IMPACTS OF FIRE AND ELK HERBIVORY IN THE MONTANE ECOREGION OF JASPER NATIONAL PARK, ALBERTA, CANADA

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ABSTRACT

Complex interactions exist among ungulates, predators, humans, and vegetation in Jasper National Park, Alberta, Canada. Fire and herbivory are key parts of the interactions among these ecosystem components. Significant increases in human use, exclusion of fire, and thriving populations of elk (*Cervus elaphus*) within the montane region are impacting the ecological integrity of this vital area. Prescribed fire is being used to help restore ecosystem structure and natural processes with the goal of maintaining a landscape of open-canopy lodgepole pine (*Pinus contorta*) forest and grassland. As part of this management activity, the impact of prescribed burning, elk herbivory, and elk–fire interactions on the montane vegetation are being measured.

Permanent vegetation plots were established in 1998, some of which involved fenced exclosures to exclude elk. Plots were placed in unburned control areas and in an 80-ha area burned in the spring of 1999, representing both closed forest canopy and open forest–grassland sites. The prescribed fire was lit as lines using hand torches, and fire behavior was measured in these plots. Pre- and post-burn vegetation sampling included measurements of tree height, condition, and diameter at breast height; shrub height and crown diameter; and ground cover vegetation percent cover by species. About two-thirds of the pine trees were killed, and mortality estimates were in the range of model predictions. Almost all of the shrubs were burned, but Canada buffaloberry (*Shepherdia canadensis*) is regenerating. Little impact on ground cover vegetation cover and phytodiversity has been detected, but this may change as the tree canopy opens up and the solar radiation environment changes. Vegetation recovery is being monitored annually, although a few more years may be needed to detect the full impact of fire and elk interactions. The information is being used to evaluate the success of prescribed burning as an ecosystem management tool in this ecoregion.

keywords: Alberta, *Cervus elaphus*, elk, exclosures, fire effects, herbivory, Jasper National Park, montane, prescribed fire, tree mortality.

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INTRODUCTION

Fire has been a historically important process in the montane ecoregion of Jasper National Park, Alberta, Canada. Before 1913, fire return intervals for much of this area ranged from about 5 to 10 years (Tande 1979). Many of these fires were low intensity and resulted in a range of forest stand types and ages. With the inception of effective fire suppression and exclusion, forest stands >100 years old have increased from 21% to 78% of the montane area since 1930 (Andison 2000). In particular, lodgepole pine and Douglas-fir (*Pseudotsuga menziesii*) communities have expanded through the grassland areas with open grasslands decreasing by more than 50% (Rhemtulla 1999).

National park policies (Heritage Canada 1994) call for the restoration of natural processes and the application of fire management as a key means to achieve this. Consequently, prescribed fire has been used as an ecosystem management tool in the montane ecoregions of the park since the mid-1980s. The park management plan (Parks Canada 2000) sets a goal to restore the montane ecoregion to at least 50% of the long-term fire cycle. Recently, this goal has been refined to express how fire should be apportioned on

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the landscape according to ecosites, watersheds, ecological subregions, and vegetation groups (Achuff et al. 2001). Park policies also state that ecological restoration activities will be based on scientific research and be carefully monitored (Heritage Canada 1994), but fire managers recognize that monitoring to guide fire use is sometimes lacking.

Some Jasper lodgepole pine communities were burned in 1977 (Dubé 1978) and 1988 (McCallum 1989), and it was shown that prescribed fire could effectively reduce tree canopy cover and rejuvenate grasslands. However, the ecological impacts have not been followed over time, especially related to fire intensity and severity. In addition, elk (*Cervus elaphus*) populations are high in the park and their impacts have been documented in many other areas (e.g., Kay 1997, Kay and Bartos 2000), especially relative to fire (Peck and Peek 1991, Coughenour et al. 1996, Singer and Harter 1996, Bork et al. 1997, Hillis and Applegate 1998, White et al. 1998, White 2001). These studies have shown that fire can increase vegetation growth, but elk browsing can greatly reduce regrowth of trembling aspen (*Populus tremuloides*). However, the responses are largely site-specific and it is difficult to extrapolate vegetation responses in other geographic areas and plant communities. This is especially true for ground cover vegetation because species dynamics are a function of incumbent vegetation, local seed sources, and their responses to grazing and fire.

