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Postfire grazing management effects on mesic sagebrush-steppe vegetation: Mid-summer grazing☆

Patrick E. Clark*, C. Jason Williams, Patrick R. Kormos, Frederick B. Pierson

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

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ABSTRACT

Postfire livestock grazing management in the higher-elevation, mesic portions of the sagebrush steppe lacks a firm scientific foundation to support decision making. Following a prescribed fire conducted in fall 2002, we evaluated effects of different lengths of rest from mid-summer (July) cattle grazing on postfire ground cover and plant species diversity responses in mesic sagebrush steppe. Treatment levels representing no rest; 1, 2, or 3 years of rest from grazing, and a burned-ungrazed control, all had similar effects on graminoid and forb basal cover and plant species density and frequency. However, grazing reduced litter cover and increased bare ground exposure relative to the control. A synthesis of this and other case studies of postfire grazing in sagebrush steppe indicates multiple years of rest from grazing are not strictly necessary for effective and timely recovery of vegetation but, on sloping terrain where potential runoff and erosion hazards exist, multiple years of rest may be needed to promote sufficient rates of litter recovery and reduction of bare ground exposure. These findings support calls for increased flexibility in policies governing postfire grazing management on federal lands and, as such, could influence ecosystem health, livestock production and other services throughout much of the western US.

1. Introduction

While the effects of postfire livestock grazing on ecological patterns and processes have been fairly well studied in some rangeland ecosystems (e.g., prairie grasslands), these effects are rather poorly understood in others. Fire and postfire herbivory can have interactive effects on plant vigor, potentially causing reduced productivity and, in some cases, mortality. Suppressed reproduction and even localized loss of intolerant species may result from adverse combinations of fire and livestock grazing (Bunting et al., 1998; Jirik and Bunting, 1994; Wright and Bailey, 1982). The physical environment can also be impacted by these combined disturbances. Fire generally increases bare ground exposure and postfire grazing can reduce or delay recovery of litter and ground cover thus prolonging elevated risks of soil erosion (Moody et al., 2013; Pierson et al., 2002) and promoting excessive solar heating and soil moisture losses to evaporation (Hulbert, 1969; Facelli and Pickett, 1991; Villegas et al., 2010). Fire and livestock grazing are common place on most rangelands throughout the world. Consequently, selection of appropriate and effective postfire livestock grazing strategies, which may include rest or deferment from grazing, changes in stocking rates, and other adaptations is an important management consideration for rangelands worldwide.

Unfortunately, a scientific foundation to inform and support postfire grazing management and decision making is lacking or inadequate for some rangeland ecosystem types. While the effects of fire and herbivory have received much study in grassland ecosystems such as the tall grass prairie (Allred et al., 2011; Collins and Smith, 2006), mixed-grass prairie (Collins and Barber, 1986), short-grass steppe (Augustine and Derner, 2014; Augustine et al., 2010) and Mediterranean grasslands (Noy-Meir, 1995); studies in shrub-grass ecosystems like the sagebrush steppe are more rare. Some postfire-grazing research, however, has been conducted on several sagebrush-dominated rangelands (Bates and Davies, 2014; Bates et al., 2009; Bruce et al., 2007; West and Yorks, 2002). This past work has tended to focus in the lower-elevation, drier sagebrush ecosystems dominated by Wyoming big sagebrush (*Artemisia tridentata* Nutt. subsp. *wyomingensis* Beetle & Young) and other dryland sagebrush species. Very little postfire grazing research has been done in the higher-elevation, mesic portions of the sagebrush steppe dominated by mountain big sagebrush (*Artemisia tridentata* Nutt. subsp. *vaseyana* [Rydb.] Beetle) along with antelope bitterbrush (*Purshia tridentata* [Pursh] DC.), and mountain snowberry (*Symphoricarpos oreophilus* A. Gray).

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

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Mesic sagebrush steppe occupies about 7 million ha and thus represents a substantial proportion of sagebrush-dominated rangelands in the western US (Comer et al., 2002). The term "mesic" is used here in a relative sense as these systems can still be rather dry, e.g., cumulative evapotranspirative losses commonly offset much of the annual precipitation input (Kormos et al., 2017). The vast majority of these mesic sagebrush-steppe lands are federally-managed and are grazed by livestock (Knick et al., 2003). Federal land managers commonly employ prescribed fire to control juniper (*Juniperus* spp.) and pinyon pine (*Pinus* spp.) tree encroachment into mesic sagebrush steppe. These fires are intended to rehabilitate or restore vegetation structure and diversity, ecosystem health and function, and delivery of ecosystem services such as wildlife habitat and livestock grazing (Connelly et al., 2000; Davies et al., 2011; McIver et al., 2014; Miller et al., 2014). Pinyon and juniper are native to the western US, but have encroached mesic sagebrush rangelands due to multiple factors including improper land-use practices, lack of fire, climate variability, etc. (Miller et al., 2005; Romme et al., 2009). Effective use of fire to manage conifer encroachment on grazed mesic sagebrush steppe necessitates a rigorous understanding of the interaction of fire and post-fire grazing effects on vegetation structure and diversity.

