rates were an effective alternative to grazing deferment after dormant-season prescribed fire on the shortgrass steppe. However, under the conditions of our study, applying postfire grazing without two growing seasons of rest reduced surface litter recruitment by about 7% and increased the amount of bare ground by as much as 12% compared to the effects of prescribing burning alone. Consequently, we echo the conservative tone expressed by Bates et al. (2009) and stress that each postfire grazing case should be carefully evaluated, individually, based on its unique characteristics. In situations where litter recovery is expected to be slow (e.g., low prefire bunchgrass cover or evident fire-caused bunchgrass mortality), some level of postfire deferment from grazing would be a wise precaution. Finally, the reader is cautioned to remember the findings reported herein were derived from a case study and extrapolation of our results to other areas should be done only after careful consideration.

Acknowledgments

Funding for this research was provided by the USDA Agricultural Research Service. Prescribed burning was expertly conducted by the Boise District, USDI Bureau of Land Management, Boise, ID.

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jaridenv.2015.10.022.

References

Allred, B.W., Fuhlendorf, S.D., Engle, D.M., Elmore, R.D., 2011a. Ungulate preference for burned patches reveals strength of fire-grazing interaction. Ecol. Evol. 1, 132–144.
Allred, B.W., Fuhlendorf, S.D., Hamilton, R.G., 2011b. The role of herbivores in Great Plains conservation: comparative ecology of bison and cattle. Ecosphere 2 art26.
Anderson, E.W., Scherzinger, R.J., 1975. Improving quality of winter forage for elk by cattle grazing. J. Range Manag. 28, 120–125.
Archibald, S., Bond, W.J., 2004. Grazer movements: spatial and temporal responses to burning in a tall-grass African savanna. Int. J. Wildland Fire 13, 377–385.
Archibald, S., Bond, W.J., Stock, W.D., Fairbanks, D.H.K., 2005. Shaping the landscape: fire-grazer interactions in an African savanna. Ecol. Appl. 15, 96–109.
Augustine, D.J., Derner, J.D., 2014. Controls over the strength and timing of fire-grazer interactions in a semi-arid rangeland. J. Appl. Ecol. 51, 242–250.
Augustine, D.J., Derner, J.D., Milchunas, D.G., 2010. Prescribed fire, grazing, and herbaceous plant production in shortgrass steppe. Rangel. Ecol. Manag. 63, 317–323.
Bates, J.D., Rhodes, E.C., Davies, K.W., Sharp, R., 2009. Postfire succession in big sagebrush steppe with livestock grazing. Rangel. Ecol. Manag. 62, 98–110.
BLM, 2003. Prescribed Fire Management. USDI Bureau of Land Management. Available at: http://www.blm.gov/style/medialib/blm/nifc/directives.Par.71728.File.dat/imof&a2004003a1.pdf (accessed 29.10.15.).
Boerner, R.E.J., 1982. Fire and nutrient cycling in temperate ecosystems. Bioscience 32, 187–192.
Bonham, C.D., 2013. Measurements of Terrestrial Vegetation, second ed. John Wiley & Sons, Ltd., Hoboken, NJ.
Bruce, L.B., Perryman, B., Conley, K., McAdoo, K., 2007. Case study: grazing management on seeded and unseeded post-fire public rangelands. Prof. Anim. Sci. 23, 285–290.
Bunting, S.C., Robberecht, R., Defosse, G.E., 1998. Length and timing of grazing on postburn productivity of two bunchgrasses in an Idaho experimental range. Int. J. Wildland Fire 8, 15–20.
Burkhardt, J.W., 1996. Herbivory in the Intermountain West: an overview of evolutionary history, historic cultural impacts, and lesson from the past. Station Bulletin 58. Idaho Forest, Wildlife, and Range Experiment Station, College of Natural Resources. University of Idaho, Moscow, ID.
Cerdà, A., Doerr, S.H., 2008. The effect of ash and needle cover on surface runoff and erosion in the immediate post-fire period. Catena 74, 256–263.
Christensen, N.L., 1985. Shrubland fire regimes and their evolutionary consequences. In: Pickett, S.T.A., White, P.S. (Eds.), The Ecology of Natural Disturbance and Patch Dynamics. Academic Press, Inc., New York, pp. 85–100.
Clark, P.E., Krueger, W.C., Bryant, L.D., Thomas, D.R., 1998. Spring defoliation effects on bluebunch wheatgrass: I. Winter forage quality. J. Range Manag. 51, 519–525.
Clark, P.E., Krueger, W.C., Bryant, L.D., Thomas, D.R., 2000. Livestock grazing effects on forage quality of elk winter range. J. Range Manag. 53, 97–105.
