The Effects of Prescribed Fire and Ungulate Herbivory 6 and 7 Years Postburn in the Upland Bitterbrush (*Purshia tridentata*) Communities of Rocky Mountain National Park, Colorado[1]

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Abstract

A controlled, manipulative study utilizing exclosures was initiated in 1994 by Zeigenfuss and others (2002) to assess the effects of prescribed fire and ungulate herbivory in the bitterbrush communities of Rocky Mountain National Park (RMNP). Four sites were chosen randomly from all available bitterbrush communities in RMNP wherein exclosures were erected (March/May 1995) and paired grazed plots established. Prescribed burns were performed on half of each exclosed and grazed area in 1995 and 1996. Using the sites and exclosures already present in the bitterbrush communities, additional data were collected from July 2001 to August 2002 to determine the communities' responses to prescribed fire and herbivory 6 and 7 yr post-burn. Bitterbrush canopy volume and estimated annual production remain lower 6 and 7 yr post-burn in burned treatments as compared to their corresponding unburned treatments. Total shrub canopy area, volume, and estimated annual production did not significantly differ due to burning, but differed due to grazing at least 1 yr of the study (p <0.10). Data support that the ambient level of herbivory present in RMNP is affecting post-burn successional patterns by impeding shrub regeneration. We caution against the use of any type of prescribed fire that would burn a substantial portion of the shrub component in these communities while ungulate herbivory remains high due to the alteration of post-fire successional patterns that could result in the loss of a major component of the community.

Introduction

The effects of prescribed fire on the plant communities within Rocky Mountain National Park (RMNP) are not well understood (NPS 1992). Because prescribed fire is utilized as a management tool within RMNP, it is important not only to understand the effects of prescribed fire on the plant species within the Park, but what impact the resulting changes in vegetation may have on the wildlife within the Park. The upland bitterbrush communities within Rocky Mountain National Park comprise 3.3 km^2 of the winter range for ungulates on the Park's east side and are characterized by antelope bitterbrush (*Purshia tridentata*), ponderosa pine (*Pinus ponderosa*), rabbitbrush (*Chrysothamnus viscidiflorus*), mountain muhly (*Muhlenbergia montana*), and needle-and-thread grass (*Hesperostipa comata*). Despite their small area, bitterbrush communities are important habitat for many species of wildlife found in the Park, such as mule deer (*Odocoileus hemionus*) and the green-tailed towhee (*Pipilo chlorurus*). Thus, proper management of bitterbrush communities is

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essential not only to conserve community biodiversity within RMNP, but also for conservation of wildlife populations that depend on the communities.

Historically, bitterbrush communities were characterized by patchy, low-intensity surface fires (NPS 1992, Pyne and others 1996). There is much variation in the literature regarding the fire frequency for bitterbrush communities, but the RMNP Fire Management Plan cited a fire frequency of 30 yr (NPS 1992). According to the 1992 Fire Management Plan for RMNP, bitterbrush communities fall into the fire suppression zone where prescribed fire is utilized. The type of prescribed fire used in these communities involves a majority of the shrub canopy during burning and does not mimic the type of fire thought to occur historically.

Due to the fact that bitterbrush and other plant species in these communities may have limited tolerance to fire, RMNP managers need to assess the use of widespread prescribed burns as a valid management tool in the upland bitterbrush communities. As stated previously, bitterbrush communities are utilized by many animal species within the Park. For example, bitterbrush is a main source of winter forage for mule deer in RMNP, accounting for approximately 24 percent of mule deer winter diets, with rabbitbrush accounting for 13 percent and wax currant (*Ribes cereum*) 7 percent (Stevens 1980). Therefore, Park managers must take into account all effects prescribed fire may have on the productivity and physiognomy of the plant community and in turn what these effects will have on animals in the Park.

Confounding the use of prescribed fire in RMNP is an increase in Park elk numbers. The elk (*Cervus elaphus*) population wintering in RMNP is estimated at 1,069 individuals, with an additional 1,975 individuals outside RMNP in the Estes Valley that utilize park winter ranges during spring and fall migrations (Lubow and others 2002). The elk population in RMNP is estimated to have increased nearly three-fold since being released from artificial controls within the Park in 1968, and heavy herbivory and plant damage to winter range vegetation has been documented (Olmsted 1997, Zeigenfuss and others 1999). However, the mule deer population within the Park has decreased from a count of 1,021 mule deer in 1937 to 1938 to approximately 200 individuals in RMNP and 300 individuals in the surrounding Estes Valley (Conner and others 2002, Stevens 1980). Herbivory at moderate to heavy levels (>50 percent herbaceous offtake), as occurs on the winter range of RMNP, could alter successional patterns after prescribed burning by decreasing shrub regeneration, resulting in an additive effect between prescribed fire and herbivory.