Clark et al. (2016b) examined the vegetation cover and plant species diversity responses to differing periods of rest from spring (May) cattle grazing following a fall prescribed fire in mesic sagebrush steppe. Even at light stocking rates, however, they found spring grazing during the early reproductive or "boot" phenological stage of forage grasses reduced litter accumulation rates on burned areas thereby prolonging the exposure of bare soils to erosion hazards. To address this concern, we hypothesized cattle grazing during mid-summer when forage grasses like bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] A. Love) and squirreltail (*Elymus elymoides* [Raf.] Swezey) have progressed into the less palatable, seed-formation and seed-ripening phenological stages may have less impact on litter accumulation than spring grazing. We tested this hypothesis in a postfire grazing study conducted following a fall prescribed fire at a study area nearby to that of Clark et al. (2016b) and under a mid-summer (July) cattle grazing regime. Specific objectives for this case study were to: (1) determine if vegetation cover and plant species diversity responses differed among different levels of postfire rest from lightly-stocked, mid-summer cattle grazing; (2) evaluate whether these cover and diversity responses to grazing rest differed among the principal sagebrush-steppe communities or vegetation types present; and (3) compare cover and diversity results from this mid-summer grazing study with those of Clark et al. (2016b).

2. Methods

2.1. Study area

The research was conducted in the Breaks prescribed-fire study area (43° 6′ 29″ N, 116° 46’ 37” 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 (1965–1975, 2002–2014) mean annual precipitation at the Breaks gauges (#145) was 588 mm (NWRC, 2016). Typically about 1/3 of this precipitation occurs as snow (Hanson, 2001). Annual precipitation during the study (2003–2007) ranged from a low of 463 mm in 2007, 543 mm in 2003, 543 mm in 2004, 655 mm in 2006, and a high of 773 mm in 2005. 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, 2016). Mean daily air temperatures during the study ranged from 8.6 °C in 2005, 8.8 °C in 2004 and 2006, 9.4 °C in 2003, and 9.6 °C in 2007, all of which exceeded the long-term mean.

The Breaks study area (176 ha) is a fenced rangeland pasture encompassing the toe-slopes and narrow stream terraces and riparian areas occurring near the base of an east-facing hillslope. Elevation at Breaks ranges from 1547 to 1761 m. Slopes range from flat to very steep (107.5% or 47.1° 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).

Three vegetation cover types: (i) mountain big sagebrush – mountain snowberry, (ii) antelope bitterbrush – mountain big sagebrush, and (iii) native bunchgrass dominate the study. In addition to the 2 dominant species, the mountain big sagebrush-mountain snowberry type includes western juniper, yellow rabbitbrush (*Chrysothamnus viscidiflorus* [Hook.] Nutt.), Utah serviceberry (*Amelanchier utahensis* Koehne), 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 more complete species list). Other components of the antelope bitterbrush-mountain big sagebrush type include western juniper, native bunchgrasses and biscuitroots (*Lomatium* spp. Raf.). Bluebunch wheatgrass, Sandberg bluegrass, squirreltail, Idaho fescue, and needlegrasses (*Achnatherum* spp. Beauv.) dominate the native bunchgrass cover type. Cheatgrass (*Bromus tectorum* L.) has a minor to common presence within all three of these dominant vegetation types. A perennial stream, Reynolds Creek, flowed through the study area. The riparian zone associated with this stream contained a black cottonwood (*Populus balsamifera* L. ssp. *trichocarpa* [Torr. & A. Gray ex Hook.] Brayshaw) overstory; a peachleaf willow (*Salix amygdaloides* Andersson), redosier dogwood (*Cornus sericea* L. ssp. *sericiea*), and Woods' rose (*Rosa woodsii* Lindl.) shrub layer; and a sedge (*Carex* spp. L.) and Kentucky bluegrass (*Poa pratensis* L.) understory.

2.2. Prescribed fire

On 24 September 2002, the Breaks prescribed fire (81 ha) burned about 34 ha of the study area pasture (176 ha). The intent of this vegetation treatment was to decrease shrub cover, enhance availability of herbaceous forages, and kill as many encroaching western juniper trees as possible without adversely impacting 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). Wind speeds during the burn averaged 2.2 m s^{-1} with a range of 0.66–3.5 m s^{-1}. Wind direction remained relatively constant. Air temperature ranged from 20.8 to 22.6 °C and relative humidity 18.1–20.0%. The prescribed burn was successful in that it met the intended results and the results are typical of fall burns. The fire produced a mosaic of areas with low (6 ha), moderate (23 ha), and high (5 ha) burn severity. About 6 ha of unburned areas also remained within the outer perimeter of the fire. These areas of differing burn severity were digitized as polygons using a dual-channel GPS unit (Trimble Pro XRS, Trimble Navigation, Inc., Sunnyvale, California) immediately after the fire was extinguished. Generally, heavier woody fuels promoted longer fire residence times in the antelope bitterbrush-mountain big sagebrush type, consequently, these stands tended to be burned with moderate severity (Clark unpublished data). Finer fuels promoted shorter residence times in the bunchgrass grassland and mountain big sagebrush-snowberry types, generally resulting in low burn severity. However, where burning occurred, it generally killed or greatly suppressed mountain big sagebrush and bitterbrush shrubs and juvenile juniper trees.

Herbaceous vegetation on burned areas recovered quite rapidly. Compared with prefire conditions, perennial forbs and some grasses, particularly squirreltail, increased in cover on formerly shrub-dominated areas during the first postfire year (2003; Clark unpublished data). The postfire landscape generally appeared as a perennial grassland dotted with burned shrub and tree skeletons, occasional inclusions of unburned woody vegetation, and remained so throughout the course of this study (2003–2007).