Clark, P.E., Lee, J., Ko, K., Nielson, R.M., Johnson, D.E., Ganskopp, D.C., Chigbrow, J., Pierson, F.B., Hardgree, S.P., 2014. Prescribed fire effects on resource selection by cattle in mesic sagebrush steppe. Part 1: Spring grazing. J. Arid Environ. 100–101, 78–88.
Clark, P.E., Lee, J., Ko, K., Nielson, R.M., Johnson, D.E., Ganskopp, D.C., Chigbrow, J., Pierson, F.B., Hardegree, S.P., 2015. Prescribed fire effects on resource selection by cattle in mesic sagebrush steppe. Part 2: Mid-summer grazing. J. Arid Environ. 124, 398–412.
Collins, S.L., Smith, M.D., 2006. Scale-dependent interaction of fire and grazing on community heterogeneity in tallgrass prairie. Ecology 87, 2058–2067.
Cook, C.W., Stoddart, L.A., 1953. The quandary of utilization and preference. J. Range Manag. 6, 329–331.
Cook, C.W., Stubbendieck, J., 1986. Range Research: Basic Problems and Techniques. Society for Range Management, Denver, CO.
Despain, D.W., Ogden, P.R., Smith, E.L., 1991. Plant frequency sampling for monitoring rangelands. In: Ruyle, G.B. (Ed.), Some Methods for Monitoring Rangelands and Other Natural Area Vegetation. Division of Range Management, College of Agriculture. The University of Arizona, Tucson, AZ, pp. 7–25.
Doncaster, C.P., Davey, A.J.H., 2007. Analysis of Variance and Covariance: How to Choose and Construct Models for the Life Sciences. Cambridge University Press, Cambridge, UK.
Drut, M.S., Pyle, W.H., Crawford, J.A., 1994. Technical note: diets and food selection of sage grouse chicks in Oregon. J. Range Manag. 47, 90–93.
Facelli, J.M., Pickett, S.T.A., 1991. Plant litter: it's dynamics and effects on plant community structure. Botanical Rev. 57, 1–32.

Fuhlendorf, S.D., Archer, S.A., Smeins, F.E., Engle, D.M., Taylor Jr., C.A., 2008. The combined influence of grazing, fire, and herbaceous productivity on tree-grass interactions. In: Van Auken, O.W. (Ed.), Western North American Juniperus Communities: a Dynamic Vegetation Type. Springer, New York, NY, pp. 219–238.
Fuhlendorf, S.D., Engle, D.M., 2001. Restoring heterogeneity on rangelands: ecosystem management based on evolutionary grazing patterns. Bioscience 51, 625–632.
Fuhlendorf, S.D., Engle, D.M., 2004. Application of the fire-grazing interaction to restore a shifting mosaic on tallgrass prairie. J. Appl. Ecol. 41, 604–614.
Fuhlendorf, S.D., Engle, D.M., Kerby, J., Hamilton, R., 2009. Pyric herbivory: rewilding landscapes through the recoupling of fire and grazing. Conserv. Biol. 23, 588–598.
Fuhlendorf, S.D., Harrell, W.C., Engle, D.M., Hamilton, R.G., Davis, C.A., Leslie, D.M., 2006. Should heterogeneity be the basis for conservation? Grassland bird response to fire and grazing. Ecol. Appl. 16, 1706–1716.
Ganskopp, D., Angell, R., Rose, J., 1992. Response of cattle to cured reproductive stems in a caespitose grass. J. Range Manag. 45, 401–404.
Ganskopp, D., Angell, R., Rose, J., 1993. Effect of low densities of senescent stems in crested wheatgrass on plant selection and utilization by beef cattle. Appl. Anim. Behav. Sci. 38, 227–233.
Ganskopp, D., Rose, J., 1992. Bunchgrass basal area affects selection of plants by cattle. J. Range Manag. 45, 538–541.
Ganskopp, D., Svejcar, T., Vavra, M., 2004. Livestock foraging conditioning: bluebunch wheatgrass, Idaho fescue, and bottlebrush squirreltail. J. Range Manag. 57, 384–392.
Hanson, C.L., 2001. Long-term precipitation database, Reynolds creek experimental watershed, Idaho, United States. Water Resour. Res. 37, 2831–2834.
Hanson, C.L., Marks, D., Van Vactor, S.S., 2001. Long-term climate database, Reynolds creek experimental watershed, Idaho, United States. Water Resour. Res. 37, 2839–2841.
Hobbs, N.T., Schimel, D.S., Owensby, C.E., Ojima, D.S., 1991. Fire and grazing in the tallgrass prairie: contingent effects on nitrogen budgets. Ecology 72, 1374–1382.
