

SHORT-TERM CHANGES IN VEGETATION AND SOIL IN RESPONSE
TO A BULLDOZED FIRELINE IN NORTHERN
GREAT PLAINS GRASSLANDS

by

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CHAPTER 1

GENERAL INTRODUCTION

Literature ReviewFire Effects on Vegetation

In grassland ecosystems, fire has a significant role in vegetation dynamics (Vogl 1974). Fire is the primary decomposition agent and the key nutrient cyler in many grasslands (Mutch 1970). Fire is facilitated in grasslands because they typically exhibit low growth forms, large areas of continuous vegetation, unrestricted flow of winds, intense insolation, low humidity, low precipitation, and periodic droughts (Vogl 1974).

Fire has a variety of effects on vegetation depending on intensity, severity and season (Wright 1982). Daubenmire (1968) reported from many sources that burning favors forbs, specifically perennial forbs over grasses. In northwestern Montana, a late summer burn caused significant loss of perennial grass cover in fall regrowth primarily due to damaged clumps of native perennial grasses, whereas forbs had greater cover in the burned area, although at the species level forb response was variable; some species increased considerably while others were severely reduced (Antos et al. 1983). After the prescribed burning of fescue grassland, the most common perennial forbs that responded quickly to fire were three-flowered avens (*Geum triflorum*), milk vetch (*Astragalus striatus*), yarrow (*Achillea millefolium*), and pussytoes (*Antennarianitida*) (Bailey 1978).

Fire impacts on vegetation may vary depending on burning season (Lesica et al. 2003). Burning during the spring enhances recruitment of native plants in comparison with fall burning (Cobb et al. 1996). However spring fire is more detrimental to cool-season perennial grass than fall fire. Actively growing rough fescue plants appeared to be more sensitive to fire damage than are dormant plants (Bailey 1978). Late spring or fall fire was more effective at increasing perennial native plant cover without exotic invasion than winter burns in California grassland (Meyer et al. 1999).

Fire Impacts on Soil

Numerous studies from a variety of ecosystems have been published that report on soil response to fire. The effect of fire on soil vary with the properties of the fuel, soil, and fire, especially the fire's frequency, intensity, and timing (Pyne et al. 1996).

Loss of ground cover causes soil erosion after the fire even with a less intense fire. Ground cover facilitates infiltration by impeding overland flow, thereby increasing the frequency and depth of ponding, and by protecting the soil surface from compaction and from sealing effects that can inhibit infiltration (Wilcox et al. 1988; Bryan 2000). Ground cover also protects against detachment and entrainment of soil particles by shielding the soil surface from direct transfer of kinetic energy from raindrops (inter-rill erosive forces) and from shear stress of overland flow (Lane et al. 1997; Bryan 2000).

Another factor besides ground cover that can affect post-fire runoff and sediment yield is soil alteration resulting from fire. In particular, water-repellent soils can develop during fire when organic matter at the soil surface is volatilized (figure 1.1). The volatilized organic matter can move downward as vapor and condense as a hydrophobic

coating on soil particles, thereby reducing infiltration (DeBano 1981). Such reductions in infiltration are thought to change hydrologic response, producing greater than initial runoff from sites where hydrophobicity is a factor than from sites where it is not (Robichaud et al. 2000). Other effects of fire on soil properties include combustion of organic matter (Hester et al. 1997) and reductions in soil aggregate sizes (Emmerich et al. 1994), both of which can affect soil resistance to erosive forces.

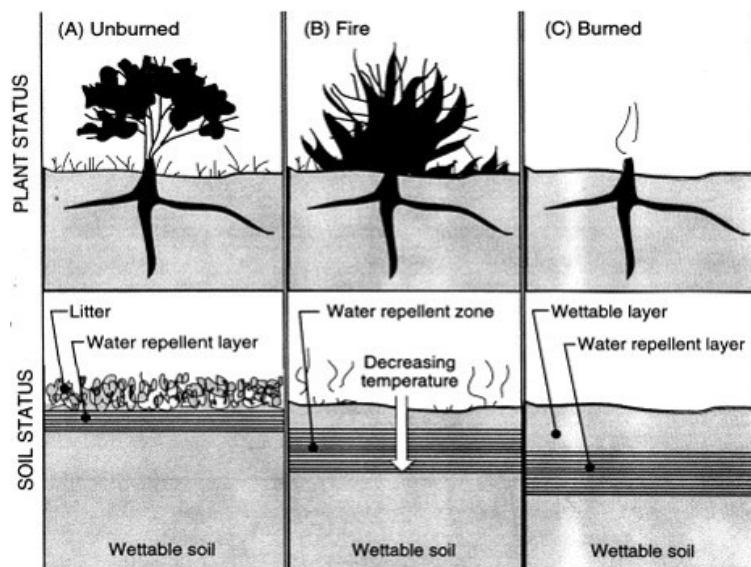


Figure 0.1. Response of water-repellent layer to fire. A) Soil water repellency in unburned brush is found in the litter, duff, and mineral soil layers immediately beneath the shrub plants. (B) When fire burns, hydrophobic substances are vaporized, moving downward along temperature gradients. (C) After the fire has passed, a water repellent layer is present below and parallel to the soil surface on the burned area (adapted from DeBano 1981)

Fire Suppression Impacts on Ecosystem.

