

Effects of Post-fire Logging on Elk Habitat During the First 3 Years Post-Fire: A Case Study of the Dogrib Creek Fire in the Eastern Slopes of Alberta.



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EXECUTIVE SUMMARY

In the fall of 2001, the Dogrib fire burned 102 km² of pine and spruce forests southeast of the Ya Ha Tinda Ranch, home to one of Alberta's largest elk herds. Following the fire, Sundre Forest Products (formally Sunpine Forest Products) salvage logged 25% of the burn. Few studies, and none in Alberta, have examined how ungulate and ungulate forages respond to salvage, or post-fire, logging. We examined the effects of post-fire logging on elk within the Dogrib burn. We compared plant responses in three post-fire treatments (burned only, burned and logged, and pre-fire cutblocks that were burned) during the first three years post-burn for differences between: 1) forage biomass (amount of food for elk), 2) plant species diversity, 3) conifer (pine) seedling germination, and iv) ground cover and temperature. We also examine how elk responded to the three treatments using telemetry locations and counts of pellet groups.

Post-fire logging reduced the biomass of forbs in all 3 years relative to the other treatments, but reduced graminoids only during the first 2 years. By the third year after the fire, graminoids had increased 5 times, such that the total amount of herbaceous (graminoids + forb) forage was equal among burned forest, burned clearcut and post-fire logged treatments. Shrub biomass (browse) did not differ among treatments in any year. Previous cutblocks (<10 years old) that were burned had the highest amount of herbaceous forage (both graminoids and forbs) after fire. Post-fire logging tended to reduce species diversity in favour of a few dominant species in some areas, but had no influence on diversity or the number of species. Post-fire logging did, however, reduce pine seedling densities by 20% compared to burned treatments.

Despite minimal differences in forage between burned and salvaged sites, elk (and deer and moose) selected un-logged burns and avoided post-fire logged sites. Based on other studies of elk and forestry across western North America, we infer that elk avoided salvaged cutblocks because: 1) post-fire logging would reduce habitat effectiveness through removal of dead standing tree cover, and/or 2) elk avoided salvaged areas due to predation risk by humans and wolves associated with logging roads built for salvage logging. Cover has been shown to be important for increasing elk habitat effectiveness in Alberta and western North America. In addition, in ongoing studies of wolf predation on elk at the Ya Ha Tinda and in the nearby Clearwater-Rocky forest, risk of being killed by wolves increased for elk near roads. Moreover, human-caused mortality may have increased in association with logging roads: in 2002, at least 3 elk were poached in the Dogrib burn from roads. Therefore, managers must consider that if access is not effectively restricted following salvage logging, post-fire logging may reduce habitat available to ungulate species such as elk. We recommend increased on-site mitigation measures for post-fire logging and stronger access management policy and enforcement in Alberta that may increase the value of this treatment for enhancing ungulate habitat.

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1. INTRODUCTION

Wildfire is an important natural disturbance in much of the world's forested temperate ecosystems, and can have profound effects on terrestrial ecological processes (Wright and Bailey 1982; McRae et al. 2001). In Canada, wildfire is the dominant ecological disturbance for forested ecosystems in terms of area affected and frequency of disturbance (Rowe & Scotter 1973; Weber & Flanigan 1997). Fire affects temperate ecosystems by renewing forest succession, affecting nutrient cycling, sediment transport, and wildlife habitat, and the effects of fire vary across spatial scales of forest fire patch size and temporal scales of fire return intervals (reviewed in Wright and Bailey 1982; Smith 2000; McRae et al. 2001). In the temperate forests of North America, fire is a critical factor shaping the diversity and abundance of songbirds, vascular plants, insects, and terrestrial vertebrates (Nappi et al. 2004; Morissette et al. 2002; Smith 2000). In temperate regions, one important group of terrestrial vertebrates that are strongly influenced by fire as a natural disturbance are ungulates. Ungulate species such as moose (*Alces alces*), elk (*Cervus elaphus*), bighorn sheep (*Ovis canadensis*), mule (*Odocoileus hemionas*) and white tailed deer (*Odocoileus virginianus*) consume herbaceous or woody plant species known to increase following fire (Sachro et al. 2005; Smith 2000; Peek 1974; Merrill et al. 1982). Vegetation response following fire can affect ungulate populations through increased forage quality and quantity (Wu et al. 1996; Boyce & Merrill 1996; Turner et al. 1994).

In Yellowstone National Park (YNP), positive effects of the 1988 fires contributed to a 20% increase in the elk population for 2-5 years post burn (Taper & Gogan 2002), and simulation studies (Wu et al. 1996; Boyce & Merrill 1996) projected similar long-lasting effects of fire on elk population dynamics. In areas dominated by mature conifer with fewer grasslands, the effects of fire on ungulates may be expected to be more important. Ecosystem management

plans including fire as a tool for restoration of terrestrial ecosystems often make restoration of ungulate habitat potential as an important response variable (Svancara et al. 2004; Lehmkuhl et al. 2001). However, restoration of fire is complex in modern landscapes with a high proportion of the landscapes often allocated for timber extraction (Timoney 2003), and forest-harvesting companies can incur substantial economic losses of merchantable timber by wildfire. Current predictions of the effects of global climate warming include increased frequency of wildfires and extension of the forest fire season in North America (Brown et al. 2004; Gillett et al. 2004).

Following fires in commercial timber areas, companies must often quickly extract burned wood through post-fire harvesting, often known as “salvage logging” in an attempt to recover these economic losses (Nappi et al. 2004). Thus, because of the potential for increased wildfires due to lengthening of the fire season, large-scale post-fire salvage operations may increase. Numerous studies have documented the disruptive effects of post-fire logging on natural processes such as fuel accumulation, hydrology, sediment transport, and on select wildlife species (Lindenmayer et al. 2004; Smith 2000; McIver & Starr 2000; Beschta et al. 2004; Nappi et al. 2004). The removal of large woody structures typical in post-fire logging operations can change plant species composition, reduce plant species diversity, and increase conifer growth in the first years after logging (McIver & Starr 2000; Beschta et al. 2004). Post-fire harvesting removes critical habitat for species, such as cavity-nesting mammals, birds, and insects that depend on burned wood (Lindenmayer et al. 2004). Removal of large woody structure has caused declines in the abundance of several cavity-nesting bird species, including mountain bluebird (*Sialia currucoides*), and black-backed (*Picoides arcticus*), hairy (*P. villosus*), and three-toed woodpeckers (*P. tridactylis*) (Hutto 1995; Hoyt & Hannon 2002; Morissette et al. 2002; Hobson & Schieck 1999).

Although a number of studies have examined the effect of post-fire logging on wildlife, most have examined effects on species that depend on snags created by wildfire such as woodpeckers or cavity nesting marsupials (Morissette et al. 2002; Lindenmayer & Possingham 1996). Unfortunately, the effects of post-fire logging on wildlife that are not obligate snag-dependent species are largely unknown and remain a priority for land management agencies (McIver & Starr 2000). Post-fire logging will influence post-fire vegetation patterns in ways that may potentially benefit species such as ungulates that depend on early seral forests for forage. Because there is a paucity of information regarding the effect of post-fire logging on ungulates, we examined the effects of post-fire logging on forage from an ungulate management perspective, focusing on elk (*Cervus elaphus*). We investigated effects of post-fire logging on plant species diversity, elk forage biomass, conifer seedling densities, and elk use of different treatments within the Dogrib fire. The Dogrib fire occurred late in the fall of 2001 in the foothills area of central Alberta adjacent to Banff National Park (BNP). Following the burn, approximately 25% of the Dogrib burn in total was post-fire logged that immediate winter. Thus, the post-fire landscape was a patchwork 1) burned and uncut, 2) burned and cut (post-fire logged), and 3) cut and burned. We adopt the term post-fire logging specifically to avoid confusion with other types of salvage logging (e.g., following insect outbreaks), and for clarity in application in situations when post-fire harvest is not done for monetary reasons (McIver & Starr 2000).

The Dogrib burn provides an ideal opportunity to examine the influence of post-fire logging on ungulate forage dynamics and habitat selection because the burn lies adjacent to the Ya Ha Tinda (YHT) elk herd, one of Alberta's largest elk herds (Gunson 1997). This elk herd migrates between winter ranges at the YHT Ranch in the Red Deer River valley to upper

elevation alpine ranges in front ranges of the Rocky Mountains including the Dogrib Fire (Morgantini & Hudson 1988). Our main objectives were to investigate the effects of post-fire logging relative to burned uncut areas on (1) herbaceous and shrub forage biomass, (2) plant species composition and diversity, (3) conifer seedling density, and (4) elk resource selection during the first three years post-burn. We also tested for differences between treatments in soil surface growing temperature during 1-growing year to examine potential mechanisms for post-fire vegetation differences. Further, during the second year, 2003, we included a treatment of pre-burn cutblocks that were opportunistically burned in the Dogrib burn in a three way comparison among post-fire logging and burned treatments. In these analyses, we controlled for the effects of the following independent variables: fire severity, distance to seed sources, terrain features, soil moisture potential, and years since burn.

2. STUDY AREA

The Dogrib Fire is located in west-central Alberta (51°38'N/-115°30'W) along the Upper Red Deer River just outside the eastern boundary of Banff National Park (BNP) (Fig 1). The burn straddles the upper foothills and montane ecoregions of Alberta, with elevations ranging from 1500 to 2350 m in the western mountainous part of the burn. The western part of the burn is comprised of lodgepole pine (*Pinus contorta*), englemann spruce (*Picea engelmanni*) grading to white spruce (*Picea glauca*) at lower elevations, interspersed with shrub meadows (*Salix* spp. and *Betula glandulosa*), grasslands, and some alpine herbaceous communities. The eastern and lower Dogrib burn is a predominantly forested area of low hills comprised of a mosaic of lodgepole pine and white spruce, black spruce (*Picea mariana*)/tamarack (*Larix laricina*) bogs, low gradient riparian areas, and an increased component of deciduous trembling aspen (*Populus*

tremuloides) stands. Adjacent to BNP, the area is home to diverse mammal community, including elk (*Cervus elaphus*), white-tailed deer (*Odocoileus virginianus*), mule deer (*O. hemionus*), and feral horses (*Equus caballus*) (Morgantini 1995).

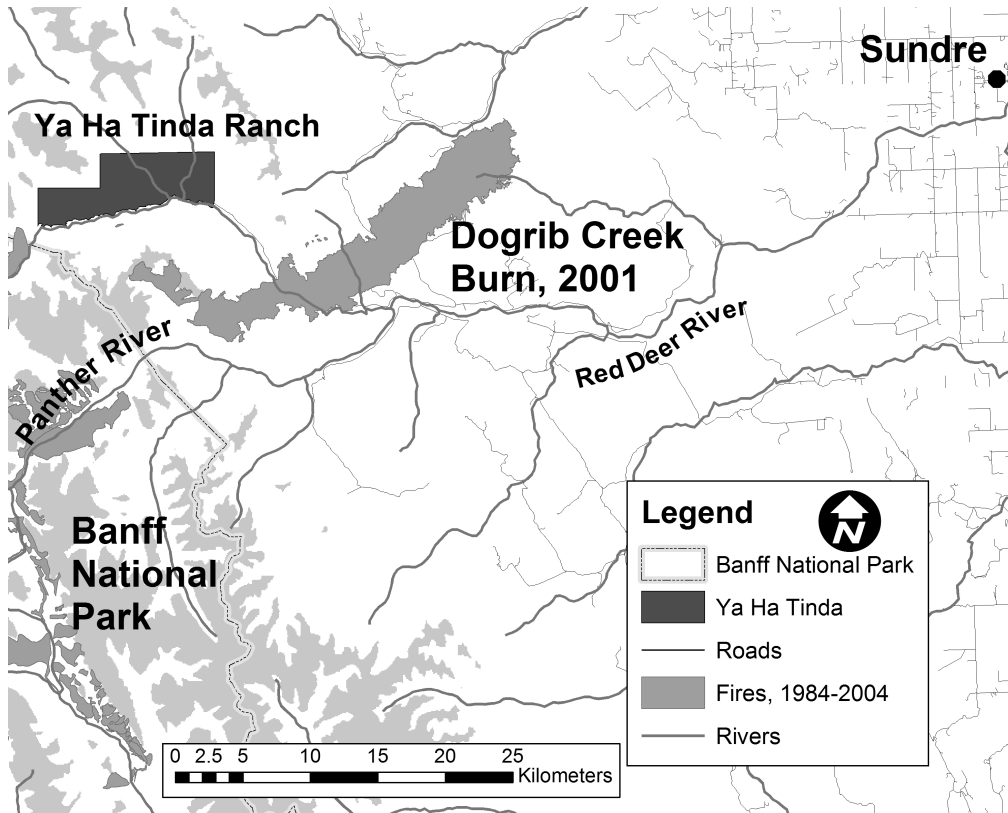


Figure 1. Location of the Dogrib burn within the eastern slopes of Alberta just east of Banff National Park.

Mean summer daily temperature for the period 1970-2004 was 9.2°C, and mean summer (June and July) precipitation is 179.7 mm (Environment Canada, Blue hill tower climate station ID 3050725, located 5 km SE of the Dogrib Burn, 51 42'N / 115 13'W, 1951m). In comparison to this 30-year average, the period of study was significantly drier than average during June and July (Fig. 2, standardized values for rainfall during 2002, 2003, and 2004 compared to the 30-year mean were -1.24, -1.21, and +0.66 standard deviations from the mean). In fact, 2002 was

the 2nd driest and 2003 the 6th driest summers on record since 1970. However, the study period was of average temperature (standardized temperature values for 2002-2004 compared to the 30-year mean were 2002, -0.64, 2003, -0.32, and 2004, 0.32).

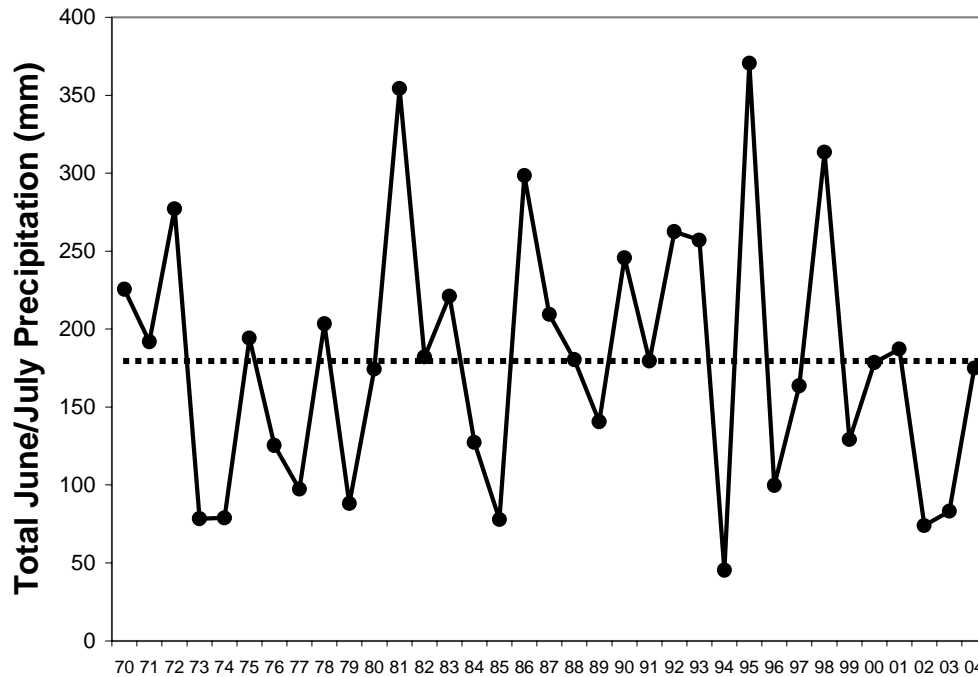


Figure 2. Total Precipitation (mm) in June and July recorded at Blue Hill Tower Weather Station (51 42'N / 115 13'W, 1951m, Env.Cdn. ID 3050725) 1970 – 2004. 2002 was the first year post-burn.

The Dogrib burn started in the upper Dogrib Creek area on or around Sept 10, 2001 from an unattended camping fire in the upper subalpine zone. It spread downslope and east through Dogrib creek quickly but was declared contained and allowed to burn because it was in a zone 1 prime protection wildlife management area (Anonymous 1986). However, on Oct 16th, 2001, outside the normal fire season for this area (C. White, Parks Canada, personal communication), the fire sparked and spread 20km to the east across the Red Deer, Wildhorse and Yara Creeks.

Fire spread was naturally contained by snow on Oct 18, 2001. The total fire size was 10,200 ha, or 102 km². The fire was very severe, and was classified as a crown-sustained fire fanned by dry, Chinook winds. The median fire severity was 0.69 and 78% of the fire was classified as a high severity burn as calculated using the normalized burn ratio of Key and Benson (2003)(see methods for details).

The area burned by the Dogrib burn included both mature and overmature (at the NE end of the fire) unharvested forest and previously harvested forests up to 20-years in age. Previous cutblocks were classified for some analyses, where enough samples enabled such comparisons, into young and old cutblocks using the cutblocks of 5-years of age as the cutpoint between young and old. Approximately 25% of the Dogrib burn was post-fire harvested for salvage purposes that following winter by Sundre Forest Products (SFP, formerly Sunpine Ltd). These differing land-uses thus created a patchwork of landscape treatments including 1) burned and uncut (58% of the total burn), 2) burned and cut (post-fire logged, 25%), 3) cut and burned (17%), and 4) adjacent, unburned, uncut habitats (Fig. 3). In addition, only the eastern portion of the burn east of the Red Deer River is within Sunpine's FMA, and this boundary divides what we refer to as the lower and upper Dogrib Burn.

