CSIRO PUBLISHING

International Journal of Wildland Fire
http://dx.doi.org/10.1071/WF15055

Winter grazing can reduce wildfire size, intensity and behaviour in a shrub-grassland

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Abstract. An increase in mega-fires and wildfires is a global issue that is expected to become worse with climate change. Fuel treatments are often recommended to moderate behaviour and decrease severity of wildfires; however, the extensive nature of rangelands limits the use of many treatments. Dormant-season grazing has been suggested as a rangeland fuel treatment, but its effects on fire characteristics are generally unknown. We investigated the influence of dormant-season (winter) grazing by cattle (*Bos taurus*) on fuel characteristics, fire behaviour and area burned in Wyoming big sagebrush (*Artemisia tridentata* subsp. *wyomingensis*) shrub-grassland communities in south-eastern Oregon, USA. Winter grazing was applied for 5 years before burning and compared with ungrazed areas. Winter grazing decreased fine fuels and increased fine fuel moisture, which reduced flame height and depth, rate of spread and area burned. Winter-grazed areas also had lower maximum temperature and heat loading during fires than ungrazed areas, and thereby decreased risk of fire-induced mortality of important herbaceous functional groups. These results suggest that winter grazing may be a fuel management treatment that can be applied across vast shrub-grasslands to decrease wildfire risk and fire intensity to mediate climate change effects on wildfire activity.

Additional keywords: cattle grazing, fire behaviour, fire management, fuel management, rangeland, sagebrush, wildfire suppression.

Received 27 February 2015, accepted 16 May 2015, published online 11 August 2015

Introduction

A general increase in wildfires and occurrences of large-scale wildfires (mega-fires) has become a global issue (Krawchuk *et al.* 2009; Adams 2013). In the United States in the last 15 years, 7 of the 11 westernmost states have experienced their largest wildfire since European settlement (National Oceanic and Atmospheric Administration (NOAA) 2012). These mega-fires and extreme wildfire seasons have resulted in billions of dollars expended annually on suppressing wildland fires (Calkin *et al.* 2005; National Interagency Fire Center (NIFC) 2013). Perhaps even more concerning is that recent modelling efforts suggest extreme wildfire seasons and the associated incidence of mega-fires will probably become more common in the future owing to climate change (Fulé 2008). Yue *et al.* (2013) estimated that by mid-century, the wildfire season will increase by more than 3 weeks and area burned will likely double for much of the western United States. Warmer and earlier springs have already increased wildfire activity in the western United States (Westerling *et al.* 2006). Elevated atmospheric CO_2 levels may further exacerbate the issue. Increases in CO_2 can increase plant production and decrease plant litter biodegradation, further increasing fuel loads between fires and the potential for frequent large fires (Ziska *et al.* 2005).

Besides being extremely costly, unprecedented large and frequent wildfire can negatively impact wildlife habitat. Large-scale and frequent wildfires have been identified as one of the primary threats to sage-grouse (*Centrocercus urophasianus*) (USFWS 2013), a species of conservation concern in the western United States and Canada that has experienced range-wide population declines (Connelly *et al.* 2000; Knick *et al.* 2003). For instance, in 2012, wildfires burned over 10% of the priority sage-grouse habitat in Oregon (Murphy *et al.* 2013). These areas may not be productive sage-grouse habitat for decades to even a century. When fires burn landscapes completely, sage-grouse and other sagebrush obligate wildlife will not likely occupy burned areas until sagebrush *Artemisia* L. recovers (Connelly *et al.* 2000). Recovery of sage-grouse habitat after fire is estimated to take 35 to 120 years (Baker 2006). With widespread loss of sagebrush habitat and severe decline in sagebrush obligate wildlife species (Connelly *et al.* 2000; Crawford *et al.* 2004; Davies *et al.* 2011), there is a critical need to reduce the risk, size and intensity of wildfires in sagebrush-dominated communities. Similarly, wildfire activity needs to be reduced in some other ecosystems to decrease the cost of fire suppression and protect ecosystem services.

The extensive nature of rangelands may limit the use of costly intensive fuel-management treatments (Davies *et al.* 2015). Therefore, management of fuel in rangelands is generally limited to treatments that can be applied broadly such as grazing, or techniques that can be strategically applied at smaller scales such as fuel breaks and green strips (Omi 1979; Pellant 1994; Diamond *et al.* 2009). Fuel breaks and green strips may create opportunities to spatially contain wildfire and provide staging areas for wildfire suppression depending on their placement in the landscape (Pellant 1994; Agee *et al.* 2000). However, these treatments do not influence fire behaviour across the landscape or decrease the risk of an ignition source propagating a wildfire. Strategic livestock grazing is probably the most, if not only, logistically feasible fuel treatment that can be applied at the scales required to potentially influence fire behaviour and intensity across vast rangeland landscapes.