2.3. Postfire grazing-rest treatments

This study area had traditionally been grazed during mid-to late-summer (July and/or August) after vegetation production had peaked and forage plants like bluebunch wheatgrass had begun to ripen and senesce. During this postfire study, the pasture was not rested but was grazed by 14 lactating beef cows and their calves for 30 days during mid-summer (July) of each study year (2003–2007). Assuming each cow and her calf represented 1.15 metabolic Animal Unit Equivalents (AUEs), the stocking rate in the pasture during the study was about 0.0915 AUEs ha^{-1} or 10.9 ha AUM^{-1}. Typically, this would be considered a fairly light stocking rate for mesic sagebrush steppe rangelands. Prior to the fire, this pasture was moderately stocked at about 4–7 ha AUM^{-1} which is within the range of rates typically used by the USDI Bureau of Land Management (BLM) and other public land management agencies for this type of rangeland. Our research, however, was intended to provide both private and public land managers with information useful in developing postfire cattle grazing management strategies. Private lands, such as this study area, often are not rested from grazing during the initial growing seasons after a prescribed fire. In contrast, managers of federal agency lands in the USA (e.g., BLM) typically follow a guideline of resting burned pastures from livestock grazing for at least 2 postfire growing seasons. The intent of this guideline is to allow burned vegetation to recover vigor before being subjected to the stresses of postfire herbivory (Wright and Bailey, 1982). This postfire-rest 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. Use of conservative stocking rates as an alternative strategy to postfire deferment or rest from livestock grazing is supported by research from the short-grass steppe (Augustine et al., 2010) but the concept has not been rigorously tested in the sagebrush steppe.

Postfire grazing-rest treatment levels in this study were applied using sets of livestock exclosures within the study area pasture. Six sets of exclosures were established during the postfire fall and winter 2002–2003. Three exclosure sets were randomly located on sites formerly occupied by mountain big sagebrush-snowberry stands and which had received low burn severity. Random sites within former antelope bitterbrush-mountain big sagebrush stands which had received moderate burn severity were selected for the remaining 3 exclosure sets. Areas burned at moderate and high severity in the mountain big sagebrush-snowberry type or at low or high severity in the antelope bitterbrush-mountain big sagebrush type were considered atypical and thus excluded from selection.

Each exclosure set consisted of 4 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. Each exclosure or unexclosed area within a set represented one level of the 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 July for postfire years 2003–2007. 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 (2004) thus exposing the vegetation to cattle grazing during July for study years 2004–2007. The second exclosure of each set, which represented the 2-Year Rest treatment level, began being grazed by cattle during July of the third postfire year (2005). Fences of the third exclosure per set were removed so cattle could graze the interior starting the fourth post-fire year (2006) thus representing the 3-Year Rest treatment level. 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 (2003–2007). Wild herbivores (e.g., mule deer [*Odocoileus hemionus* Rafinesque], elk [*Cervus elaphus* Linnaeus], and rabbits [*Lepus* spp.]) were capable of jumping or passing through the exclosures fence and thus had access to all exclosed and unexclosed areas throughout the study duration.

2.4. Vegetation sampling

Forage utilization was assessed using the caged paired-plot method (Cook and Stubbendieck, 1986) in each treatment area exposed to cattle grazing. Multiple utilization assessments were made in July, at about 15-day intervals using movable cages, to minimize issues associated with differential plant growth (Cook and Stoddart, 1953). Utilization was determined on a dry-weight basis as the percentage removal of herbaceous biomass. Although not formally sampled, 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. Wildlife utilization was rarely detected inside the exclosures and then only in trace amounts.

Basal cover by growth form, litter cover, and bare ground percentage within treatment areas were assessed using point-intercept sampling. Four 1-m^2 quadrats were randomly positioned within each exclosure or unexclosed area. Sampling with a 20-pin frame was applied at 5 stations per 1-m^2 quadrat thus yielding a total of 100 pins per 1-m^2 area. Pin-point basal contacts with 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 was conducted just prior to the 30-day grazing trial of 2003 (pre-treatment assessments), during treatment application years 2004 and 2005, and during the post-treatment assessment year (2007). Sampling of the 3-Yr Rest and Control levels was not conducted in 2005 due to logistical constraints. No sampling was not conducted in 2006.

Plant species density was measured in each 529-m^2 exclosure or unexclosed treatment area by conducting a timed search for all macroscopic plant species within the exclosure or areal boundaries. Species were 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, using a nested frequency approach (Bonham, 2013) at the same random locations used for the basal-cover sampling (see above). Five different plot sizes, ranging from 1 m^2 to 0.0008 m^2, were used. All macroscopic plant species rooted in the nested plots were recorded. From this larger set, several target graminoid, forb, and shrub species were selected for analysis. Target species were selected based on documented importance as livestock forage; as wildlife forage, browse, and cover; or as having special status such as being an invasive exotic (e.g., cheatgrass). We applied the 20–80 percent rule (Despain et al., 1991) for choosing which plot size to use in the frequency calculations. In almost all cases, presence data from the 1-m^2 plot size was selected for calculation. The 0.0625-m^2 plot size, however, was used to assess frequency of cheatgrass.

2.5. Statistical analysis

A split–plot experimental design was used in this case study. The three exclosure sets within each of the two formerly shrub-dominated, vegetation cover types were considered whole plots and the exclosures

Table 1
Least-squares mean estimates and 95% confidence limits for percentage basal cover and bare ground responses to 5 different levels of rest from postfire cattle grazing as measured in 2007, 5 years after application of the Breaks prescribed fire in the Owyhee Mountains of southwestern Idaho.

