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Original Research

Managing Medusahead Using Dormant Season Grazing in the Northern Great Basin☆

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ABSTRACT

The invasive annual grass, medusahead, infests rangelands throughout the West, from the Columbia Plateau to the California Annual Grasslands and the Great Basin. Dominating secondary succession in the sagebrush steppe, medusahead can degrade the habitat of threatened species such as the greater sage-grouse. This research explores the potential of dormant season grazing as an applied management strategy to reduce the negative impacts of medusahead while promoting recovery of perennial vegetation at the landscape scale. In particular, it assessed grazing with four treatments from 2018 to 2020: traditional grazing (May–October), dormant season grazing (October–February), traditional + dormant season grazing (May–February), and no grazing. After 2 yr of grazing treatments, biomass, density, cover, and fuel continuity did not differ between treatments ($P > 0.05$). However, biomass measurements were significantly different between years, which is likely due to greater than normal precipitation in 2019 and 2020. Between 2018 and 2019, annual grass biomass increased by 81% (666–1 212 kg ha^{-1}) and perennial grass biomass increased by 165% (118–313 kg ha^{-1}). Litter biomass decreased by approximately 15% in every year since 2018 (2 374, 2 012, and 1 678 kg ha^{-1} in 2018–2020). There were not significant differences in cover or density of annual and perennial grasses between treatments and years. Our results indicate that 2 yr may not be adequate time for dormant season grazing treatments to be effective in reducing the abundance of medusahead and that after 2 yr of treatments, dormant season grazing does not have a detrimental effect on perennial vegetation.

Introduction

Medusahead (*Taeniatherum caput-medusae* [L.] Nevski) is an introduced annual grass primarily from the western Mediterranean region of Eurasia (Young 1992). Medusahead is capable of dominating secondary succession of western rangelands from the Great Basin to the Columbia Plateau, and estimates suggest it has invaded > 2 million ha of rangeland across the western United States (Davies and Johnson 2008; Duncan et al. 2004). Medusahead and other invasive annual grasses, like cheatgrass (*Bromus tectorum* L.), pose major problems for rangeland health including, but not limited to, decreased species diversity, diminished forage quality, and increased accumulation of litter resulting in a combination of more fine fuels and reduced fuel moisture content (Davies 2011; Davies and Johnson 2008; Davies and Nafus 2013; Duncan et al. 2004; Young 1992). Perhaps the most significant threat is the development of an annual grass–fire cycle resulting in more frequent fire. For example, Whisenant (1992) observed fire frequency increasing from 0.1 fires yr^{-1} to 0.5 fire yr^{-1} when introduced annual grass cover increased from 40% to 90%. This in-

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crease in fire frequency further perpetuates the dominance of invasive annual grasses, including medusahead, while degrading big sagebrush (*Artemisia tridentata* Nutt.) rangeland (Davies and Svejcar 2008; Nafus and Davies 2014; Young 1992; Young and Evans 1970).

One of the key reasons medusahead can dominate postfire recovery is due to unique phenological characteristics. Like other winter annuals, an established medusahead seedbank can germinate and emerge in the fall after a moisture event (Young 1992). With fall germination, medusahead possesses rapidly developing roots substantially deeper than perennial bunchgrass seedlings, providing a competitive advantage during the spring and summer when soil water becomes a limiting factor on plant growth (Harris 1977; Hironaka 1961).

Medusahead-dominated rangelands develop a thick layer of litter composed of the prior year's growth (Evans and Young 1970; Mariotte et al. 2017; Torell et al. 1961; Young et al. 1971). This persistent litter layer allows for medusahead seeds to remain suspended in, or covered by, litter, creating advantageous microclimates for germination (Evans and Young 1970; Mariotte et al. 2017; Young et al. 1971). The litter also reduces competition from native plant species (Davies and Svejcar 2008; Young and Evans 1970) by preventing native seeds from contacting mineral soil, which is crucial for germination and seedling establishment (Torell et al. 1961). Reducing the negative impacts of the litter layer should be one of the first priorities when improving annual grass–dominated rangelands (Perryman et al. 2018). Observations from Spackman (2019) have shown a positive feedback cycle; when cattle graze medusahead seedlings, they also consume litter, reducing the amount of dead material on the ground.

Additionally, a large abundance of litter is problematic because it increases fuel continuity across invaded landscapes and reduces fuel moisture content (Davies and Nafus 2013). For example, Davies et al. (2015) found that fuel moisture in annual grass–dominated rangelands may be < 20% once perennial plants reach maturity in July, whereas native-dominated plant communities may not reach this level of desiccation until late August. Lower fuel moisture can contribute to increased fire ignition and spread (Chuvieco et al. 2004; Cruz et al. 2015; Krueger et al. 2016). Hence, reducing herbaceous biomass and fuel continuity can decrease wildfire probability, size, and intensity (Davies et al. 2015; Davies et al. 2016).

The loss of native vegetation in the Northern Great Basin exacerbates recent concern over native sagebrush obligate wildlife such as greater sage-grouse *(Centrocercus urophasianus).* One of the largest threats to greater sage-grouse is habitat loss and fragmentation (Connelly et al. 2000; Stiver et al. 2015). Productive sage-grouse habitat should exceed 15% perennial grass cover and 15–25% sagebrush cover (Connelly et al. 2000; Stiver et al. 2015). Medusahead-dominated rangelands do not allow for these conditions to be met, in some cases reducing perennial grass cover by > 90% (Connelly et al. 2000; Nafus and Davies 2014). Additionally, loss of habitat due to medusahead invasion is detrimental to other species of wildlife, including introduced granivores such as the chukar partridge *(Alectoris chukar).* Due to medusahead's physical properties, such as a high silica concentration, chukar are less able to use medusahead when compared with other non-native grasses such as cheatgrass (Connelly et al. 2000; Davies and Svejcar 2008; Savage et al. 1969).

Medusahead can reduce the grazing capacity of rangelands by > 50% (Hironaka 1961; Young 1992). Medusahead is unique in that it possesses a high insoluble ash content, of which a large portion is silica (> 11%; Bovey et al. 1961; Swenson et al. 1964). This, combined with stiff awns featuring silicate barbs, make the grass palatability low at maturity (Swenson et al. 1964; Villalba and Burritt 2015; Young 1992). However, at the leaf stage medusahead is palatable with > 10% crude protein (Bovey et al. 1961; Young 1992). For comparison, dormant bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] Á. Löve) and crested wheatgrass (*Agropyron cristatum* [L.] Gaertn.) crude protein is about 4% in the fall (Ganskopp et al. 2007; Ganskopp and Bohnert 2001).

When considering these factors, it is apparent that an effective means to reduce the landscape-scale impact of medusahead on western rangelands is needed. Livestock grazing is often considered as a cost-effective management strategy to control invasive annual species (Bernues et al. 2014; Sheley et al. 2014). Specifically, dormant season grazing has been demonstrated to be effective at managing medusahead at the pasture level (~800–1 000 ha); however, there has been no research investigating its effectiveness at larger scales (e.g., 10 000 ha; James et al. 2015). The majority of research investigating grazing-based control of medusahead has been conducted during the spring-summer growing season before medusahead plants reach maturity, with little research focusing on fall-winter dormant season grazing (Davy et al. 2015; DiTomaso et al. 2008; James et al. 2015). Existing research also supports fall grazing as a method to limit the impact of annual grasses (Davies et al. 2021); fall grazing after perennial grasses have gone dormant is minimally detrimental and can actually increase the abundance of perennial grasses in sagebrush stands (Daubenmire 1940; Laycock 1967; McLean and Wikeem 1985). By using fall-winter dormant season grazing, cattle may be able to graze newly emerged medusahead plants, reducing the impacts of medusahead on rangelands.