Fire is an essential disturbance that cycles nutrients, regulates succession by selecting and regenerating plants, maintains diversity, reduces biomass, triggers and regulates interaction of vegetation and animals and maintains biological and biochemical

processes (Crutzen 1993). Fire exclusion removes the roles of fire from the ecosystem. Native fire regimes create shifting mosaics of patches, processes and habitats (Agee 1993). Fire exclusion increases stem density, biomass and number of woody species (Ogle 1997; Tilman et al. 2000). For example the fescue grassland in central Alberta was almost pure rough fescue but it has virtually disappeared because of either cultivation or tree encroachment (Bailey 1978). Moss (1955) stated that native fire regime had effectively stopped tree advance in the Alberta aspen parkland and predicted that in the absence of fire, succession would soon produce a considerable extension of the aspen (*Populus tremuloides*) forest.

The suppression of fires could result in the loss of native vegetation and favor noxious weeds and non-native species. Removal of fire from the grasslands accumulates fine fuel and aggressive and invasive plants can seize the opportunity to claim habitat and displace native species (Wright 1982). Fire suppression removes the advantage of a competitive edge of the plants that have evolved with fire. A proliferation of weeds leads to a loss of biodiversity. Over time, the loss of native species can lead to monocultures of plants rather than the rich biodiversity that once existed. The new monocultures of weeds may offer little value to wildlife in terms of food (Wright 1982).

In the past decade, the number and severity of wildfires in the western United States has increased and is predicted to continue rising (National Interagency Fire Center 2008). Because of these rising threats of wildfire to natural resources (forest, grassland, water), infrastructure (road, power-lines) and properties (houses and barn), wildfire is suppressed with various techniques depending on site and time specific factors including

weather and fuel conditions. Unfortunately, suppression efforts generally result in disturbances that can generate long-term consequences for natural resources (table 1.1) (Benson et al. 1995).

An increasing amount of literature indicates that every fire suppression activities damage ecosystems to a certain extent. Generally, all the attempts to weigh the costs of fire suppression against potential losses due to fire and fire suppression activity, fail to acknowledge the ecological cost, despite the fact that adverse effects from suppression activities may be substantial and persistent and, in some instances, may exceed impacts attributable to the fires themselves (Mohr 1989; Pyne et al. 1996).

Table 1. Soil impacts associated with fire-suppression activities (adapted from Backer et al. 2004)

Impacts	Potential sources
Soil compaction	Fire camps Firelines Helibases Incident command posts Road construction
Erosion	Firelines Road construction
Nonnative species	Fire camps Firelines Helibases Incident command posts Rehabilitation activities (seed mixes, straw-bale, check dams)
Litter and waste	Fire camps Firelines Extinguished fuses, line explosives, aerial ignition devices Rehabilitation activities
Habitat reduction	Contour-felled logs (rehabilitation activities) Snag removal (mop up activities)
Soil contamination	Fuel spillage

The most common and visible adverse effects of fire suppression activity are associated with soil disturbance. Soil erosion can be initiated or exacerbated by both fires and fire-management activities, including the construction of firelines, temporary roads, and helicopter pads, as well as post-fire rehabilitation activities (Robichaud et al. 2000). Heavy equipment such as tractors, bulldozers, and wheeled skidders produce soil compaction, the extent of which depends on soil type, soil moisture content, and the number of times an area is traversed (Froehlich et al. 1983; Cullen et al. 1991; Reisinger et al. 1992). Soils typically recover from compaction, although recovery times may be as long as 45 years (Reisinger et al. 1992).

A fireline is a break in fuel continuity and these may be constructed with hand tools, heavy equipment, or fireline explosives (Pyne et al. 1996). Firelines are used during initial attack to stop advancing fire fronts and as anchor points for starting a backfire. Effects on the environment also vary with the type of device used to plow the line (Arno 1996). Although manual construction of fire lines provides a useful mechanism to control the width of the line, it requires a well-trained hand-line crew, which is often not available.