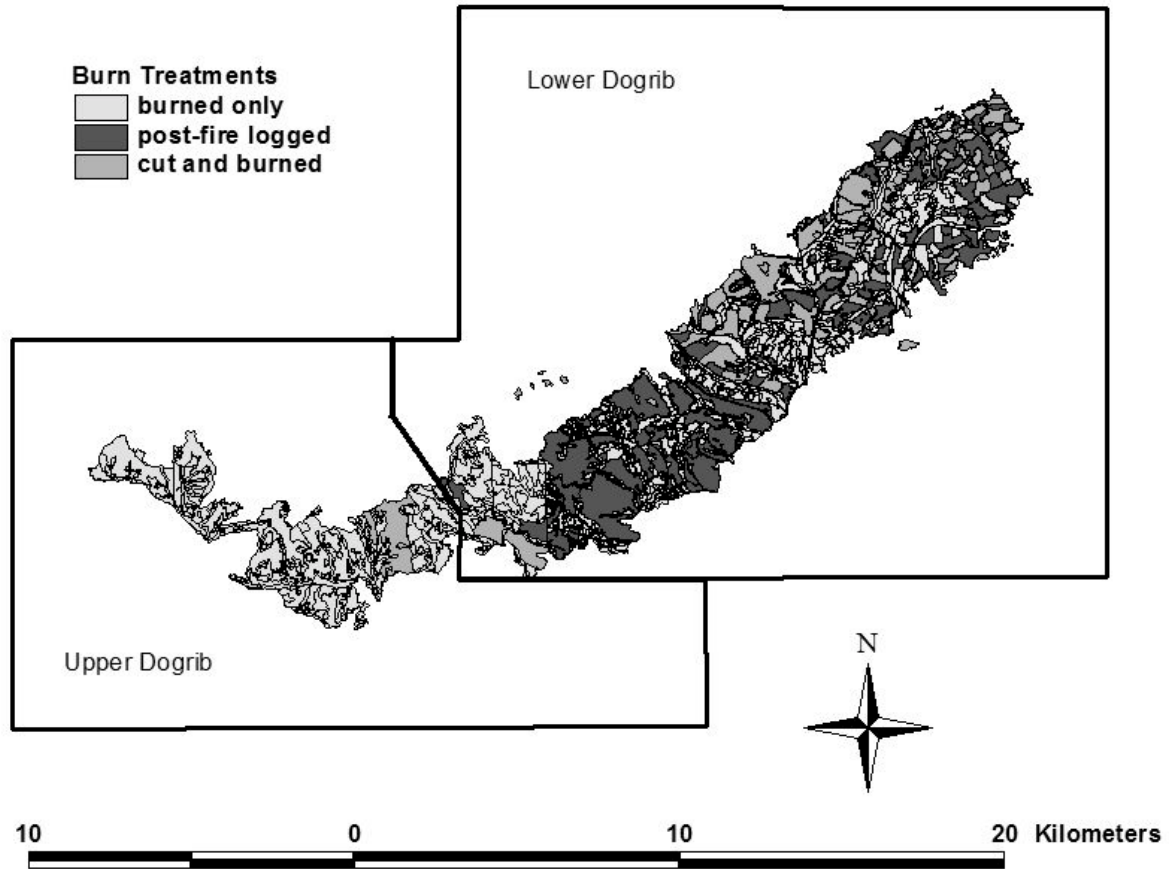


Figure 3. Post-fire landscape treatments within the Dogrib burn including burned, post-fire logged, and previously cut and burned treatments. Upper and lower Dogrib areas roughly correspond to the boundary of Sundre Forest Products Ltd. FMA, which represents the lower Dogrib burn.

3. METHODS

3.1. Elk forage sampling

3.1.1. *Sampling design*

We used a proportional allocation random sampling design (Krebs 1989) following the methods of (Sachro et al. 2005). We randomly sampled within the following strata:

- 1) Fire treatment: 3 strata within the Dogrib burn to design sampling: 1) burned and cut (post-fire logged), 2) burned and uncut (burn), and 3) previously cut and burned.
- 2) Slope categories of flat (0-5°), moderate (5-30°), and steep (>30°)
- 3) Aspect classes of north (0-112.5° and 292.5-360°), south (112.5° – 292.5°), and flat (overlapping with flat slopes).

We selected random locations <1km from roads and trails throughout the study area to facilitate economic sampling using ARCVIEW 3.2 (ESRI), and sampled as many sites as possible during each year. We allocated random locations to strata combinations, and ensured selection of random locations was in proportion to availabilities of strata in the landscape. Because of spatial inaccuracies in GIS location of post-fire logged areas, pre-allocation was often unsuccessful (31% of the time). Field crews navigated to the assigned random plot location by GPS (Garmin GPS 12XL). When the allocated sampling treatment for a sample plot was incorrect, we attempted to match the allocated sampling strata for that site by searching <500m radii. If this proved unsuccessful, we sampled the putative random location regardless. In subsequent years, we attempted to resample all plots repeatedly, but because plant cover quadrats within sample plots were not permanently marked, our design was repeated at the site but not the plot scale.

3.1.2. *Vegetation sampling methods*

Following the rules described above for mismatches between putative and actual strata combination, a final sampling location was selected. At each sampling location we laid out a 30m transect perpendicular to the dominant slope or at a random direction if flat, and ground cover plots located on the upslope side of the transect. Within the sampling location we recorded vegetation data in a hierarchical fashion following Fig 4.

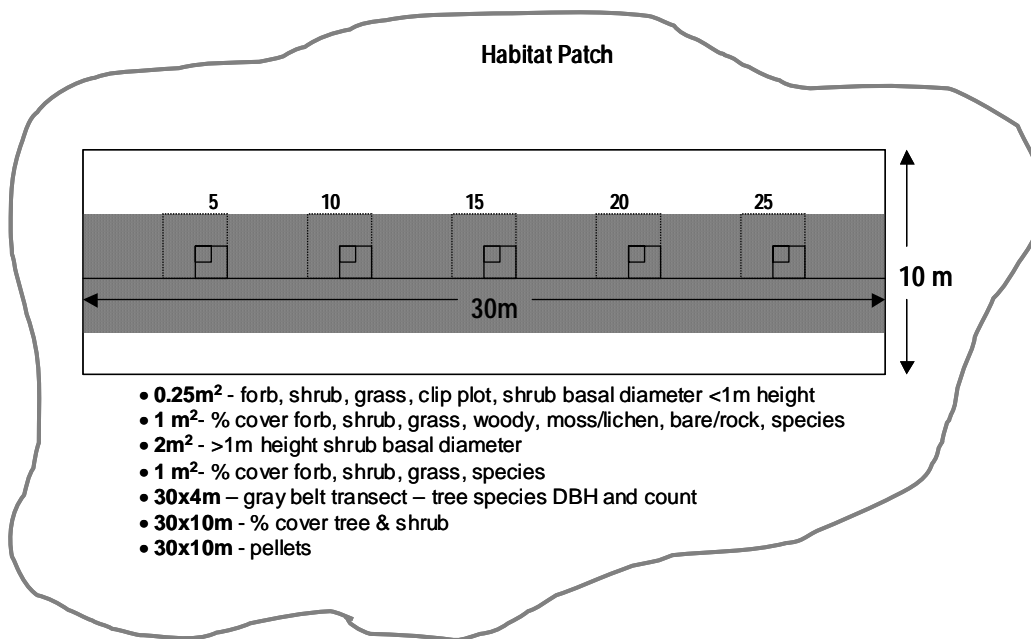


Figure 4. Field sampling protocol for vegetation sampling plots. Note schematic is not to scale. A habitat patch was considered to be one of the strata combinations used for sampling.

Macro-site characteristics included actual location (UTM NAD 83), transect bearing (°), elevation (m), slope (° or percent), aspect (°), canopy cover (%), canopy height (m), and total (winter and summer) ungulate pellet groups were counted. Actual location and elevation were recorded with GPS. Transect bearing, slope, and aspect were recorded using compass and/or

clinometers. For aspect, we recorded both at 0, 15, and 30m along the transect and used the average value for each plot, and took one slope measurement/plot. Canopy cover was estimated with a spherical densiometer or ocular estimate following standardization within field personnel. Canopy height of the top 3 canopy species was recorded by taking clinometer readings at a fixed 20m distance from an average individual species representative of each species. Ungulate pellet groups were counted for summer and winter elk, moose, deer spp., horse, and bighorn sheep within a 30x10-m area (300m²) following the transect, and were converted to pellets/m² for reporting.

Overstory/tree vegetation was recorded in a 4-m wide belt surrounding the 30-m transect (120m²). For each species, we recorded the number of stems with a >5 DBH measurement (i.e., excluding shrubs) within 5 DBH classes: 5-15cm, 15-20 cm, 20-25cm, 25-30cm, >30cm. We assessed shrub cover within a 30x10m area surrounding the transect within two height strata, 1-2m and >2m to approximate shrubs available as ungulate browse within the 1-2m stratum (Lundberg & Danell 1990). At the ground cover layer, we estimated canopy cover / composition of forage classes (forb, graminoid, shrub deciduous, shrub-deciduous), moss and lichen, woody debris, and bare/rock/litter cover categories within five 1m² plots spaced at 5, 10, 15, 20, and 25m. Within composition plots cover of grass, sedge, and forb was recorded to species. Species composing <1% cover were not included, and unknown species recorded as most likely genera.

3.1.3. *Herbaceous biomass*

We estimated herbaceous biomass by clipping standing herbaceous vegetation (green and cured) at the 5, 15, and 25 m plots within a 0.25 m² quadrat at 2-3 cm above ground level to approximate biomass available to ungulates. We recorded wet weight and oven dry weight to the

nearest 0.01g after drying samples at 50 degrees for 48 hours. We report herbaceous biomass in g/m².

3.1.4. Woody shrub biomass

We estimated biomass using an indirect approach based on measuring shrub basal diameters (Brown 1976; MacCracken & Van Ballenberghe 1993). We recorded the total count and basal diameters (mm) of all shrub species <1m in height in 0.25m² plots at the 5, 15, and 25m plots. We used empirical predictive equations to predict twig (current annual growth, CAG, and previous years) and leaf biomass based on basal diameter (mm) for the common shrub species. For shrub species >1m in height, we recorded the basal diameter within a larger 2m² plot to ensure we sampled an adequate number of lower density larger shrubs. Predictive equations for shrub basal diameter to biomass were adapted from (Visscher et al. 2005) in the adjacent central east slopes elk study (CESWES) area. Visscher et al. (2004) collected 20 to 60 individual stems for dominant shrub species that had >10 basal diameter measurements (0.01% prevalence) in 2003. For each stem Visscher et al. (2004) measured basal diameter, stripped and separated biomass into CAG, old twigs <7 mm and leaves, dried samples at oven-dried at 100 °C for 72 hours and weighed to the nearest 0.01 g. Twigs ≤ 7 mm were considered available to elk based on diameter at point of browse for moose (Lundberg and Danell 1990), in contrast to approaches that just assume CAG is available as biomass. Total twig biomass is CAG and old twig.

Shrub biomass per stem was modeled for individual species based on basal diameter. Species-specific regression models (Visscher et al. 2004) for the relationship between basal diameter and biomass of the individual stems varied in form (linear, power and exponential) and overall achieved r² values ranging from 0.43 to 0.93. Shrubs were initially divided into low and high browsing pressure; however, Visscher et al. (2004) showed no effect of browsing on

biomass, thus, browsed and unbrowsed samples were combined. We did not develop predictive equations for some rare species or for species that were not common elk forages. We considered *Spirea* spp. and *Symphoricarpus* spp. to be similar to *Rosa* spp., *Menziesia* spp. to *Sherherdia* spp., and *Philadelphus* spp. as *Salix* spp. Finally, we did not consider *Picea* spp., *Abies* spp., etc., nor low-creeping mat growth form shrubs such as *V. scoparium*, *Juniperus* spp., and *Arctostaphylos* spp. because basal diameter is unrelated to shrub biomass in these species. While *V. scoparium* was clipped and treated as a forb (see above), other creeping shrub species (*Juniperus* spp.) were not important elk foods (Hebblewhite, unpubl. data, Cook 2002). For “unknown” shrubs, we created a model using all the species data to create an “average” shrub. Using these predictive equations, we converted basal diameter to twig, leaf, and total biomass for each stem measured. Biomass is reported as g/m².

3.2. Diversity, evenness, and number of species

Species diversity was calculated for each site using Simpson’s index of diversity: $1-D = 1 - \sum(\rho_i)^2$, where ρ_i = proportion of individuals of species i in the community (Krebs 1989). We used Simpson’s index because it is most sensitive to changes in abundant species and we were interested in minimizing effects of misidentifying and under-sampling rare species (Krebs 1989). Evenness at each site was calculated as follows: $\text{Evenness} = D/D_{\text{max}}$, where D = observed index of species diversity, and D_{max} = maximum possible index of diversity, given S species and N individuals ($D_{\text{max}} = 1/S$) (Krebs 1989).

3.3. Ground cover

We tested for differences in the percent cover of bare ground, mineral soil, and woody debris between the post-fire and burned treatments to test for potential mechanisms of plant competition

that may help explain differences in forage responses between treatments. We only recorded ground cover within the above categories during summers 2003 and 2004.

3.4. *Growing temperatures*

In 2003, we deployed iBUTTON® (MAXIM Dallas Semiconductors, Dallas, TX, USA) temperature thermistors to investigate temperature differences between the two main treatments (burned, post-fire logged). We deployed iBUTTON's from 1 July to 1 Oct within 25 randomly selected sites (following our sampling design above) in each of the post-fire logged and burned treatments. iBUTTONS recorded hourly temperature. iBUTTON's were affixed to a survey stake and positioned on the north side of a tree stem (or trunk in the case of post-fire logged treatments) >30 cm from any coarse woody debris. iBUTTONS were deployed so that the iBUTTON was at soil level, below the duff layer. Figure 5. illustrates a typical iBUTTON deployment site in a post-fire logged area.

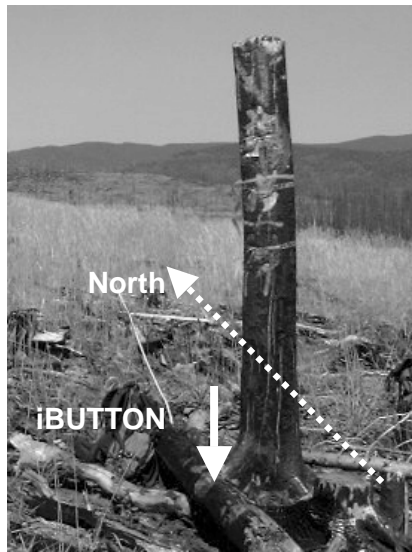


Figure 5. Typical field deployment of an iBUTTON thermistor on a post-fire logged site.

We examined the influence of treatment on the following temperature measures: 1) average and total growing degree days accumulated, 2) average maximum temperature recorded, 3) average minimum temperature, and 4) average range of temperature. Growing degree-days

(GDD) was defined as average daily temperature minus 5°C, following Cesaraccio et al. (2001). We also investigated differences between summer and fall, defined as from deployment until Sept 5th, the date of the first frost, to examine seasonal differences. We analyzed temperature data using ANOVA with season and treatment as the main effects, and include the interaction between season and treatment.

3.5. Conifer seedling regeneration

Conifer seedling germination success, hereafter referred to as regeneration, was assessed during winter 2003/04, 2-years post burn, as part of routine Sundre Forest Products (SFP) germination surveys, which Barry McElhinney, Silviculture Technician, made available. Conifer data were only collected in the lower Dogrib burn within Sundre Forest Products FMA (Fig. 6). Data were collected in two periods: one before snowfall from Sept 22-Nov 1, 2003, and after snowmelt, April 29 – May 29, 2004. Conifer seedling survival was assumed =1 during winter. Sampling design followed a proportional allocation design in proportion to expected variance in conifer seedling regeneration in the following strata: 1) % pine canopy in three classes; >30% pine, mixed conifer (<30% pine), and other conifer (spruce); 2) post-fire logged, burned, and cut and burned and 3) age of pre-burn cutblock (< or > 10-years old), for a total of eight stratum combinations. Sampling intensity decreased with increasing pine canopy composition because increasing pine was assumed to increase successful pine seedling regeneration (Turner et al. 2004), SFP, unpublished data). Because the effect of post-fire logging was uncertain, sampling was higher within these areas. Similarly, seedling regeneration was expected to be lower in younger cutblocks that were burned post harvest compared to older (~10 year old) cutblocks, so we increased sampling in younger cutblocks. Sampling intensity was high overall, and ranged

from 0.25 plots/ha to 1 plot/ha according to expected variance within the above strata (SFP, unpublished data).

Within strata combinations, survey transects were started at a random starting location and azimuth and one plot was conducted every 50m until the predetermined sampling number was reached within that stratum (SFP, unpublished data). Plots were circular with 1.78 metre radius, or 10m². Within each plot, the number of conifer (undifferentiated to species) seedlings were counted from 0-4, and then classified as ≥ 5 . Counts were converted to density of seedlings/ha. Percent cover for grass, forbs, shrubs, organic matter, and mineral soil totalling 100% was determined at each plot, and the % of the seedbed favourable for seedling recruitment was estimated. While seedling regeneration was not identified to species, in summer 2004 we collected an additional 150 plots following the same stratification and determined *Pinus* seedlings constituted >96%, and *Picea* spp. <4% of all seedling. Thus, we effectively consider conifer seedling density as *Pinus* seedling density hereafter.