Livestock grazing to manage fuels must be done carefully because improper grazing, especially in ecosystems that have not recently evolved with high grazing pressure, can damage native plants and, in the sagebrush ecosystem, promote exotic annuals (Daubenmire 1940; Mack and Thompson 1982; Reisner *et al.* 2013). In the sagebrush ecosystem, grazing in the winter, when herbaceous vegetation is not actively growing, is less likely to damage native perennial grasses and forbs than grazing during the growing season when defoliation reduces photosynthetic tissue and can place grazed plants at a competitive disadvantage with ungrazed plants (Caldwell *et al.* 1987; Briske and Richards 1995; Holechek *et al.* 1998). Winter grazing may also alleviate livestock distribution problems caused by limited water sources. Livestock's water demands are lower in winter (Winchester and Morris 1956) and ephemeral water sources are often available during the winter in Intermountain West and Great Basin sagebrush rangelands because most precipitation occurs in winter and evapotranspiration is low. Therefore, winter grazing may be a landscape-scale fuel treatment to alter fire behaviour and intensity.

Livestock grazing in the sagebrush ecosystem has been demonstrated to decrease fine fuel loads, continuity and height (Davies *et al.* 2010, 2015; Bates and Davies 2014). Herbivory has also been demonstrated to affect fuel load and fires in other systems (Van Langevelde *et al.* 2003; Waldram *et al.* 2008; Leonard *et al.* 2010). Winter grazing, compared with ungrazed areas, can also increase fine fuel moisture during the wildfire season (Davies *et al.* 2015). The effects of winter grazing on fine fuel may influence fire behaviour and intensity. Fine fuel loads, height, continuity and moisture content influence fire behaviour and severity (Rothermel 1972; Bradstock and Gill 1993; Blackmore and Vitousek 2000; Thonicke *et al.* 2001). However, shrubs in shrub-grasslands may drive fire behaviour and severity during a fire (Strand *et al.* 2014). Therefore, to evaluate the potential for winter grazing to serve as a fuel treatment in shrub-grassland ecosystems, it is critical to investigate the effects of winter grazing on fire behaviour, intensity and area burned.

The purpose of the present study was to evaluate the effects of winter grazing by cattle (*Bos taurus*) on fire behaviour and intensity in shrub-grasslands. We evaluated winter-grazing effects on fire behaviour and intensity in Wyoming big sagebrush (*Artemisia tridentata* Nutt. subsp. *wyomingensis*) Beetle & Young–bunchgrass communities because there is a current management emphasis on limiting fires in these plant communities to protect sage-grouse habitat and to limit exotic annual grass invasion (Connelly *et al.* 2000; Chambers *et al.* 2007; Davies *et al.* 2011; USFWS 2013). We hypothesised that flame height and depth, rate of spread and area burned would be lower in winter-grazed areas compared with ungrazed areas. We also hypothesised that burn temperature and heat loading at the meristematic crown region of bunchgrasses would be lower in winter-grazed compared with ungrazed areas. However, we did not expect burn temperature and heat loading under shrubs to vary between treatments because shrub cover was not influenced by winter grazing (Davies *et al.* 2015) and shrubs significantly influence fire characteristics (Strand *et al.* 2014).

Materials and methods

Study area

The study was conducted near the Diamond Craters in southeastern Oregon, USA (43°04′N, 118°40′W). Study sites receive on average 250 to 280 mm of precipitation annually (Natural Resource Conservation Service (NRCS) 2013). Precipitation mainly occurs in the winter and spring, and summers are typically dry and hot. The wildfire season generally spans from early summer to mid- to late-September (Davies and Nafus 2013; NIFC 2013). Topography was generally flat and elevation was ~1450 m. Study sites were Sandy Loam 10–12 PZ (R023XY213OR) and Droughty Loam 11–13 PZ (R023XY316OR) Ecological Sites (NRCS 2013). Plant communities were shrub-grasslands, with *Artemisia tridentata* subsp. *wyomingensis* being the dominant shrub at all study sites. Dominant perennial bunchgrass was *Achnatherum thurberianum* [Piper] Barkworth or *A. thurberianum* co-dominant with *Pseudoroegneria spicata* [Pursh] A. Löve depending on study site. Other common bunchgrasses on the study sites included *Elymus elymoides* [Raf.] Swezey, *Achnatherum hymenoides* [Roem. and Schult.] Barkworth and *Poa secunda* J. Presl. *Bromus tectroum* L., a naturalised exotic annual grass, was found in low abundance across the study area. Historical fire returns intervals are estimated to be 50–100+ years for these sagebrush communities (Wright and Bailey 1982; Mensing *et al.* 2006). The wildfire season in this region was active in 2014, with the Buzzard mega-fire burning 160 000 ha within 25 km of the study area.