Plant communities respond variably to the effects of fire and herbivory over time. Long-term data are needed to ascertain the effects of fire and herbivory on a specific community and then make effective management decisions for the entire community. If long-term data can be collected on the effects of fire and herbivory in upland bitterbrush sites, stronger inference of the effects on plant species productivity and community structure will be possible. This study collected 2 yr of post-treatment data that assessed the condition of the vegetation 6 and 7 yr post-burn and provides Park managers with a more reliable representation of the impacts of prescribed fire and herbivory on the upland bitterbrush communities.

Materials and Methods

To assess the impacts of prescribed fire and herbivory on Rocky Mountain National Park's upland bitterbrush communities a controlled, manipulative study

began in 1994. Four sites were chosen randomly from all available bitterbrush communities in the Park using GIS (Zeigenfuss and others 2002). The site locations lie in the areas of Aspenglen Campground, Deer Ridge, Hollowell Park, and Beaver Meadows Entrance Station (*table 1*). In March-May of 1995 researchers set up 30.5 × 45.7 m exclosures on the four sites with paired plots that were allowed to be grazed (Zeigenfuss and others 2002). A prescribed burn was done on half of the exclosed and paired plots at each site in late fall 1995 or early April 1996, resulting in the treatment combinations of grazed-burned, grazed-unburned, exclosed-burned, and exclosed unburned.

The previous researchers collected 1 to 2 yr pre-burn data and 2 years post-burn vegetative data. We collected an additional 2 yr post-burn data, 6 and 7 yr post-burn in the summers of 2001 and 2002 on the same parameters as the previous researchers. One year of winter offtake data was collected in the spring of 2002. Our methodology follows that of the previous researchers (Zeigenfuss and others 2002).

Table 1—*Site descriptions for the locations of the four study sites in RMNP, Colorado.*

	Aspenglen Campground	Hollowell Park	Beaver Meadows Entrance	Deer Ridge
Elevation (m)	2558	2625	2544	2648
Ave. slope[1] (°)	10	22	8	14
Aspect (° from N)	131	162	170	166
Soil type[2]	Rofork-Chasmfalls complex	Isolation gravelly sandy loam	Rofork-Chasmfalls complex	Rofork-Chasmfalls complex

[1]Slope taken at the middle of each treatment then averaged.
[2] Unpublished data on file at Rocky Mountain National Park, Estes Park, Colorado.

Measurement of Shrub Parameters

Shrub data were collected from three randomly placed 9.3 m^2 circular plots in each treatment. Data on species-specific estimated annual shrub production (kg ha^{-1}), canopy area (m^2 ha^{-1}), and canopy volume (m^3 ha^{-1}) were collected in July/August of 2001 and 2002 for all treatments at each site. Estimated annual shrub production data were collected and calculated following Peek (1970). Log-log regression equations were developed for each shrub species each year using site, treatment, and canopy volume as predictors of production. Adjusted R^2 values ranged from 0.41 to 0.97 in 2001 and from 0.54 to 0.87 in 2002. In addition to shrub plot data, three bitterbrush plants per treatment per site were tagged from the previous research. These tagged plants had all data collected all years.

Measurements of shrub offtake by ungulates from the previous winter were collected in May 2002 from the shrub plots in the grazed treatments only. Percent leader use was calculated by dividing the number of browsed leaders by the total number of leaders and multiplying by 100. Percent twig winter utilization was calculated following Jensen and Urness (1981). Total consumption was estimated by multiplying percent leader use by percent twig utilization.

Measurement of Herbaceous Parameters

Data for annual herbaceous biomass were collected in July/August 2001 and 2002 for all treatments at each site. Three randomly located 0.25 m^2 circular plots per

exclosed treatment were used for sampling. To ascertain annual herbaceous biomass on grazed treatments, three 1 m^2 moveable grazing cages were randomly placed in each grazed treatments in April, and samples were collected using the same 0.25 m^2 circular plots from under the grazing cages in July/August. All herbaceous biomass within the circular plots was clipped 5 cm above ground level and sorted by species. The litter, graminoids, and forbs were dried at 55°C for 48 hr and weighed. Percent cover data were collected from the same 0.25 m^2 circular plots used to measure annual herbaceous biomass.