We examined the effects of the three post-fire treatments, while controlling for GIS derived covariates described below, on conifer seedling density (stems/ha). The first GIS variable we derived was the distance from lightly burned or unburned forests on the number of conifer seedlings. Distance to lightly burned or unburned forests can influence conifer regeneration because these areas can act as seed sources after stand-replacing burns (Turner et al. 1994). Lightly burned areas were identified in a GIS based on a reclassification of the fire intensity surface layer using the Jenks natural breaks clustering algorithm in ArcGIS 9.0 (ESRI), which classified fire intensities of 0-0.45 as low severity (see below, Fig. 7). We then combined low intensity areas and surrounding unburned forest matrix and calculated distance to this combined layer to assess the effects on conifer regeneration (Fig 7).

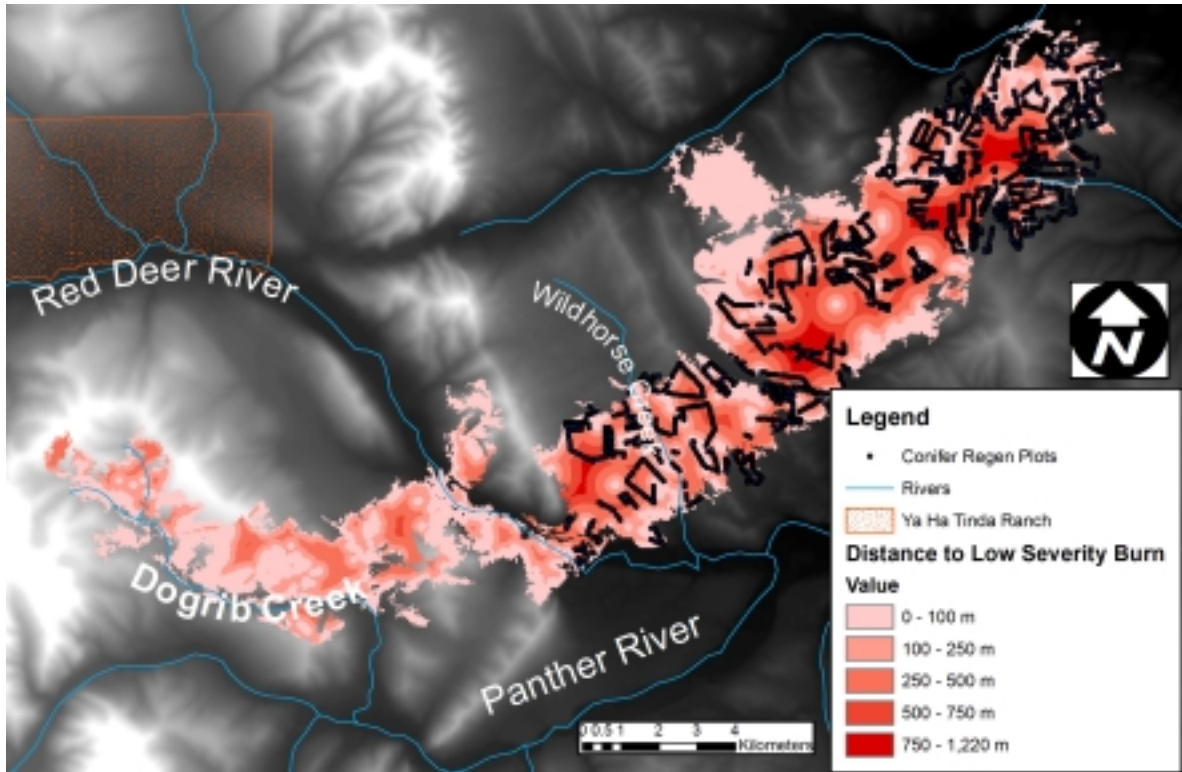


Figure 6. Conifer seedling germination plots and distance to low severity burn used in modeling factors influencing conifer seedling density within the Dogrib burn.

3.6. *Elk resource selection*

3.6.1. *Elk diet composition*

We assessed elk food habits during July 2002 (first year post-burn) to investigate plant species diet composition by elk using the Dogrib Burn. We collected pellet groups from 2-5 different pellet groups each within each site, and created two composite diet samples from 20 different sites within the Dogrib burn during July 2002. We analyzed these using fecal plant fragment analysis conducted at the Washington State University Wildlife Habitat Nutrition Lab.

3.6.2. *Elk telemetry*

We assessed patterns of adult female elk resource selection within the three Dogrib burn treatments using telemetry location data from Global Positioning Systems (GPS) collars and

standard VHF collars deployed on elk captured during winter at the Ya Ha Tinda ranch. All capture methods were approved under University of Alberta Animal care protocol ID#353112. VHF collared animals were located every 7-14 days throughout the year from the ground or via aerial telemetry, and GPS collars were programmed to take a location every 1-2 hours. For this analysis, we restricted our scope of inference to just the Dogrib burn, and selected all GPS and VHF locations which occurred within the burn. We assessed general trends of use of the Dogrib burn by radiocollared elk by: 1) calculating the proportion of total radiocollared elk within each year using the Dogrib burn to assess any trends over time, 2) the seasonal frequency of use of the Dogrib burn, averaged for each individual elk, and 3) resource selection patterns for the three treatment types within the Dogrib burn. Detailed analysis of elk resource selection patterns for fires by the YHT elk herd are presented elsewhere (Hebblewhite, in prep).

We assessed resource selection for each individual elk following Manly (2002)'s study design II where individual animals were identified and considered the sample unit, and use is compared to treatments available throughout the entire burn. We calculated the selection ratio using:

$$\beta_j = \ln \left(\frac{(u_{ij} / m_i)}{\sum_{i=1}^n (u_j / m_i)} \right) \quad \text{Eq. 1.}$$

where β_j = the selectivity coefficient for treatment type $j=1$ to 3 (burned, post-fire logged, cut and burned), u_{ij} = the number of locations for animal $i=1$ to n in treatment type j , and m_i = the total number of radio telemetry locations for individual i , and n = the total number of radiocollared animals used in the sample (adapted from Manly 2002). Equation 1 calculates the resource selection ratio for each animal first, which controls for sampling intensity differences between animals resulting from different sampling intensities in the VHF and GPS collared

samples, then estimates the population average selection coefficient, β_j . Standard error's were calculated following Manly (2002).

3.6.3. Elk pellet data

We tested for differences in the total number of elk pellet groups in the three main treatments detected on all biomass plots using negative binomial regression, NBREG, in STATA 8.0 (StataCorp 2004)(*see statistical analyses section below*). We included the same covariates as in ANOVA models, including GIS variables, and year effects in models.

3.7 GIS derived landscape covariates

We calculated landscape covariates using a 30-meter resolution digital elevation model, from which we derived elevation (m), slope (degrees), aspect, hillshade (evaluated at 45 degrees inclination at an aspect of 225 degrees to represent xeric sites with high values, and mesic sites with low values, see Nielsen et al. (2002)), and a soil moisture index (compound topographic index, CTI (Gessler et al. 1995). We obtained spatial coverage's of the different treatment areas following post-fire harvest from Sundre Forest Products Ltd.

We calculated fire severity using a remote sensing approach (White et al. 1996) following the normalized burn ratio of Key and Benson (2003) derived from LANDSAT 5/7 images obtained before (09/07/1998, Path 43 / Row 24, LANDSAT 5) and after (06/22/2002, LANDSAT 7) the Dogrib burn (D.Zell, BNP Informatics Specialist, unpubl.data). We followed the approach of Key and Benson (2003) in calculating the normalized burn ratio. The Dogrib burn was characterized by overall high fire severity (Fig. 7, 8).

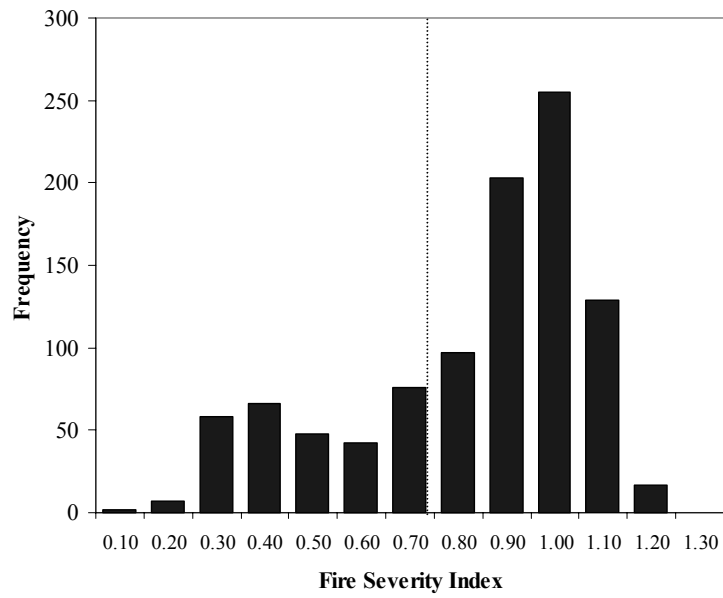


Figure 7. Frequency distribution of normalized fire severity values for the Dogrib creek burn, Fall 2001. The mean fire severity value was 0.77 (dashed line), and the median 0.86.

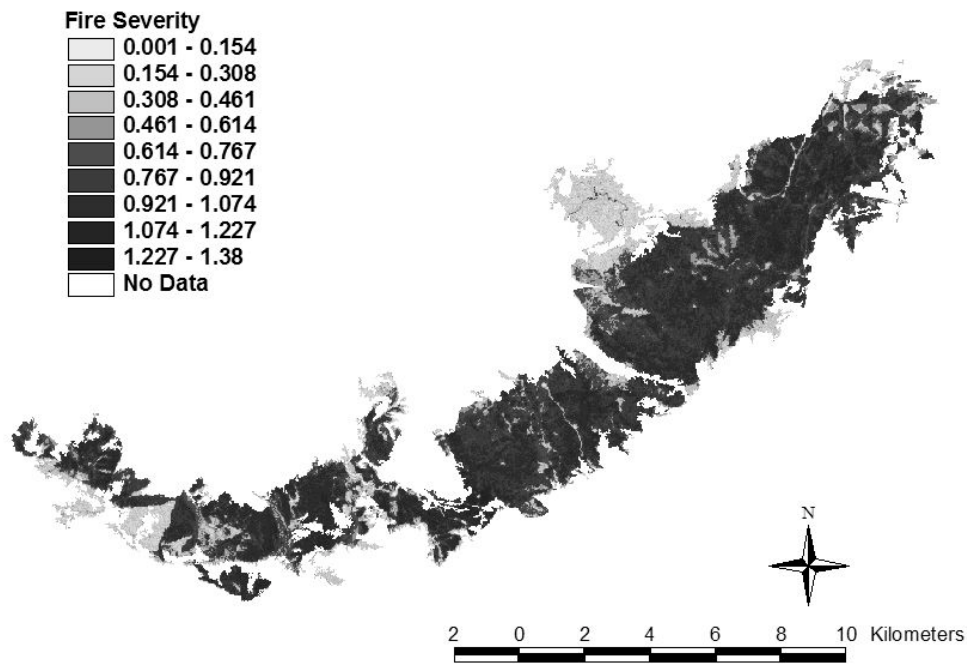


Figure 8. Fire severity of the Dogrib burn calculated using the normalized fire severity index of Key and Benson (see text for details).

3.8 Statistical analyses

Because natural disturbances are unpredictable and unplanned, it is usually not possible to study their effects with a well-balanced design (Bennett & Adams 2004; Wiens & Parker 1995). Furthermore, true replicates are often not possible because the only true level of independent replication is the disturbance event itself, in our case, the entire Dogrib burn. We believe, however, that if interpreted appropriately, comparison of post-fire vegetation dynamics among the management treatments within the Dogrib burn is a valid method for studying effects of disturbance. In this context, evaluation of the Dogrib burn represents a ‘management experiment’ at a useful spatial and temporal scale to provide guidance for land management. As an added check, we examined potential bias between years in our sampling design examining differences between years and treatments for sampled locations using ANCOVA for continuous covariates such as elevation, fire severity, wetness index, or degrees slope.

First, we examined the effects of post-fire logging using a two-factor ANCOVA design with YEAR and TREATMENT type (post-fire logged, burned) as main effects in SPSS 11.0, and refer to this throughout as the Year*Treatment ANCOVA. We conducted a different ANCOVA for each of the dependent variables;

- a) Total herbaceous, forb, graminoid, and total shrub elk forage biomass (g/m^2)
- b) Simpsons diversity index
- c) Simpsons evenness
- d) Total number of species
- e) Ground cover composition (in 2003 & 2004)

To test the hypothesis of treatment differences over time we included a treatment*year interaction. The distribution of forage biomass was heavily skewed to the left because of a high

proportion of low or zero biomass values. Therefore, we transformed all biomass variables to satisfy normal distribution assumptions of ANCOVA using a natural logarithm ($\ln+0.5$). Most diversity related variables were normally distributed, or required a natural logarithm transformation to meet parametric assumptions. We tested for normality using Levene's test (Sokal and Rohlf 1995).

We conducted a second ANCOVA analysis for 2003 data, during which we included the cut and burned treatment in sampling. We used a one-way ANCOVA with treatment as a main fixed effect to test for differences between post-fire logged, burned, and cut and burned treatments. We analyzed the same suite of dependent variables (a) through (e) described above. We refer to this analysis as the 2003-treatment ANCOVA throughout.

We considered the following potential covariates for inclusion in both the Year*

Treatment and 2003-Treatment ANCOVA analyses:

- 1) Aspect class (northeast and southwest facing),
- 2) Slope class (flat, steep),
- 3) Elevation (m),
- 4) Slope (degrees),
- 5) Fire severity measured by the normalized burn ratio (see above),
- 6) Hillshade
- 7) Site soil moisture index (CTI wetness).

Prior to statistical modeling, covariates were screened to reduce collinearity. We used a backwards-stepwise model selection procedure, starting with a full-list of all non-collinear variables, and proceed backwards to the most parsimonious ANCOVA, or, where no covariates were important, a simple two-way ANCOVA design.

Dependent variables whose distributions could not be analyzed using ANOVA were conifer seedling density and elk pellet counts. Both of these variables were modeled using Negative binomial regression using NBREG in STATA 8.0 (StataCorp 2004), appropriate for count data with inflated numbers of low counts or zeros's (Hilbe 1999). We used pellet group count and conifer seedling count as dependent variables in negative binomial regression models, and included the main effects of interest (i.e., burned, post-fire logged, etc) using indicator coding, and the same continuous covariates described for ANOVA models as independent variables. We used a backwards-stepwise model selection procedure to select the most parsimonious model, and tested for adequacy of NBREG model versus the Poisson or zero-inflated negative binomial model using a likelihood ratio test and the Vuong test (StataCorp 2004, Hilbe 1999; Vuong 1989), respectively. We report negative binomial regression parameter estimates, and, for figures, back-transform means and presented them with standard errors calculated from negative binomial regression.

For the conifer analyses, we conducted two negative binomial regression analyses, one for the effects of treatment (burned, post-fire logged) and stand type (pine, mixed conifer, spruce), and a second analyses between the treatment types of burn, post-fire logged, and old (>5 years) and new (<5 years) cut and burned areas. We did not include cut and burned in the full ANCOVA because stand type for the cut and burned treatments were not available. In addition to the covariates described above, for conifer regression models we also included distance from unburned forest, % mineral soil exposed, and % grass cover as continuous covariates.

4. RESULTS

4.1. Forage biomass plots

We collected a total of 132 forage biomass plots over the first three years post-burn. During 2002, we began sampling with 6 and 7 plots in post-fire logged and burned respectively. During 2003, we increased sampling to 27, 28, and 22 in post-fire logged, burned, and cut and burned. During 2004, we sampled 19 plots in logged and 22 plots in burned treatments.

4.1.1. Sampling assessment

There were no large differences in sampled plots between years or treatments for elevation (m), fire severity, wetness index, or degrees slope. While the difference between elevation approached statistical significance, effects sizes were minimal (Table 1), with only a 70 m difference between years in mean elevation, and likewise, only a 63 m difference between treatments. Given the much wider range of elevations present within the burn (see study area description, 1500-2350m range), we felt these differences in elevation were trivial. Fire severity and wetness index values did not differ at all between treatments or years (Table 1). While the difference in slope between treatments and years was significant (Table 1), we felt the difference was trivial compared to those available within the burn (0-45 degrees). In 2003, there were also no differences in the distribution of samples across the three treatment types for elevation, fire severity, wetness index and degree slope (Table 2). In conclusion, despite variation in study design between years and treatments, we felt sampling differences were minor enough not to confound our main analyses.

Table 1. Post-hoc assessment of differences in sampling plot locations between year and treatment for continuous covariates elevation (m), fire severity, soil moisture index, and degrees slope in the Dogrib Creek fire, eastern slopes of Alberta, 2002-2004. Shown are results of two-way ANOVA's showing results for overall and main effects, as well as treatment and yearly mean values for each covariate. Bolded SE's indicate significant differences at p=0.05 between categories.