Experimental design

We evaluated the effect of winter grazing on fire characteristics using a randomised complete block design with five blocks (sites). Treatments were winter-grazed or ungrazed 50 × 50-m plots that were burned in September 2014. The five sites differed in site and vegetation characteristics, but treatment plots within sites did not differ when the study was initiated in 2009. Winter-grazed and ungrazed treatments were randomly assigned to two 50 × 50-m plots that were separated by 10 m at each site. The ungrazed treatment was applied by erecting exclosures inside large (~800 to 1000 ha) pastures. Winter grazing was applied at the operation (pasture) level using cattle to ensure that results were representative of management scenarios. Cattle were rotated through the pastures between November and early April

Table 1. Weather conditions during prescribed burns for both winter-grazed and ungrazed treatments

Site	Air temperature (°C)	Relative humidity (%)	Wind speed (km h^{-1})
Block 1	26.1	20	2.4–8.8
Block 2	26.1	19	7.4–20.4
Block 3	22.8	25	11.3–19.3
Block 4	23.8	35	2.7–12.9
Block 5	27.2	17	6.4–13.0

to apply the winter grazing treatment before current year's growth of bunchgrasses (i.e. cattle were consuming forage from previous growing seasons). Rotation among pastures varied across years as this was an operational-level grazing treatment. Forage consumption by cattle was between 40 and 60% based on biomass. Grazing treatments were applied for 5 consecutive years before burning. Prescribed burns were applied on 23 September 2014 during the wildfire season. The Bone Creek and Blitzen wildfires, 40 to 50 km south of the study area, were still burning at this time and required active suppression. The week prior, the Glass Butte wildfire, ~90 km to the north and west of the study area, ignited and required suppression efforts to keep it from spreading. Prescribed burns were applied using drip torches to ignite a head fire on both treatments within a block simultaneously. During the prescribed burns, relative humidity varied between 17 and 35%, air temperature ranged from 22.8 to 27.2°C, and wind speed varied from 2.4 to 20.4 km h^{-1} (Table 1).

Measurements

Average sagebrush height in plots was estimated by measuring 50 randomly selected plants in each plot. Sagebrush above-ground biomass was estimated using allometric relationships between sagebrush volume and biomass. Within-plot sagebrush canopy volume was estimated by measuring height and two perpendicular canopy widths of 50 randomly selected sagebrush plants in each plot. The longest canopy width was measured and the second canopy width was measured at the centre of the first canopy width. The allometric relationship between volume and biomass was determined using the same height and sagebrush canopy measurements on 292 sagebrush plants outside the treatment plots. The 292 sagebrush plants were harvested, dried and then weighed to determine biomass. Simple linear regression was then used to correlate canopy volume (cm^3) with sagebrush biomass (g):

$$\text{Sagebrush biomass} = 176.9 + 0.0151(\text{sagebrush volume})$$

$$P < 0.001, \ R^2 = 0.754$$

The preceding equation was used to estimate the mean biomass of a sagebrush plant in each plot and then average biomass was multiplied by sagebrush density to determine sagebrush biomass per hectare. Sagebrush density was determined by counting all sagebrush plants rooted in four 2 × 45-m belt transects in each plot. Shrub cover was determined using the line intercept method (Canfield 1941) along four 45-m transects in each plot. Current and previous years' grass height was measured on 50 randomly selected bunchgrasses in each plot. Litter biomass and standing crop biomass were determined by clipping 15 randomly placed 1-m^2 quadrats in each plot. Harvested biomass was oven-dried and then weighed. Fine-fuel moisture content was calculated as a percentage of dry weight and determined by clipping fine fuels in five 0.2-m^2 quadrats in each plot on 23 September 2014. Harvested biomass was weighed, oven-dried and weighed again to determine moisture content.

Temperatures during the fires were measured at 1-s intervals using type-K high temperature Inconel overbraided ceramic fibre insulated thermocouples (Omega Engineering, Stamford, CT) attached to a Campbell Scientific 21X micrologger with an SM4M storage module (Campbell Scientific, Logan, UT). Fifteen thermocouples were placed in each plot with approximately half in the interspace between shrubs and the other half placed underneath the canopy of shrubs. Thermocouples were placed inside the centre of the meristematic crown region of randomly selected bunchgrasses immediately before burning. Thermocouples were left in place for 20 min following maximum temperature. Thermocouple data were used to measure maximum temperature and calculate heat load (degree seconds at ≥60°C), which is the cumulative result of temperature and duration of elevated temperature at and above 60°C. Temperatures ≥60°C were used because for heat to be detrimental to plants, tissues must reach ~60°C (Yarwood 1961; Wright and Bailey 1982; Hartford and Frandsen 1992). Heat load was determined at 5-min intervals from 5 to 20 min. Area burned was measured immediately post fire by recording the perimeter of burned and unburned areas within each plot with a Trimble Geo XT Global Positioning System (GPS) unit (Trimble, Sunnyvale, CA). Unburned areas were measured if they were 50 cm in diameter or larger. Area of unburned and burned polygons was then determined using *ArcMap ver. 10.2* (ESRI, Inc., Redlands, CA). Rate of spread was determined by placing two sets of metal stakes, one set at a 2-m spacing and the other set at a 4-m spacing, parallel with the prevailing wind. The length of time between when the flame base crossed the first stake to when it crossed the second stake was recorded with a stopwatch. The rate of spread was averaged for both sets of stakes. The intervals between the metal stakes were used to estimate flame depth (depth of active flaming front). Maximum flame height was estimated to the nearest 0.25 m using 6-m vertical poles segmented into 1-m intervals with alternating white and red heat-resistant paint.