Hobbs, N.T., Spowart, R.A., 1984. Effects of prescribed fire on nutrition of mountain sheep and mule deer during winter and spring. J. Wildl. Manag. 48, 551–560.
Jirik, S.J., Bunting, S.C., 1994. Post-fire defoliation response of *Agropyron spicatum* and *Sitanion hystrix*. Int. J. Wildland Fire 4, 77–82.
Keane, R.E., Cary, G.J., Davies, I.D., Flannigan, M.D., Gardner, R.H., Lavorel, S., Lenihan, J.M., Li, C., Rupp, T.S., 2004. A classification of landscape fire succession models: spatial simulations of fire and vegetation dynamics. Ecol. Model. 179, 3–27.
Kenward, M., Roger, J., 1997. Small sample inference for fixed effects from restricted maximum likelihood. Biometrics 53, 983–997.
Kerby, J.D., Fuhlendorf, S.D., Engle, D.M., 2007. Landscape heterogeneity and fire behavior: scale-dependent feedback between fire and grazing processes. Landsc. Ecol. 22, 507–516.
Knapp, A.K., Blair, J.M., Briggs, J.M., Collins, S.L., Hartnett, D.C., Johnson, L.C., Towne, E.G., 1999. The keystone role of bison in North American tallgrass prairie. BioScience 49, 39–50.
Mack, R.N., Thompson, J.N., 1982. Evolution in steppe with few large, hooved manmals. Am. Nat. 119, 757–773.
Magurran, A.E., 1988. Ecological Diversity and its Measurement. Princeton University Press, Princeton, NJ.
Manier, D.J., Hobbs, N.T., 2007. Large herbivores in sagebrush steppe ecosystems: livestock and wild ungulates influence structure and function. Oecologia 152, 739–750.
McNaughton, S.J., 1984. Grazing lawns: animals in herds, plant form, and coevolution. Am. Nat. 124, 863–886.
Milchunas, D.G., Lauenroth, W.K., 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments. Ecol. Monogr. 63, 327–366.
Milchunas, D.G., Sala, O.E., Lauenroth, W.K., 1988. A generalized model of the effects of grazing by large herbivores on grassland community structure. Am. Nat. 132, 87–106.
Moody, J.A., Shakesby, R.A., Robichaud, P.R., Cannon, S.H., Martin, D.A., 2013. Current research issues related to post-wildfire runoff and erosion processes. Earth-Science Rev. 122, 10–37.
Noy-Meir, I., 1995. Interactive effects of fire and grazing on structure and diversity of Mediterranean grasslands. J. Veg. Sci. 6, 701–710.
NWRC, 2014. Public Database for the Reynolds Creek Experimental Watershed. Boise, Idaho: Northwest Watershed Research Center. U.S. Department of Agriculture, Agricultural Research Service. Available at: ftp://ftp.nwrc.ars.usda.gov/publicdatabase/reynolds-creek/ (accessed 29.10.15.).
Ojima, D.S., Schimel, D.S., Parton, W.J., Owensby, C.E., 1994. Long- and short-term effects of fire on nitrogen cycling in tallgrass prairie. Biogeochemistry 24, 67–84.
Pannkuk, C.D., Robichaud, P.R., 2003. Effectiveness of needle cast at reducing erosion after forest fires. Water Resour. Res. 39, 1333.
Peek, J.M., Riggs, R.A., Lauer, J.L., 1979. Evaluation of fall burning on bighorn sheep winter range. J. Range Manag. 32, 430–432.
Pyle, W.H., Crawford, J.A., 1996. Availability of foods of sage grouse chicks following prescribed fire in sagebrush-bitterbrush. J. Range Manag. 49, 320–324.
Pierson, F.B., Bates, J.D., Svejcar, T.J., Hardegree, S.P., 2007. Runoff and erosion after cutting western juniper. Rangel. Ecol. Manag. 60, 285–292.
Pierson, F.B., Carlson, D.H., Spaeth, K.E., 2002. Impacts of wildfire on soil hydrological properties of steep sagebrush-steppe rangeland. Int. J. Wildland Fire 11, 145–151.
Pierson, F.B., Moffet, C.A., Williams, C.J., Hardegree, S.P., Clark, P.E., 2009. Prescribed-fire effects on rill and interrill runoff and erosion in a mountainous sagebrush landscape. Earth Surf. Process. Landforms 34, 193–203.
Pierson, F.B., Williams, C.J., Hardegree, S.P., Clark, P.E., Kormos, P.R., Al-Hamdan, O.Z., 2013. Hydrologic and erosion responses of sagebrush steppe following juniper encroachment, wildfire, and tree cutting. Rangel. Ecol. Manag. 66, 274–289.