Use of a bulldozer is an efficient and faster way to construct a fireline when fire intensity is high and heavy fuel load must be removed (Phillips et al, 1984). Regardless of the methods or equipment used to construct firelines, extensive rehabilitation efforts may be required (Lott 1975). Federal agencies suggest if possible, the use of natural barriers for line construction to minimize the impact of suppression activities (U.S. Fish and Wildlife Service 2002).

Bulldozed fireline refers to strips of exposed bare ground where top soil rich in organic matter “A” horizon is displaced (National Wildfire Coordinating Group 2012). According to the National Wildfire Coordinating Group (2012) bulldozed firelines are limited on unstable soil and steep terrain $\leq 60\%$ slope due to its destructive effect on soil.

Ecological impacts associated with bulldozed firelines may persist long term and promote non-native vegetation invasion (D’Antonio et al. 1999; Young et.al 2002; Merriam 2009,) which is one of the unintended consequence of constructing bulldozed fireline (Merriam et al 2006; Moroney et al 2013). Bulldozed firelines have a large edge effect which often encourages spontaneous seedling recruitment from adjacent communities (Young et al., 1995). If the adjacent vegetation community happens to include highly invasive species such as cheatgrass (*Bromus tectorum*), succession on bulldozed fireline may be truncated by exotic weeds. In the process of fireline construction, bulldozer blade creates disturbed soil which is ideal condition for nonnative invasion (Young et al 2002). In Northern Montana, frequent wildfires, which make more resources available by killing some fire intolerant plants, makes an ecosystem susceptible to cheatgrass invasion. If the fireline is not returned to dominance by desirable perennial species, it will serve as a corridor for the movement of invasive species (Rejmanek 1989; Hobbs 1992). Disturbance corridors often promote invasion of surrounding areas by non-native plants (Tyser et al. 1992, Gelbard et al. 2003). Areas adjacent to the fireline may be susceptible to invasion following the larger scale disturbance such as a subsequent fire (D’Antonio et al. 1990; Brooks et al. 2004; Merriam et al 2006; Moroney et al 2013).

Another consequence of bulldozed firelines is exposed bare ground, possibly with disrupted soil properties. Removal of vegetation cover and litter layer can decrease surface storage and result in decreased infiltration rates, thereby increasing surface runoff (Robichaud et al. 1994). A bulldozed fireline that equaled 1% of the burned area contributed 20% of total erosion in Appalachian region (Christie et al. 2013). Christie et al. (2013) also found that as fireline slopes increased, potential erosion rates also increased. Flat sloped firelines (0-5% slope) contributed 2% of total run off, Intermediate slope (6-29% slope) contributed 33%, and steep (>30% slope) contributed 65% of total run from fireline (Christie et al. 2013).

The majority of the research regarding firelines has concentrated on vegetation impacts while very few studies have addressed soil erosion. In addition most studies of fireline impacts have occurred in forest ecosystems while on grasslands have not received much attention.

Federal and state land management agencies and researchers have been investigating the impacts of fire suppression activities, particularly adverse effects associated with fireline in forest or taiga ecosystems and none of the research has been done on ecological impacts of fireline constructed in grassland ecosystem. Therefore this study seeks to quantify environmental effects of firelines on soil and vegetation attributes in Northern Great Plains grassland ecosystems. Understanding these effects will aid land managers and local land owners in deciding whether to construct a fireline or choose an alternative option.

CHAPTER 2

SHORT-TERM VEGETATION RESPONSE TO BULLDOZED FIRELINE

Contribution of Authors and Co-Authors

Manuscripts in Chapters 2, 3

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Co-Author: Craig A. Carr

Contributions: Helped conceive and implement the study design. Provided field expertise and funding. Provided feedback on drafts of the manuscript.

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SHORT-TERM VEGETATION RESPONSE TO BULLDOZED FIRELINE IN NORTHERN GREAT PLAINS GRASSLANDS

The following work is currently in progress to be submitted for publication

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Abstract

Bladed fire line construction is a fire suppression technique that limits fire spread by altering fuel continuity through vegetation removal and mineral soil exposure. Anecdotal evidence suggests bladed fire lines can cause long-term changes in soil and vegetation properties; however, relatively little research has been performed to corroborate these changes. In this study we compared vegetation properties among burned, unburned, and bladed fire line conditions (treatments) at three locations in north-central Montana and three locations in southwest Montana. Vegetation cover, standing biomass, and functional group abundance were quantified and comparisons made using Analysis of Variance (ANOVA) ($\alpha=0.05$). Perennial grass, shrub, and litter cover and standing biomass were lower on fire lines while annual grasses and annual forbs were higher on firelines in comparison to the other treatment types. Perennial forb cover and standing biomass were similar on all three treatment types with some exceptions. However, by the second year, burned area recovered up to same level with unburned area. These data support the contention that bladed fire lines could cause long-term

ecological change associated with altered successional dynamics associated with substantial increase in dominance on non-native annual species.