The objectives of this study were to evaluate the effectiveness of fall-winter dormant season cattle grazing across large landscapes (~10 000 ha) to reduce annual grass (i.e., medusahead) fine fuels and promote perennial bunchgrass abundance.

Methods

Study area

Research was conducted on the Three Fingers Allotment (56 170 ha, 138 800 acres) in southeastern Oregon (43°25′N, 117°8′W), approximately 70 km west of Boise, Idaho, United States (Fig. 1). The Three Fingers Allotment is administered by the Vale District Bureau of Land Management (BLM). Typical of the Northern Great Basin, the study area experiences cool, wet winters with hot, dry summers. Average water year (October 1–September 30) precipitation over the past 30 yr (1991–2020) was 258 mm (10.2 inches), with the majority of the precipitation occurring in the winter and spring months (Western Regional Climate Center 2021). Topography is variable, with flat valleys and steep, rocky hillsides. Elevation ranges from 950 m to 1 400 m (3 100–4 600 ft) with an average slope of 12% (US Geological Survey 2017). Soil texture is generally loamy to clay, and aspect varies by research site (Table 1). Livestock have been present in the area since the late 19th century, with a deferred rotational grazing system used during the summer grazing period (May 1–September 30) since the 1980s (personal communication, local rancher 2019).

The plant community in the Three Fingers Allotment is typical of the Northern Great Basin (Appendix A). Since 1985, 17 wildfires have burned 32 090 ha (79 296 acres) of the Three Fingers Allotment at least once (Fig. 2). Within the burned areas where study plots are located, there was a minimal component of shrubs (1% cover), annual and perennial forbs (4% cover), and perennial grasses (9.5% cover). The burn scars are often still visible as the affected areas have become dominated by medusahead, and nearly 19% (10 571 ha, 26 122 acres) has burned at least twice since 1985.

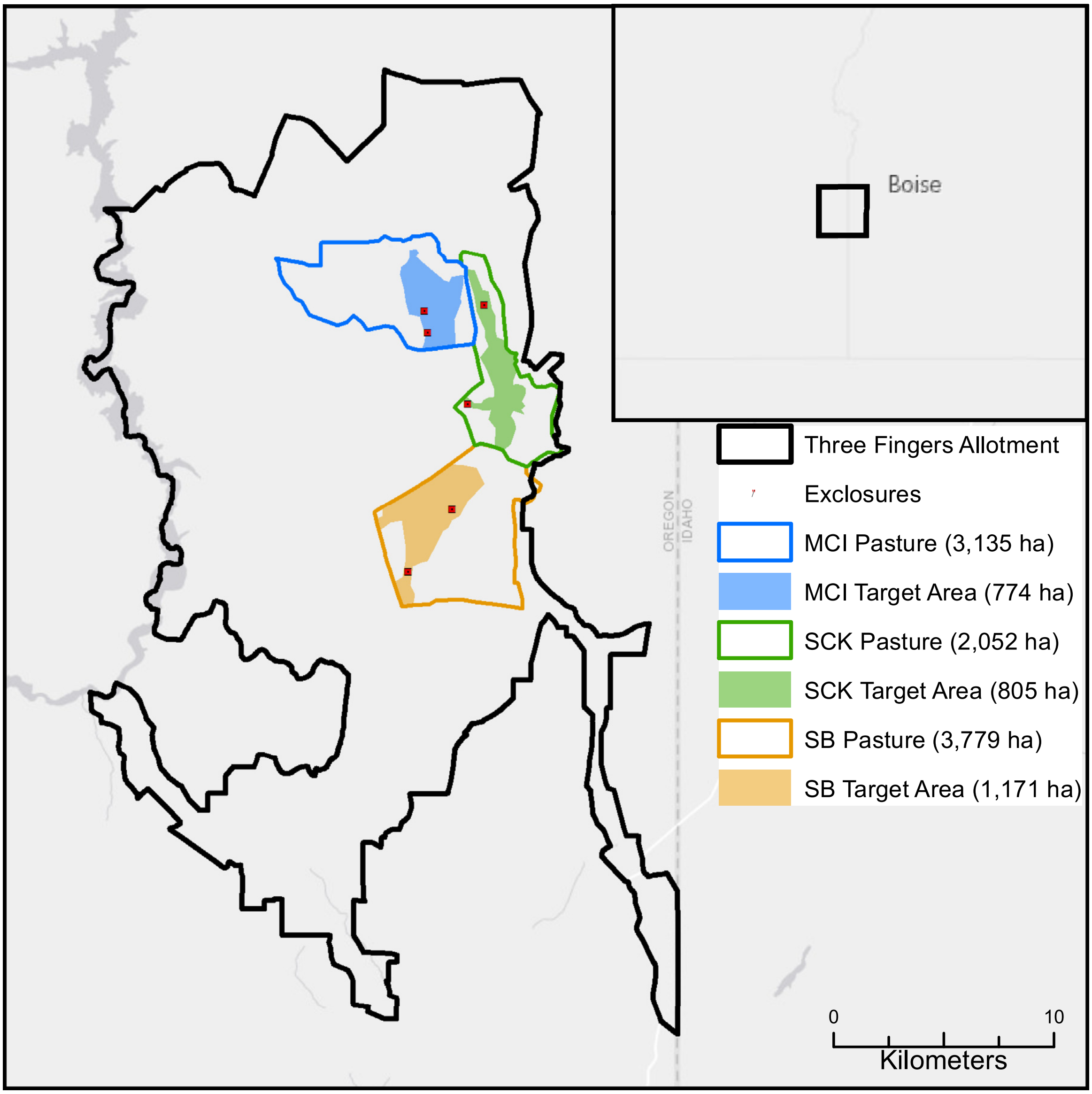

Figure 1. Map of the Three Fingers Allotment and three pastures (SCK: South Camp Kettle; MCI: McIntyre; SB: Saddle Butte) and target areas within pastures where dormant season grazing was concentrated. Target areas are defined as areas within the larger pasture that are dominated by medusahead with reduced ecological function. Within each target area, there were two research sites, north (N) and south (S). Each research site was a four-paddock exclosure where the grazing treatments were applied.

Experimental design

The study was conducted on three pastures within the Three Fingers Allotment: McIntyre (MCI), Saddle Butte (SB), and South Camp Kettle (SCK). Within each pasture, treatments were concentrated within specified target areas (see Fig. 1) that were selected because of the dominance of medusahead with minimal perennial vegetation, recurrent fire history, and proximity to core greater sage-grouse habitat (USDI 2018).