		Basal cover		
Category	Treatment	Estimate	Lower CL	Upper CL
		%		
Graminoid	No-Rest	5.63[a†]	3.67	8.01
	1-Yr Rest	3.59 [a]	2.08	5.53
	2-Yr Rest	3.62 [a]	2.08	5.58
	3-Yr Rest	3.19 [a]	1.77	5.02
	Control	2.75 [a]	1.44	4.46
Forb	No-Rest	0.212 [a]	0.0264	0.574
	1-Yr Rest	0.174 [a]	0.0142	0.511
	2-Yr Rest	0.344 [a]	0.0829	0.783
	3-Yr Rest	0.533 [a]	0.187	1.06
	Control	0.217 [a]	0.0278	0.585
Litter	No-Rest	40.1 [a]	34.0	46.2
	1-Yr Rest	39.5 [a]	33.4	45.5
	2-Yr Rest	48.8 [a]	42.7	54.8
	3-Yr Rest	49.5 [a]	43.5	55.5
	Control	63.9 [b]	57.9	69.9
Bare Ground	No-Rest	48.2 [a]	41.8	54.6
	1-Yr Rest	52.0 [a]	45.7	58.4
	2-Yr Rest	43.6 [a]	37.2	50.0
	3-Yr Rest	40.8 [ab]	34.4	47.1
	Control	29.2 [b]	22.8	35.6

[†]Within each of four categories, 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.

or unexclosed areas representing the five treatment levels within exclosure sets were sub-plots. While locations of the exclosure sets within vegetation types were completely randomized, logistical constraints dictated that the treatment levels be systematically rather than randomly assigned among the exclosures within each set. However, given the randomization at the exclosure-set level, the potential for biasing in the systematic treatment assignment is probably quite low.

Mixed-model analysis of covariance (ANCOVA) was used to evaluate basal cover, plant species density, and frequency responses to the five postfire grazing-rest treatment levels. Vegetation type, treatment, and their interaction were considered fixed effects. Nesting of exclosure sets within vegetation types was a random effect. Pre-treatment conditions, measured prior to cattle entry in 2003, were included as a continuous covariant in the ANCOVA. Contrasts of the treatment vs. covariate regression slopes for each treatment level were used to test the ANCOVA independence assumption (Doncaster and Davey, 2007). The ANCOVA model was fitted using the Mixed procedure in SAS (SAS Institute Inc., 2011) and significance was reported at the 0.05 alpha level. Denominator degrees of freedom for F-tests of fixed effects were adjusted using the Kenward-Roger method (Kenward and Roger, 1997). 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 graminoid and forb basal cover data to obtain normality of ANCOVA model residuals. Results for these responses were back transformed for presentation.

The reader should note that application of the different treatment levels within the experimental design was executed in stages. Cattle grazing under the No-Rest treatment level was applied during July of the first postfire year (2003) while grazing was first applied to the 1-Year Rest treatment level during the second postfire year (2004), and so on. Consequently, to avoid missing any short-term responses which might have occurred during the staged treatment application, we conducted three separate ANCOVA, two for treatment application years 2004 and 2005, and one final analysis using data from post-treatment year, 2007.

3. Results

3.1. Utilization levels

Light to moderate utilization of herbaceous plants was observed within the treatment areas exposed to grazing during each year of treatment application. During 2003, the first growing season following fire, mean utilization in the No-Rest Treatment level was 31.8 ± 2.45% SD. Mean utilization rates were 28.2 ± 1.67% SD in the No-Rest level and 25.9 ± 3.75% SD in the 1-Yr Rest level during 2004. In 2005, mean utilization rates were 19.3 ± 4.82% SD in the No-Rest level, 18.4 ± 3.78% SD in the 1-Yr Rest, and 17.7 ± 4.22% SD for the 2-Yr Rest. Mean utilization rates of 22.6 ± 2.74% SD, 29.5 ± 3.23% SD, 23.8 ± 3.90% SD, and 19.4 ± 5.29% SD were observed in 2006 for the No-Rest, 1-Yr Rest, 2-Yr Rest, and 3-Yr Rest treatment levels, respectively. Utilization was not formally assessed in 2007 because sampling of other vegetation responses was conducted before any cattle grazing commenced during this post-treatment year. An informal survey confirmed little or no wildlife herbivory had occurred within the treatment areas prior to response sampling in 2007.

Utilization during 2005 was generally somewhat less than that obtained during other study years. Spring 2005 (May and June) was quite wet and graminoid production was more than twice that of 2003 and 2004 (Clark et al., 2016a). It is likely a heavy crop of reproductive culms, which cattle tend to avoid (Ganskopp et al., 1992, 1993) depressed forage utilization rates in 2005.

Utilization in treatment areas exposed to grazing tended to be about 9–16 percentage points higher than that detected in burned areas within the remainder of the pasture (Clark unpublished data). This discrepancy was also noted in a previous postfire study conducted under a spring grazing regime (Clark et al., 2016b). In both cases, cattle seemed to be attracted to the treatment areas, perhaps to the exclosure fencing, sampling stakes, etc. Consequently, the reader should take note that this study was conducted under higher utilization levels than might be expected from the light stocking rates we used.

3.2. Basal cover and bare ground

Graminoid basal cover was not affected by the postfire grazing-rest treatment during 2004 ($P = 0.2402$), 2005 ($P = 0.4541$), and 2007 ($P = 0.1632$; Table 1), i.e., the second, third, and fifth postfire years, respectively. The pretreatment covariate ($P = 0.0340$) was the only significant term in the 2007 ANCOVA model thus indicating that most of the variability in graminoid basal cover could be explained by pretreatment conditions (e.g. spatial variability in graminoid cover among exclosures) rather than treatment effects.

A treatment effect was detected in forb basal cover during the 2005 ($P = 0.0063$) but not in 2004 ($P = 0.2870$) or 2007 ($P = 0.5807$; Table 1). In 2005, forb cover was greater in the 3-Yr Rest treatment level (3.58 ± 0.0615% SE) which had not yet received postfire grazing, than in the 1-Yr Rest level (0.841 ± 0.0615% SE). The only significant term in the 2007 ANCOVA model was the pretreatment covariate ($P = 0.0045$).