Introduction

An increasing amount of research indicates that every fire suppression activity results in some level of ecosystem damage. Generally, in the attempts to weigh the costs of fire suppression against potential losses due to fire and fire suppression activities the ecological cost is often not acknowledged, despite the fact that adverse effects from suppression activities may be substantial and persistent and, in some instances, may exceed impacts attributable to the fires themselves (Mohr 1989; Pyne et al. 1996). The use of a bulldozer is an efficient way to construct a fireline when fire intensity is high and heavy fuel load must be removed (Phillips et al, 1984). A Bulldozed fireline refers to strips of exposed bare ground with displaced topsoil (National Wildfire Coordinating Group 2012). According to National Wildfire Coordinating Group (2012) bulldozed firelines are limited on unstable soil and steep terrain $\leq 60\%$ slope due to its destructive nature.

Ecological impacts associated with bulldozed firelines may persist long-term and promote non-native vegetation invasion (D'Antonio et al. 1999; Young et al. 2002; Merriam 2009) which is one of the unintended consequence of constructing bulldozed firelines (Merriam et al. 2006; Moroney et al. 2013). Exposed bare ground encourages seedling recruitment from adjacent communities (Young et al. 1995). If the adjacent vegetation community happens to include highly invasive species such as cheatgrass, succession on bulldozed fireline may be truncated by exotic weeds. In the construction

process of firelines, the bulldozer blade creates a disturbed soil which is ideal for nonnative invasive (Young et al. 2002). If a fireline is not returned to dominance by desirable perennial species, it may serve as a corridor for the movement of invasive species (Rejmanek 1989; Hobbs 1992) and promote invasion of surrounding areas by non-native plants (Tyser et al. 1992; Gelbard et al. 2003). Areas adjacent to a fireline may also be susceptible to invasion following a larger scale of disturbance such as a subsequent fire (D'Antonio et al. 1990; Brooks et al. 2004; Merriam et al. 2006; Moroney et al. 2013). Fireline impacts on grassland have not been done except in California shrubland, all the studies have been done in forest ecosystem. Thus our study examined the short-term ecological impacts associated with bulldozed firelines in Northern Great Plains grasslands. More specifically, this study focused on determining the vegetation response to bulldozed fireline in comparison with burned and unburned sites. To understand the ecological impacts we investigated vegetation cover and biomass by functional group including perennial grass, perennial forb, annual grass, annual forb, shrub and litter. Specifically we hypothesized that construction of fireline will result a reduction of desired species abundance such as perennial grasses and an increase of exotic non-natives species abundance such as cheatgrass, on bulldozed fireline relative to unburned.

Methods

Site Description

We investigated two firelines in two different locations, Redbluff Agricultural Research and Teaching Ranch, 35 miles from Bozeman and the Thackarey Ranch, 15 miles south of Havre, MT (figure 2.1). Both locations had grassland fires in August 2012 which were suppressed by constructing bulldozed firelines.

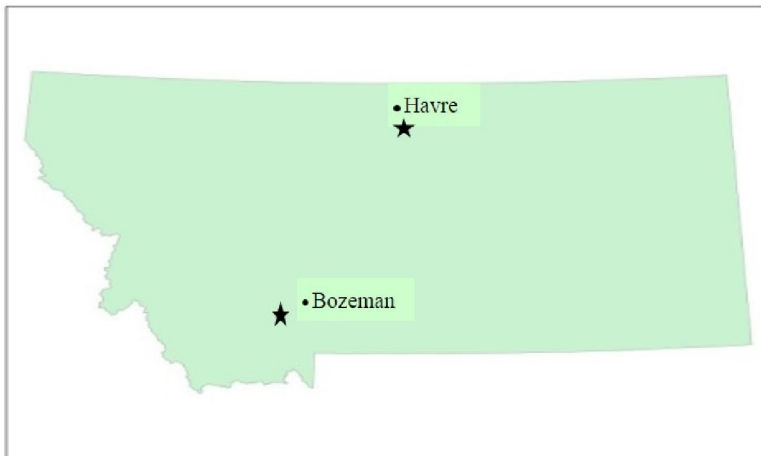


Figure 2.2. Study site locations

The Red Bluff Ranch, a 10,803 acre property, is located near Norris in Madison County, Montana, along the west side of the Madison River. Most of this land is rangeland, with limited hay meadows along the valley bottoms. Elevations range from 4,600 feet to 6,200 feet above sea level.

Soils mapped in the study area include Crago Gravelly loam (USDA NRCS 2011). Soils of these associations are Loamy skeletal, carbonatic, frigid Aridic Calciustepts. The soil surface is often very cobbly, stony, or gravelly, medium textured,

has moderately slow water intake rates, low water holding capacity, rapid runoff, and are well drained (USDA NRCS 2003).