Effect	Elevation (m)	Fire Severity	Wetness	Slope (degrees)
ANOVA	F _{5,101} =2.815	F _{3,69} =1.56	F _{3,103} =1.00	F _{3,111} =11.97
P-value	P=0.039	P=0.206	P=0.39	P<0.0005
R ²	0.065	0.023	0.001	0.24
YEAR	F _{2,103} =2.719	F _{2,69} =2.22	F _{2,103} =0.22	F _{1,101} =14.8
P-value	0.071	P=0.12	P=0.80	P=0.009
Mean (S.E.)				
2002	1736 (52.1)	0.67 (0.085)	10.6 (0.85)	3.3 (2.47)
2003	1668 (31.6)	0.87 (0.048)	10.2 (0.51)	8.1 (1.51)
2004	1738 (29.7)	0.803 (0.035)	10.07 (0.45)	5.1 (1.30)
TREATMENT	F _{1,102} =4.386	F _{1,69} =0.01	F _{1,103} =2.65	F _{1,101} =7.21
P-value	0.04	P=0.92	P=0.11	P=0.009
Estimates				
<i>Burned & Cut</i>	1675 (29.9)	0.72 (0.076)	10.9 (0.45)	1.3 (1.42)
<i>Burned</i>	1738 (27.8)	0.804 (0.035)	10.07 (0.48)	5.1 (1.31)

Table 2. Post-hoc assessment of differences in sampling plot locations between treatments for continuous covariates elevation (m), fire severity, soil moisture index, and degrees slope in the Dogrib Creek fire, eastern slopes of Alberta, 2003. Shown are results of the ANOVA for overall and main effects, as well as treatment and mean values for each covariate.

Effect	Elevation (m)	Fire Severity	Wetness	Slope (degrees)
ANOVA	F _{2,74} =0.960	F _{2,59} =1.42	F _{2,74} =1.33	F _{2,74} =0.114
P-value	P=0.39	P=0.251	P=0.270	P=0.114
R ²	0.001	0.013	0.009	0.031
TREATMENT				
Estimates (S.E.)				
Post-fire logged	1626 (31.5)	0.85 (0.052)	10.7 (0.68)	9.0 (2.07)
Burned	1650 (31.3)	0.89 (0.049)	10.6 (0.68)	12.9 (2.06)
Cut & Burned	1606 (23.4)	0.81 (0.038)	9.68 (0.51)	9.9 (1.54)

4.2. Total herbaceous, forb, grass, and shrub biomass

4.2.1. Plant species composition

Species composition differed slightly between the three treatments, with post-fire logged and burned sites being qualitatively the most similar. For example, burned and logged sites were dominated by *Elymus* spp. and *Epilobium angustifolium*, compared to cut and burned sites, which were dominated by *Calamagrostis* spp. and fireweed (Table 3). Graminoids were most abundant in the cut and burned treatment (Table 3). Post-fire logged and burned sites were more similar than compared to cut and burned treatments, which had greater forbs (e.g. *Petasites* spp. and *Geranium* spp.) more typical of moister site conditions. Finally, *Picea* and *Pinus* seedling cover appeared higher than either other treatment (Table 3).

Table 3. Top 20 species based on average % cover in post-fire logged, burned, and cut and burned treatments.

Rank	Post-fire logged		Burned		Cut and Burned	
1	<i>Elymus</i> spp.	7.35	<i>Elymus</i> spp.	8.00	<i>Calamagrostis</i> spp.	7.05
2	<i>Epilobium</i> spp.	3.64	<i>Epilobium</i> spp.	4.30	<i>Epilobium</i> spp.	6.72
3	<i>Bryophyte</i> spp	2.13	<i>Aster</i> spp.	2.84	<i>Rubus idaeus</i>	3.59
4	<i>Carex</i> spp	0.98	<i>Picea</i> spp.	1.34	<i>Geranium bicknellii</i>	2.77
5	<i>Juniper</i> spp.	0.95	<i>Betula glandulosis</i>	1.06	<i>Elymus</i> spp.	2.76
6	<i>Rubus</i> spp.	0.71	<i>Arnica</i> spp.	1.03	<i>Rosa acicularis</i>	1.10
7	<i>Aster</i> spp.	0.67	<i>Caltha</i> spp.	0.98	<i>Carex</i> spp.	1.05
8	<i>Arnica</i> spp	0.62	<i>Salix</i> spp.	0.93	<i>Aster</i> spp.	0.64
9	<i>Picea</i> spp.	0.56	<i>Rosa acicularis</i>	0.75	<i>Agrostis</i> spp.	0.53
10	<i>Caltha</i> spp.	0.45	<i>Shepherdia canadensis</i>	0.74	<i>Cornus canadensis</i>	0.51
11	<i>Arctostaphylos</i> spp.	0.45	<i>Carex</i> spp.	0.67	<i>Arnica</i> spp.	0.49
12	<i>Geranium bicknellii</i>	0.42	<i>Geranium bicknellii</i>	0.61	<i>Mertensia paniculata</i>	0.45
13	<i>Hedysarum</i> spp.	0.41	<i>Hedysarum</i> spp.	0.60	<i>Equisetum</i> spp.	0.41
14	<i>Rosa acicularis</i>	0.31	<i>Pinus</i> spp.	0.59	<i>Petasites</i> spp.	0.37
15	<i>Linnaea borealis</i>	0.26	<i>Linnaea borealis</i>	0.58	<i>Dracocephalum parv.</i>	0.17
16	<i>Lichen</i> spp.	0.25	<i>Bryophyte</i> spp.	0.47	<i>Populus tremuloides</i>	0.15
17	<i>Corydalis aurea</i>	0.23	<i>Arctostaphylos</i> spp.	0.40	<i>Spirea betulifolia</i>	0.11
18	<i>Potentilla</i> spp.	0.21	<i>Potentilla</i> spp.	0.39	<i>Salix</i> spp.	0.11
19	<i>Shepherdia canadensis</i>	0.18	<i>Dryas</i> spp.	0.38	<i>Streptopus amplex.</i>	0.10
20	<i>Populus tremiloides</i>	0.18	<i>Achillea milliforum</i>	0.34	<i>Taraxacum</i> spp.	0.08

4.2.2. *Herbaceous total biomass*

Year*Treatment: The overall ANCOVA model for total herbaceous biomass explained 47% of the variance and was significant (Table 4). While there was no effect of treatment on herbaceous biomass ($F_{1,99}=1.23$, $P=0.27$), biomass increased over time ($F_{2,99}=39.9$, $P<0.0005$, Table 4, 5, Fig. 6). The only covariate that influenced overall total herbaceous biomass was hillshade, with biomass increasing on more xeric, southwestern facing slopes ($F_{1,99}=3.65$, $P=0.06$, $\beta=0.014$, Table 4, 5). Although there was no significant Year*Treatment interactions, Fig. 9 suggests that the effects of post-fire logging were not constant over time. By the second year post-burn, the burned sites had approximately three times more herbaceous biomass than logged sites, however, by the third year this difference had disappeared.

2003-Treatment: The overall ANCOVA model for treatment differences in 2003 was not statistically significant ($F_{3,52}=2.04$, $P=0.12$), and explained only 12% of the variance in total herbaceous biomass. Fire severity had a negative impact on total herbaceous biomass within 2003 ($F_{1,52}=4.32$, $P=0.04$, $\beta=-3.90$, Table 6, 7). Although, TREATMENT did not have a significant impact on herbaceous biomass (Table 6), both burned/not cut and cut/burned sites had more biomass than post-fire logged areas (Fig. 10).

4.2.3. *Forb biomass*

Year*Treatment – The overall forb ANCOVA model explained 51% ($F_{5,97}=14.62$, $P<0.0005$) of the variance in forb biomass (Table 4). Post-fire logging reduced forb biomass consistently across all three years compared to burned sites ($F_{1,97}=3.29$, $P=0.07$, $\beta=-0.315$, Table 5, Fig. 9), but the differences were significant only in the first two years post-burn. There was no interaction between treatment and year ($F_{2,97}=1.70$, $P=0.19$). Forb biomass increased consistently

over time ($F_{2,97}=30.92$, $P<0.0005$). Aspect was the only covariate retained in the final model, forb biomass declined on north aspects ($\beta=-1.10$, Table 5).

2003-Treatment: - The forb biomass model for 2003 treatment differences was significant ($F_{2,74}=6.00$, $P=0.004$) but only explained 14% of the variance in forb biomass, suggesting substantial unexplained variance. Post-fire logging reduced forb biomass during 2003 compared to both burned and cut and burned treatments ($F_{2,74}=6.00$, $P=0.004$, Post-fire $\beta=-2.08$), which post-hoc tests revealed were not statistically different from each other ($S<BNC=CB$, Table 7, Fig. 10). No covariates were retained in the final model of forb biomass (Table 6).

4.2.4. Graminoid biomass

Year*Treatment – The graminoid model fit well ($F_{6,99}=10.87$, $P<0.0005$) and explained 40% of the variance in graminoid biomass (Table 4). While treatment type did not have a main effect on graminoid biomass by itself ($F_{1,99}=0.18$, $P=0.67$, Table 3), the effect of treatment did interact with year such that post-fire logging reduced graminoid biomass in 2002 and 2003, but graminoid biomass was higher in logged sites during 2004 ($F_{2,99}=3.12$, $P=0.05$). Thus, for the first two years post-burn, burn sites exceeded logged sites in grass biomass, but by the third year logged sites exceeded burn sites (Fig. 9). Again, similar to total biomass, graminoid biomass increased on more xeric steep slopes, as indexed by hillshade ($F_{1,99}=7.88$, $P=0.006$, $\beta=0.022$, Table 4, 5).

2003-Treatment: As with total herbaceous biomass, while the overall model was significant ($F_{3,52}=3.29$, $P=0.03$), the model only explained 14% of the variance in total herbaceous biomass, and total graminoid biomass was not statistically different between any of the three treatments during 2003 ($F_{2,52}=0.18$, $P=0.83$). Regardless of statistical significance, post-fire logging had the lowest graminoid biomass of the three treatments during 2003 (Fig. 10). Of

the covariates, only fire severity was statistically important, with graminoid biomass decreasing on more severely burned sites ($F_{1,52}=9.00$, $P=0.003$, $\beta=-5.88$, Table 4, 5).

4.2.5. Shrub total biomass:

Year*Treatment Shrub biomass was extremely variable among sites and over time in the Dogrib burn ($F_{4,29}=0.48$, $P=0.75$, Fig. 9), with nothing significantly affecting shrub biomass, either between years or across treatments (Table 4, 5, Fig. 9). Despite this variation, shrub biomass did appear to increase over time for both treatments, and there did not appear to be important differences between treatments. We discuss sources of shrub variability below.

2003-Treatment: Similarly, substantial variation in shrub biomass between the three treatments in 2003 masked main or covariate effects (Table 4, 5). While shrub biomass did not differ significantly among treatments ($F_{2,31}=1.74$, $P=0.19$), shrub biomass in post-fire logged and burned sites was generally higher, and more variable in response following fire than the cut and burned treatment (Fig. 10).

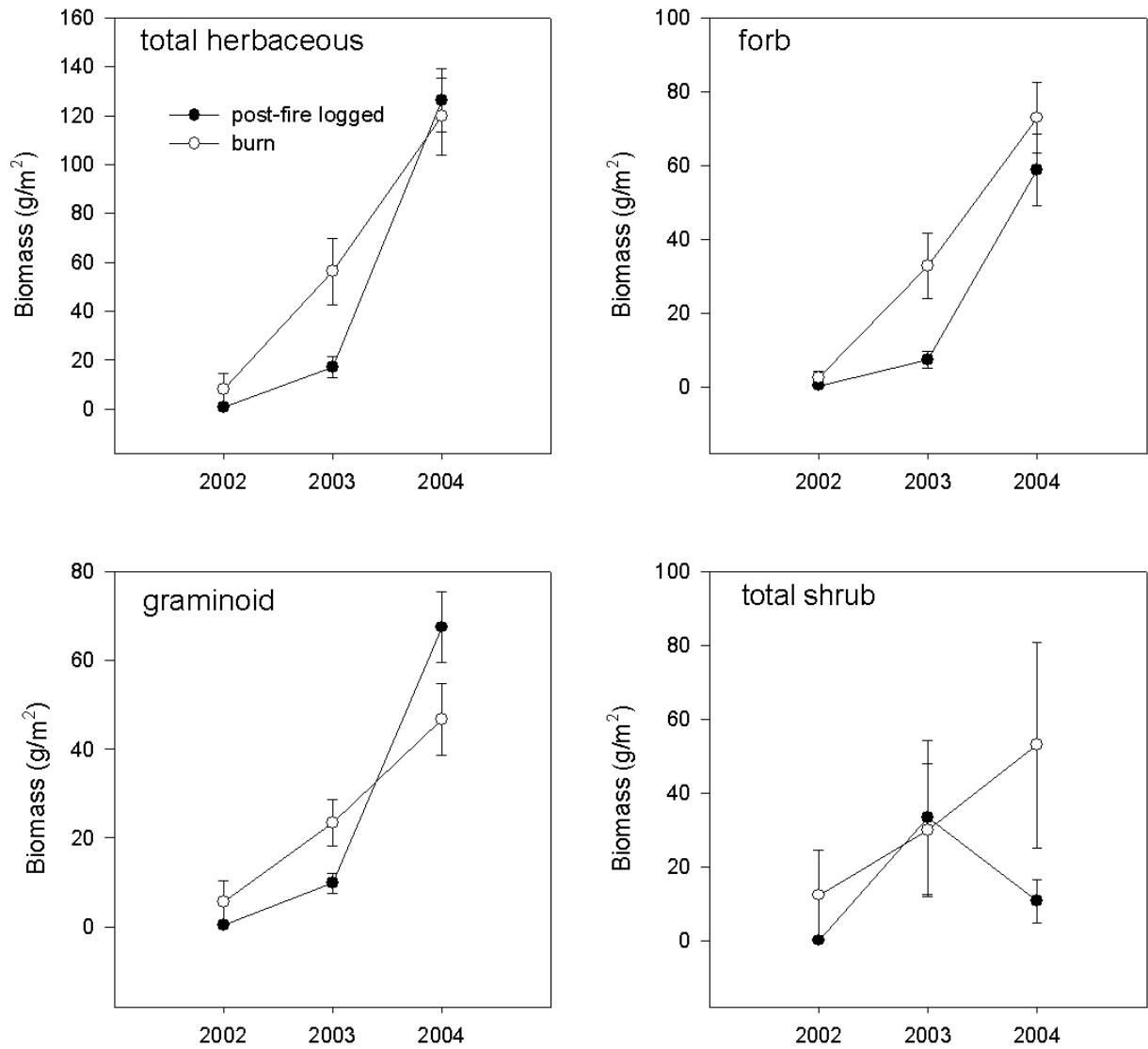


Figure 9. Responses of total, forb, graminoid, and shrub biomass (g/m²) to post-fire logged and burned treatments in the first three years post-burn in the Dogrib Fire, 2002-2004.

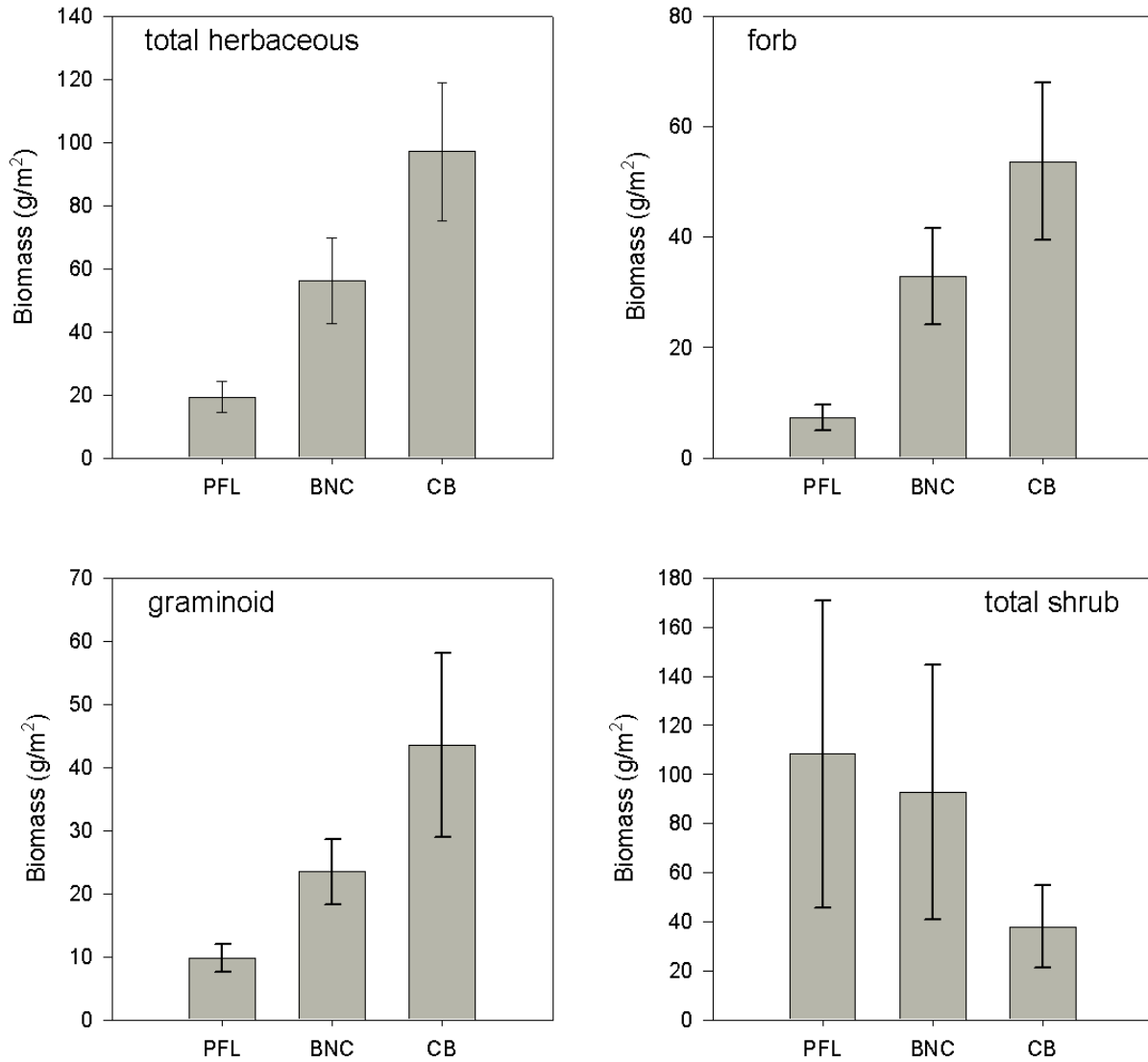


Figure 10. Responses of total, forb, graminoid, and shrub biomass (g/m²) to post-fire logged (salvaged, S), burned (BNC), and cut and burned (CB) treatments in the Dogrib Fire, 2003.