Litter cover was affected by treatment in 2004 ($P = 0.0003$) and 2007 ($P < 0.0001$) but not in 2005 ($P = 0.5588$). During 2004, the 2-Yr Rest treatment level (45.4 ± 3.05% SE) which was yet to be grazed, had more litter cover than the No-Rest (26.2 ± 3.08% SE) and Control (30.9 ± 3.05% SE) treatment levels which were similar. In 2007, after all treatment levels had been fully implemented, more litter cover was present in the control than all other treatment levels (Table 1). None of the other terms in the 2007 ANCOVA model were significant.

The amount of bare ground was also affected by treatment in 2004 ($P = 0.0058$) and 2007 ($P < 0.0001$) but not in 2005 ($P = 0.1460$). In 2004, the 2-Yr Rest treatment level (54.8 ± 3.02% SE) had less bare ground exposed than the No-Rest (67.4 ± 3.01% SE) and Control (67.1 ± 3.02% SE) levels. During 2007, the Control had less bare ground than all the other levels except the 3-Yr Rest level (Table 1). The pretreatment covariate was the only other significant term in the 2007 ANCOVA model thus indicating most of the remaining variability in bare ground exposure was due to pretreatment conditions.

3.3. Species density by functional group

The postfire grazing-rest treatment did not affect species density in any of the four functional groups evaluated during any study year (Table 2). The vegetation type by treatment interaction term was significant in the 2007 ANCOVA model for density of annual forb species ($P = 0.0183$). In the bitterbrush-mountain big sagebrush type, the 3-Yr Rest treatment level (4.71 ± 1.05 species treated area^{-1} SE) had lower species density than the 1-Yr Rest level (7.75 ± 1.06 species treated area^{-1} SE) while, in the mountain big sagebrush-snowberry type, species density was similar among all treatment levels. Vegetation type was the only significant term in the 2007 ANCOVA model for shrub species density ($P = 0.0007$). The bitterbrush-mountain big sagebrush type had greater shrub species density (5.26 ± 0.254 species treated area^{-1} SE) than the mountain big sagebrush-snowberry type (3.74 ± 0.254 species treated area^{-1} SE). Table 3 provides a summary of species counts by year and functional group across all postfire treatment levels. A full species list for this study is provided in the online Supplemental Materials associated with this paper.

3.4. Species frequency

Four perennial and one exotic annual graminoid species were evaluated for frequency of occurrence under each treatment level (Table 4). The postfire grazing-rest treatment did not affect the frequency of any of these species during any study year analyzed. In most cases, the only significant term in the ANCOVA models for these graminoid species was the pretreatment covariate indicating the variability in frequency of occurrence could generally be explained by pretreatment conditions.

Frequencies of the following 6 perennial forb species were analyzed including common yarrow (*Achillea millefolium* L.), pale agroseris (*Agroseris glauca* [Pursh] Raf.), tapertip hawksbeard, western stoneseed, silvery lupine, and longleaf phlox (*Phlox longifolia* Nutt.). These species were selected because they are typically dominant in mesic sagebrush-steppe rangelands and some are important in wildlife diets (e.g., sage-grouse [*Centrocercus urophasianus* Bonaparte]) (Barnett and Crawford, 1994; Drut et al., 1994; Klebenow and Gray, 1968; Pyle and Crawford, 1996). In almost all cases, the postfire grazing-rest treatment did not affect the frequency of these perennial forb species (Table 5). In 2007, however, the frequency of silvery lupine ($P = 0.0052$) was greater under the 2-Yr Rest treatment level than under the No-Rest and 1-Yr Rest levels. Vegetation type affected the frequency of tapertip hawksbeard in 2007 when this species occurred much more frequently in the mountain big sagebrush-snowberry type (62.7 ± 8.87% SE) than in the bitterbrush-mountain big sagebrush type (6.27 ± 8.87% SE). Generally, for all species and most study years, pretreatment conditions explained much of the variability in frequency of occurrence. 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.

Table 2
Least-squares mean estimates and 95% confidence limits for species density responses within 4 functional groups to 5 different levels of rest from postfire cattle grazing as measured in 2007, 5 years after application of the Breaks prescribed fire in the Owyhee Mountains of southwestern Idaho.

		Species density		
Functional group	Treatment	Estimate[a]	Lower CL	Upper CL
		# of Species treated-area^{-1}		
Perennial[b] Graminoids	No-Rest	5.01	4.48	5.55
	1-Yr Rest	5.45	4.92	5.99
	2-Yr Rest	5.45	4.92	5.99
	3-Yr Rest	5.23	4.69	5.77
	Control	5.19	4.65	5.73
Perennial Forbs	No-Rest	9.27	7.23	11.3
	1-Yr Rest	11.3	9.23	13.3
	2-Yr Rest	11.3	9.23	13.3
	3-Yr Rest	10.7	8.68	12.8
	Control	10.8	8.77	12.8
Annual Forbs	No-Rest	7.83	6.14	9.52
	1-Yr Rest	8.05	6.34	9.75
	2-Yr Rest	8.18	6.49	9.88
	3-Yr Rest	7.33	5.64	9.03
	Control	8.28	6.57	9.99
Shrubs	No-Rest	4.17	3.37	4.97
	1-Yr Rest	4.83	4.03	5.63
	2-Yr Rest	4.34	3.53	5.15
	3-Yr Rest	5.16	4.34	5.98
	Control	4.00	3.20	4.81

[a] Within each functional group, least-squares mean estimates were found to be similar among all treatment levels.
[b] 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.