The current dominant plant community is Idaho fescue (*Festuca idahoensis*) with bluebunch wheatgrass (*Elymus spicatum*) and fringed sagebrush (*Artemisia frigida*). Long term temperature and precipitation data were obtained from the Norris weather station (45.4855° N, 111.632° W, 4915ft)(US Climate Data 2014) located approximately 15 km from the study site. The 30-year (1981-2010) mean annual temperature was 10.2°C ranging from a low of 4.2°C and a high of 16.2°C. Mean yearly precipitation was 42.9 cm, and has ranged between 30.4 cm and 52.7 cm over the last 4 years. However, for the first 11 months of 2014, total precipitation was 52.7 cm, while for the same period in 2013, total precipitation was 45.4 cm. Much of the precipitation comes as snow from November to April with local thunderstorms accounting for scattered summer precipitation.

The Thackeray Ranch is located about 17 miles south of Havre in Hill County, Montana, in the Bear's Paw Mountains and comprises approximately 4,000 acres of foothill rangeland. Elevation ranges 3080ft to 6060ft and slope ranges between 25%-70%. Soils mapped in the study area include the Whitelash-Perma-Rock outcrop association (USDA NRCS 2011). Soils of these associations are loamy skeletal, mixed, superactive Typic Haploborolls. Parent materials are weathered from igneous rocks, and soils are very deep and associated with rocky outcrops. The soil surface is often cobbly loam or gravelly loam, medium textured, has moderately slow water intake rates, low water holding capacity, rapid runoff, and are well drained (USDA NRCS 2003).

The current dominant plant community is rough fescue, prairie junegrass, Sandberg bluegrass, Idaho fescue, needleandthread, thickspike or western wheatgrass,

fringed sagewort, and increaser forbs such as arrowleaf balsamroot and goldenpea. There are remnant amounts of some of the late-seral species such as bluebunch wheatgrass, and green/Columbia needlegrass present. In some situations, non-native grasses such as Kentucky bluegrass may also occur, sometimes comprising up to about 50 percent of the vegetation.

Long term temperature and precipitation data were obtained from the Havre weather station (48°54' N, 109°76' W, 789 m); (US Climate Data 2014) located approximately 25 km from the study site. The 30-year (1981-2010) mean annual temperature was 6.36°C ranging from a low of -0.9°C and a high of 13.6°C. Mean yearly precipitation was 28.34 cm and has ranged between 27.7 cm and 48.6 cm over the last 4 years. For the first 11 months of 2014, total precipitation was 27.7cm, while for the same period in 2013, total precipitation was 45.4 cm. Much of the precipitation comes as snow from November to April with local thunderstorms accounting for scattered summer precipitation.

Site Selection

We limited our selection to portions of fireline within a common ecological site to ensure that we could replicate sampling under the same site conditions. According to Natural Resource Conservation Service (NRCS), an ecological site is described as “a distinctive kind of land with specific soil and physical characteristics that differs from other kinds of land in its ability to produce distinctive kinds and amounts of vegetation, and in its ability to respond similarly to management actions and natural disturbances”

(USDA NRCS 2003). We selected three ecological sites at each location. In Havre, our ecological sites were:

- 1). Havre North site (hereafter refer to HN): loamy steep ecological site on the north facing slope with 25% grade, the “A” horizon depth of 21-25cm. Dominant grasses were rough fescue (*Festuca campestris*), bluebunch wheatgrass (*Elymus spicatus*), prairie junegrass (*Koeleria macrantha*) needle and thread (*Heterostipa comata*), Idaho fescue (*Festuca idahoensis*), kentucky bluegrass (*Poa prantesis*). Dominant forbs were goatsbeard (*Tragopogon* species), silky lupine, chickweed (*Stellaria species*), prairie goldenbean (*Thermopsis rhombifolia*), western yarrow (*Achillea millefolium*) and some creeping juniper (*Juniperus horizontalis*).
- 2). Havre West site (hereafter refer to HW): clayey ecological site on the northwest facing slope with 18-20% grade, the “A” horizon depth of 35-40cm. Dominant grasses were kentucky bluegrass (*Poa prantesis*), rough fescue (*Festuca campestris*), bluebunch wheatgrass (*Elymus spicatus*), prairie junegrass (*Koeleria macrantha*) needle and thread (*Heterostipa comata*). Dominant forbs were goatsbeard (*Tragopogon* species), silky lupine, chickweed (*Stellaria species*), prairie goldenbean (*Thermopsis rhombifolia*), western yarrow (*Achillea millefolium*) and some snowberry(*Symphoricarpos albus*).
- 3). Havre South site (hereafter refer to HS): shallow clayey ecological site on the north facing slope with 22% grade, the “A” horizon depth of 15-20cm. Dominant grasses were idaho fescue (*Festuca idahoensis*),bluebunch wheatgrass (*Elymus spicatus*), rough fescue (*Festuca campestris*), prairie junegrass (*Koeleria macrantha*) needle and thread (*Heterostipa comata*),kentucky bluegrass (*Poa prantesis*). Dominant forbs were