Table 4. Statistical results summary of ANCOVA for effects of TREATMENT (L=post-fire logged, B=burned), YEAR, Aspect, and continuous covariates on 1) graminoid biomass (g/m²), 2) forb biomass (g/m²), 3) total herbaceous biomass (g/m²), 4) total shrub biomass (g/m²), 5) Simpsons diversity (H'), 6) Simpson evenness and 7) total number of species within the Dogrib Creek Burn, east slopes of Alberta, 2002-2004. Bolded cells represent those that were statistically significant at P<0.50, whereas N.S. represent non-significance.

Effect	Total Herbaceous	Forb	Graminoid	Total Shrub	Diversity	Evenness	Species Number
ANOVA Model	F _{6,99} =14.74	F _{5,97} =14.62	F _{6,99} =10.87	F _{4,29} =0.48	F _{6,65} =4.02	F _{5,100} =0.93	F _{6,65} =2.47
P-value	P<0.0005	P<0.0005	P<0.0005	P=0.75	P=0.002	P=0.47	P=0.03
R ²	0.47	0.51	0.40	0.62	0.27	0.04	0.19
TREATMENT	F _{1,99} =1.23 P=0.27 N.S.	F _{1,97} =3.29 P=0.07 L<B	F _{1,99} =0.18 P=0.67 N.S.	F _{1,29} =0.10 P=0.75 N.S.	F _{1,65} =1.65 P=0.20 N.S.	F _{1,100} =0.44 P=0.51 N.S.	F _{1,65} =1.12 P=0.30 N.S.
YEAR	F _{2,99} =39.87 P<0.0005	F _{2,97} =30.92 P<0.0005	F _{2,99} =27.63 P<0.0005	F _{2,29} =0.52 P=0.60	F _{2,65} =2.75 P=0.002	F _{2,100} =0.85 P=0.43	F _{2,65} =1.69 P=0.19
TREATMENT* YEAR	F _{2,99} =1.71 N.S.	F _{2,97} =1.70 N.S.	F _{2,99} =3.12 P=0.05	F _{1,29} =1.08 N.S.	F _{1,65} =0.01 N.S.	F _{1,100} =1.47 N.S.	F _{1,65} =0.12 N.S.
ASPECT	N.S.	F _{2,97} =2.95 P=0.07	N.S.	N.S.	N.S.	N.S.	N.S.
North	N.S.		N.S.	N.S.	N.S.	N.S.	N.S.
South	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.
ELEVATION	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.
FIRE SEVERITY	N.S.	N.S.	N.S.	N.S.	F _{1,65} =3.45 P=0.07	N.S.	F _{1,65} =4.43 P=0.04
DISTLOW	N.S.	N.S.	N.S.	N.S.	F _{1,65} =5.72 P=0.02	N.S.	N.S.
WETNESS	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	F _{1,65} =6.77 P=0.01
HILLSHADE	F _{1,99} =3.65 P=0.06	N.S.	F _{1,99} =7.88 P=0.006	N.S.	N.S.	N.S.	N.S.

Table 5. Parameter estimates from ANOVA/ANCOVA models of the effects of TREATMENT, YEAR (1=2002, 2= 2003, 3= 2004), ASPECT (N=north, S=south), and continuous covariates on 1) graminoid biomass (g/m²), 2) forb biomass (g/m²), 3) total herbaceous biomass (g/m²), 4) total shrub biomass (g/m²), 5) Simpsons diversity (H'), 6) Simpsons evenness (Ddmax), and 7) total number of species within the Dogrib Creek Burn, east slopes of Alberta, 2002-2004.

Parameter	Total Herbaceous	Forb	Graminoid	Total Shrub	Diversity	Evenness	Species Number
Intercept	2.00 (1.42)	4.37 (0.525)	-0.921 (1.49)	3.51 (0.616)	0.797 (0.140)	0.808 (0.092)	5.85 (4.72)
TREATMENT							
Post-fire logged	0.131 (0.571)	-0.315 (0.554)	1.00 (0.599)	-0.871 (0.974)	-0.0670 (0.072)	0.104 (0.133)	-1.49 (1.60)
Burned	Intercept	Intercept	Intercept	Intercept	Intercept	Intercept	Intercept
YEAR							
2002	-4.89 (0.790)	-4.75 (0.807)	-3.72 (0.828)	0.939 (1.95)	-0.367 (0.113)	-0.146 (0.184)	-4.38 (2.43)
2003	-1.66 (0.521)	-1.61 (0.587)	-1.02 (0.546)	-0.088 (0.871)	0.037 (0.062)	0.007 (0.122)	-0.191 (1.36)
2004	Intercept	Intercept	Intercept	Intercept	Intercept	Intercept	Intercept
TREATMENT* YEAR	N.S.	N.S.		N.S.	N.S.	N.S.	N.S.
Post-fire logged *2002	-0.518 (1.27)	N.S.	-1.67 (1.33)	1.33 (1.31)		0.142 (0.296)	
Post-fire logged logged *2003	-1.38 (0.753)	N.S.	-1.94 (0.789)		0.011 (0.097)	-0.242 (0.175)	0.721 (2.13)
ASPECT							
N	N.S.	-1.10 (0.470)	N.S.	N.S.	N.S.	N.S.	N.S.
S	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.
FIRE SEVERITY	N.S.	N.S.	N.S.	N.S.	-0.298 (0.161)	N.S.	-7.45 (3.54)
DISTLOW	N.S.	N.S.	N.S.	N.S.	0.0002 (0.0001)	N.S.	N.S.
WETNESS	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	0.852 (0.328)
HILLSHADE	0.014 (0.007)	N.S.	0.022 (0.008)	N.S.	N.S.	N.S.	N.S.

Table 6. Statistical results summary of ANCOVA for effects of treatment (L=post-fire logged, BNC= burned & not cut, CB=cut & burned), aspect, and continuous covariates on 1) graminoid biomass (g/m²), 2) forb biomass (g/m²), 3) total herbaceous biomass (g/m²), 4) total shrub biomass (g/m²), 5) Simpson diversity 6) Simpson evenness and 7) total number of species within the Dogrib Creek Burn, in 2003.

Effect	Total Herbaceous	Forb	Graminoid	Total Shrub	Diversity	Evenness	Species Number
ANOVA	F _{3,52} =2.04	F _{2,74} =6.00	F _{3,52} =3.29	F _{2,31} =1.74	F _{4,51} =2.32	F _{2,74} =2.54	F _{4,51} =3.74
P-value	P=0.12 *N.S.	P=0.004	P=0.03	P=0.19 *N.S.	P=0.07	P=0.09	P=0.01
R2	0.11 *N.S.	0.14	0.14	0.10 *N.S.	0.15	0.07	0.22
TREATMENT	F _{2,52} =0.74 P=0.48	F _{2,74} =6.00 P=0.004 L<BNC=CB	F _{2,52} =0.18 P=0.83	F _{2,31} =1.74 P=0.19	F _{2,51} =1.19 P=0.31	F _{2,74} =2.54 P=0.09 CB>L=BNC	F _{2,51} =0.74 P=0.49
ELEVATION	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.
FIRE SEVERITY	F _{1,52} =4.32 P=0.04	N.S.	F _{1,52} =9.00 P=0.003	N.S.	F _{1,51} =6.98 P=0.01	N.S.	F _{1,51} =9.93 P=0.03
DISTLOW	N.S.	N.S.	N.S.	N.S.	F _{1,51} =3.03 P=0.09	N.S.	F _{1,51} =3.90 P=0.05

Table 7. Parameter estimates from ANCOVA models of the effects of treatment, aspect and continuous covariates on 1) graminoid biomass (g/m²), 2) forb biomass (g/m²), 3) total herbaceous biomass (g/m²), 4) total shrub biomass (g/m²), 5) Simpson diversity (D-1), 6) Simpson evenness (D/D_{max}), and 7) total number of species within the Dogrib Creek Burn, east slopes of Alberta, 2003.

Parameter	Total Herbaceous	Forb	Graminoid	Total Shrub	Diversity	Evenness	Species Number
Intercept	6.36 (1.59)	2.82 (.471)	6.67 (1.62)	2.62 (0.421)	0.912 (0.144)	0.975 (0.098)	2.23 (0.626)
TREATMENT							
Post-fire logged	-0.80 (0.718)	-2.08 (0.634)	-0.225 (0.731)	1.26 (0.702)	0.027 (0.065)	-0.298 (0.132)	-0.189 (0.187)
Burned and Not Cut	-0.107 (0.671)	-0.558 (0.629)	0.204 (0.683)	0.802 (0.702)	0.090 (0.061)	-0.160 (0.131)	0.007 (0.174)
Cut & Burned	Intercept	Intercept	Intercept	Intercept	Intercept	Intercept	Intercept
FIRE SEVERITY	-3.90 (1.87)	N.S.	-5.88 (1.91)	N.S.	-0.468 (0.177)	N.S.	-1.52 (0.483)
DISTLOW	N.S.	N.S.	N.S.	N.S.	0.0002 (0.0001)	N.S.	0.098 (0.050)

4.3. Diversity, Evenness, and Number of Species

4.3.1. Diversity

Year*Treatment: The ANCOVA model for species diversity was significant ($F_{6,65}=4.02$, $P=0.0002$), and explained 27% of the variance in species diversity. There were no differences between post-fire logged and burned treatments in species diversity ($F_{1,65}=1.65$, $P=0.20$), and no interactive effects of treatment and year ($F_{1,65}=0.01$, $P=0.92$) despite the appearance of post-fire logging having increased species diversity during 2002 compared to burned sites. During 2003 and 2004, diversity was similar for both treatments (Fig. 11). Plant species diversity increased further from low intensity burns ($F_{1,65}=5.72$, $P=0.02$, $b=0.2$, Table 5), but decreased in more severely burned areas ($F_{1,65}=5.72$, $P=0.02$, $b=0.2$, Table 5).

2003-Treatment: The overall ANCOVA model for diversity in 2003 was marginal (Table 6) and explained only 15% of the variance in diversity. While it appeared that burned sites had higher diversity compared to post-fire logged and cut and burned (Fig. 11), these differences were not statistically significant ($F_{2,51}=1.19$, $P=0.31$, Table 6). Similar to the year*treatment ANCOVA, diversity tended to increase further from increasing distance to low intensity burns ($F_{1,51}=3.03$, $P=0.09$, $\beta=0.02$, Table 5), but fire severity had a stronger depressing impact on diversity ($F_{1,51}=6.98$, $P=0.01$, $b=-0.47$, Table 5).

4.3.2. Evenness

Year*Treatment: There were no consistent effects of treatment or year on species evenness in the final model ($F_{5,100}=0.93$, $P=0.47$), suggesting patterns of species dominance were variable over time (Fig. 11). Moreover, no covariates influenced species evenness over the first three years post-burn (Table 4, 5).

2003-Treatment: In contrast to the year*treatment model, during 2003, evenness was moderately influenced by treatment ($F_{2,74}=2.54$, $P=0.09$), with cut and burned having the highest species evenness, and post-fire logged areas the lowest (Table 7, Fig. 11). No other covariates were related to species dominance patterns (Table 6, 7).

4.3.3. *Total number of species:*

Year*Treatment: The number of species did not differ between treatments ($F_{1,100}=0.44$, $P=0.51$) or year ($F_{2,100}=0.43$) in the final ANCOVA model (Fig. 11). However, the total number of species decreased in areas of higher fire severity ($F_{1,65}=4.43$, $P=0.04$, $\beta=-7.45$), and increased in areas with higher soil moisture, indexed by wetness ($F_{1,65}=6.77$, $P=0.01$, $\beta =0.85$).

2003-Treatment: Results were similar for the 2003 treatment model, where species number was the same for all three treatments ($F_{2,51}=0.74$, $P=0.49$, Fig. 11). Species number decreased with increasing fire severity ($F_{1,51}=9.93$, $P=0.03$) and increased further from low intensity areas ($F_{1,51}=3.90$, $P=0.05$).

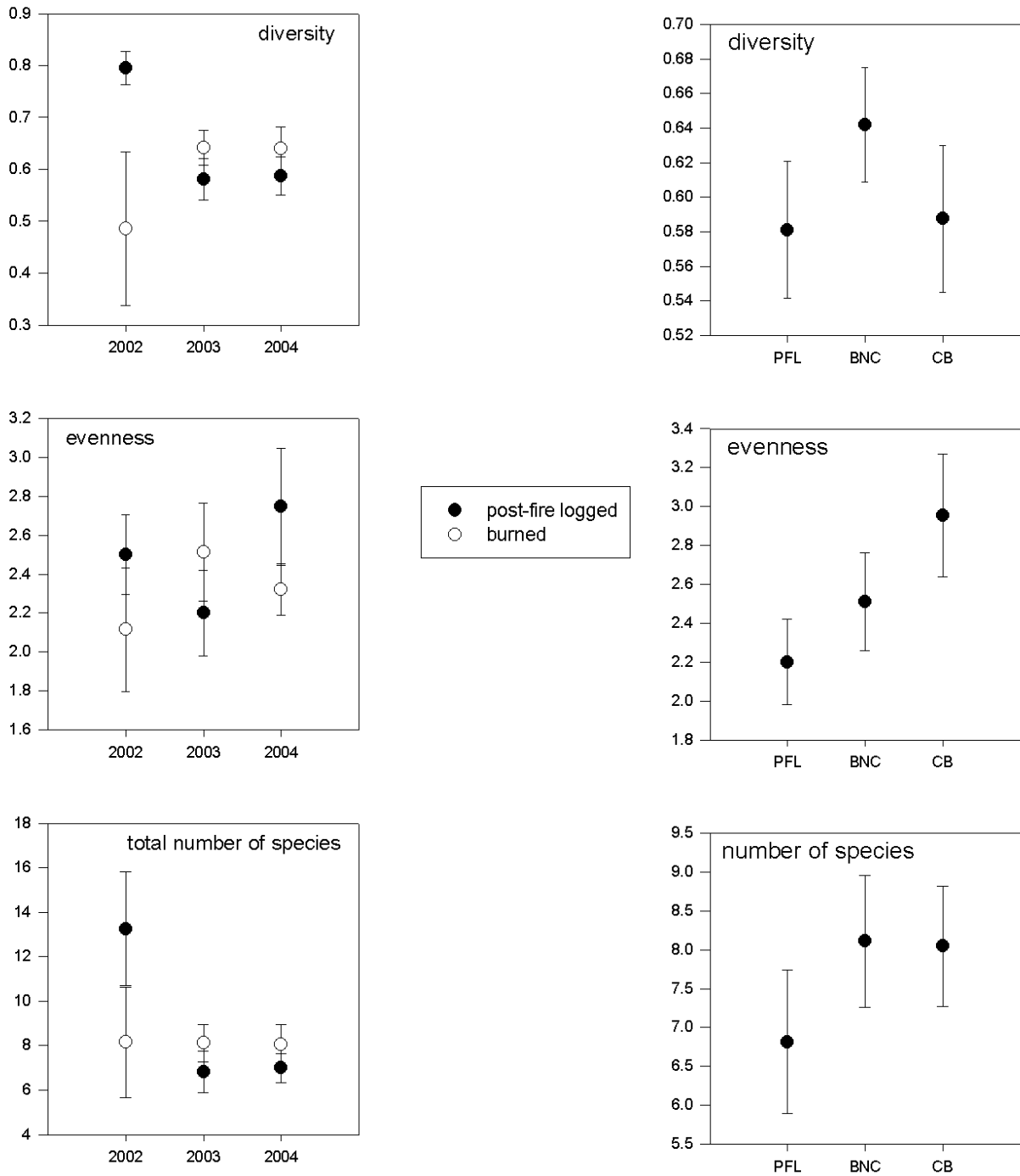


Figure 11. Relationships between diversity, evenness, and total number of plant species a) over time between post-fire logged and burned treatments, and b) between the three treatments, post-fire logged (S), burned (BNC), and cut and burned (CB), during 2003. Shown are means with SE's.

4.4. Ground cover composition

Year*Treatment: Percentage cover of woody debris and bare rock was higher in post-fire logged areas than burned areas during both 2003 and 2004 (Fig. 12), and both decreased from 2003 and 2004. Other differences in total cover, such as reduced forb cover in post-fire logged areas, are consistent with results of ANCOVA models (Table 4, 5, Fig. 6).