Statistical analysis

Repeated-measure analysis of variance using the mixed models procedure (*Proc Mixed SAS ver. 9.1*, SAS Institute, Cary, NC) was used to determine the effects of winter grazing on variables repeatedly measured with time as the repeated variable. Random variables were site and site by treatment interactions. Akaike's Information Criterion (Littell *et al.* 1996) was used to select the appropriate covariance structure for repeated-measures analysis of variance. Analysis of variance using the mixed models procedure was used to determine the effects of winter grazing on variables that were not repeatedly measured. Data that did not meet assumptions of analysis of variance were either log- or square-root-transformed. All figures and text present original, non-transformed data. Means are reported with standard errors

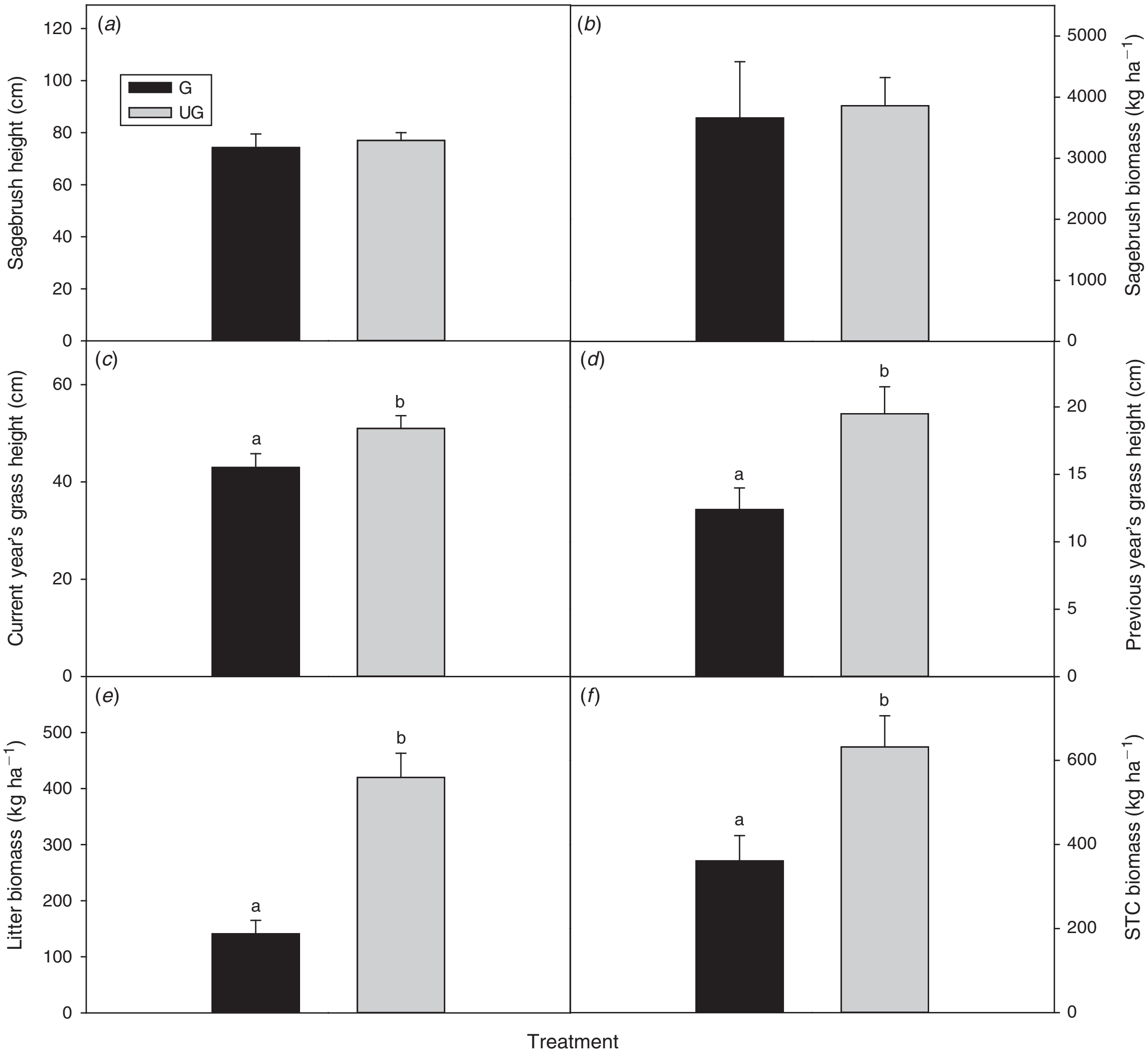

Fig. 1. Fuel characteristics (mean + s.e.) in winter-grazed (G) and ungrazed (UG) treatments before prescribed burning. STC = Standing crop. (*a*) Sagebrush height; (*b*) sagebrush biomass; (*c*) current year's grass height; (*d*) previous years' grass height; (*e*) litter biomass; and (*f*) standing crop biomass. Lower-case letters represent significant differences between treatments ($P \leq 0.05$).

in figures and text. Treatment means were considered different at $P \leq 0.05$.

Results

Fuels

Sagebrush height and aboveground biomass did not differ between the winter-grazed and ungrazed treatments (Fig. 1*a* and *b*; $P = 0.607$ and 0.823 respectively). Average sagebrush height was ~75 cm in both treatments. Sagebrush biomass averaged 3660 ± 920 and 3860 ± 460 kg · ha^{-1} in the winter-grazed and ungrazed treatment respectively. Average shrub cover was 11% in both treatments (11 ± 0.6 and 11 ± 