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

	Graminoids		Forbs		
Years	Perennial	Annual	Perennial	Annual	Shrubs
2003	16	2(2)[a]	43	33	9
2004	12	2(2)	35	21	10
2005	10	2(2)	39	27	10
2007	9	1(1)	30	22	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 four perennial bunchgrass species and one exotic, annual grass species during two treatment application years (2004 and 2005) and the post-treatment year (2007) under 3 or 5 levels of postfire grazing-rest treatment at the Breaks prescribed fire study area within the Reynolds Creek Experimental Watershed in the Owyhee Mountains of southwestern Idaho.

		Perennials				Annual
Year	Level	ELEL5[a]	LECI4	POSE	PSSPS	BRTE
		%				
2004	No-Rest	51.2 ± 9.69[b]	10.6 ± 3.57	72.6 ± 8.02	49.4 ± 8.20	33.2 ± 11.4
	1-Yr Rest	42.9 ± 9.69	8.67 ± 3.63	58.0 ± 7.62	43.9 ± 8.02	24.5 ± 11.1
	2-Yr Rest	56.2 ± 9.58	7.57 ± 3.55	61.8 ± 7.58	50.4 ± 8.07	19.2 ± 11.7
	3-Yr Rest	45.3 ± 9.66	NA[c]	58.4 ± 7.73	33.3 ± 8.23	19.9 ± 11.0
	Control	42.0 ± 9.84	10.6 ± 3.57	70.1 ± 7.58	39.7 ± 8.02	24.1 ± 11.0
2005[d]	No-Rest	43.3 ± 19.4	41.3 ± 16.5	62.3 ± 10.5	38.4 ± 5.24	38.7 ± 10.6
	1-Yr Rest	30.8 ± 19.4	37.9 ± 16.5	44.6 ± 9.97	35.4 ± 5.27	31.3 ± 10.3
	2-Yr Rest	59.2 ± 19.4	25.0 ± 16.5	72.3 ± 9.64	47.1 ± 5.19	25.8 ± 11.0
2007	No-Rest	53.7 ± 15.1	13.6 ± 6.25	76.8 ± 8.83	65.2 ± 8.88	83.2 ± 13.5
	1-Yr Rest	53.7 ± 15.1	−0.76 ± 6.22	67.6 ± 8.40	60.8 ± 8.69	56.0 ± 13.1
	2-Yr Rest	55.6 ± 15.0	16.9 ± 6.20	70.3 ± 8.35	58.2 ± 8.75	70.1 ± 13.7
	3-Yr Rest	63.4 ± 15.1	2.45 ± 6.32	65.0 ± 8.51	50.9 ± 8.91	41.2 ± 13.0
	Control	61.1 ± 15.3	5.31 ± 6.25	62.0 ± 8.35	56.6 ± 8.69	66.2 ± 13.0

[a] Symbols for graminoid species where ELEL5 = squirreltail, LECI4 = basin wildrye, POSE = Sandberg bluegrass, PSSPS = bluebunch wheatgrass, and BRTE = cheatgrass.
[b] All frequency values within species and year combinations were similar between or among treatment levels at the 0.05 alpha level based on SMM-adjusted multiple comparisons.
[c] The 3-Yr Rest treatment level was dropped from the 2004 ANCOVA model for basin wildrye (LECI4) to satisfy treatment-covariate independence assumptions.
[d] Sampling of the 3-Yr Rest and Control levels was not conducted in 2005 due to logistical constraints.

Table 5
Least-squares mean estimates and standard errors for the frequency responses of six perennial forb species during two treatment application years (2004 and 2005) and the post-treatment year (2007) under 3 or 5 levels of postfire grazing-rest treatment at the Breaks prescribed fire study area within the Reynolds Creek Experimental Watershed in the Owyhee Mountains of southwestern Idaho.

	Treatment	Perennial forb species					
Year	Level	ACMI2[a]	AGGL	CRAC2	LIRU4	LUAR3[b]	PHLO2
		%					
2004	No-Rest	24.3 ± 6.68	5.25 ± 9.38	47.9 ± 8.05	21.8 ± 5.63	38.3 ± 7.38	NA[c]
	1-Yr Rest	28.2 ± 6.65	21.9 ± 9.38	47.5 ± 7.88	34.7 ± 5.66	33.1 ± 7.87	11.6 ± 7.30
	2-Yr Rest	24.3 ± 6.68	30.8 ± 9.43	49.9 ± 8.16	34.3 ± 5.63	58.3 ± 7.66	21.8 ± 7.40
	3-Yr Rest	40.9 ± 6.64	30.8 ± 9.43	59.5 ± 7.92	41.7 ± 5.69	47.9 ± 8.01	19.0 ± 7.46
	Control	32.3 ± 6.73	27.9 ± 9.45	61.9 ± 7.79	38.4 ± 5.63	47.4 ± 7.60	10.2 ± 7.27
2005[d]	No-Rest	31.1 ± 6.54	13.3 ± 9.72	49.4 ± 10.4	32.1 ± 6.10	50.5 ± 8.81	3.35 ± 5.65
	1-Yr Rest	33.7 ± 6.67	25.9 ± 9.72	34.5 ± 9.72	27.6 ± 6.11	40.1 ± 8.69	9.15 ± 5.65
	2-Yr Rest	26.9 ± 6.54	31.8 ± 9.78	45.3 ± 10.1	40.4 ± 6.10	63.5 ± 8.59	NA[c]
2007	No-Rest	16.4 ± 7.85	0.774 ± 6.89	31.2 ± 9.12	34.0 ± 8.03	46.5 ± 8.29[b]	13.6 ± 6.49
	1-Yr Rest	0.180 ± 7.83	9.11 ± 6.89	30.4 ± 8.95	37.4 ± 8.06	39.0 ± 8.88[b]	11.4 ± 6.54
	2-Yr Rest	28.9 ± 7.85	10.7 ± 6.93	27.5 ± 9.21	42.4 ± 8.03	85.7 ± 8.63[a]	13.3 ± 6.63
	3-Yr Rest	8.29 ± 7.81	6.53 ± 6.93	40.2 ± 8.99	27.2 ± 8.11	54.8 ± 9.05[ab]	22.7 ± 6.74
	Control	4.57 ± 7.91	2.06 ± 6.94	24.9 ± 8.88	50.7 ± 8.03	57.3 ± 8.55[ab]	9.84 ± 6.53