Two sites were randomly placed within each of the three target areas for a total of six research sites (see Fig. 1). Each site consists of four 150 × 150 m grazing exclosures (Fig. 3). The exclosures at each site were randomly assigned to one of the four grazing treatments: nongrazed (NG), traditional grazing from May 1 to September 30 (T), dormant season grazing from October 15 to February 28 (D), and both traditional and dormant season grazing (T + D). The NG treatment exclosure was constructed of a permanent four strand fence, with three strands of barbed wire and a strand of smooth wire on the bottom, on all four sides. The T and D treatment exclosures have permanent barbed wire fence on two sides and a lay-down fence as described in Turner (1960) on two sides to allow cattle access during the prescribed treatment period. The T + D treatment is unfenced year-round. Protein supplements were strategically placed next to roads and at least 0.4 km (0.25 miles) away from study sites throughout each of the target areas during the dormant season.

Grazing was applied at the pasture level during the traditional season (May 1–September 30) in accordance to BLM permits. Dormant season grazing was permitted for a maximum of 1 700 cows across the three pastures, at any one time, from October 15 through February 28. Cows used for this study were bred, commercial cows that calve between March and April. The calves are weaned before exposing the cows to the experimental pasture. Within target areas we strategically located water sources and protein supplementation to draw cattle to that region of the pasture. Target utilization during the dormant season within the target areas was 60%.

Table 1
Soil texture, slope, aspect, and full soil taxonomic name at each of the six research sites. Soil data were collected in June 2020. Texture was obtained using the laboratory hydrometer method (Gavlack et al. 2005).

Research site	Horizon depth (cm)	Texture	% Slope	Aspect	Taxonomic classification
McIntyre North	1-12	Silt Loam	16	W	Fine Smectic Mesic Vitrixerandic Haplargid
	12-25	Silt Loam			
	25-38	Clay Loam			
	38-63	Clay			
McIntyre South	1-10	Silt Loam	12	NE	Fine Smectic Mesic Vitrixerandic Haplargid
	10-30	Clay Loam			
	30-46	Clay Loam			
	46-60	Clay			
Saddle Butte North	1-10	Loam	10	N	Fine Smectic Mesic Vitirixerandic Paleargid
	10-40	Clay			
	40-50	Clay			
	50-60	Clay			
Saddle Butte South	1-6	Loam	20	NE	Very-fine Smectic Mesic Vitrixerandic Paleargid
	6-25	Clay Loam			
	25-44	Clay			
	44-58	Clay			
South Camp Kettle North	1-9	Loam	15	NE	Fine Smectic Mesic Vitrixerandic Haplargid
	9-22	Clay Loam			
	22-40	Clay			
	40-58	Clay			
South Camp Kettle South	1-6	Sandy Loam	12	S	Coarse-loamy Mixed Superactive Vitrixerandic Haplocambid
	6-24	Silt Loam			
	24-38	Loam			
	38-62	Loam			

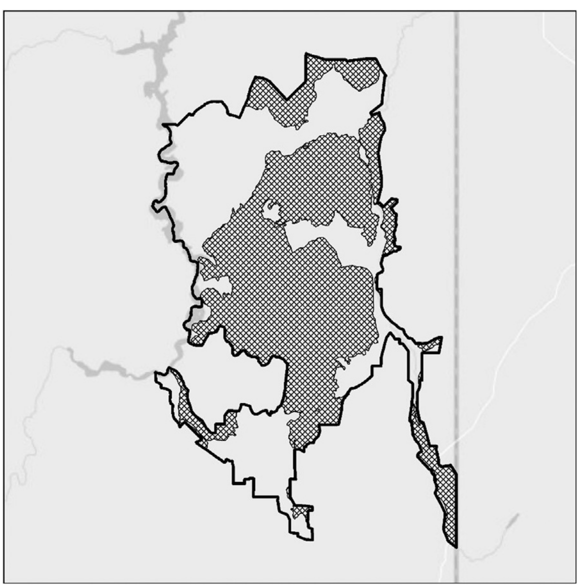

Figure 2. Map showing the footprint of the 17 wildfires (line texture) that have affected 32 090 ha (57%) of the Three Fingers Allotment since 1985. These wildfires have also resulted in 10 571 ha burning at least twice.

Measurements

Fuels and vegetation data were collected in late June of 2018, 2019, and 2020. Fuels data were collected using a modified Fire Effects Monitoring and Inventory System (FIREMON) protocol (Lutes et al. 2006) and modified BLM Assessment, Inventory, and Monitoring Program (AIM) sampling methods (Taylor et al. 2014). Within each exclosure, vegetation and fuel were measured using a 50×50 m plot consisting of three 50-m transects. The three transects were arrayed parallel to each other and spaced 25 m apart (Fig. 4). To ensure destructive sampling points were not measured in repeated years and to capture the heterogeneity of the treatment exclosures, plots were moved 20 m in a random direction each year from the center point used the previous year. Fine fuel measurements consisted of herbaceous biomass and fuel continuity. Vegetation measurements consisted of cover and density of both herbaceous and woody species.

Herbaceous biomass was measured every 10 m per transect using a 0.20-m^2 rectangular quadrat, for a total of 15 samples per plot. Biomass was collected by the following functional groups: annual grass, perennial grass (including standing dead), forbs, and litter. Plants were clipped to 1 cm above ground level, and all litter within the quadrat was collected. Samples were dried for 48 h at 60°C, weighed, and used to calculate total biomass on a kg ha^{-1} basis.

Fuel continuity was assessed using canopy gap between all species regardless of life-span. Along all transects, gap lengths that were devoid of vegetation for at least 20 cm were recorded (Herrick et al. 2005). Percent gap per transect was calculated by adding the total gap in cm and dividing by 5 000 cm (total area measured). Mean gap size and number of gaps was used to characterize fuel continuity.

The line-point intercept method (Herrick et al. 2009) was used to estimate cover of functional groups. A 1-m long metal pin was dropped every 1 m along each transect from a height of 5 cm above the herbaceous canopy, for a total of 150 points for each plot. Live and dead plant interceptions (by species) and litter were recorded. Ground cover was recorded as one of the following soil surfaces: rock, moss, lichen crust, mineral soil, or in the case of the pin intercepting the base of a plant, the species. Each species could be recorded once for each pin drop; hence, when aggregated into functional groups, cover may be greater than 100%. Functional groups consisted of annual grass, perennial grass, Sandberg bluegrass (*Poa secunda* J. Presl), annual forbs, perennial forbs, and litter (Appendix A). Cover was calculated by taking the total number of interceptions per functional group and dividing by 150, then multiplying by 100.

Shrub cover was collected using a 2-m belt transect along each of the three transects. Data were collected for sagebrush species and all other woody species that were > 15 cm in height. Extending 1 m to either side of each transect, the height of the tallest leaf (excluding inflorescences), longest diameter (D1), and perpen-

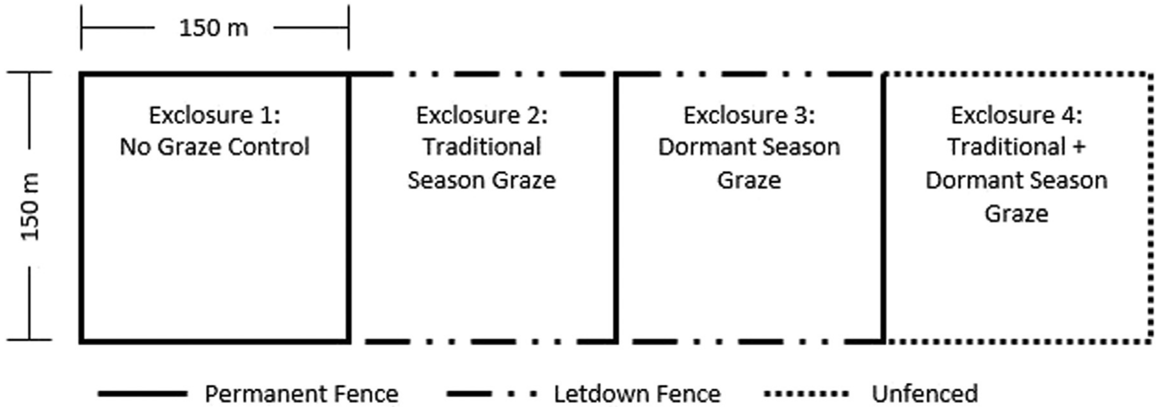

Figure 3. Diagram of the exclosure layout at one of the research sites. Grazing treatments were randomly applied at each research site. Permanent fences are a four-strand fence with three strands of barbed wire and the bottom strand smooth wire to allow for the passage of wildlife. Letdown fences are laid down during the seasons when grazing is allowed and put up during the seasons when grazing is excluded.