1.5% in grazed and ungrazed areas) and did not differ between treatments ($P = 0.928$). In contrast, fine (herbaceous) fuel characteristics differed between winter-grazed and ungrazed treatments (Fig. 1). Current year's grass height was ~8 cm taller in the ungrazed than winter-grazed treatment (Fig. 1*c*; $P = 0.012$). Previous years' grass height was 1.6 times taller in the ungrazed compared with winter-grazed treatment (Fig. 1*d*; $P = 0.005$). Litter biomass was three times greater in the ungrazed compared with the winter-grazed treatment (Fig. 1*e*; $P = 0.002$). Standing-crop biomass in the ungrazed treatment was nearly twice as great as in the winter-grazed treatment (Fig. 1*f*; $P = 0.029$). Fine fuel moisture was 1.4 times greater in the winter-grazed compared with the ungrazed treatment ($P = 0.020$). Fine fuel moisture was 19.7 ± 2.4 and 14.2 ± 1.7% in the winter-grazed and ungrazed treatment respectively.

Fire characteristics

Winter grazing reduced burn temperatures compared with ungrazed areas (Figs 2 and 3). Maximum temperature at the

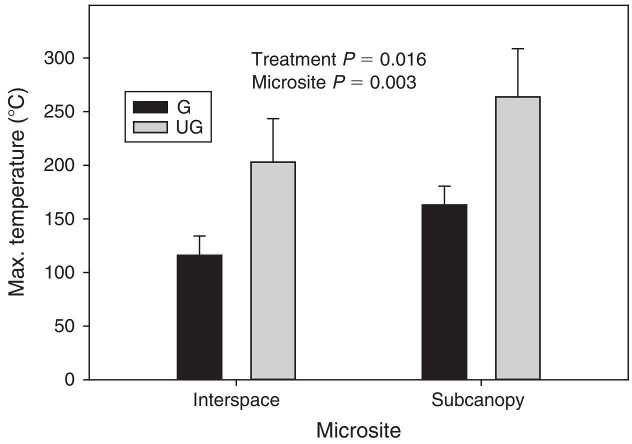

Fig. 2. Average maximum (mean + s.e.) temperature during prescribed burns in the centre of the meristematic crown region of bunchgrasses in winter-grazed (G) and ungrazed (UG) treatments in interspace (between shrubs) and subcanopy (under shrub) microsites.

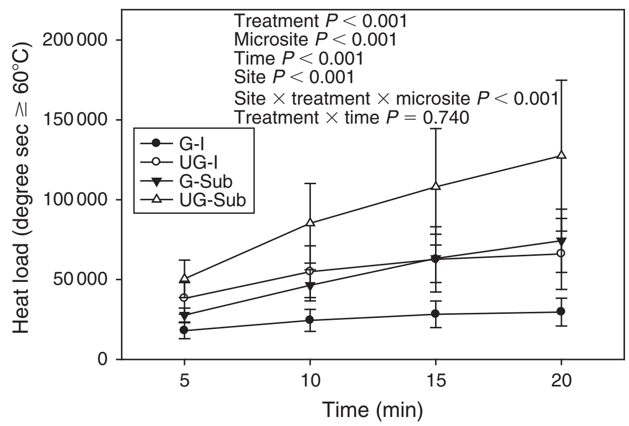

Fig. 3. Heat load (mean ± s.e.) during prescribed burns in the centre of the meristematic crown region of bunchgrasses in winter-grazed and ungrazed treatments in interspace (between shrubs) and subcanopy (under shrub) microsites. G-I = winter-grazed interspace; UG-I = ungrazed interspace; G-Sub = winter-grazed subcanopy; and UG-Sub = ungrazed subcanopy. Data were summarised across sites.