In 1998, we began an investigation in support of Jasper National Park's prescribed fire management plan to follow the ecological effects of fire on the landscape: in essence, using planned burns as experiments in an adaptive management approach. Improved scientific information will help to develop fire prescriptions to aid managers in their use of prescribed fire. The current project is part of a larger prescribed fire program that also involves Douglas-fir communities in the montane ecoregion. We are studying the effects of fire intensity on vegetation and using exclosures to investigate ungulate impacts and interactions with fire. Here, we report data collected during the first 3 years of the project.

METHODS

The experimental site is a flat glaciofluvial terrace at an altitude of about 1,020 m within the montane ecoregion of the Athabasca river valley (52°59′N, 118°03′W) in Jasper National Park, Alberta, Canada. The 250-ha experimental area is a north–south valley, bordered by a river to the east and a highway to the west. The experimental design consists of three treatments: no-burn control (60 ha), low-intensity fire (80 ha), and high-intensity fire (110 ha). For each of these treatments, sites were selected subjectively based on tree density representing closed forest canopy (>75% closure) and open forest canopy (<25% closure). Both of these canopy types occur in this montane area, as well as some areas that are open grassland. Paired 40 × 15-m plots were established for each treatment and site type, one of which was enclosed with a 2.5-m-high metal fence (exclosure plots). This design of 12 plots (6 exclosures), allows testing of the effects of fire intensity, ungulate exclusion, and tree density.

Prescribed fires were ignited using a hand drip torch in strips ranging from about 10 m to 50 m wide, depending on the need for flame-length control. The strips were lit sequentially upwind and the plots were burned as a head fire. Flame lengths were measured against pre-established marked posts and fire rate of spread was timed manually in each plot. Fire spread was along the long axis of each plot.

Ecological measurements began in 1998 before the fires and were made annually in late July following the fires. The vegetation was divided into four types: trees, shrubs, regenerating seedlings–saplings, and ground cover vegetation. The diameter at breast height (DBH), total height, health, and height to live crown were measured for all trees (>5 cm DBH) in each 40 × 15-m plot. Height and crown diameter of clumped shrubs, such as Canada buffaloberry (*Shepherdia canadensis*) and common juniper (*Juniperus communis*), were measured. The browse condition (none, light [1%–10%], medium [11%–50%], or heavy [>50% of twigs]) was recorded to estimate ungulate usage. The height of non-clumping saplings (e.g., *Rosa acicularis*) >15 cm tall was measured along a 2 × 40-m strip through the center of the plot. Ground cover vegetation was measured by estimating percent cover of each species in permanently marked 0.2 × 0.5-m quadrats at 2-m intervals along three transects in each plot, spaced 4 m apart ($n = 60$). Floral taxonomy is based on Moss (1983). Graminoids were defined as grass and sedge species, with dead graminoids including all dead and cured material. Statistical differences among treatments were tested using paired t-tests with the individual ground cover vegetation quadrats as replicates (SYSTAT 1997). The change in shrub height was tested among plots using analysis of variance (ANOVA) (SYSTAT 1997).

Counts of elk and deer winter pellet groupings were made each spring along a 100 × 2-m belt transect next to each plot. Counted pellets were removed to ensure that they were not recounted in future years.

RESULTS AND DISCUSSION

Fire Behavior

The fire-behavior parameters for both low- and high-intensity fires are given in Table 1. The 1999 low-intensity fire was essentially a slow-moving surface fire under the tree canopy with some single trees with low branches in open areas candling and being completely engulfed. The 2001 high-intensity fire supported a crown fire in continuous patches of canopy, with flame lengths being 2–3 tree heights in some cases. We concentrate on the low-intensity burns of 1999, since only a short post-fire period has followed the 2001 high-intensity burns.