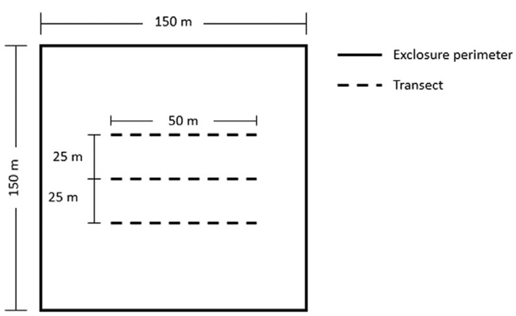

Figure 4. Diagram of the transect layout in one of the grazing exclosures.

dicular diameter (D2) were recorded for all woody species rooted within the belt transect. Plants with < 10% live canopy were not measured. Shrub cover was calculated by adding the area (calculated using the ellipse formula $= \pi \cdot D1/2 \cdot D2/2$ where diameters were divided by 2 to estimate the radius) of each shrub (m^2) in the belt transects and dividing by the total area of the belt transects (300 m^2).

Herbaceous density was measured using a 0.20-m^2 rectangular quadrat every 5 m along each transect, for a total of 30 measurements per plot. Individuals were counted for each life form category: perennial tall grass, perennial short grass (i.e., Sandberg bluegrass), perennial forbs, and annual forbs. Shrub seedlings < 15 cm in height were counted by species. Density of shrubs > 15 cm in height was collected using the same belt transect used for shrub cover. Frequency of invasive annual grasses was recorded by measuring the presence or absence of plants in each quadrat.

Statistical analysis

Mixed-effects model analysis of variance (ANOVA) was used to determine if there was a difference among grazing treatments (JMP, Version 14. SAS Institute Inc., Cary, NC). Fixed variables were treatment, year, and treatment-by-year interactions. Random variables were block, block-by-treatment, block-by-year, and block-by-treatment-by-year interactions. Model parameters were estimated with restricted maximum likelihood. Data were not transformed. Means were reported with standard errors (mean ± standard error) and considered different when $P \leq 0.05$. A Tukey-Kramer HSD test was used to further investigate differences when the ANOVA indicated significant main or interactive effects.

Results

Grazing

Utilization varied between 18% and 62% depending on pasture and year due to differences in forage availability, weather, and access to water (Table 2). Percent utilization was calculated by dividing the current year's biomass (animal unit mo) measured by the previous year's biomass (animal unit mo). Although a somewhat crude measurement of utilization, this method allowed us to capture the regrowth of the plants each season rather than a relative use estimation. Utilization calculations also assumed that all cows concentrated in the target areas within each pasture during the dormant season; hence, estimates presented may be more than actual use in target areas.

Precipitation

Total precipitation (Fig. 8) for the 2018 water yr (October 2017–September 2018) was 195 mm (7.7 inches), 75% of normal. Total precipitation in 2019 was 378 mm (14.9 inches), and in 2020 it was 311 mm (12.2 inches); these values are 147% and 121% of normal, respectively. In 2018, May and June monthly precipitation was 68% (29 mm, 1.1 inches) and 47% (11 mm, 0.4 inches) of the monthly normal, respectively. April and May of 2019 received 176% (54 mm, 2.1 inches) and 248% (107 mm, 4.2 in) of the monthly normal, respectively. May and June of 2020 received 198% (85 mm, 3.3 in) and 358% (86 mm, 3.3 in) of the monthly normal.

Fuels

Grazing treatments had no significant impact on annual grass, perennial grass, forbs, litter, or total biomass ($P = 0.96$, 0.24, 0.14, 0.86, and 0.81, respectively; Fig. 5). There was a significant year effect for annual grass, perennial grass, and forb biomass ($P = 0.049$, 0.022, and < 0.001; Fig. 6). Mean annual grass biomass increased by 81% from 2018 to 2019 (666 kg ha^{-1} and 1 212 kg ha^{-1}, respectively). However, in 2020 annual grass biomass (1 011 kg ha^{-1}) was not different than either 2018 or 2019 (see Fig. 6). Perennial grass biomass increased by 165% from 2018 to 2019 (118 kg ha^{-1} to 313 kg ha^{-1}) and 8% between 2019 and 2020 (313 kg ha^{-1} to 339 kg ha^{-1}; see Fig. 6). There were only trace amounts of forbs in 2018; however, forb biomass was 183 kg ha^{-1} in 2019 and 141 kg ha^{-1} in 2020. There was not a difference between years for litter and total biomass ($P = 0.11$ and 0.17). While not statistically different,

Table 2
Animal unit mo (AUM) remaining after the conclusion of the traditional grazing season. Use (AUMs) is the number of AUMs used during the dormant season (October 15–February 28), and utilization is the percent utilization for the 2018–2019 and 2019–2020 dormant seasons in the three target areas.

Target area	2018–2019 Dormant season			2019–2020 Dormant season		
	AUMs	Use (AUMs)	Utilization	AUMs	Use (AUMs)	Utilization
McIntyre	2 145	1 259	59%	2 968	1 743	59%
Saddle Butte	691	430	62%	5 080	1 047	21%
South Camp Kettle	1 903	348	18%	2 940	815	28%