goatsbeard (*Tragopogon* species), silky lupine, chickweed (*Stellaria* species), prairie goldenbean (*Thermopsis rhombifolia*), arrowleaf balsamroot (*Balsamorhiza sagittata*), western yarrow (*Achillea millefolium*) and some prairie wild rose (*Rosa arkansana*).

We established 3 replicate plots within each ecological site and each treatment (unburned, bladed and burned) for a total of 54 plots in two locations. Plots were 50-150 meters apart away from each other to insure independence between replicates. Plot sizes were 16.7m x 3m in Redbluff and 16m x 4m in Havre. Plots on unburned and burned treatments were established 2 to 10 meter apart from fireline and parallel to it (figure 2.2). Plot widths were dictated by the fireline width. In 2013, we collected vegetation data from only 2 sites in Redbluff because cattle were put on RBE site before we were able to collect the data. However, in 2014, we collected data from all three sites in Redbluff. All soil data was collected from all 3 sites both in 2013 and 2014 from Redbluff.

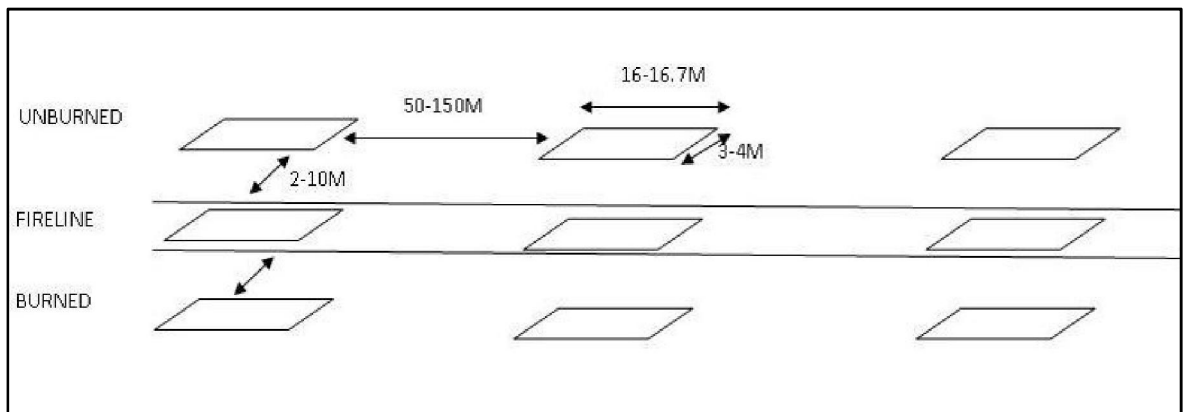


Figure 2.3. Study design for an ecological site

Vegetation Assessment

Percent Cover. Plant community composition, foliar cover, and basal cover were quantified on each treatment using line-point intercept (Floyd 1987; Herrick et al. 2005). To evaluate the number of sample points required for this study, a sample size calculation based on estimating dominant perennial plant foliar cover was performed. Three transects with points every 30cm were determined to be the minimum sample size, total of 165 and 159 points per plot in Redbluff and Havre respectively. The first transect was randomly placed within the fireline and the remaining two transects placed 1m and 1.3m apart from each other in Redbluff and Havre, respectively. The initial transect placement was selected based on a random location within the first 100 cm of fireline width at Redbluff and 130cm at Havre.

Biomass. Biomass clipping was done to determine aboveground biomass production for both 2013 and 2014 (Pieper 1988). We clipped three 50cmx50cm frames per transect for a total of 9 frames per plot in the summer 2013 in Redbluff. To reduce the labor requirements, we reduced the number of frames per plot to six frames per plot at Havre in 2013 and 2014 and in Redbluff in 2014. Systematic randomization technique was employed to place our clipping frames along the transects. The first frame was placed on randomly selected distance between 1 to 8m along the transect and second frame was located on 8 m interval. Frames were placed 20cm apart from and parallel to transects. Aboveground biomass was sorted by functional group, such as annual grass, annual forb, perennial grass and perennial forb, shrub and litter, and stored in paper bags. Plant biomass samples were dried at 65⁰C for 48 hours and then weighed.