Effects on Trees

Tree mortality and survival were measured in 2000 and 2001 for the low-intensity burn. Figure 1 shows mortality and survival for four size classes, comparing the open-canopy and closed-canopy plots. Almost all of the trees (only 1 survivor) were killed in the open-canopy plots, largely because of abundant lower branches on these trees, which promoted candling, and because of increased wind speeds in the open with drier conditions caused by more available sunlight. Percent crown scorch has been shown to be one of the best mortality indicators for lodgepole pine (Petersen and Arbaugh 1986). Grouping all trees, we observed that 74% of the dead trees had >50% of their crowns scorched, whereas 83% of the surviving trees had <50% crown scorch. Almost all of these trees are still standing, and most still retain the dead foliage. Thus, there has not been an increase of sunlight penetration to the forest floor, and the trees still maintain the same structural characteristics that they had before the fire. Tree fall will probably occur in several more years, a result of high winds creating blowdown as trees weaken from insects and disease.

Several models have been proposed to estimate lodgepole pine mortality after fire. We compared our sample of 100 trees with the equations of Ryan and Reinhardt (1988) and Petersen and Arbaugh (1986). Using both bark thickness and percent crown scorch, the models of Ryan and Reinhardt (1988) predicted that all of our study trees would have >50% probability of being killed because of the relatively thin bark (based on the equation of Faurot [1977] as given by Ryan and Reinhardt), even if there is no crown scorch. Petersen and Arbaugh (1986) estimate 49% of the trees would experience a >50% probability of mortality using crown scorch alone as a parameter, or 47% of the trees would die if we include bark thickness as a second parameter. Three growing seasons following the fire, 62% of our study trees in the closed-canopy plots and 96% of our study trees in the open-canopy plots have died. Hence our data lie between the model predictions of Ryan and Reinhardt (1988) and Petersen and Arbaugh (1986) for all plots.

Effects on Shrubs

We measured height and crown cover of Canada buffaloberry, the most common shrub in these communities. Most clumps regenerated from their roots following the 1999 fire, and measurements of height and crown volume indicated recovery success (Figure 2). Shrub height was significantly greater in the exclosed open-canopy plots than in the non-exclosed plot (mean exclosed = 0.4 m, non-exclosed = 0.29 m; ANOVA, $P = 0.001$). Twig browsing was medium to heavy in all non-exclosed plots. Shrub height was not significantly different between exclosed and non-exclosed closed-canopy control plots.

Effects on Regenerating Saplings

Very few pine seedlings (<5 per year) were found in the ground cover vegetation quadrats, indicating little

Table 1. Parameters of two prescribed fires in Jasper National Park, Alberta, Canada. Fire parameters are based on the Canadian Fire Weather Index System (Van Wagner 1987).

Parameter	Low-intensity fire	High-intensity fire
Date of fire	6 May 1999	23 May 2001
Fine Fuel Moisture Code	91	93
Duff Moisture Code	35	40
Drought Code	225	602
Fire Weather Index	19	36
Air temperature (°C)	21	30
Relative humidity (%)	20	15
Wind speed (km/h)	10	<20
Rate of spread (m/min)	2–5	10
Flame length	10–20 cm; occasional candling	Surface 10–20 cm; crown fire to 30 m

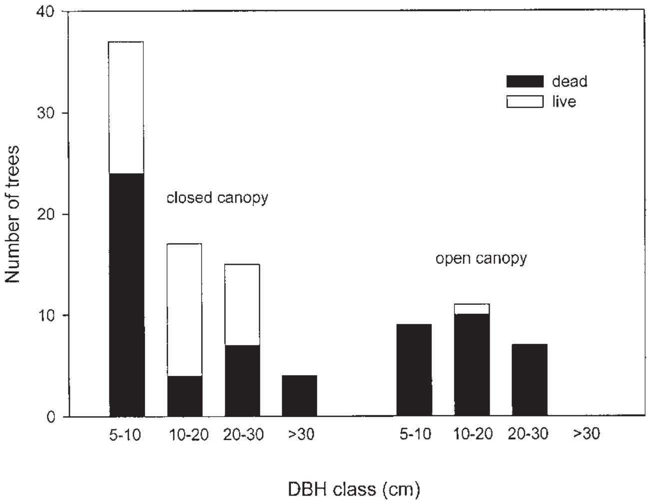

Figure 1. Tree survival and mortality by diameter at breast height (DBH) class in closed- and open-canopy plots for the 1999 low-intensity prescribed burn, Jasper National Park, Alberta, Canada. Total number of trees is given within two plots for each canopy closure type (1,200 m^2 each).