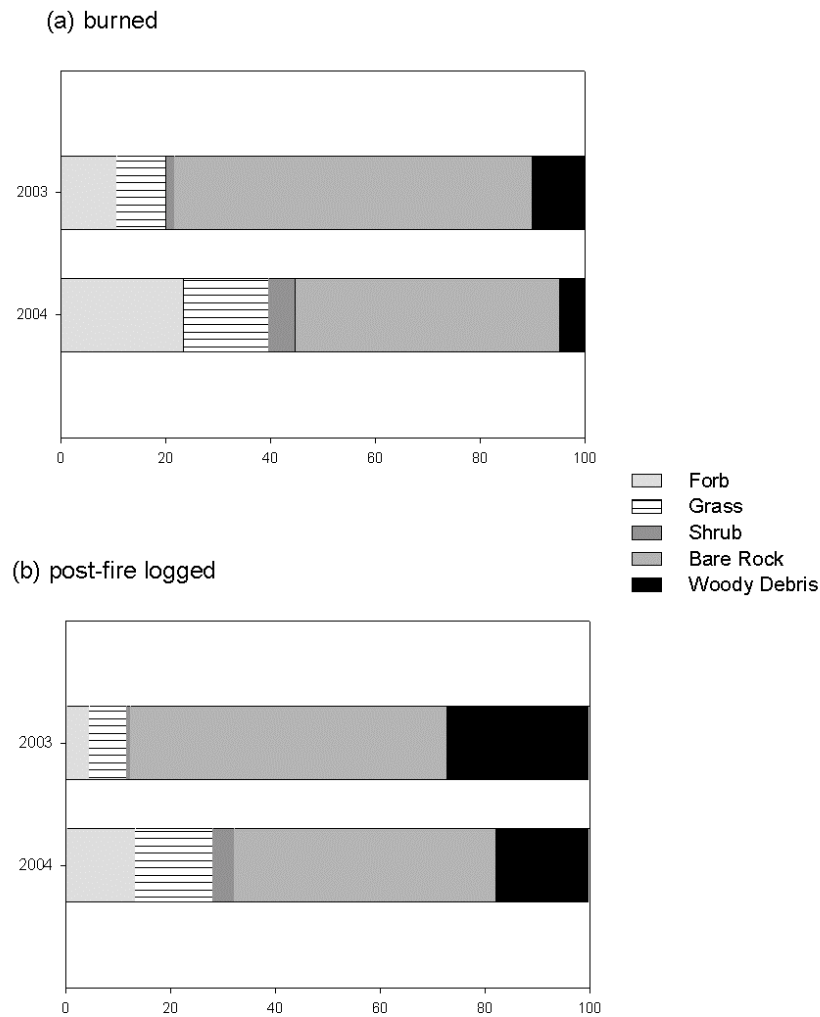


Figure 12. Ground cover composition by class for post-fire logged and burned sites during 2003 and 2004 in the Dogrib burn.

2003-Treatment: The percentage of woody debris cover was highest in post-fire logged areas compared to the other 2 treatments (Fig. 13). Whereas the % of bare ground was similar between post-fire logged and burned areas, but highest in the cut and burned treatments during 2003. Other differences, for example, reduced forb cover in post-fire logged areas, are consistent with ANCOVA models presented above (Table 5, 6, Fig. 7).

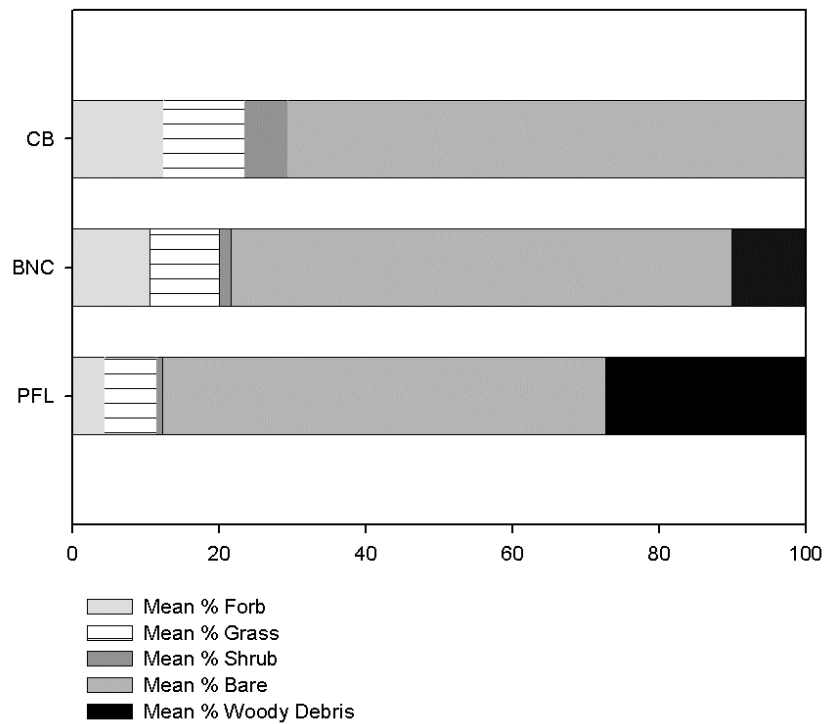


Figure 13. Ground cover in post-fire logged, burned, and cut and burned treatments during 2003 in the Dogrib burn.

4.5. Growing season temperature

We successfully recovered 46 of the 50 deployed iBUTTON thermistors, which recorded 49,139 hourly temperatures, or an average of 1055 temperatures each over the course of 77 days. Temperature differences were slight for the 4 parameters measured, but there were some

important trends. Average daily growing degree-days (GDD) did not differ between treatments ($F_{1,151}=0.53, P=0.46$) with post-fire logged areas showing only a -0.29°C difference compared to burned areas. Total accumulated GDD over the time period was greater for burned areas, 657.8°C , than logged areas, which had a total GDD of 636.1°C . Average maximum daily temperature was 1.1°C warmer in post-fire logged areas regardless of season, but this difference was not statistically significant ($F_{1,151}=1.19, P=0.28$). Average minimum temperature was significantly lower for logged areas in both seasons (average -1.1°C , $F_{1,150}=14.44, P=0.0002$), but the effect was greater during the fall when average minimum temperature in logged areas was -3.3°C lower than burned areas (Treat*Season interaction $F_{1,150}=14.44, P=0.001$). Similarly, average daily temperature range was 1.9°C greater in post-fire logged areas compared to burned areas ($F_{1,151}=3.72, P=0.05$). Summary statistics for iBUTTONS are reported in Table 8.

Table 8. Average daily iBUTTON temperature data for burned and burned and cut (post-fire logged) treatments in the Dogrib Burn, July 1 to Oct 1, 2003.

	GDD	StDev	Max	StDev	Min	StDev	Range	StDev
Treatment								
Burn - Summer	11.8	2.52	36.9	5.64	4.30	2.26	32.6	6.35
Post-fire - Summer	11.5	2.52	38.3	6.42	4.51	2.71	33.8	6.81
Burn - Fall	1.5	2.19	17.7	5.44	0.44	2.46	17.2	5.67
Post-fire - Fall	1.2	4.76	17.8	4.76	-2.83	4.87	20.6	6.77
Burn mean	8.5	5.36	30.9	10.55	3.10	2.93	27.8	9.43
Post-fire mean	8.3	5.32	31.9	11.28	2.22	4.89	29.7	9.14

4.6. Conifer Regeneration

We used 1514 conifer regeneration plots in negative binomial regression with values for all covariates, with sampling focused on post-fire logging treatments ($n=1033$) vs. burned treatments ($n=482$) in the first analysis of stand by treatment effects. Conifer seedling density was strongly affected by stand type, aspect class, and the interaction between stand type and treatment (Table 9). In pine stands and mixed conifer stands, post-fire logging consistently

reduced conifer seedling density, whereas in spruce stands, post-fire logging increased conifer seedling density (Table 9, Fig 14). Conifer regeneration was reduced on north facing aspects compared to flat and south facing slopes. Conifer regeneration was greater in areas of high fire severity, while plots further from unburned and lightly burned forest had reduced conifer regeneration. More xeric sites, indexed by higher hillshade values (south-west facing aspects) decreased conifer regeneration (Table 9). On the ground, increasing percentage of the soil that was exposed mineral soil increased conifer regeneration, while increased grass cover decreased conifer regeneration (Table 9).

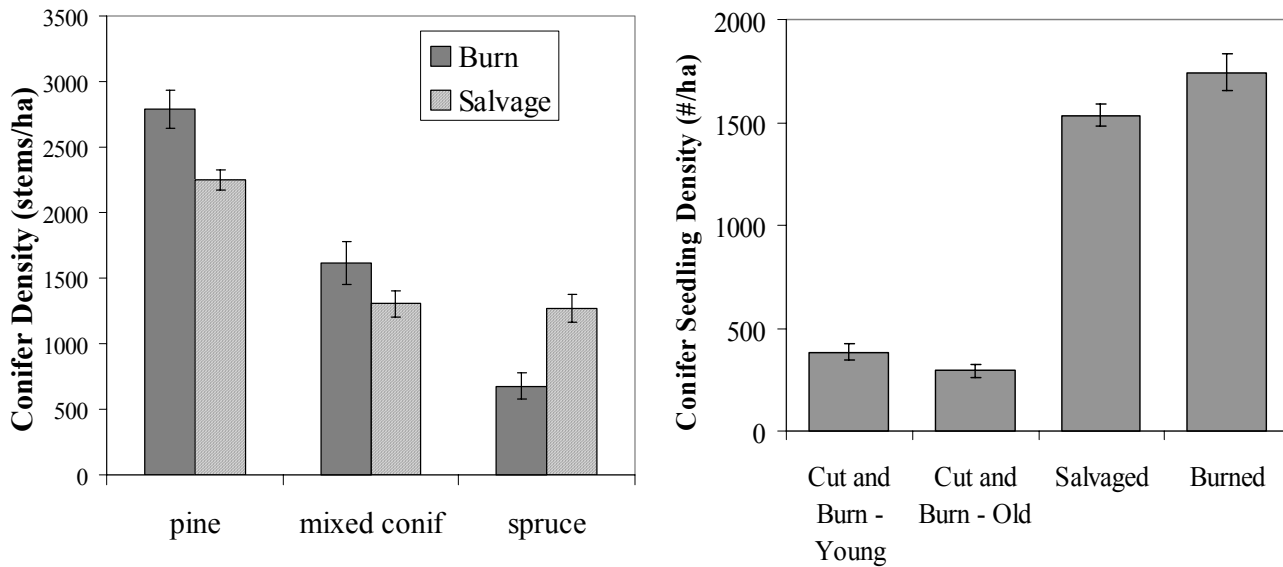


Fig 14. Conifer (*Pinus* spp) seedling density (stems/ha) by a) treatment and stand types, and b) between treatment types including cutblocks burned post-harvest in the lower

Table 9. Negative binomial regression model for conifer seedling count in the Dogrib burn, 2-years post-burn. Shown are parameter estimates and standard errors for all statistically significant parameters retained in the final model.

Parameter	β	SE	Parameter	β	SE
Constant**	1.87	0.411	% Grass Cover	-0.012	0.002
Post-fire logged	-0.18	0.08	% Mineral Soil	0.004	0.001
Mix Conifer	-0.60	0.081	Hillshade	-0.007	0.002
Spruce	-1.33	0.154	Fire Severity	0.80	0.265
Logged * Spruce	0.60	0.190	North Slopes	-0.38	0.093
Distance to Low Severity Burn (km)	-0.58	0.101			

* Note that with indicator coding, the constant represents the reference category Burned Pine on south or flat aspects, against which all parameters are in reference to.

Following the analysis of treatment type by stand type, we examined differences between post-fire logged, burned, and old and young cutblocks that were burned post-harvest in a second analysis for conifer density. We included the same covariates, but because their parameter estimates are essentially identical to those in Table 9, we only show seedling density differences between major treatment types in Fig 14. Holding the effects of stand type constant (e.g., Fig. 14), cutblocks that were cut and then burned afterwards had substantially lower conifer seedling recruitment (effect of cut and burned on seedling density $\beta=-1.85$, $SE=0.10$) than the average for post-fire logged and burned stands. Moreover, cutblock age did not make a difference in conifer seedling density; both older (>5 years) and younger (<5 years) aged cutblocks had similarly low regeneration ($\beta=-0.06$, $SE=0.139$).

4.7. Elk resource selection patterns

4.7.1. Elk diet composition

During summer 2002, elk diet consisted primarily of graminoids (75% of total diet), followed by forbs (18%), and shrubs (7%). Dominant graminoids included *Carex* spp., *Elymus* spp., *Poa* spp., and *Calamagrostis* spp.; dominant forbs included *Epilobium* spp. and *Astragalus*

spp. (Fig. 15). It should be noted that fecal plant fragment analysis is biased against forbs, so it is likely that the actual diet composition of forbs and shrub leaves was higher in the diet (Hobbs 1987).

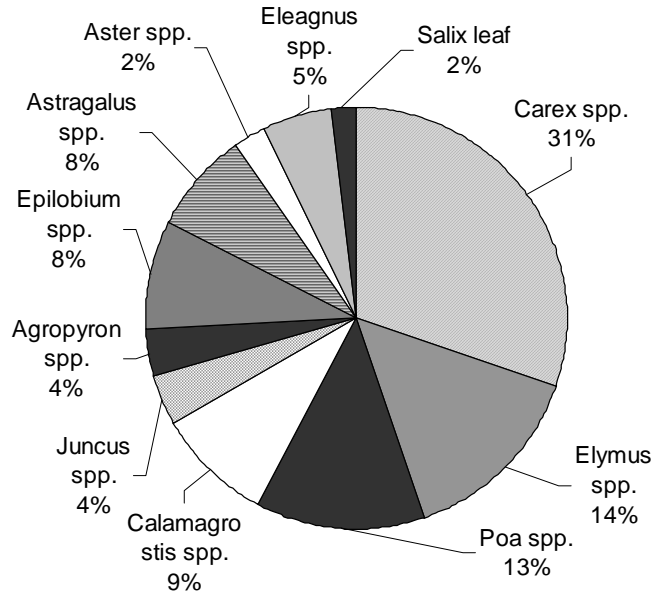


Figure 15. Summer 2002 Elk diet composition within the Dogrib burn, as determined by fecal plant fragment diet composition analysis.

4.7.2. Radiotelemetry

Between March 2002 and Dec 2004, we collected a total of 86 VHF telemetry locations from 21 individual VHF collared elk within the Dogrib burn area (mean=5), and 2068 GPS telemetry locations from 10 GPS collared elk (mean = 206) (Fig. 16). Use by the YHT elk herd was restricted to the western areas of the burn; no radiocollared elk from YHT ventured east of Yara creek (Fig. 16). Thus, because most post-fire logging occurred in the eastern portion of the burn, and Ya Ha Tinda elk occurred mostly in the western zone, our scope of inference for telemetry data is valid only for Ya Ha Tinda elk, and biased against postfire logging treatments. Seasonal use of the Dogrib burn peaked in July, when over 35% of the elk use of the Dogrib burn

occurred (Fig. 17). A small peak in October of seasonal use reflected use by migratory elk from the Panther and Cascade river areas of BNP that remained near the Dogrib burn before completing their migration back to the YHT. Use of the Dogrib burn increased over time during the first three years post burn, as measured by the proportion of total radiomarked elk in the YHT elk herd using the Dogrib burn (Fig. 18). Some caution in interpreting Fig. 18 is warranted because new elk were radiocollared each year; some of the apparent increase may have just been because of sampling error during elk capture. Regardless, it does provide evidence that elk use of the Dogrib burn has increased over the first three years post-burn. Elk from the Ya Ha Tinda selected for burned and uncut treatments, while avoiding both the post-fire logged and cut and burned treatment types (Fig. 19). Avoidance was strongest for the post-fire logged areas, and patterns were similar for both GPS and VHF collared elk.

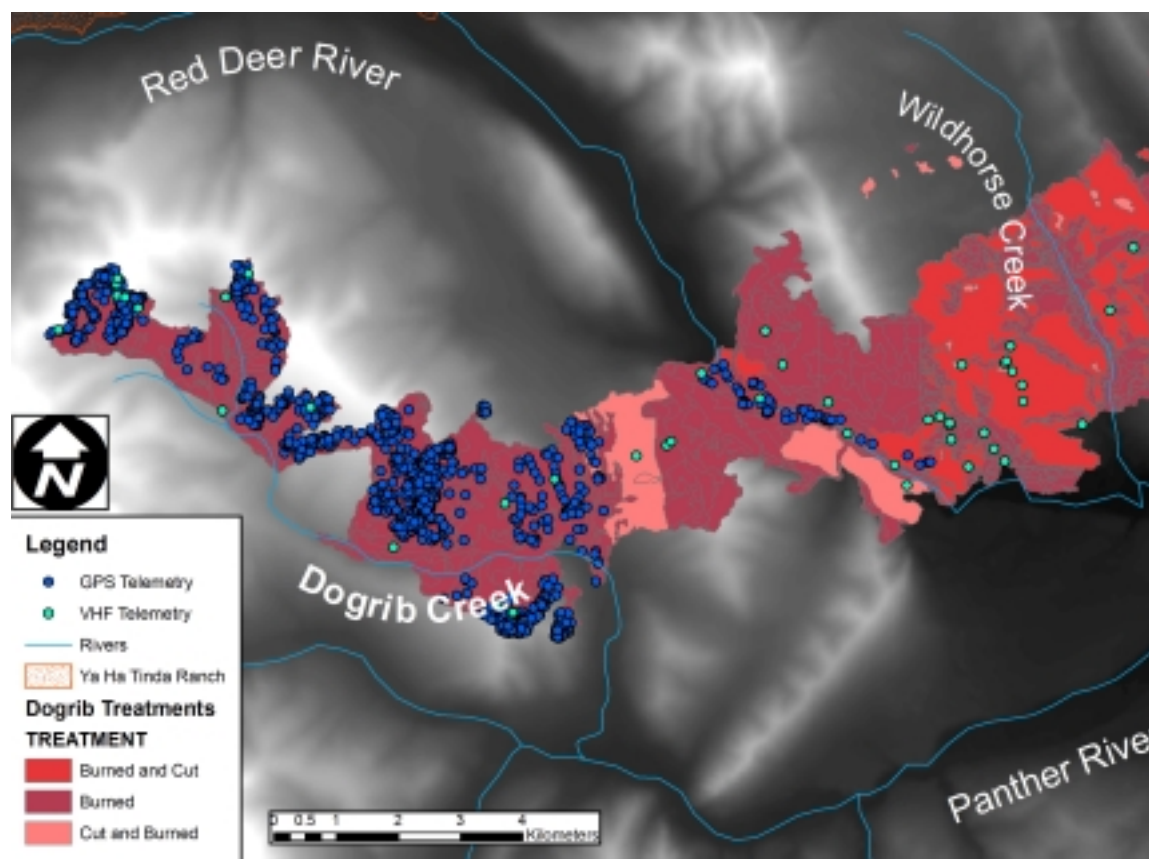


Figure 16. GPS and VHF telemetry locations in the Dogrib burn, Feb 2002 to Dec 2004.