Data Analysis

If certain assumptions are met, a normal linear model can be used for examining evidence that at least one treatment has a different mean parameter than the others (Ramsey 2012). If we consider the three treatments burned, unburned and bladed as three groups, we can test the null hypothesis that the mean value for a certain parameter for the three treatments after accounting for site and year are equal versus the alternative hypothesis that at least one is different from the others. Under the null hypothesis, the mean value for the certain parameter for each treatment is the same and our equal-means and separate-means model should explain approximately the same amount of variation after accounting for site and year. The full model with separate means among treatments can be written as:

$$\mu\{Y_i/Treatment, Site, Year\} = \beta_0 + \beta_1 BU_i + \beta_2 UB_i + \beta_3 Site2_i + \beta_4 Site3_i + \beta_5 Site4_i + \beta_6 Site5_i + \beta_7 Site6_i + \beta_8 Year2014_i$$

Where BU_i and UB_i are indicator variables for observations in burned and unburned treatment and the $Site_i$ variables are indicators for observations in the different ecological sites. Site 2 and 3 are located in Havre and site 4 and 5 is located in Redbluff. Note that this model is using bladed fireline as the reference treatment and site 1 as the reference for all sites as well as year 2013. The normal linear model requires three assumptions (Ramsey 2012):

1. Residuals are independent. This will be the assumption to be most carefully controlled. Our model accounts for sites and nesting of treatments within each site, but some spatial correlation may exist if sites closer together are more similar than sites

farther apart. This may be present but it would also be difficult to quantify because of the small number of sites in the study. If positive correlation is present then the p-values of our analysis may be too small and the confidence intervals too narrow. However, since we have a small number of sites there is not much that can be done to check this assumption.

2. The residuals have constant variance. This can be checked by looking for patterns in the residuals versus a fitted values plot. If it is found that the residuals have a fanning shape, one remedy is to apply a log or square root transformation to the response variable which can help equalize the spread of the residuals. Variation between treatments were high, ranging between zero to thousand kg for biomass data and 0 to 100 percent for cover. The proper transformation was taking the square roots.

3. The residuals are approximately normally distributed. This can be checked with a normal QQ plot. If we have a problem with our constant variance assumption then we will most likely have a problem assuming normality. A log transformation on the response variable can sometimes remedy both situations depending on the spread of the residuals. Plotting a normal QQ plot with randomly generated standard normal observations with the same sample size as our data can provide a way to compare how far our residuals deviate from truly normal residuals.

Result

Biomass

Perennial Grass. Perennial grass production on bladed site was significantly lower relative to burned (p -value=0.0001) and unburned sites (p -value=0.0001), while perennial grass production was similar on burned and unburned sites (p -value=0.96) in both 2013 and 2014 (Figure 2.3). Perennial grass production increased in 2014 relative to 2013 (p =0.02). Only significant difference between years was found in the burned treatment (p -value=0.001) (Table 2.1).

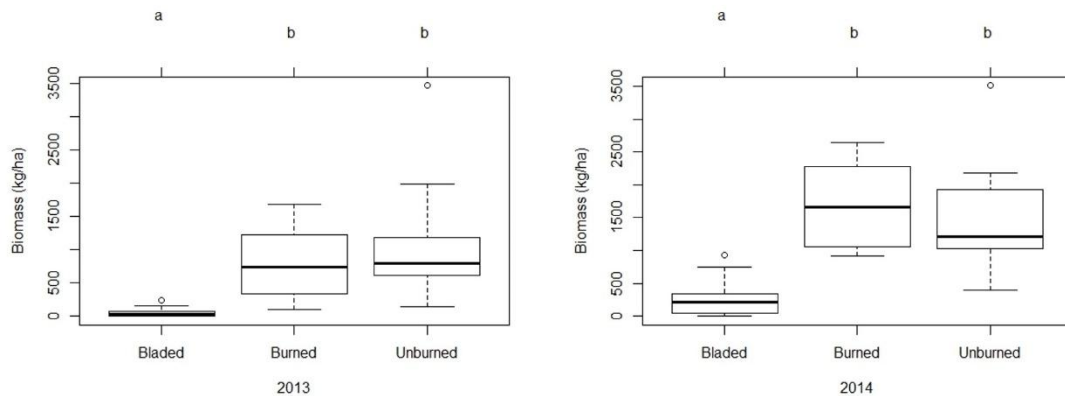


Figure 2.4. Perennial grass production year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

Table 2. Mean perennial grass production in year 2013 and year 2014 (kg/ha). (*- indicates significant difference between years)

Treatment	Year of 2013	Year of 2014
Bladed	60	266
Burned	808	1671* ($p=0.001$)
Unburned	1048	1450

The bulldozed fireline impact on perennial grass production was the greatest on all the study sites and across both years. Perennial grass production was always lowest on bladed treatment. However treatment effects were not consistent across all the study sites. Construction of fireline substantially reduced perennial grass on all study sites while burning generated both increases and decreases. There was no clear pattern of burning effect on perennial grass biomass one year after the fire. By the second year, perennial grass production on fireline remained significantly lower, while burned sites had higher or equal production than unburned treatment on all sites except HW relative to unburned, but not statistically significant (Figure 2.4).