meristematic tissue of perennial bunchgrasses was greater in the ungrazed compared with the winter-grazed treatment (Fig. 2; $P = 0.016$). Ungrazed interspace (areas between shrub canopies) reached maximum temperatures that were 87°C hotter than winter-grazed interspaces. Maximum temperatures during the burns were 101°C hotter in ungrazed subcanopy microsites compared with winter-grazed subcanopy microsites. Maximum temperature was also greater in subcanopy than interspace microsites ($P = 0.003$). Heat load was influenced by the three-way interaction of treatment–site–microsite (Fig. 3; $P < 0.001$). General patterns were similar, with all sites having greater heat load in ungrazed areas and subcanopy microsites, but the magnitude of differences varied among sites. Heat load at the meristematic tissue of bunchgrasses was 1.7 to 1.8 times greater in the subcanopy microsite and 2.0 to 2.2 times greater in the interspace microsite in the ungrazed compared with the winter-grazed treatment (Fig. 3; $P < 0.001$). Heat load was also greater in the subcanopy than the interspace microsite ($P < 0.001$). Heat load increased with time ($P < 0.001$), but the time–treatment interaction was not significant ($P = 0.740$).

Area burned during fires was 2.4 times greater in the ungrazed compared with the winter-grazed treatment (Fig. 4*a*; $P = 0.033$). Rate of fire spread was 3.2 times faster in the ungrazed than winter-grazed treatment (Fig. 4*b*; $P = 0.005$). Flame depth was nearly four times greater in the ungrazed treatment compared with the winter-grazed treatment (Fig. 4*c*; $P = 0.022$). Flame height was 3.1 times greater in the ungrazed compared with the winter-grazed treatment (Fig. 4*d*; $P = 0.015$). Maximum flame height was 4.3 and 2.3 m in the ungrazed and winter-grazed treatment respectively.

Discussion

Winter-grazing effects on fuel and fire characteristics demonstrate that it has the potential to be used as a fuel treatment to reduce the size of wildfires, increase the likelihood of effective suppression, and decrease fire intensity in some *Artemisia tridentata* subsp. *wyomingensis* steppe communities and likely other shrub-grasslands. Prior research (Davies *et al.* 2015) suggested that fire characteristics may differ between winter-grazed and ungrazed areas based on herbaceous fuel differences. However, shrubs, which were not influenced by winter grazing, may ultimately determine fire characteristics (Strand *et al.* 2014). By burning during the wildfire season, we were able to determine that winter grazing can alter wildfire characteristics, even though shrub characteristics were not affected by winter grazing, in shrub-grasslands. We suspect that increased fire behaviour in ungrazed plots was associated, at least in part, with increased herbaceous fuel loads promoting greater engagement (i.e. combustion) of energy-dense woody fuels.

Winter grazing resulted in lower herbaceous fuel amounts and heights and increased fuel moisture compared with ungrazed areas, which likely cumulatively reduced fire behaviour, area burned and fire intensity. Other studies have similarly reported that grazing reduced fine-fuel amounts and heights (e.g. Blackmore and Vitousek 2000; Briggs *et al.* 2002; Davies *et al.* 2010; Bates and Davies 2014). Less and shorter herbaceous fuels in winter-grazed areas likely contributed to the observed slower rate of fire spread and decreased flame depth and height in the present study (Bradstock and Gill 1993; Blackmore and Vitousek 2000). Prior research (Davies *et al.* 2015) also established that fine-fuel moisture is greater in winter-grazed areas compared with ungrazed areas from late June through August. Our current study demonstrated that winter-grazed areas compared with ungrazed areas have greater fuel moisture content in September. The greater moisture content of fine fuel in winter-grazed areas likely also contributed to more moderate fire behaviour. As fuel moisture increases, more energy is required to cause ignition, which reduces the rate of spread (Rothermel 1972; Thonicke *et al.* 2001; Chuvieco *et al.* 2004). Greater fuel moisture also decreases the likelihood of successful ignition and reduces the potential area burned (Chuvieco *et al.* 2009).