Herbaceous overwinter offtake by ungulates was measured in late April 2002 in grazed treatments. Grazing cages were randomly relocated in August after annual herbaceous biomass collection. The procedure follows that of collection for annual herbaceous biomass in grazed treatments with 3 additional measurements per treatment taken outside the grazing cages. Clippings were sorted by vegetation type (forb, graminoid, litter), and percent differences in dry weight between grazed and caged locations were determined to measure winter consumption by ungulates.

Statistical Analyses

The data were analyzed for significant differences between treatments in SAS Statistical Software Package, version 8 (alpha=0.10). Voucher specimens were collected and archived at the Rocky Mountain National Park Working Herbarium (ROMO). All plant identifications were made using Weber and Wittmann (2001).

Results and Discussion

Overwinter herbaceous offtake for 2001 to 2002 was 64 percent, with 21 percent total shrub consumption and 49 percent leader use. A significant negative effect due to grazing on total estimated annual shrub production was detected in 2002 (p=0.07), with no effects detected due to burning both years. From 2001 to 2002, both exclosed treatments increased estimated annual shrub production, while both grazed treatment decreased. Also, the grazed-burned treatment had the lowest estimated production value both years. Tagged plant production in unburned treatments was twice as great as their corresponding burned treatments in 2001 and four times as great in 2002 (*fig. 1*, p=0.004; p=0.01). These data suggest a long-term negative effect due to prescribed burning on individual bitterbrush plants. Tagged plant data reveal more about individual plant responses to treatments, because shrub plot data are confounded by dispersion of shrubs within and among sites. Precipitation in 2002 was five times less than 2001 from May to September, as measured at RMNP.

Grazing decreased total shrub canopy volume in 2001 and 2002 (p=0.06; p=0.04) with no effect detected due to burning. Moreover, the exclosed-burned treatment averaged 25 percent lower canopy volume values than the exclosed-unburned treatment in 2001 and 2002, while the grazed-burned treatment averaged 50 percent lower values than the grazed-unburned treatment both years (*fig. 2*). Grazing decreased total shrub canopy area in 2002 (p=0.04) with no effects in 2001 detected due to grazing or burning. However, the grazed-burned treatment had the lowest value for both parameters both years and burned treatments had lower values than their corresponding unburned treatments both years (*fig. 2*).

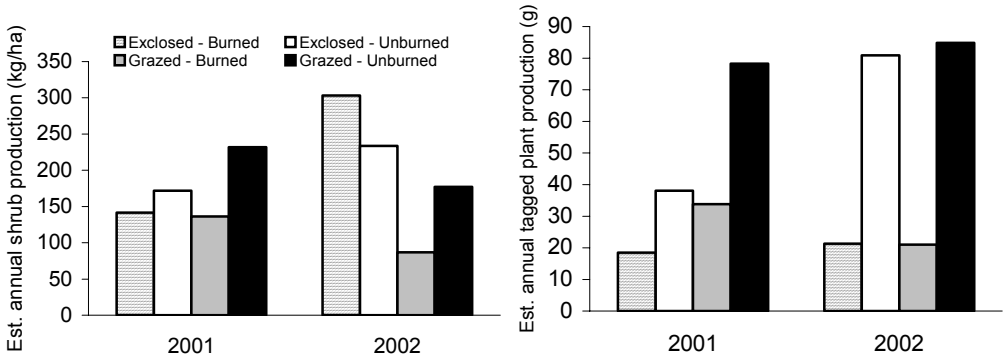

Figure 1—Estimated annual production for total shrub (kg/ha) and tagged plant (g) parameters 6 and 7 years post-burn.