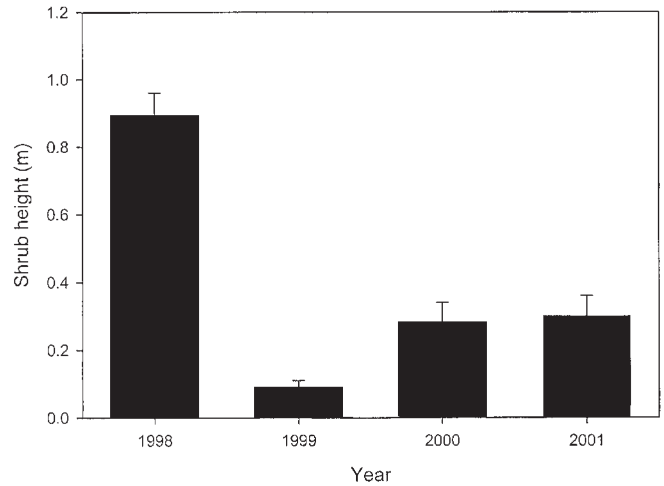

Figure 2. Fire effects on height of Canada buffaloberry (*Shepherdia canadensis*) shrubs, Jasper National Park, Alberta, Canada. The mean ± 1 SD is shown (n = 4 plots, including exclosed and non-exclosed). The low-intensity prescribed burn was in May 1999.

successful reseeding following the fire. However, this lodgepole pine population continuously regenerates without fire and is not obligatorily serotinous. Essentially all of the pine saplings were killed by the fires. Similarly, white spruce (*Picea glauca*) has not re-established, but it made up only 7% of the pre-fire tree canopy.

Elk browsing has severely hampered aspen regeneration and contributed to the decline of aspen stands throughout the montane ecoregion of Jasper National Park (Nietvelt 1999). There are very few mature aspen trees in our study area, which is pine dominated. No aspens were present within our plots, but some were located within the root radius of the plots. Although there were a few aspen suckers within one of our plots, no aspen have yet appeared following the low-intensity burn in 1999 within the regeneration transect along the center of the plot axis. However, more aspen suckers have regenerated following the high-intensity burn in 2001 and are surviving within the exclosures. Within two 40 × 2-m belt transects, we counted 30 aspen saplings with an average height of 16 cm.

Effects on Ground Cover Vegetation

The mean number of ground cover vegetation species per plot was similar among the plots and changed little following the low-intensity fire (Figure 3). The percent ground cover of ground species (vascular non-graminoid species) was reduced in all plots immediately following the fire in 1999, but recovered fully in subsequent years (Figure 3). Figure 4 shows the dynamics of live and dead graminoids, most of which are grasses. The greatest fire effect was the reduction of graminoid thatch immediately after the fire but graminoids recovered soon after the fire, so that amounts of live and dead graminoids by the third growing season were similar to pre-fire amounts. These figures included data from both exclosed and non-exclosed plots.

Paired t-tests were used to compare the pre-fire 1998 to the post-fire 2001 ground cover data for all species that had a percent cover of >0.1% in each plot (Table 2). The amount of detritus consistently increased over this period, whereas moss cover decreased. *Anemone multifida* increased in three plots. *Linum lewisii* and *Solidago spathulata* each increased in a single plot. The other species shown in Table 2 decreased in some plots; *Astragalus dasyglottis* decreased in three plots. The species listed in Table 2 were not very sensitive to time of sampling since their abundance remained similar through a period of several weeks near late July when we sampled. We believe that these small changes are consistent, although we expect some interannual variation.

Few changes in ground cover species could be attributed to open versus closed canopy or to the presence of exclosure (Table 2). The exception is the amount of live graminoids, which increased in the closed-canopy plots, with no change or a decrease in the open-canopy plots. Grass in the closed-canopy plots was dominated by *Elymus innovatus*, whereas there was more *Koeleria macrantha* in the open plots.