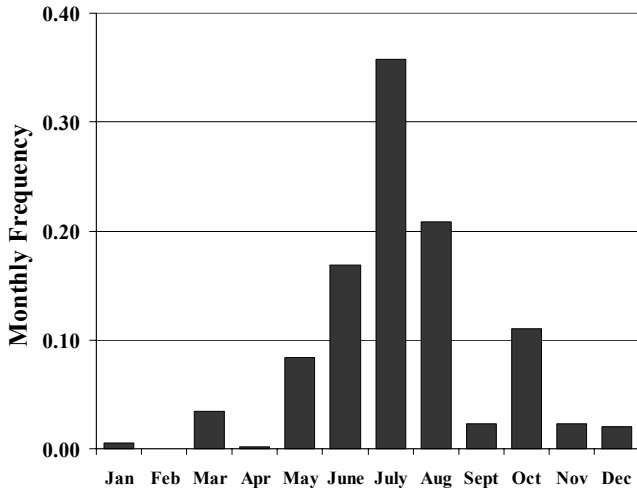


Figure 17. Average monthly frequency of use for each elk, Dogrib burn, Feb 2002 to Dec 2004.

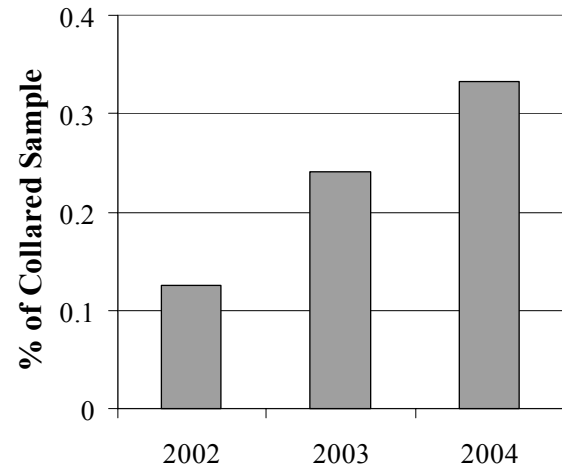


Figure 18. Proportion of the total collared elk using the Dogrib burn.

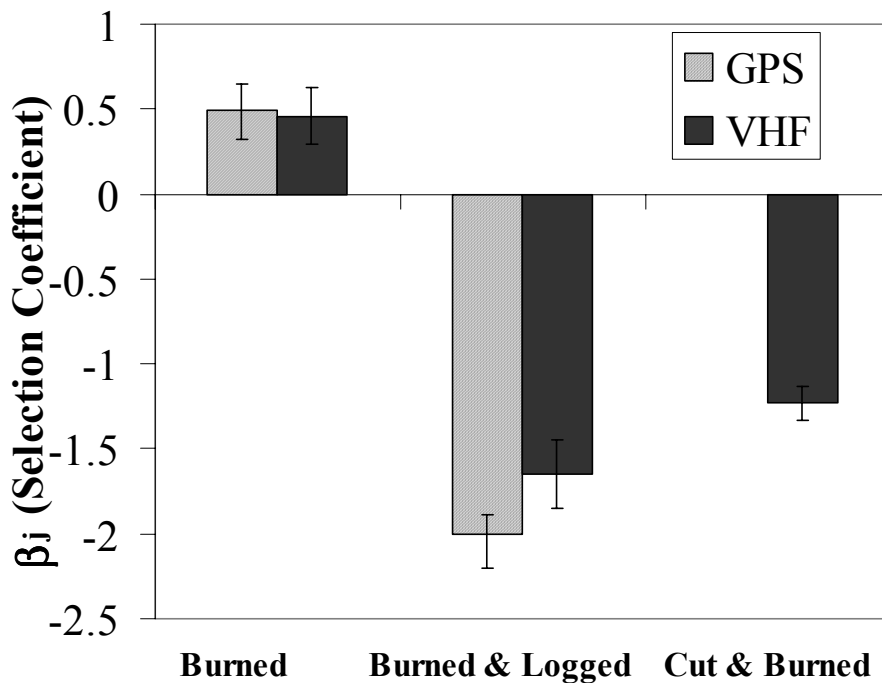


Figure 19. Selection coefficients of VHF and GPS collared elk from the Ya Ha Tinda population for the three post-fire treatment types in the Dogrib burn, March 2002- to Dec 2004. Positive values represent statistical selection for a treatment type, negative values statistical avoidance. Standard error bars are shown. Note, there were no GPS locations within the cut and burned treatment.

4.7.3. Elk pellet groups.

In comparison to the analysis of telemetry data which was restricted inferentially to just the Ya Ha Tinda elk population, pellet surveys were stratified across the entire burn and provided an unbiased estimate of ungulate response to post-fire treatments. Deer were the most abundant ungulate within the Dogrib burn (52% of all pellet groups counted), followed by elk (20%), moose (16%), and other (12%). Other included wild horses, domestic cattle, and bighorn sheep, and we were unable to distinguish between mule deer and white-tailed deer with pellet data. For elk, deer, and moose, negative binomial models fit pellet data better than Poisson or Zero-inflated Poisson models (all Likelihood ratio-test for adequacy over Poisson P -values <0.0001 , and all Vuong test P -values for need for the ZINB model were <0.0005). Backwards model selection resulted in final models for elk, deer, and moose only retaining treatment effects; all other effects, including year and GIS variables were not related to pellet group counts. Results of negative binomial regression models for treatment and species, with standard errors derived from regression models, are shown in Fig. 20. All three species pellet counts were the lowest in the post-fire logged treatment, and this difference was statistically significant (Table 10). Elk pellet counts were zero or low in post-fire logged and burned treatments, but were significantly higher in the cut and burned treatment (Table 10, Fig. 20). Deer pellet abundance was higher than burned and cut treatments in both burned and cut and burned treatments (Table 10, Fig. 20). Finally, moose pellet counts were lower than deer and elk in all treatments, and appeared higher in cut and burned, and burned treatments (Table 10, Fig. 20).

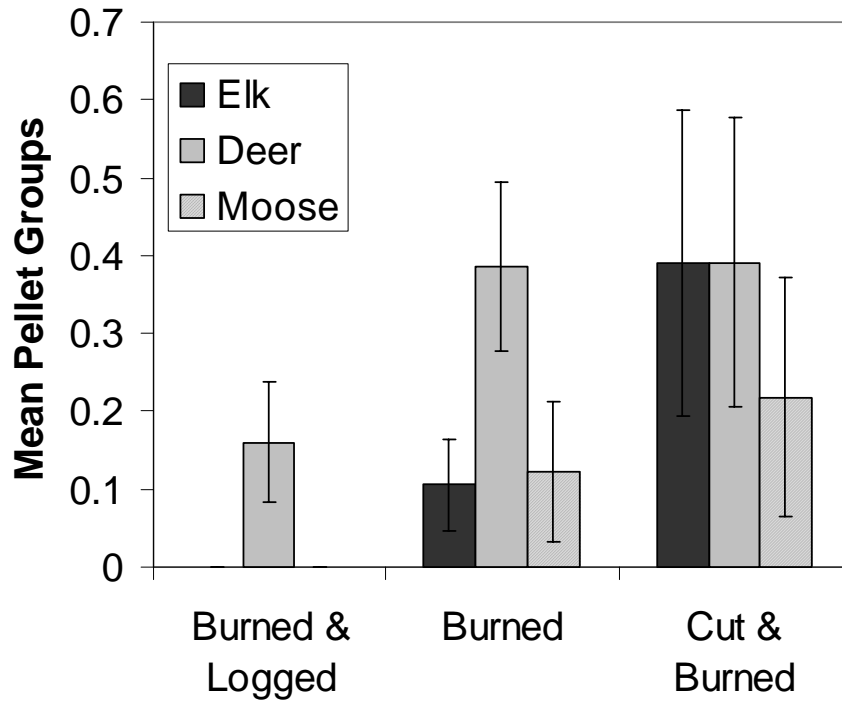


Figure 20. Mean pellet groups counted in the three different treatments, 2002-2004, Dogrib burn. Pellets were counted within a 300m² transect. Standard errors calculated from Negative binomial regression (see text for details).

Table 10. Negative binomial regression parameter estimates for the effects of treatment type on pellet group counts for elk, deer *spp.*, and moose from the Dogrib burn, 2002-2004.

Species	β_0^*	Post-Fire Logged	Cut & Burn	LR χ^2 , P-value	Pseudo-R ²
Elk	-2.25 (0.504)	-1.66 (1.17)	1.32 (0.643)	11.59, P<0.001	0.09
Deer <i>spp.</i>	-0.95 (0.314)	-0.88 (0.423)	0.14 (0.584)	18.72, P<0.005	0.12
Moose	-2.09 (0.730)	N/A	0.57 (1.301)	29.10, P<0.0001	0.09

* Note that β_0 represents the reference treatment burned and uncut.

Bold indicates significant at P=0.10

5. DISCUSSION

In the first three-years post-burn in the Dogrib Fire, post-fire logging had negative, but transient effects on elk forage biomass relative to sites that were only burned. During the first two-years, post-fire logging reduced graminoid and forb biomass by one-third of biomass in sites that were only burned. Forb biomass was reduced in all three years following post-fire logging, although by the third year, the negative effects of logging on forbs had declined considerably. Indeed, by the third year post-burn, graminoid biomass recovered to the point where total herbaceous biomass was equal between post-fire logged and burned treatments. Moreover, effects of post-fire logging on species diversity, evenness, and number of species were weak or transitory such that by the third-year post burn, vegetation community parameters differed little between treatments. Thus, by the third year-post burn, there were few differences between post-fire logged and burned treatments from strictly a vegetation perspective. Of the covariates used to control for variation between sites and treatments, few had consistent or strong effects on forage biomass. Fire severity reduced forb biomass in 2003, diversity and total number of species overall, and increased distance to seed sources in low severity burns reduced diversity. Moreover, there were no major differences between sites in continuous covariates suggesting that our sampling design was adequate despite not being a repeat measures design. Thus, the main differences we observed were due to treatment effects of post-fire logging.

The differences we did observe, including reduced forb biomass, reduced herbaceous biomass during the first two years, reduced diversity, increased evenness, and increased ground cover of woody debris are consistent with the sparse literature on post-fire logging from western North America (Reviewed in McIver & Starr 2000). Post-fire logging reduced herbaceous groundcover in Arizona Ponderosa Pine (*Pinus ponderosa*) stands (Blake 1982). In Idaho, also in

Ponderosa pine, post-fire logging reduced herbaceous biomass and plant species diversity during the first two-years post burn (Sexton 1998). In California Douglas Fir (*Psuedotsuga menziesii*) stands subjected to post-fire logging, the effects of reduced shrub and forb biomass persisted from the first year post burn to 11-years post post-fire (Stuart et al. 1993). Although we found no published studies on post-fire logging in lodgepole pine, these previous studies support our findings of at least short-term negative effects of post-fire logging on herbaceous forage.

We suggest the mechanism of reduced forage, especially for forb biomass, in post-fire logged areas may be a result of 1) higher and more variable temperatures within post-fire-logged sites, 2) significantly reduced summer rainfall during the first two years post burn (Fig.2), and 3) competition for space with woody debris left over after post-fire operations. Although our iBUTTONS revealed a modest temperature difference of +1.1°C in average daily maximum temperature in post-fire logged areas, these differences are perhaps high enough, especially in combination with the 2 drought years that followed the Dogrib burn, to suppress forb biomass production in post-fire logged sites. Forbs are sensitive to reduced moisture levels, increased temperatures, and drought conditions, which reduce forb development in montane regions (Walker et al. 1995). Although we did not measure spring phenology, fall temperatures fluctuated more widely in post-fire logged areas. If this pattern is also observed in the spring in post-fire-logged areas, this could have delayed spring forb production. Thus, unharvested burned trees provided some degree of temperature shading (Amaranthus et al. 1989). Regardless, that herbaceous forage production in post-fire logged areas was equivalent to burned areas during 2004, a normal or slightly wetter summer season, supports the limiting effects of reduced precipitation on herbaceous forage biomass production in the salvage logged areas in combination with the hotter temperatures caused by post-fire logging. Despite the overall

drought conditions, logging still reduced biomass relative to burned stands. Thus, given that future forest managers will not be able to predict the occurrence of droughts following salvage logging, reduced forage biomass following salvage logging during the first 2 years post burn should be expected.

We found that woody debris cover was higher in post-fire logged areas, up to 26% in 2003, than burned areas, and that increased woody debris reduced forage biomass. Thus, an important mechanism for depressed forage biomass is simple competition for growing space in post-fire-logged sites. We did not measure woody debris cover during the first year post fire, but by the second year, it was still 26% in post-fire logged areas. Potential mitigation measures for woody debris competition could be similar to site preparation treatments used in forestry practices. At present, the Alberta government does not prescribe any post-post-fire site preparation treatments, such as mounding, trenching, etc., such as for conventional cutblock logging. Further, work on potential application of site preparations to post-fire logged areas for mitigating negative effects on vegetation and conifer regeneration (see below) should be an important area of applied research. The effects of woody debris are likely to be more severe and long lasting surrounding operational logging landing areas and near the edges of haul roads where harvested wood is processed, and site preparation treatments for these areas should be developed following similar guidelines for conventional forestry.

While our main objectives were focused on comparing burned and post-fire logged areas, our work on previously cut and burned treatments demonstrates the importance of pre-burn condition on post-fire vegetation response (Wright and Bailey 1982). Cut and burned sites had the highest forage biomass during 2003, almost 40g/m² more herbaceous biomass, with most of the difference composed of greater forb biomass in cut and burned sites (Fig. 10). Plant diversity

also was lower, although not statistically significant, in cut and burned sites than unburned sites, and evenness, a measure of species dominance, was greater in cut and burned sites. Fireweed and *Calamagrostis* spp. were dominant in cut and burned sites. Hence, our cut and burned sites were similar to previous studies focusing on post-harvest application of fire to clearcuts showing species dominance is common (McRae et al. 2002).

Despite similarities between post-fire and burned treatments for overall vegetation response, conifer regeneration on post-fire-logged sites was lower than on burned sites, both of which were much higher than on cut and burned sites. The same potential mechanisms related to reduced herbaceous biomass following post-fire logging, i.e., competition with woody debris, could be related to diminished conifer regeneration in post-fire logged areas. Seedling mortality, and subsequent lack of a seed source or seed bank were likely responsible for substantially reduced regeneration in cut and burned sites. Given present stem densities in cut and burned treatments, forest regeneration will be slow and these sites are likely to persist as herbaceous meadows with sparse regeneration. The differences between post-fire logged and burned sites in stem densities/ha amounted to 200 fewer stems/ha because of post-fire logging. However, placing our conifer stem densities in comparison to the literature reveals that overall, conifer stem density within the Dogrib burn was low. For example, in YNP, 5-years post-fire, stem density ranged from 0 to 535,000 stems/ha, with most sites having between 5000 and 10,000 stems/ha (Tinker et al. 1994; Turner et al. 2004). By comparison, our average stems/ha were between 1500 and 2000, on average, for post-fire logged and burned sites. These regeneration levels, assuming 50% seedling mortality, are actually within operational targets for regeneration densities that would not require extensive forest thinning (recommended mature stocking densities of ~900 stems/ha, Barry McElhinney, SFP). However, Sundre Forest Products regeneration surveys only recorded

whether 5 or more conifer seedlings were present in 10m²-plots, imposing a potential negative bias in stem density counts. Regardless of this bias, however, we feel that conifer regeneration densities were still low in the Dogrib burn compared to other burns such as in YNP. Thus, in areas with higher conifer sapling density, the magnitude of post-fire logging on reducing conifer densities may be proportionately higher. In such situations, foresters may have to make trade-offs between increased costs of re-planting post-fire logged areas.

Reasons for overall low conifer regeneration may be related to drought in the first 2-years post-fire or other environmental factors. For example, fire severity positively affects conifer regeneration (Turner et al. 2004), but the majority of the Dogrib burn was high or very high fire severity, thus, regeneration should have been high already, opposite to our findings, and supporting the role of drought. Alternately, low regeneration may be related to low pre-fire serotiny levels in lodgepole pine stands (e.g., Tinker et al. 1994; Schoenagel et al. 2003; Turner et al. 2004). For example, lodgepole pine sapling density was strongly related to pre-fire heterogeneity in pinecone serotiny in YNP, and declined with increasing elevation, but increased with increasing pine stand age (Tinker et al. 1994, Schoenagel et al. 2003). Elsewhere, red squirrel density influenced serotiny through squirrel herbivory on seeds (Benkman et al 2004). These recent studies are among the first research to describe factors related to pre-fire serotiny of lodgepole pine. Other factors, such as soil types or regional climatic trends may also influence pre-fire patterns in serotiny, and hence, conifer regeneration. If both prescribed and wildfires are expected to be an increasing component of Alberta's protected and managed forest landscapes, additional research on landscape controls on lodgepole pine serotiny may help enhance post-fire forest management. If forest managers are interested in such regional level applied research, we encourage field methods that record absolute numbers of regenerating conifers.