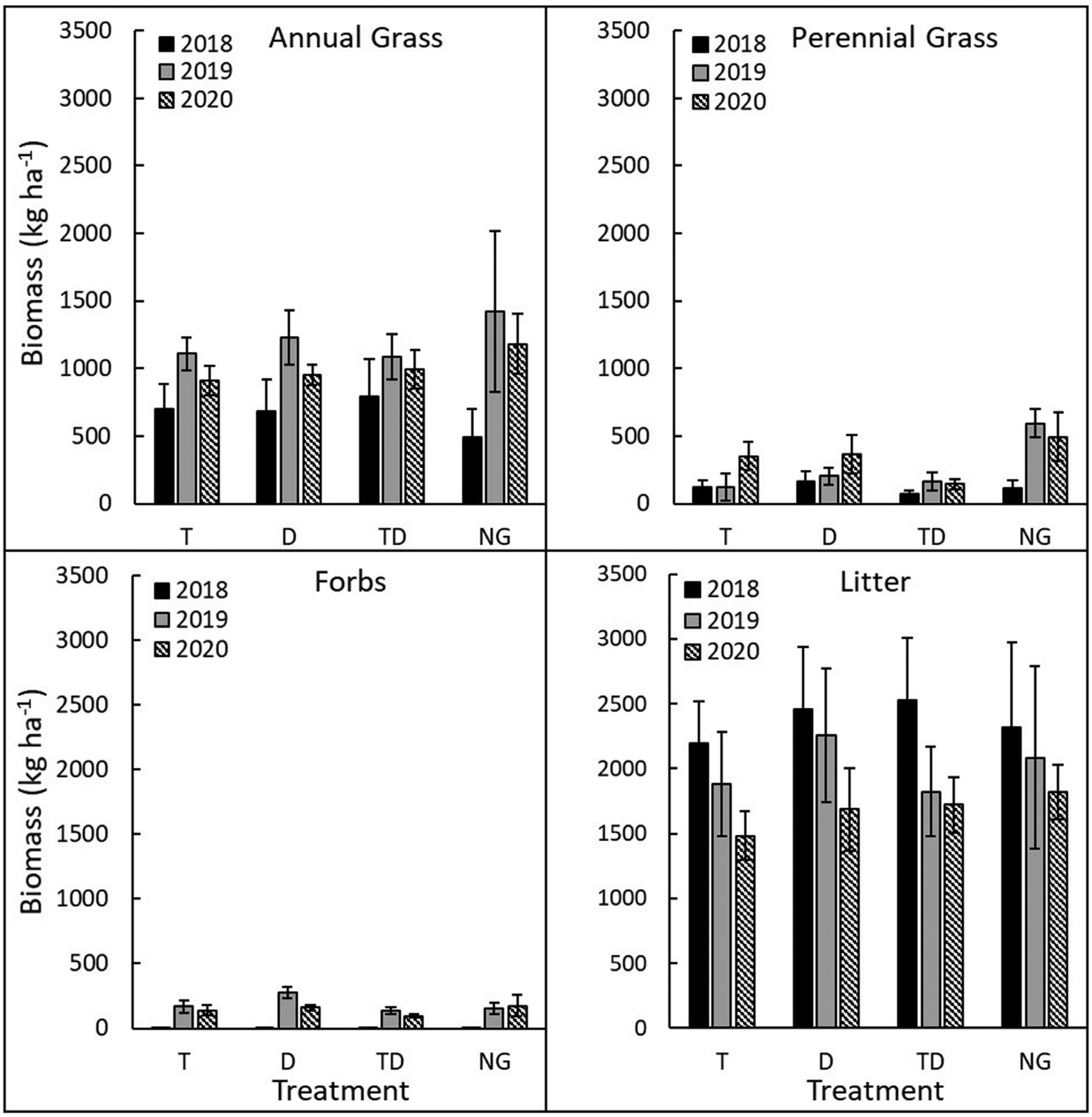

Figure 5. Biomass (mean ± standard error) of four functional groups for each of the four treatments collected in 2018−2020. Functional groups: annual grass, perennial grass, forbs, and litter. Treatments: traditional graze (T), dormant season graze (D), traditional plus dormant season graze (TD), and no graze (NG). Treatment means by year within functional groups were not significantly different (P < 0.05) using the Tukey-Kramer Honestly Significant Difference test. Reported values are the means of each treatment in each year.

it should be noted that litter biomass on average numerically decreased each year since 2018 (2 374 kg ha^{-1} in 2018, 2 012 kg ha^{-1} in 2019, and 1 678 kg ha^{-1} in 2020; see Fig. 6).

Fuel continuity, measured by gap, was not different among treatments, year, or treatment-by-year interactions (P = 0.44, 0.14, and 0.54). Overall fuel continuity was high as percent gap was < 2.5% all 3 yr (Fig. 7). The mean gap size was 27 cm, and there was an average of 4.7 gaps per plot.

Cover

Annual grass and perennial grass cover did not differ among treatments (P = 0.33 and 0.40), year (P = 0.15 and 0.07), or treatment-by-year interactions (P = 0.16 and 0.30; Table 3). Medusahead was the dominant annual grass comprising 82%, 80%, and 73% of the annual grass cover across all treatments and sites in 2018, 2019, and 2020, respectively. Cheatgrass made up 10%, 16%,

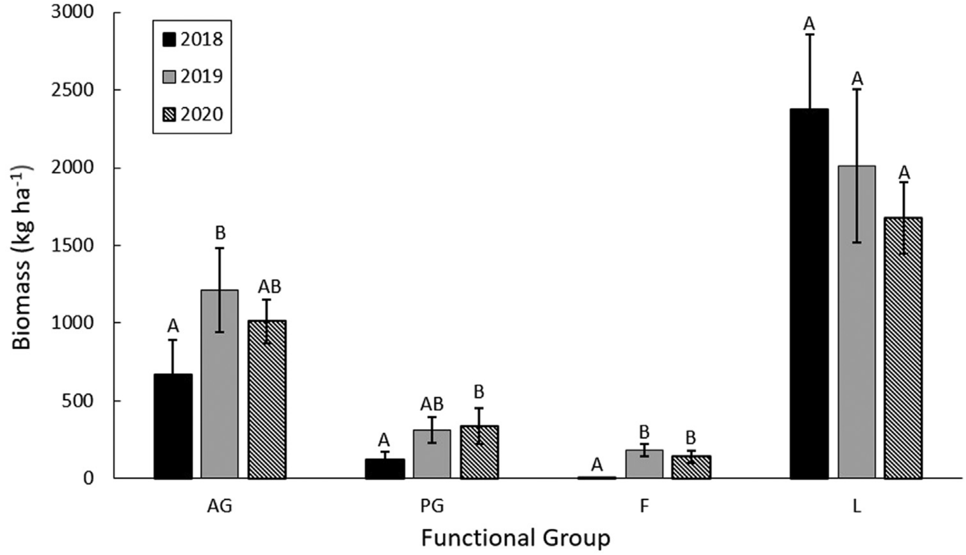

Figure 6. Biomass (mean ± standard error) of four functional groups collected in 2018–2020; functional groups: annual grass (AG), perennial grass (PG), forbs (F), and litter (L). Significant differences between means within a functional group are indicated by different letters using the Tukey-Kramer Honestly Significant Difference test. Differences were considered significant when $P \leq 0.05$. Reported values are the means across all four treatments in each year.

Table 3
Cover (mean ± standard error) of plant functional groups collected in 2018–2020. Values reported: yearly mean of the four treatments, traditional grazing (T), dormant season grazing (D), traditional + dormant season grazing (T + D), no graze (NG). Differences between means are considered significant when $P \leq 0.05$, and a Tukey-Kramer HSD was performed when the *P* value indicated that there was a significant difference. Different letters indicate a significant difference between years.