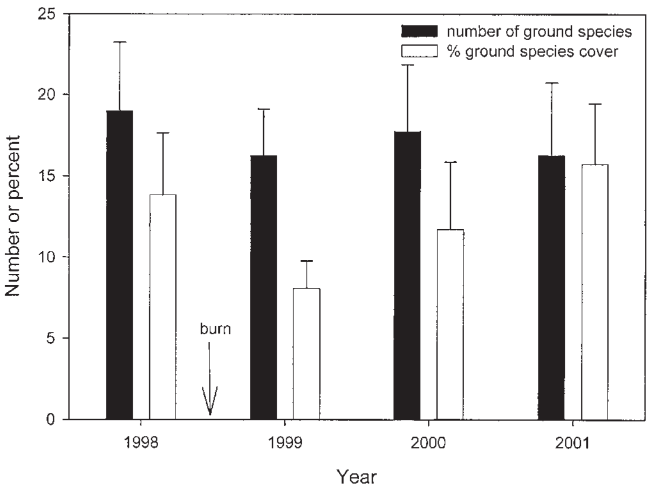

Figure 3. Effect of the 1999 low-intensity prescribed burn on non-graminoid ground cover vegetation species, Jasper National Park, Alberta, Canada. Means ± 1 SD are shown (*n* = 4 plots). The burn occurred in May 1999.

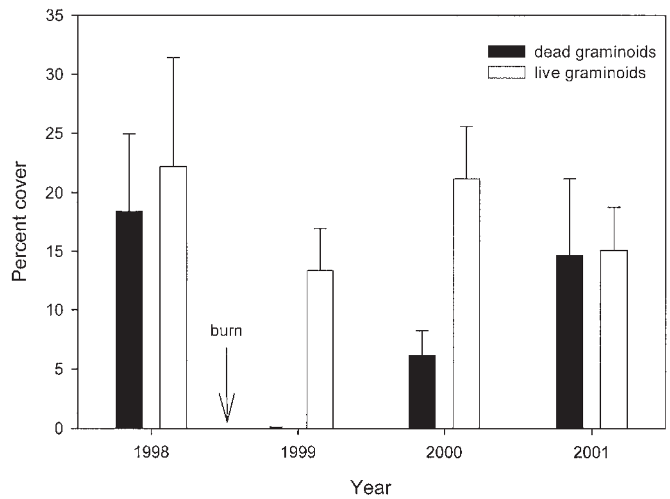

Figure 4. Effect of the 1999 low-intensity prescribed burn on graminoids, Jasper National Park, Alberta, Canada. Means ± 1 SD are shown (*n* = 4 plots). The burn occurred in May 1999.

As mentioned previously, it is likely that a large change in the ground cover vegetation may not occur until the tree canopy disappears and allows more solar radiation to reach the forest floor. So far, we have not seen invasions of nonnative plants, which is a potential concern in the montane ecoregion of the park.

Effects of Ungulates

Observations in an exclosure established following a 1998 fire on a similar site near our experimental area show dramatic differences between exclosed and non-exclosed areas. The exclosure has poplar (*Populus balsamifera*) and aspen growth about 2 m high with substantial amounts of grasses and herbs, especially leguminous species. The non-woody species also reach maturity and distribute seeds. However, this exclosure is in a community that had many mature *Populus* trees, which sprouted as root suckers. In contrast, our experimental area had very few mature *Populus* trees. We have observed little sprouting within

Table 2. Ground cover vegetation changes in closed- and open-canopy plots between 1998 (pre-fire) and 2001 (post-fire) for the 1999 low-intensity prescribed burn, Jasper National Park, Alberta, Canada. Species or cover elements with greater than 0.1% cover and statistically significant changes in at least one plot are shown. Values in bold indicate significant differences between years at $P < 0.05$ (paired *t*-test).