Another aspect of post-fire management that may have influenced our results but we did not study was the impact of cattle grazing. In the summer of 2003, in the second year post-burn, ~160 Cattle cow:calf pairs were grazed under lease in the Dogrib burn for ~4 months. Cattle in this area had been usually pastured in the Yara creek grazing allotment. The decision to move the cattle to the Dogrib burn was based in part on the need to reduce grazing at the Yara creek grazing area (Darren Labonte, AB-SRD, personal communication). Although we have no quantitative data on cattle grazing effects on forage biomass, or cattle preferences within the Dogrib burn, our field observations and review of the literature suggest several ways in which cattle grazing might have negative impacts in burned areas (*sensu* Ramirez-Marcial 2003). We observed cattle grazing damaging conifer seedlings through trampling and consumption, damage to plant root systems that were consumed because of loose soil structure, and potential spread of exotic plant species (Ramirez-Marcial 2003). To determine how burned and post-fire logged areas compared to productivity of other Alberta rangelands, we compare our estimates to those of Willoughby(2001). Our estimates for herbaceous biomass in post-fire logged areas were only ~150 KG/ha in the second year, rising to 1270 kg/ha by the third year post burn. Third-year estimates compare favourably with Willoughby (2001)'s estimates for cutblocks in Alberta. Willoughby estimated 724 KG/ha of herbaceous forage biomass in Pine-spruce cutblocks, 1997 KG/ha in overgrazed *Poa-Phleum* cutblocks, 1206 KG/ha in seeded cutblocks with creeping red fescue (*Festuca rubra*), 1656 KG/ha in subalpine fir cutblocks, 1837 KG/ha in Pine-spruce-pinegrass cutblocks, and 1647K KG/ha in Pine – *Elymus* cutblocks. Thus, compared to Willoughby's estimates for older cutblock types, many of which were noted as being overgrazed in his report, post-fire logged areas had average or slightly lower forage production by the third year post-burn. Therefore, we recommend against grazing livestock within burned areas before at

least the third-year post burn because of the potential risks for overgrazing during the sensitive revegetation period following burn. We urge managers to make careful decisions regarding trade-offs between range management benefits and potential negative effects of cattle on conifer regeneration and plant communities following post-fire logging. At the very least, research is needed on this applied subject.

Use of the Dogrib burn by elk and ungulates in general showed some consistent and some apparently contradictory patterns when measured by elk telemetry and pellet groups. Elk (as measured by elk telemetry and elk pellet groups) avoided and occurred the least in the post-fire logged treatment, and deer and moose pellet counts mirrored these results. Higher use and selection of burned uncut stands were also consistent for elk telemetry and pellet groups, and for deer and moose pellets as well. However, telemetry indicated elk avoided cut and burned habitats whereas elk pellet groups indicated elk pellet group counts were highest in cut and burned treatments. Why the inconsistency? A careful consideration of the scope of inference of the pellet and telemetry studies provides an answer. Telemetry on radiocollared adult female elk from the Ya Ha Tinda revealed that elk did not use the lower Dogrib burn (Fig. 16) where most of the cut and burned treatments and indeed post-fire logged areas occurred (Fig. 3). Thus, it was not avoidance per se of the cut and burned and burned and cut treatments, but instead overall avoidance of the areas of the Dogrib burn with many cut and burned areas by elk from Ya Ha Tinda. This reflects the pre-burn distribution of the Ya Ha Tinda elk herd which also avoided the lower Dogrib area (Morgantini 1988). Historically, elk only used the upper Dogrib burn area, and so the avoidance we observed of radiocollared Ya Ha Tinda elk of cut and burned and burned and cut treatments really reflect the larger spatial scale avoidance of the areas of the burn with these treatments. While use of the Dogrib burn by the Ya Ha Tinda elk population has

tended to increase over time (Fig. 15), it remains to be seen whether the Ya Ha Tinda elk will expand to the far northeast corners of the burn where cut and burned areas predominate. Pellet groups, however, were collected from across the Dogrib burn, and are a reliable measure for inferences applied to the entire burn. Thus, the apparent discrepancy in the cut and burned treatment can be explained in terms of whether we are making inferences about just the Ya Ha Tinda elk (who avoided the cut and burned areas) or elk in general that used the entire Dogrib burn. Moreover, inferences that across the entire Dogrib burn, elk, deer, and moose pellet counts were the lowest in post-fire logged areas indicate that avoidance of post-fire logged areas was real for three ungulate species.

Why then did elk (and other ungulates including deer and moose) avoid the burned and cut treatments? By the third year, forage between the post-fire logged and burned treatments was almost equal. While elk may have avoided logged treatments during the first year or two because of vegetation, we found no differences between years in patterns of avoidance of post-fire logged areas (unpublished data). We believe the importance of other factors besides forage is paramount in explaining elk avoidance of post-fire logged areas. Elk preference for cover (Lyon 1983), avoidance of roads (Rowland et al. 2000), and refuge from predation risk by humans and predators (Frair et al. 2005) has been documented for over three decades of research on elk in North America (Rowland et al. 2000; McCorquodale 2000; Toweilli and Thomas 2002; Roloff et al. 2001; Lyon 1983; Holthausen et al. 1994; Frair et al. 2005). Assuming that burned trees also provide some measure of cover for elk, this previous research suggests that avoidance of post-fire logged areas within the Dogrib burn by elk (and other ungulates) occurred because of the absence of hiding cover in large post-fire cutblocks. Previous research also suggests that construction of logging roads to facilitate post-fire harvest may have increased access by human

and other predators within post-fire logged areas. Mean post-fire cutblock size was 19 hectares with a maximum salvage block size of 246 ha (see Appendix B, Fig. B1), twice that of cut and burned areas (Fig. 3). In the adjacent central east slopes of Alberta, elk use of areas close to cover (~30 m on average), either in response to predation risk or disturbance from humans, potentially restricted use of large open cuts to patch edges (Frair et al. 2005). In addition, the management guidelines for salvage logging limit logging to slopes <30 percent slope for hydrological reasons. Because of this, shallow flat slopes will be disproportionately harvested following fire, and large open post-fire cutblocks will more common. Because predation risk for elk declines with increasing elevation and steepness (Frair et al. 2005; Hebblewhite et al. 2005), post-fire logging guidelines may increase predation risk for elk in post-fire logged landscapes. Thus, these same mechanisms creating elk avoidance of areas with low cover could be responsible for avoidance of post-fire logged areas.

Our field observations also support the role of predation factors influencing elk movements and use of the Dogrib burn. In fall of 2002, radiocollared elk #101 and up to 2 other elk were poached within wildlife management unit 417 inside one of the Wildhorse creek post-fire logged areas (in Appendix Fig. B1). Despite aggressive road closure measures by Sundre Forest Products in 2003, once humans get beyond the first 1-kilometer past the road closure, access is relatively unrestricted. Wolf predation also interacts with roads; we found wolves near the Ya Ha Tinda selected to use areas near roads, increasing risk of elk being encountered and subsequently killed by wolves in areas near roads (Hebblewhite et al. 2005), and that total predation risk for elk would be a function of wolf density. The Wildhorse creek area is a centre of activity of the Wildhorse wolf pack (Hebblewhite, unpublished data), which denned during 2002-2004 near the burned areas during summer. High local wolf density would interact with

easier access due to roads (James & Stuart-Smith 2000; James 1999) to cause elk to avoid roads within post-fire logged areas. Similar to the effects of road building associated with post-fire logging operations on water quality, erosion, and overall hydrology (Beschta et al. 2003, McIver and Starr 2000), the most important effects of post-fire logging for ungulates may result more from road building and human access issues. It is important to note that it is not the roads themselves that are potentially being avoided, but the predators (humans and otherwise) that make use of these roads to encounter and kill elk. Thus, a strong policy focus on human access management is needed with enhanced enforcement and increased penalties for violation of access management plans. Without access management, effects of post-fire logging on ungulate species such as elk may be to reduce elk habitat effectiveness.

Of course, any short-term study of a long process such as forest succession following fire has the fundamental drawback of having too short a sampling frame (Bennett & Adams 2004). In our study, we followed post-fire responses for the first three-years post burn. We found limited long-term effects of post-fire logging on elk forage dynamics by the third year post-burn. However, we did find evidence that despite similarity in vegetation, elk and other ungulates responded very differently to post-fire logged and burned forests. While this is obviously a short period to make final conclusions about the effects of post-fire logging on elk and elk habitat, our research makes a substantial contribution to the current limited knowledge of the effects of post-fire logging on wildlife in the literature (McIver & Starr 2000; Nappi et al. 2004). Nevertheless, we recommend continuing to monitor post-fire vegetation responses in the future in the Dogrib burn.

6. SUMMARY AND CONCLUSIONS

Based on our research on the effects of post-fire logging within the Dogrib burn we draw the following conclusions and make the following management recommendations.

- 1) Post-fire logging reduced forage biomass for first 2-years, especially for forbs, following harvest. By the third year post-harvest, however, post-fire logged areas had similar forage biomass as unharvested, burned sites. This increase was mostly due to increase in grass production as forb biomass was still reduced by the third year post-burn.
- 2) A potential mechanism for reduced forage response in the post-fire logged sites was competition for space with woody debris left over from salvage logging operations and because of more variable and hotter maximum temperatures. Temperatures also likely interacted with drought conditions within the salvage logged sites, but salvage logging still reduced forage biomass despite the drought.
- 3) Conifer regeneration by the second year post burn was low in the Dogrib burn compared to other stem densities reported in the literature but within the target range for forestry stocking densities. Post-fire logging reduced conifer regeneration, likely for the same reasons it reduced forage biomass. Conifer regeneration was the lowest in previously cut and burned sites that will likely persist as meadows in the future.
- 4) Despite forage biomass being comparable between burned and burned-and-logged sites following the Dogrib fire, elk use and selection differed between these treatments. Across the whole Dogrib burn, elk selected burned only sites, and strongly avoided post-fire logged sites. These patterns were evident for deer and moose as well. Elk from the Ya Ha Tinda did not appear to shift their distribution eastwards into the

whole burn, instead using areas of the burn that they previously did, although some indication was found that use is increasing of the burned areas by this herd.

- 5) Other factors besides forage availability, such as lack of cover and increased predation risk by humans and predators such as wolves are the most likely explanations for why elk strongly avoided the post-fire logged sites. Roads used for salvage operations likely facilitate human and wolf predation on elk, and the large size of post-fire logged cutblocks leaves larger areas of open habitats that elk are known to avoid because of lack of security cover.

MANAGEMENT RECOMMENDATIONS

- 1) Continue to monitor vegetation response and ungulate use of the Dogrib burn because of the short-term nature of the results of our study over just the first three years post-burn. Repeat sampling of 50-100 of our forage biomass plots/year or perhaps bi-tri-annually would provide a longer-term view of the effects of post-fire harvest on vegetation dynamics and ungulate response.
- 2) Avoid using post-fire logged areas as grazing leases for domestic livestock until at least after the third year post burn based on our comparison of forage biomass levels during the first three years to provincial range assessments conducted for other cutblock types in Alberta (Willoughby 2001). We also recommend applied research into the potential effects of cattle grazing within post-fire logged and burned stands on vegetation dynamics, conifer regeneration, and invasive species.
- 3) Investigate post-harvest site preparation treatment similar to those that occur within conventional cutblocks, such as mounding or trenching, to break up woody debris left following post-fire harvest. Presently, no post-harvest site preparation treatments are

applied to salvage logged areas. Of particular importance will be landing and haul-road side areas.

- 4) Recommend that the absolute number of conifers seedlings and the species be recorded in their post-fire regeneration surveys. A regional study of landscape controls of post-fire serotiny of lodgepole pine stands would aid forest managers in predicting post-fire vegetation dynamics with potential commercial applications.
- 5) Consider applying the same wildlife and elk habitat guidelines for post-fire cutblock layout as for conventional cutblock. For example, maximizing the perimeter to area ratio, varying cutblock boundaries and shape, and leaving patches of burned trees within the cutblocks for cover will help to reduce elk avoidance of these post-fire logged sites based on distance to cover.
- 6) Recommend enhanced road mitigation measures following post-fire logging operations. Our observations of elk avoiding post-fire logged sites could be a direct result of enhanced predation risk by both humans and wolves following road construction for salvage logging. Increased efforts and governmental policy support for road decommissioning, obstruction, and restriction to access will be critical to enhance elk use of post-fire logged areas. Further, government policy support for enhanced enforcement of access management guidelines and policy should be a priority.

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Appendices

Appendix A – FMF Quicknote Communication on Post-fire Logging and Ungulate Habitat Recommendations

Appendix B. Field Photographs of the Dogrib burn and 3 main treatments.

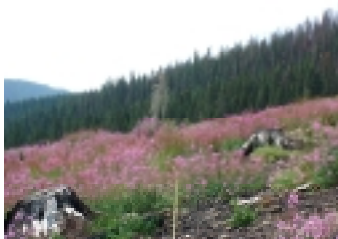
Elk Habitat is More Than Just Forage in Salvaged Burns



Salvaged



Burned



Cut then Burned

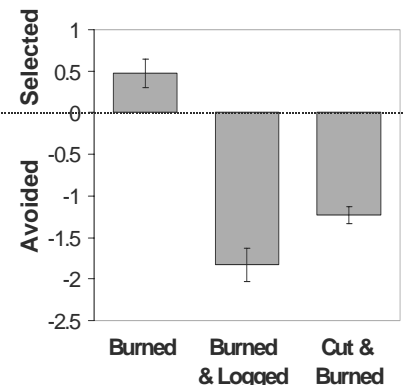
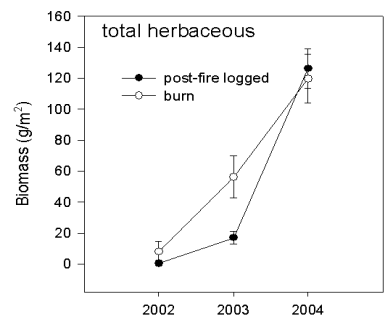
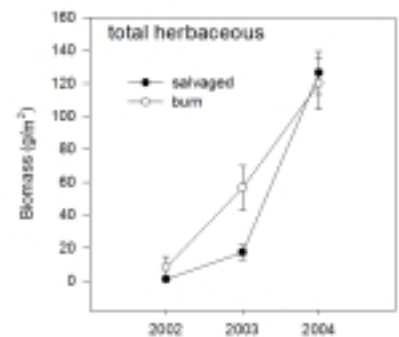
In the Fall of 2001, the Dogrib fire burned 102 km² southeast of the Ya Ha Tinda Ranch, home to one of Alberta's largest elk herds. Following the fire, 25% was salvage logged. Few studies in Alberta have examined how ungulates respond to salvage logging. We examined the effects of salvage logging on elk forage and use in the Dogrib burn during the first 3-years post-burn. We compared forage biomass, plant diversity, conifer regeneration, and ground cover and temperature across 3 treatments: 1) burned, 2) salvage logged, and 3) pre-fire cutblocks that were burned. We examined how elk responded to these treatments using telemetry and pellet counts.

Salvage logging consistently reduced the amount of forbs, but only reduced grasses during the first 2-years compared to the other 2 treatments. By the third year post-fire, grass biomass had increased such that the total amount of herbaceous (grass + forb) forage was equal between burned and salvaged treatments (top figure). Previous cutblocks (CB) that were burned had the highest amount of herbaceous forage (middle figure). Salvage logging reduced species diversity in favour of a few dominant species. Salvage logging slightly reduced pine regeneration, but cutblocks that were burned had the lowest conifer regeneration.

Despite relatively minor and short-term effects of salvage logging on forage, elk (and deer and moose) selected for burned sites and avoided salvaged sites (bottom figure). Ungulates may avoid salvaged sites because of reduced cover and/or increased mortality from humans and wolves. During 2002, several elk were poached in the Dogrib burn from roads. Also, in two other studies in the mountains and foothills region, risk of being killed by wolves increased for elk near roads.

We make the following recommendations from our research:

- 1) *Access management* must be considered when evaluating the effects of salvage logging on elk. If access management does not occur following salvage logging, habitat effectiveness for elk may decline.
- 2) *Cattle grazing* should be delayed until at least the third-year post burn. Forage biomass levels in salvaged logged were lower than other rangelands used for domestic cattle grazing until at least the third-year post-burn.
- 3) *Wildlife habitat guidelines for salvage logging* need to be developed in parallel to guidelines for conventional harvesting, such as retention patches, irregular boundaries to enhance cover, and post-harvest site preparation to reduce woody debris competition.



Appendix B. Field Photographs of the Dogrib burn and 3 main treatments.



Figure B1. Aerial view of the Dogrib burn looking north across Wildhorse meadow with Wildhorse creek on the right hand side. This photo illustrates the large, open, and roaded salvage logged cutblocks that were strongly avoided by elk. This particular cutblock was approximately 246 hectares in size.

Appendix B. Field Photographs of the Dogrib burn and 3 main treatments.



Figure B2. Salvage logged area during the second year post-burn just south of the Ya Ha Tinda road west of Wildhorse creek.

Appendix B. Field Photographs of the Dogrib burn and 3 main treatments.



Figure B3. Previously cut (old, ~ 8 years previous) area burned by the Dogrib fire in the upper Yara creek area of the Dogrib burn. Photo taken during the second year post-burn.

Appendix B. Field Photographs of the Dogrib burn and 3 main treatments.



Figure B4. Burned pine stand unharvested during the second-year post burn.