	Annual grass (% cover)			Perennial grass (% cover)		
	2018	2019	2020	2018	2019	2020
Mean	66.6 ± 11.2	79.9 ± 8.2	72.3 ± 7.7	9.4 ± 3.8	13.0 ± 3.9	12.6 ± 4.6
T	77.8 ± 10.2	84.3 ± 6.5	69.1 ± 9.5	10.4 ± 4.9	16.0 ± 5.2	13.2 ± 5.7
D	72.9 ± 9.8	79.1 ± 5.8	70.0 ± 6.4	8.2 ± 2.8	12.2 ± 3.1	15.9 ± 4.4
T + D	66.0 ± 9.6	79.1 ± 7.1	85.4 ± 5.8	8.9 ± 3.7	9.4 ± 2.1	5.1 ± 1.6
NG	49.6 ± 15.1	76.9 ± 13.2	64.7 ± 9.0	10.0 ± 3.9	14.4 ± 5.1	16.2 ± 6.8
P values	Treatment = 0.33, yr = 0.15, treatment · yr = 0.16			Treatment = 0.40, yr = 0.07, treatment · yr = 0.54		

	Sandberg bluegrass (% cover)			Annual forbs (% cover)		
	2018	2019	2020	2018	2019	2020
Mean	5.9 ± 2.2	3.2 ± 1.3	4.8 ± 2.0	1.6 ± 0.7^{A}	8.8 ± 3.7^{B}	3.6 ± 1.6^{AB}
T	4.2 ± 1.1	1.4 ± 0.6	4.7 ± 1.4	1.1 ± 0.5	6.0 ± 2.5	4.9 ± 2.3
D	4.7 ± 1.8	1.3 ± 0.5	4.6 ± 2.0	1.3 ± 0.6	13.2 ± 7.0	3.6 ± 1.2
T + D	7.3 ± 3.0	4.4 ± 0.7	3.8 ± 1.5	2.2 ± 0.9	8.1 ± 2.3	4.0 ± 2.1
NG	7.3 ± 3.0	5.4 ± 3.2	6.3 ± 3.2	1.6 ± 0.7	7.7 ± 3.2	2.1 ± 1.0
P values	Treatment = 0.44, yr = 0.13, treatment · yr = 0.73			Treatment = 0.59, yr = 0.042, treatment · yr = 0.37		

	Perennial forbs (% cover)			Litter (% cover)		
	2018	2019	2020	2018	2019	2020
Mean	2.5 ± 1.2	3.6 ± 1.4	3.6 ± 1.6	89.8 ± 6.1	86.0 ± 4.7	90.9 ± 2.9
T	3.1 ± 1.4	3.8 ± 1.5	3.2 ± 0.9	93.8 ± 4.1	88.0 ± 4.4	90.4 ± 3.1
D	2.7 ± 1.3	6.2 ± 1.7	3.7 ± 1.6	84.2 ± 10.7	88.2 ± 4.0	89.1 ± 3.3
T + D	2.4 ± 1.3	1.7 ± 1.0	3.1 ± 2.1	92.0 ± 3.1	87.7 ± 4.3	92.7 ± 1.8
NG	1.8 ± 0.9	2.7 ± 1.2	4.3 ± 1.8	89.3 ± 6.6	80.1 ± 6.3	91.3 ± 3.3
P values	Treatment = 0.28, yr = 0.50, treatment · yr = 0.34			Treatment = 0.71, yr = 0.26, treatment · yr = 0.63		

and 25% of the annual grass cover each year. Bluebunch wheatgrass was the most common perennial grass, accounting for an average 29% of the perennial grass cover in all 3 yr of data collection. Western wheatgrass cover made up 14%, 25%, and 32% of perennial grass cover in the 3 yr of data collection.

Annual forb cover was the only functional group with a significant change, increasing from 1.6% in 2018 to 8.8% in 2019 and 3.6% in 2020; however, 2020 was not different from either 2018 or 2019 ($P = 0.042$; see Table 3). This increase in annual forb cover occurred across all treatments and sites and can be attributed to a flush of redstem stork's bill (*Erodium cicutarium* [L.] L'Hér. ex Aiton) and tall annual willowherb (*Epilobium brachycarpum* C. Presl). In 2018, neither the redstem stork's bill nor willowherb were observed at any of the sites, but in 2019 total cover was 5% and 2%, respectively.

Sagebrush and other shrub cover were not different among treatments ($P = 0.28$, 0.41), years ($P = 0.83$, .073), or treatment-by-year interactions ($P = 0.52$, 0.77; Table 4). Rubber rabbitbrush (*Ericameria nauseosa* [Pall. Ex Pursh] G.L. Nesom & Baird) was the most common woody species, accounting for 54% of the total shrub cover averaged across all treatments and sites. Big sagebrush and yellow rabbitbrush (*Chrysothamnus viscidiflorus* [Hook.] Nutt.) were the other common woody species, accounting for 31% and 18% of the total shrub cover, respectively.

Table 4
Cover and density (mean ± standard error) of sagebrush and other shrubs > 15 cm in height, measured in 2018-2020. Values reported: yearly mean of the four treatments, traditional grazing (T), dormant season grazing (D), traditional + dormant season grazing (T + D), no graze (NG). Differences between means are considered significant when $P \leq 0.05$. Sagebrush was composed of all *Artemisia* spp., and other shrubs indicates all other woody species (see Appendix A for species list).

	Sagebrush cover (%)									Other shrub cover (%)								
	2018			2019			2020			2018			2019			2020		
Mean	0.20	±	0.15	0.19	±	0.12	0.26	±	0.17	0.80	±	0.54	0.97	±	0.63	1.35	±	1.10
T	0.21	±	0.21	0.20	±	0.20	0.50	±	0.34	0.31	±	0.14	0.68	±	0.49	0.72	±	0.63
D	0.19	±	0.14	0.04	±	0.04	0.06	±	0.06	0.50	±	0.32	1.12	±	0.79	1.72	±	1.57
T + D	0.28	±	0.17	0.53	±	0.25	0.35	±	0.16	1.40	±	0.97	1.74	±	0.94	1.71	±	1.35
NG	0.11	±	0.08	0.00	±	0.00	0.14	±	0.11	0.98	±	0.72	0.36	±	0.30	1.28	±	0.84
P values	Treatment = 0.28, yr = 0.83, treatment · yr = 0.52									Treatment = 0.41, yr = 0.73, treatment · yr = 0.77								
	Sagebrush density (plants per m^2)									Other shrub density (plants per m^2)								
	2018			2019			2020			2018			2019			2020		
Mean	0.002	±	0.002	0.003	±	0.002	0.004	±	0.003	0.048	±	0.041	0.038	±	0.027	0.022	±	0.018
T	0.002	±	0.002	0.002	±	0.002	0.005	±	0.003	0.017	±	0.009	0.018	±	0.011	0.008	±	0.007
D	0.002	±	0.002	0.002	±	0.002	0.002	±	0.002	0.023	±	0.012	0.025	±	0.015	0.025	±	0.023
T + D	0.002	±	0.002	0.008	±	0.003	0.005	±	0.003	0.090	±	0.082	0.098	±	0.076	0.043	±	0.036
NG	0.002	±	0.002	0.0	±	0.0	0.003	±	0.002	0.063	±	0.061	0.012	±	0.008	0.012	±	0.008
P values	Treatment = 0.44, yr = 0.58, treatment · yr = 0.49									Treatment = 0.34, yr = 0.70, treatment · yr = 0.80								

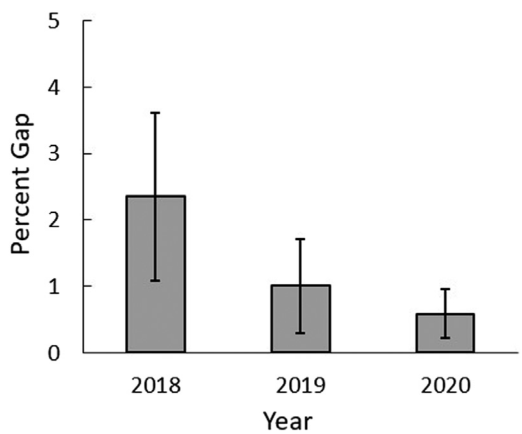

Figure 7. Percent gap (mean ± standard error) measured in 2018–2020. Values reported are the means of the four grazing treatments each year. Gaps are defined as a gap devoid of vegetative material along the transect that was at least 20 cm in length.