Winter grazing decreased the area burned from the initial ignition by more than 50%. This suggests large wildfires would be less likely and that unburned patches may be more common

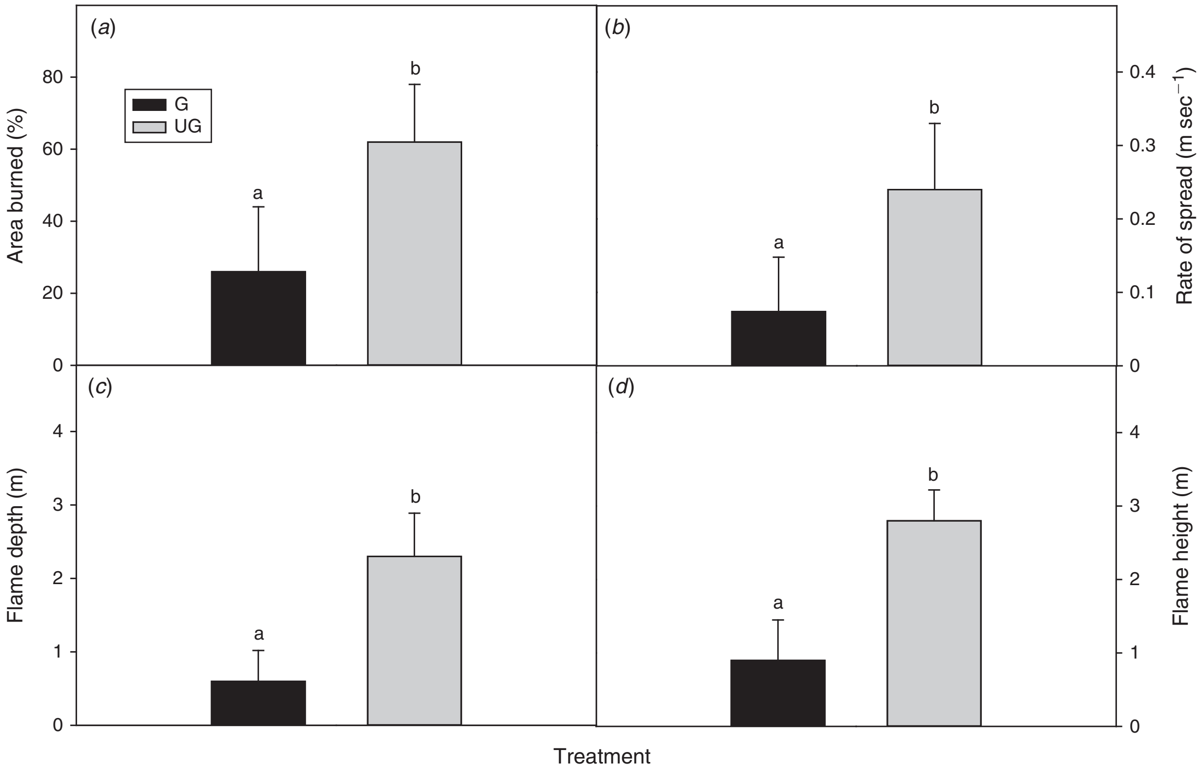

Fig. 4. Area burned and fire behaviour characteristics (mean + s.e.) during prescribed burning of winter-grazed (G) and ungrazed (UG) treatments. (*a*) Area burned; (*b*) rate of spread; (*c*) flame depth; and (*d*) flame height. Lower-case letters represent significant differences between treatments ($P \leq 0.05$).

and larger in winter-grazed compared with ungrazed areas. Wildfire suppression efforts may also be more effective in winter-grazed areas, further decreasing the probability of large fires and increasing the likelihood of unburned patches within burns. Suppression efforts are more effective and less dangerous with decreased rate of spread and reduced flame height (Fried *et al.* 2004; Moghaddas and Craggs 2007). Unburned patches in burns are important because they serve as a refuge for fire-sensitive species (Henriques *et al.* 2000; Watson *et al.* 2012), buffer the effects of wildfire on fauna (Robinson *et al.* 2013) and increase post-burn landscape heterogeneity. Smaller wildfires and increased suppression efficiency through winter grazing may reduce the cost of wildfire suppression in some areas. Extreme fire weather conditions, particularly with high amounts of woody fuels, may dictate fire behaviour (Gedalof *et al.* 2005) and override the effects of grazing on fine fuels (Strand *et al.* 2014); however, many of these fires either ignite or burn through a period with less severe weather conditions where winter-grazing effects may assist suppression efforts or reduce spread and behaviour in unsuppressed fires.

Reducing the area burned and providing unburned patches within burns in *A. tridentata* subsp. *wyomingensis* communities with winter grazing would also reduce the risk to sage-grouse and other sagebrush-associated wildlife. Fire can result in the loss of sagebrush habitat for several decades to over a century in *A. tridentata* subsp. *wyomingensis* communities (Baker 2006) and can also promote exotic annual grass invasion (Stewart and Hull 1949; Chambers *et al.* 2007; Davies *et al.* 2009). Exotic annual grass invasion can result in frequent fires (Balch *et al.* 2013) because of increased dry fine fuels during the wildfire season (Davies and Nafus 2013). Conversion to annual grass dominance likely represents a permanent loss of habitat for sagebrush-associated wildlife and is associated with decreased biodiversity (Knick *et al.* 2003; Davies and Svejcar 2008; Davies 2011). Clearly, a strategic landscape vision needs to be developed to apply winter grazing to reduce the risk of wildfire in sagebrush habitat and simultaneously maintain a wide diversity of habitats including habitats with enough residual vegetation for nesting sage-grouse. Furthermore, winter grazing treatments need to be carefully applied because overuse may negatively impact native vegetation and plant community resilience (Crawford *et al.* 2004; Davies *et al.* 2011). However, our study demonstrates that winter grazing is a potential tool that land-managers can utilise to help manage wildfire in shrub-grasslands and may be valuable to conserving habitat for species such as sage-grouse.