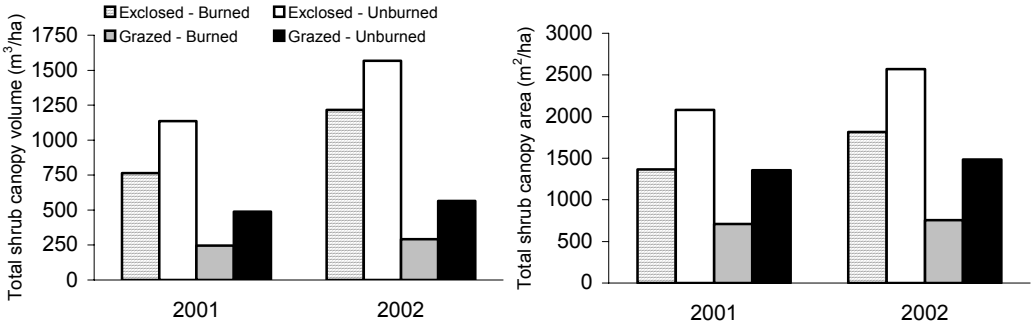

Figure 2—Total shrub canopy volume (m^3/ha) and canopy area (m^2/ha) 6 and 7 years post-burn.

Tagged plant canopy area remained decreased due to burning 6 and 7 yr post-burn (p=0.02; p=0.004) as did tagged plant canopy volume (p=0.01; p=0.03). Unburned treatments had two to four times as much tagged plant canopy area as their corresponding burned treatments and three to seven times the tagged plant canopy volume as their corresponding burned treatments. No effects on either parameter due to grazing were detected. Burning decreased shrub plot bitterbrush canopy volume in 2001 (p=0.08) and a significant effect due to grazing was detected in 2002 (p=0.04). Again, the grazed-burned treatment had the lowest treatment value both years. Burned treatments averaged 40 to 60 percent less canopy volume than unburned treatments over both years (*fig. 3*). Seven years post-burn, 54 to 61 percent of the bitterbrush plants sampled were resprouting in burned treatments. However, the resprouting plants in the grazed treatments were prostrate, with a highly pruned appearance and small canopy area and volume. This suggests that bitterbrush can resprout adequately after burning, but the ambient herbivory levels present in the Park are impeding regeneration. However, unburned treatments still have greater bitterbrush canopy volume values 6 and 7 yr post-burn than their corresponding burned treatments.

No significant effects due to burning were found for total annual herbaceous biomass or percent cover 6 and 7 yr post-burn (p>0.10). Annual herbaceous biomass ranged from approximately 30 to 50 g/m^2 in each of the treatments, with no trends present. In 2001, grazing decreased the percent cover of litter and moss and increased the percent cover of bare ground and basal graminoid cover (p<0.10).

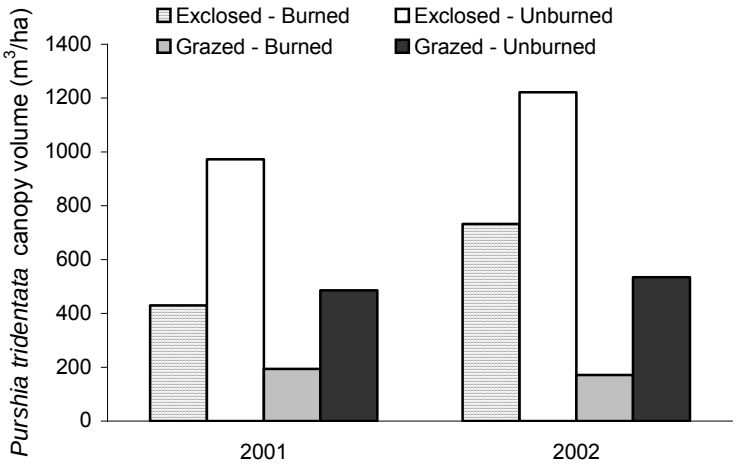

Figure 3—Shrub plot bitterbrush canopy volume (m^3/ha) 6 and 7 years post-burn.

Conclusions

The grazed-burned treatment had the lowest values for total shrub canopy area and volume, estimated annual shrub production, and bitterbrush canopy volume. This suggests that ambient herbivory levels are enough to affect post-fire successional patterns, specifically by inhibiting shrub regeneration. Due to the long-term negative effect of prescribed fire and ungulate herbivory on shrub canopy area and volume, we would not recommend the use of prescribed fire in these communities; the integrity of community structure and function can not be maintained under both influences. RMNP is considering using point-source burns in these communities that would better mimic the type of fire thought to occur historically. However, we caution against the use of any type of prescribed fire that would burn a substantial portion of the shrub component in these communities while ungulate levels are elevated. The additive effect between prescribed fire and ungulate herbivory causes the loss of essential components in the bitterbrush communities. This loss not only results in the degradation of habitat for mule deer and numerous other plant and animal species, but also a degradation of community biodiversity within the Park.

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