Density

There was no change in the density of any of the plant functional groups we measured across treatment, year, or treatment-by-year interactions. Perennial tall grass density averaged 1.08 plants per m^2, and perennial short grasses averaged 0.74 plants per m^2 over the 3 yr (Table 5). Shrub seedling (< 15 cm) density did not differ among treatment, year, or treatment-by-year interactions ($P = 0.61$, 0.91, and 0.27). Sagebrush and other shrub (> 15 cm) density did not change across treatments ($P = 0.44$, 0.34), years ($P = 0.58$, 0.70), or treatment-by-year interactions ($P = 0.49$, 0.80; see Table 4).

Discussion

After 2 yr of grazing treatments, there was no difference in plant community composition among treatments. The only detectable changes were increases among years in annual grass, perennial grass, and forb biomass. For these groups, biomass increased from 2018, a dry yr, to 2019, an above-average year (Fig. 8). This annual variability of precipitation likely decoupled grazing impacts (Young et al. 2020). In addition to total precipitation, timing of precipitation is of great importance to the success of plant growth in the Northern Great Basin and other ecosystems (Bates et al. 2006; Robinson et al. 2013). The increase in total precipitation and monthly precipitation in important spring months is likely a contributing factor in the biomass increase for annual grass, perennial grass, and forbs (Bates et al. 2006; Pilliod et al. 2017).

The decrease in litter biomass was apparent in all treatments, which can likely be attributed to greater than normal precipitation in 2019 and 2020 rather than a result of grazing. Prior studies have shown a correlation between increased precipitation and increased decomposition of litter in semiarid grasslands (Bontti et al. 2009; Epstein et al. 2002; Yahdjian et al. 2006). Higher than normal precipitation in the 2016–2017 water yr likely produced a large medusahead crop in 2017 (Poděbradská et al. 2019; Rao and Allen 2010), and due to the dry yr in 2018, the rate of decomposition of that litter was likely reduced, resulting in a large amount of litter biomass in 2018 (2 374 kg ha^{-1}; Bontti et al. 2009; Epstein et al. 2002). Subsequent wet yrs in 2019 and 2020 then provided the necessary moisture to break down greater amounts of litter biomass.

One factor that may have contributed to the lack of treatment differences is a relatively low stocking rate during the dormant season grazing period (Table 6). A utilization rate of 40%–60% during the dormant season has been shown to reduce total fine fuels (Davies et al. 2016). In our study the 2018–2019 dormant season estimated utilization within the target areas was 59% and 62% in McIntyre and Saddle Butte pastures but only 18% in the South Camp Kettle pasture (see Table 2). In the 2019–2020 dormant season, estimated target area utilization in McIntyre was 59% but only 21% in Saddle Butte and 28% in South Camp Kettle (see Table 2). One of the reasons utilization in the Saddle Butte target area was lower in the 2019–2020 dormant season can be attributed to an increase in annual grass biomass of approximately 1 000 kg ha^{-1}. Additionally, biomass production in 2018 may have been reduced due to a fungus outbreak observed at the site. Abiotic factors also impacted forage utilization during the dormant season, with decreased water availability and snow affecting the ability of the cattle to stay on the range.

Perennial grass cover and density likely did not increase (see Tables 4 and 6) because of the continued dominance of medusahead and the persistence of a robust litter layer. In order for perennial grasses to become more competitive and proliferate, the litter layer must be disrupted in order to decrease the abundance of safe sites exploited by annual grass seedlings and seedbanks (Perryman et al., 2018, 2020). Additionally, 2 yr may not be adequate time for new perennial grasses to establish.

Table 5
Density (mean ± standard error) of plant functional groups collected in 2018-2020. Values reported: yearly mean of the four treatments, traditional grazing (T), dormant season grazing (D), traditional + dormant season grazing (T + D), no graze (NG). Differences between means is considered significant when $P \leq 0.05$.

	Perennial tall grass (plants per m²)									Perennial short grass (plants per m²)								
	2018			2019			2020			2018			2019			2020		
Mean	1.30	±	0.63	1.02	±	0.50	0.90	±	0.42	0.90	±	0.40	0.80	±	0.50	0.50	±	0.20
T	1.37	±	0.71	1.04	±	0.43	1.36	±	0.77	0.60	±	0.30	0.18	±	0.06	0.31	±	0.10
D	1.60	±	0.78	1.63	±	0.97	1.09	±	0.47	0.99	±	0.25	0.45	±	0.23	0.85	±	0.43
T + D	0.95	±	0.41	0.60	±	0.20	0.46	±	0.23	0.94	±	0.34	0.89	±	0.35	0.25	±	0.07
NG	1.31	±	0.62	0.80	±	0.39	0.70	±	0.22	1.15	±	0.55	1.60	±	1.31	0.72	±	0.33
P values	Treatment = 0.45, yr = 0.37, treatment · yr = 0.66									Treatment = 0.40, yr = 0.62, treatment · yr = 0.52								
	Annual forbs (plants per m²)									Perennial forbs (plants per m²)								
	2018			2019			2020			2018			2019			2020		
Mean	0.05	±	0.03	3.86	±	1.82	2.85	±	1.19	1.43	±	0.65	0.37	±	0.14	0.53	±	0.20
T	0.03	±	0.02	2.66	±	1.06	2.13	±	0.83	1.59	±	0.53	0.33	±	0.12	0.35	±	0.06
D	0.08	±	0.04	5.95	±	2.90	4.47	±	1.83	2.00	±	1.25	0.41	±	0.10	0.95	±	0.42
T + D	0.06	±	0.04	3.51	±	1.54	2.77	±	1.19	1.33	±	0.49	0.40	±	0.17	0.41	±	0.15
NG	0.05	±	0.03	3.33	±	1.78	2.06	±	0.91	0.81	±	0.34	0.33	±	0.16	0.43	±	0.15
P values	Treatment = 0.16, yr = 0.05, treatment · yr = 0.31									Treatment = 0.42, yr = 0.047, treatment · yr = 0.88								