Prior work (Strand *et al.* 2014) has identified the importance of shrubs in dictating fire behaviour in sagebrush communities; however, our research suggests that herbaceous fuels may play an important intermediary role in determining the effects of shrubs on fire characteristics. The effects of winter grazing on herbaceous fuel altered the burn characteristics of shrubs. This can be ascertained by looking at maximum temperatures and heat loading within microsites (Figs 2 and 3). Clearly, the subcanopy microsite was influenced by the presence of woody fuels (the shrub), but interestingly, effects were vastly different depending on whether or not the area was winter-grazed or ungrazed. Winter grazing altered herbaceous fuel characteristics, which reduced the intensity of the fire as it burned the shrub. Shrubs in ungrazed areas were more engaged by the fire; thus they burned more completely and released more energy (greater maximum temperature and heat loading). This suggests that winter grazing may influence fire characteristics in shrub-grasslands with even higher levels of woody fuels than those evaluated in the current study, though grazing effects probably become less apparent as shrub fuels increase or there may be a threshold at which the influence of herbaceous fuels on fire characteristics is severely limited owing to the abundance of shrub fuels.

The higher maximum temperatures and greater heat loading during the fires (i.e. more intense fires) in ungrazed compared with winter-grazed areas suggest that ungrazed areas may experience greater fire-induced mortality of native perennial bunchgrasses. Higher temperatures and longer periods of elevated temperatures during a fire increase the likelihood of perennial grass mortality (Wright and Klemmedson 1965; Wright 1970; Odion and Davis 2000; Pelaez *et al.* 2001). Similarly, grazing in savannas reduces fire intensity, and thereby decreases damage to trees (Van Langevelde *et al.* 2003). Davies *et al.* (2009) found that native perennial bunchgrass densities were lower after burning in long-term (50+ years) ungrazed areas compared with areas grazed at moderate levels during the growing season. Though Davies *et al.* (2009) did not measure fire temperature or heat loading during burns, their results support our prediction that grazing may reduce fire-induced mortality of native perennial bunchgrasses. An elevated mortality of perennial bunchgrasses is problematic because they are the dominant herbaceous plant group in this ecosystem (Davies *et al.* 2006; Davies and Bates 2010) and are the plant group most critical to limiting exotic annual grass invasion (Chambers *et al.* 2007; Davies 2008; James *et al.* 2008). As would be expected, Davies *et al.* (2009) found that decreases in perennial bunchgrasses with burning in areas that experience long-term grazing exclusion resulted in exotic annual grass invasion, whereas grazed areas, which had lower fire-induced mortality of perennial bunchgrasses, had limited post-fire exotic annual grass presence. Therefore, winter grazing likely decreases the probability of post-fire exotic annual grass invasion and subsequent development of an exotic annual grass–fire cycle and increases the probability of post-fire recovery of the native plant community.

As mega-fires and extreme wildfire seasons become more common (Westerling *et al.* 2006; Fulé 2008; NOAA 2012; Yue *et al.* 2013) and wildfires become more severe (Fried *et al.* 2004) with climate change, it is vital to have treatments to apply before the need for wildfire suppression and post-fire restoration. Our research demonstrates that winter grazing is a pre-emptive treatment that can alter fire behaviour, area burned and fire intensity in at least some wildfires in *A. tridentata* subsp. *wyomingensis* communities and likely other shrub-grasslands. Studies in Africa (Van Langevelde *et al.* 2003; Waldram *et al.* 2008), Australia (Leonard *et al.* 2010) and the plains of North America (Fuhlendorf *et al.* 2009) have similarly concluded that herbivory influences fuels and fire behaviour. Our study also demonstrated that winter grazing can reduce fire intensity and this may increase resilience of native plant communities to fire. Obviously, the effects of winter grazing may be moderated in fires that occur under more extreme weather conditions and in plant communities with greater amounts of woody vegetation. Nonetheless, winter and other dormant-season grazing can be applied across vast rangeland landscapes where other fuel management treatments would be too expensive or impractical to apply. Further refinement and evaluation across a variety of plant community types and under varying fire weather conditions would be invaluable.

Acknowledgements

We greatly appreciate that Roaring Springs Ranch allowed us to conduct this research project on their property. Data collection by Urban Strachan, summer students and temporary employees was greatly appreciated. Mention of a proprietary product does not constitute a guarantee or warranty of the product by USDA or the authors and does not imply its approval to the exclusion of other products. USDA is an equal opportunity provider and employer.

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