[a] Symbols for forb species where ACMI2 = common yarrow, AGGL = pale agroseris, CRAC2 = tapertip hawksbeard, LIRU4 = western stoneseed, LUAR3 = silvery lupine, and PHLO2 = longleaf phlox.
[b] All frequency values within species and year combinations were similar between or among treatment levels except for silvery lupine (LUAR3) in 2007 where differing letter codes denote least-squares means which were significantly different at the 0.05 alpha level.
[c] These levels were dropped from the ANCOVA models for longleaf phlox (PHLO2) to satisfy treatment-covariate independence assumptions.
[d] Sampling of the 3-Yr Rest and Control levels was not conducted in 2005 due to logistical constraints.

The following shrub species were present in our nested frequency plots: creeping barberry (*Mahonia repens* [Lindl.] G. Don), bitterbrush, mountain big sagebrush, mountain snowberry, rubber rabbitbrush (*Ericameria nauseosa* [Pall. ex Pursh] G.L. Nesom & Baird), spineless horsebrush (*Tetradymia canescens* DC.), Utah serviceberry, yellow rabbitbrush, and Woods' rose. However, none of these species consistently occurred frequent enough within the largest plot size (i.e., 1-m^2) to meet the minimum frequency (i.e., > 20%) required for analysis. Consequently, ANCOVA models were not fitted for shrub species frequency.

4. Discussion

4.1. Relation to previous research

We found postfire cattle grazing in burned mesic sagebrush steppe, during mid-summer (July) under conservative stocking rates which resulted in 18–32% utilization, had few adverse effects on vegetation basal cover, plant species density or frequency beyond those of fall prescribed burning alone. Given the resiliency conferred by the climate and soil moisture regime of these higher-elevation, mountain big sagebrush-dominated vegetation types, postfire rest from cattle grazing was not strictly required to achieve timely and effective vegetation recovery in this case study. Working nearby in a similar mesic sagebrush-steppe landscape, Clark et al. (2016b) reported findings quite comparable to ours, under light postfire cattle grazing during spring (May). In both these studies, 4 or 5 years postfire, basal cover of graminoid and forb species was similar among a burned-ungrazed control and grazed treatment levels involving either multiple years of rest (i.e., 2 or 3 years), 1 year of rest, or no rest at all from postfire grazing. These basal cover responses indicate plant vigor was not substantially impacted by postfire grazing under the conditions of these two studies. Furthermore, graminoid and forb species-density responses from both studies were also similar among the control and all grazed treatment levels thereby indicating species diversity under postfire grazing was not reduced by excessive mortality or inflated through weedy invasion. Clark et al. (2016b) did report, however, that shrub species density was reduced in their No-Rest treatment compared treatment levels where burned areas received some quantity of rest from grazing postfire. We did not observe a grazing effect on shrub species density in our mid-summer study. Finally, species frequency of dominant perennial graminoid and forb species, in nearly all cases did not differ among the treatment levels of either study and those few differences which were detected (e.g., silvery lupine in the current study and Sandberg bluegrass in Clark et al. (2016b)) did not demonstrate any meaningful trends. While these two studies in mountain big sagebrush-dominated systems differed in terms of seasonal timing of postfire grazing, this difference did not seem to influence the cover and species diversity responses obtained.

Working on a drier sagebrush-steppe rangeland dominated by Wyoming big sagebrush, Bates et al. (2009) also found that multiple years of rest from grazing were not strictly required to obtain satisfactory postfire recovery of vegetation burned by fall prescribed fire. Under a moderate level of postfire cattle grazing (i.e., 50% forage utilization) during late summer (early August), after seed-shatter stage of the dominant perennial grasses; herbaceous production, canopy and basal cover, and species density and presence responses were similar among burned areas which received either no rest, 1 year of rest, or were burned but left ungrazed throughout the study. In most cases, similar findings were also obtained under light postfire grazing (25% utilization) during spring (early-mid May) among treatment levels involving either 1 year of rest, 2 years of rest or burning without postfire grazing; a no-rest treatment was not applied in this spring experiment (Bates et al., 2009).

Combined, the findings of all three sagebrush-steppe studies discussed above suggest postfire cattle grazing can be applied following fall prescribed fire without multiple years of rest while still obtaining effective and timely vegetation recovery as measured by cover and species diversity responses. In fact, under the conditions of these studies, application of cattle grazing during the first growing season after fire can be a viable postfire grazing strategy during spring or mid-summer on mountain big sagebrush rangelands and during late summer in Wyoming big sagebrush systems. Grazing during the first postfire year has the added benefit of allowing cattle to take advantage of increased forage quality that frequently occurs in first postfire year (Hobbs and Spowart, 1984; Willms et al., 1980; and Young and Miller, 1985). Postfire recovery, however, is not just about returning desired vegetation to the landscape. Recovery of surface litter cover and reduction of bare ground exposed by burning are also critical elements of successful postfire recovery. Any amount of postfire grazing removes at least some vegetation material that would, otherwise, have eventually become transformed to litter on the soil surface and thereby reducing bare ground exposure. Consequently, grazing effects on litter must be an important consideration in evaluating postfire grazing strategies.