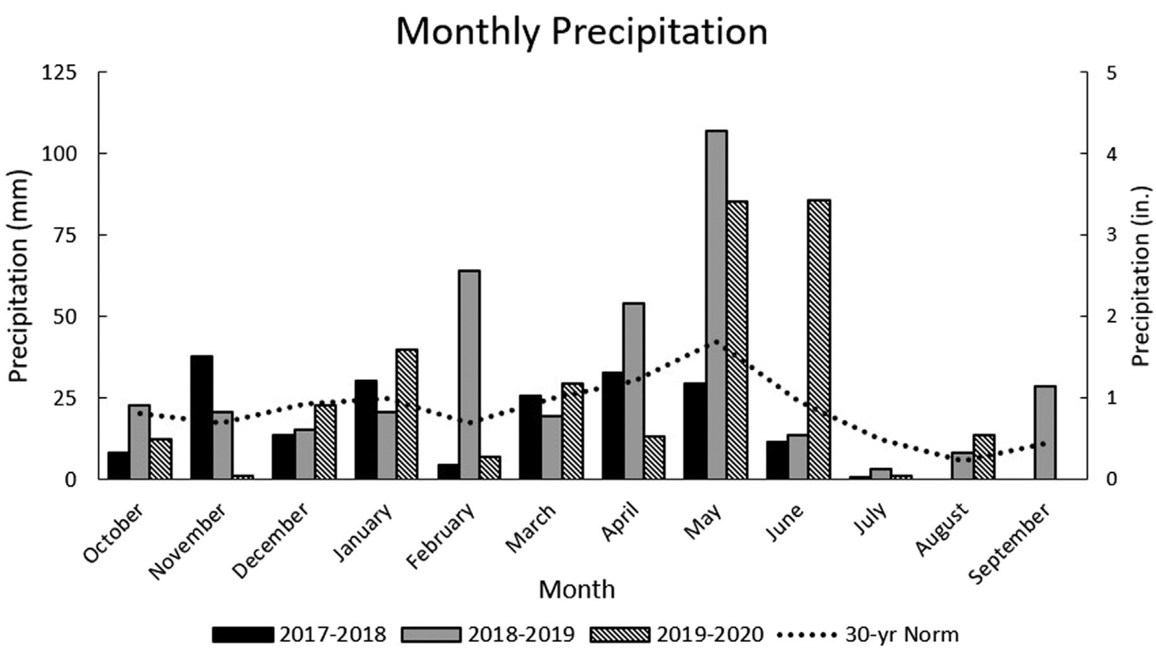

Figure 8. Monthly precipitation for the water year (October–September) at the Owyhee Ridge Remote Automated Weather Station. Water year totals were as follows: 2017–2018, 195 mm (7.7 inches); 2018–2019, 378 mm (14.9 inches); and 2019–2020, 311 mm (12.3 inches). The 30-yr normal is 254 mm (10.2 inches).

Table 6
Mean stocking rate (ha animal unit mo^{-1}) in each of the grazing periods for each of the grazing treatments. Grazing periods are divided into dormant (October–February) and traditional (May–October). Grazing treatments are traditional grazing (T), dormant season grazing (D), traditional + dormant season grazing (T + D), no graze (NG).

Grazing period	Grazing treatment			
	T	D	T + D	NG
Oct. 2018–Feb. 2019	0	5.72	5.72	0
May–Oct. 2019	4.34	0	4.34	0
2018–2019 Total	4.34	5.72	2.40	0
Oct. 2019–Feb. 2020	0	2.61	2.61	0
May–Oct. 2020	3.03	0	3.03	0
2019–2020 Total	3.03	2.61	1.40	0

Despite no increase in perennial grass density, there is no evidence that suggests dormant season grazing is detrimental to perennial vegetation after 2 yr. These findings are in agreement with other similar studies that found fall and winter grazing to have a minimal impact on perennial grasses (Davies et al. 2016, 2021). McLean and Wikeem (1985) showed that there was no difference in percent mortality, height, biomass production, or culm production when bluebunch wheatgrass was defoliated to a stubble height of 5 cm compared with an undefoliated control. While the increase in perennial grass biomass can be attributed to the above-normal precipitation in 2019 and 2020, the lack of change in perennial grass cover and density suggests that the current stocking rate is not detrimental to perennial grasses.

Sagebrush was present across the site in low amounts and was not observed to be increasing. This lack of increase can likely be attributed to competition from medusahead and low precipitation mean annual precipitation. Additionally, the extremely low abundance of sagebrush present on the sites may be due to an inadequate seed source for recovery.

After 2 yr there is no evidence showing a reduction in fine fuels (herbaceous biomass) or fuel continuity at the current utilization levels. However, Davies et al. (2015) demonstrated that dormant season grazing with utilization between 40% and 60% can reduce fine fuel biomass and continuity after 5 yr of treatment. The lack of negative impacts on perennial vegetation indicates that it is worth continuing the research to investigate future emergent impacts.

Implications

Results after 2 yr of dormant season grazing suggest that current stocking density at the research locations for all treatments will not 1) reduce annual grass fine fuels or 2) promote perennial grasses. Our results also found that the current stocking density implemented across grazing treatments was not detrimental to perennial grasses. Although these findings are important, future research with management applications must consider 1) that many of our systems in the western United States fit the nonequilibrium model where variable annual precipitation is likely to better explain the plant community dynamics than livestock grazing alone (Briske et al. 2012) and 2) higher utilization rates are needed.

While it is not possible with our collected data to fully understand the impact that higher than normal precipitation had on the results, continuing this research beyond the current 2 yr will be necessary due to the variable precipitation patterns in the Northern Great Basin. As described by Young et al. (2020), our findings support the notion that the Northern Great Basin is a disequilibrium system where primary production is driven by abiotic systems (Young et al. 2020). Hence, future research should assess the effectiveness of dormant season grazing as a tool to reduce the impact of medusahead and other introduced annual grasses, while also recognizing the limitation of prescribed grazing treatments due to variable abiotic factors such as annual precipitation.

Our results also provide evidence that the stocking rate during the dormant season should be closer to the 40–60% utilization as shown in Davies et al. (2016). Since the target areas in this study were unfenced and cattle were able to move freely throughout the pasture, we were not able to fully account for how the cattle grazed and what type of vegetation was selected. When implementing dormant season grazing to manage fine fuels, it is important to consider season, duration, intensity, and type of animal (Davies and Boyd 2020). In this study, the season of grazing was changed to promote a desirable vegetative response. However, we did not use high-intensity grazing, change the frequency of grazing, or change the class of livestock on the range.

While our results show that 2 yr is not adequate time for significant treatment effects at our utilization rates, it is necessary to determine a timeframe when land managers can expect to see treatment effects. This is critical for land managers who may be developing programs using dormant season grazing as a tool for managing annual grass fine fuels. They will need flexibility and as much information as possible to develop complete objectives and timelines. Additional work is needed to understand cattle distribution and grazing behavior across large heterogenous pastures in both traditional and dormant grazing seasons. By quantifying cattle behavior, it will be possible to make better informed decisions regarding the application of dormant season grazing as a tool for meeting landscape management goals.

Another important factor in this study is how the cattle grazing was implemented. While the cattle were permitted to remain on the range from October to February, they were removed by January 10 in both years due to lack of water and signs of declining body condition. This resulted in the potential for $>$ 2 000 more animal use equivalents (AUEs) that were not used. It may be possible to see changes in the plant community if grazing utilization was a higher priority than animal profitability. It is worth noting that in this study, cattle were provided 1.0–1.5 lb of crude protein via supplement per day to ensure nutritional requirements were met.

Targeted grazing is typically defined as the application of a specific kind of livestock at a determined season, duration, and intensity to accomplish defined vegetation or landscape goals (Launchbaugh and Walker, 2006). This brings up a dilemma, as land managers and livestock producers must make decisions regarding priorities when using grazing as a tool to manage landscapes. If making land management objectives is the priority, livestock production may be reduced in the short-term, and the opposite is true if livestock production is prioritized. More importantly, livestock producers and their partners must find a balance between livestock operation priorities and ecological outcomes that clearly considers animal welfare.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.rama.2023.04.006.

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