Pierson, F.B., Williams, C.J., Kormos, P.R., Al-Hamdan, O.Z., 2014. Short-term effects of tree removal on infiltration, runoff, and erosion in woodland-encroached sagebrush steppe. Rangel. Ecol. Manag. 67, 522–538.
Pierson, F.B., Williams, C.J., Kormos, P.R., Hardegree, S.P., Clark, P.E., Rau, B.M., 2010. Hydrologic vulnerability of sagebrush steppe following pinyon and juniper encroachment. Rangel. Ecol. Manag. 63, 614–629.
Sander, K.D., 2000. How Long Should Rangelands Be Rested from Livestock Grazing Following a Fire? University of Idaho, Moscow, ID. Miscellaneous Report.
SAS Institute Inc, 2011. Base SAS® 9.3 Procedures Guide. SAS Institute Inc., Cary, NC.
Seyfried, M.S., Harris, R., Marks, D., Jacob, B., 2001. Geographical database, Reynolds creek experimental watershed, Idaho, United States. Water Resour. Res. 37, 2825–2829.
Spasojevic, M.J., Aicher, R.J., Koch, G.R., Marquardt, E.S., Mirotchnick, N., Troxler, T.G., Collins, S.L., 2010. Fire and grazing in a mesic tallgrass prairie: impacts on plant species and functional traits. Ecology 91, 1651–1659.
Turner, M.G., Romme, W.H., Gardner, R.H., Hargrove, W.W., 1997. Effects of fire size and pattern on early succession in Yellowstone National Park. Ecol. Monogr. 67, 411–433.
Turner, M.G., Wu, Y., Wallace, L.L., Romme, W.H., Brenkert, A., 1994. Simulating winter interactions among ungulates, vegetation, and fire in northern Yellowstone Park. Ecol. Appl. 4, 472–496.
Underwood, E.C., Christian, C.E., 2009. Consequences of prescribed fire and grazing on grassland ant communities. Environ. Entomol. 38, 325–332.
Valone, T.J., Kelt, D.A., 1999. Fire and grazing in a shrub-invaded arid grassland community: independent or interactive ecological effects? J. Arid Environ. 42, 15–28.
Van Dyke, F., Darragh, J.A., 2007. Response of elk to changes in plant production and nutrition following prescribed burning. J. Wildl. Manag. 71, 23–29.
Vermeire, L.T., Crowder, J.L., Wester, D.B., 2011. Plant community and soil environment response to summer fire in the Northern Great Plains. Rangel. Ecol. Manag. 64, 37–46.
Vermeire, L.T., Mitchell, R.B., Fuhlendorf, S.D., Gillen, R.L., 2004. Patch burning effects on grazing distribution. J. Range Manag. 57, 248–252.
Villegas, J.C., Breshears, D.D., Zou, C.B., Law, D.J., 2010. Ecohydrological controls of soil evaporation in deciduous drylands: how the hierarchical effects of litter, patch and vegetation mosaic cover interact with phenology and season. J. Arid Environ. 74, 595–602.
Wallace, L.L., 1990. Comparative photosynthetic responses of big bluestem to clipping versus grazing. J. Range Manag. 43, 58–61.
West, N.E., Yorks, T.P., 2002. Vegetation responses following wildfire on grazing and ungrazed sagebrush semi-desert. J. Range Manag. 55, 171–181.
Westenskow-Wall, K.J., Krueger, W.C., Bryant, L.D., Thomas, D.R., 1994. Nutrient quality of bluebunch wheatgrass regrowth on elk winter range in relation to defoliation. J. Range Manag. 47, 240–244.
Williams, C.J., Pierson, F.B., Al-Hamdan, O.Z., Kormos, P.R., Hardegree, S.P., Clark, P.E., 2014a. Can wildfire serve as an ecohydrologic threshold-reversal mechanism on juniper-encroached shrublands. Ecohydrology 7, 453–477.
Williams, C.J., Pierson, F.B., Robichaud, P.R., Boll, J., 2014b. Hydrologic and erosion responses to wildfire along the rangeland-xeric forest continuum in the western US: a review and model of hydrologic vulnerability. Int. J. Wildland Fire 23, 155–172.
Willms, W., Bailey, A.W., McLean, A., 1980. Effects of clipping or burning on some morphological characteristics of *Agropyron spicatum*. Can. J. Bot. 58, 2309–2312.
Wright, H.A., Bailey, A.W., 1982. Fire Ecology. Wiley, New York, NY.
Zimmerman, G.T., Neuenschwander, L.F., 1984. Livestock grazing influences on community structure, fire intensity, and fire frequency within the Douglas-fir/ninebark habitat type. J. Range Manag. 37, 104–110.