	Closed canopy				Open canopy			
	No exclosure		Exclosure		No exclosure		Exclosure	
Species or cover element	1998	2001	1998	2001	1998	2001	1998	2001
Live graminoids	**12.0**	**17.8**	**9.5**	**12.7**	**16.0**	**12.1**	17.9	19.2
Dead graminoids	14.4	15.5	12.0	10.9	21.1	8.6	26.3	22.3
Detritus	**13.2**	**34.2**	**9.2**	**28.0**	**17.8**	**43.3**	**17.8**	**33.8**
Mosses	**18.8**	**1.0**	**15.1**	**0.1**	2.4	0.1	**3.8**	**0.08**
Bare mineral soil	0.3	0.7	**0.4**	**0.1**	0.2	2.5	0.45	0.33
Antennaria parvifolia	1.2	1.8	1.0	0.5	**11.0**	**3.3**	3.0	1.5
Anemone multifida	**0.48**	**0.17**	**0.2**	**0.6**	**0.51**	**1.15**	**0.07**	**0.85**
Arctostaphylos uva-ursi	**13.4**	**8.9**	2.0	1.3	0.37	0.68	4.7	5.3
Astragalus bourgovii	2.4	2.9	**1.0**	**2.7**			0.42	0.15
Astragalus dasyglottis	**3.4**	**0.8**	1.5	1.4	**14.3**	**2.8**	**10.1**	**6.2**
Gentianella amarella	**0.83**	**0.05**	0.1	0.3				
Linum lewisii	**0.02**	**0.28**	0.03	0.10	0.1	0	0.2	0.13
Solidago spathulata	1.6	1.4	**0.9**	**1.7**	**1.2**	**0.5**	0.75	0.52
Viola canadensis	**1.9**	**1.0**	1.1	1.2				

Table 3. Ungulate pellet counts (mean groupings per 100 m^2), Jasper National Park, Alberta, Canada, 1999–2001. No significant difference among treatments were detected in any given year.

Treatment	Year 1999	Year 2000	Year 2001
Control	3.2	6.0	2.4
Burned May 1999 (low intensity)	10.5	8.9	6.4
Burned May 2001 (high intensity)	5.4	7.1	3.1

our plots following the low-intensity fire. Aspen sprouting is evident within one plot following the 2001 high-intensity fire, but it will be a few years before the effect on the plant community in this plot becomes evident.

There were some differences between the exclosed and non-exclosed plots, even before the exclosures were established. For example, cover of live graminoids was lower in the exclosure of the closed-canopy plots in both 1998 and 2001 (t-test, $P < 0.03$, $n = 60$) (Table 2). Biomass measurements are being collected over the next few years within exclosed (1-m^2 range cages) and non-exclosed areas to show the overall impact of browsing on plant biomass. Overwinter use of our site by elk as measured by pellet group counts has been variable (Table 3). Annual aerial surveys by park wardens estimated elk populations in the entire Athabasca River valley to be 633 in 1999 and 533 in 2001, with subpopulations near our study area of 392 in 1999 and 291 in 2001 (Ralf 2001). This interannual variability may be caused by variable snow cover throughout the valley, recent translocation of 210 elk from the park, and hazing activities to dissuade herds from frequenting the Jasper town site, which is about 7 km away. White (2001) estimates that elk numbers producing winter pellet groupings in excess of 1/100 m^2 are sufficiently high to inhibit *Populus* species regeneration and keep biomass low in feeding areas. Our pellet group counts far exceeded White's (2001) estimate, indicating a high elk population in all treatments and years (Table 3).

MANAGEMENT IMPLICATIONS

A low-intensity prescribed fire was successful at killing about two-thirds of the trees in a lodgepole pine-dominated community, although mortality was higher in open-canopy than in closed-canopy areas. These trees were still standing 3 years after the fire and much of the canopy, although dead, is still present. Hence, there has been little change in the light regime for these plant communities, and it will likely take several years before the tree canopy disappears. Minimal changes in the number and cover of individual species have been detected to date. The immediate decrease in dead grass thatch recovered to pre-fire levels. Shrub height is about one-third of that pre-fire. The vegetation within the ungulate exclosures has not yet shown large differences compared with that in non-exclosed plots. However, aspen has sprouted in some exclosures that experienced a more recent higher-intensity fire, creating the potential for an aspen-dominated forest in the future. Although this study focused on the response of vegetation to fire, there are obvious enhancements in bird and insect life caused by the dead trees (e.g., increased beetle and woodpecker activity). Longer-term monitoring of these areas will help managers set more quantifiable objectives and better incorporate fire as an ecosystem management tool in the montane ecoregions of Jasper National Park. Information from this study is being communicated to park residents and visitors to increase public awareness and appreciation of the role of fire in montane ecosystems.

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