In both our study and that of Clark et al. (2016b), postfire grazing reduced surface litter cover and increased bare ground exposure compared to the burned but ungrazed control. Under spring grazing on burned mountain big sagebrush rangeland, treatments involving no rest or only 1 year of rest from postfire grazing had less litter cover than the control but litter cover for areas receiving 2 years of postfire rest was similar to the control (Clark et al., 2016b). Similarly, Clark et al. (2016b) found bare ground exposure was greater in the No-Rest and 1-Yr Rest treatment levels than in the 2-Yr Rest and Control. Under mid-summer grazing in the current study, we found all grazing treatments; whether receiving the no rest, 1 year, 2 years, or 3 years of rest from grazing, had less surface litter cover than the burned-ungrazed control. Only the 3-Yr Rest treatment level had bare ground exposure values similar to the control, all other grazed levels had greater exposure. Postfire reductions in litter and increases in bare ground exposure could have important implications for runoff and erosion risk on sloping rangelands, particularly, those on steep and/or highly-erodible soils (Pierson et al., 2010). Given these hazards, a further discussion of the hydrologic consequences associated with postfire grazing and its impacts on litter recovery is warranted here.

4.2. Importance of litter

Litter on the soil surface performs a number of crucial hydrologic functions. This is particularly true of sagebrush-steppe rangelands where caespitose grasses rather than rhizomatous, sod-forming grasses dominate and sloping terrain tends to promote run-off and erosion. Soil in the interspaces between the caespitose plants is often not well secured by plant roots and thus can be quite vulnerable to erosion. Surface litter shields these interspace soils from rain drop impact and consequent splash erosion (Pierson et al., 2010, 2014). Litter also helps retain precipitation where it falls by partially impeding water movement and consequently increasing the opportunity for infiltration (Pierson et al., 2007, 2013; Williams et al., 2014). These functions of surface litter are particularly important on sloping rangeland where runoff and erosion risks are potentially high. When run-off does occur, debris dams formed by surface litter function to increase flow-path tortuosity and complexity thus reducing run-off volume, velocity, and associated erosive energy (Pannkuk and Robichaud, 2003; Cerdà and Doerr, 2008). Litter also shades and insulates the soil surface and obstructs vapor diffusion thereby reducing evaporative losses of soil moisture (Hulbert, 1969; Facelli and Pickett, 1991; Villegas et al., 2010). Consequently, even short-term reductions in litter cover can expose vulnerable surface soils to erosion and excessive evaporative losses.

Both the current study and Clark et al. (2016b) indicate multiple years of rest from postfire grazing may be required to obtain litter recovery rates comparable to prescribed-burned mesic sagebrush steppe where postfire grazing is fully excluded. Prolonged rest from cattle grazing, however, can greatly complicate resource management planning and adversely impact livestock production operations which depend heavily on public lands for growing-season forages. These concerns prompted Clark et al. (2016b) to question whether deferment of grazing could be substituted for rest and still achieve desired litter recovery rates. Deferring postfire grazing until later in the season when forage plants were less palatable might reduce grazing impacts on surface litter accumulation. Under later-season conditions when the dominant graminoid forage species have entered the reproductive phenological stages (e.g., seed-formation and seed-ripening stages), cattle may tend to avoid consuming senesced flowering culms thereby leaving this material to be eventually recruited to surface litter (Ganskopp et al., 1992, 1993). Results from our current study, however, indicate deferring cattle grazing until July, even at light utilization rates, did not substantially reduce postfire grazing effects on litter cover compared to the burned-ungrazed control. Perhaps deferring grazing even later in the season, until the seed-shatter stage in August, when forage palatability is even lower may have promoted better litter recovery on burned-grazed areas. Given the complications and concerns associated with implementing 3 or more years of rest from postfire grazing, further studies to pursue the question of deferment vs. rest effects on litter recovery seem all the more relevant. In any case, it is important to note that grazing-induced reductions in litter cover and increases in bare ground can promote splash erosion and evaporative soil moisture losses and, on sloping rangelands, can decrease infiltration and elevate runoff and erosion risks.

4.3. Conclusions

Conclusions from a single case study such as ours are, by definition, limited in scope, comprehensiveness, and applicability. On the other hand, a synthesis of several comparable case studies can serve to promote and justify broader and more comprehensive and applicable conclusions, particularly, when all these studies report quite similar findings despite somewhat different settings. Based on our synthesis of postfire grazing studies in the sagebrush steppe (discussed above), we conclude that resting burned areas from postfire grazing for multiple years is not a strict necessity for effective and timely recovery of vegetation cover and diversity. In fact, in some cases, grazing burned areas without any rest at all can be a viable postfire grazing strategy for mesic sagebrush steppe. Natural resource managers and livestock producers may find a No-Rest strategy suitable for relatively flat rangelands where lack of slope moderates runoff and erosion concerns thereby allowing vegetation recovery to be the principal focus of postfire management. On sloping rangelands, however, managers and producers, must also consider the effects of postfire grazing on litter recovery and bare ground exposure. Poorly conceived grazing plans could promote increased erosion and site degradation. In areas where runoff and erosion are potential hazards, the needs for timely recovery of litter and reduction of bare ground might become the principal driver of postfire grazing management decisions. While vegetation recovery in mesic sagebrush steppe may proceed effectively with little or no rest from grazing, multiple years of rest may be required to achieve litter recovery rates similar to that of burned but ungrazed areas. Burned areas with steep slopes and/or highly-erodible soils should receive careful consideration in postfire grazing management planning. Where practically feasible, managers and producers may find it worthwhile to apply different and separate postfire grazing management plans for flat and sloping rangelands thereby accommodating both resource conservation and livestock production concerns.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jaridenv.2017.10.005.

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