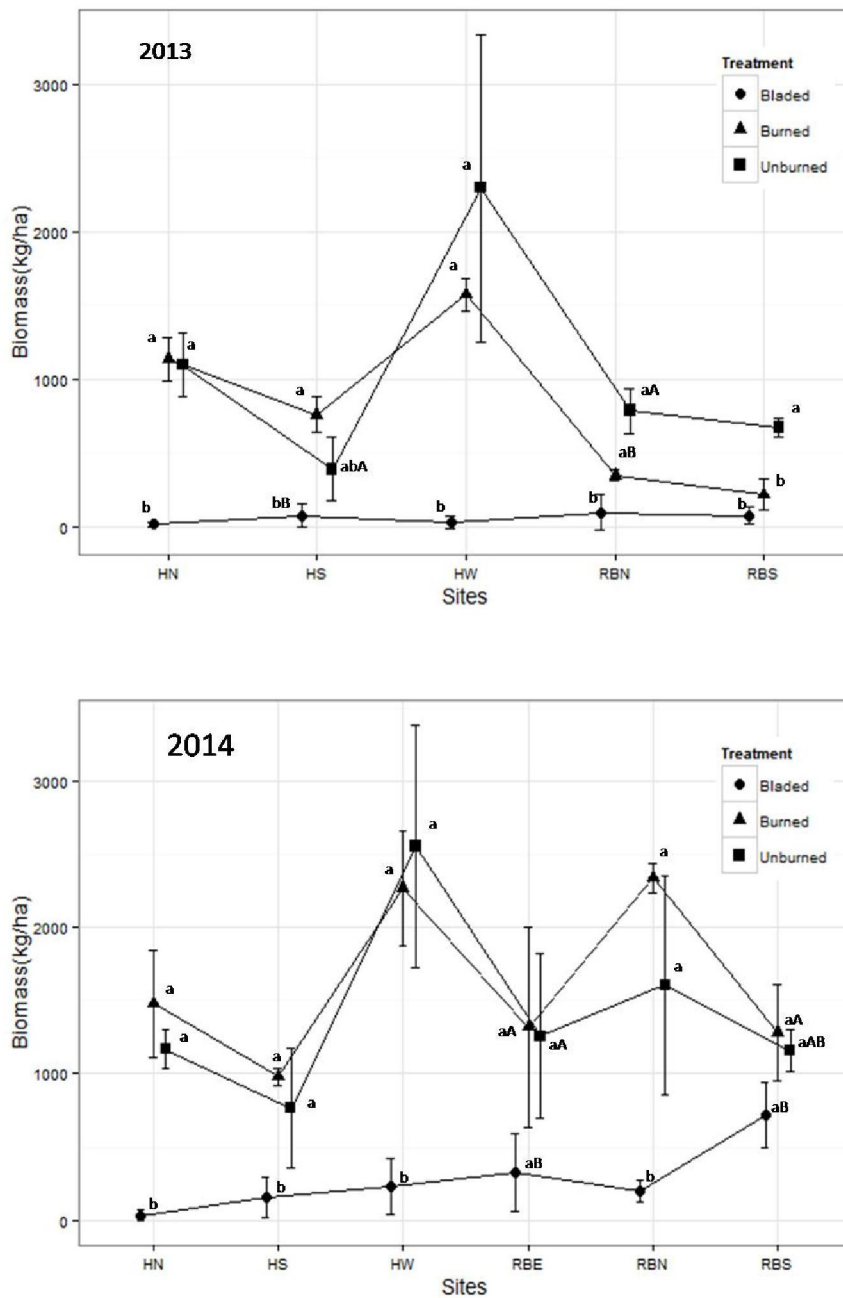


Figure 2.5. Perennial grass production by site in 2013 (above) and 2014 (below). Different lower case letters indicate significant difference at $p \leq 0.05$ and different upper case letters indicate difference at $p \leq 0.1$ within sites

Perennial Forb. Pairwise comparison results from Tukey Honest Significant

Difference (HSD) test performed on ANOVA model of perennial forb production detected significant higher production on burned relative to bladed (p -value=0.033); and year 2014 relative to 2013 (p -value=0.007). It detected higher production on unburned relative to bladed (p -value=0.2); and burned relative to unburned (p -value=0.54). Treatments were not statistically significantly different from each other within a year both in 2013 and 2014 (figure 2.5).

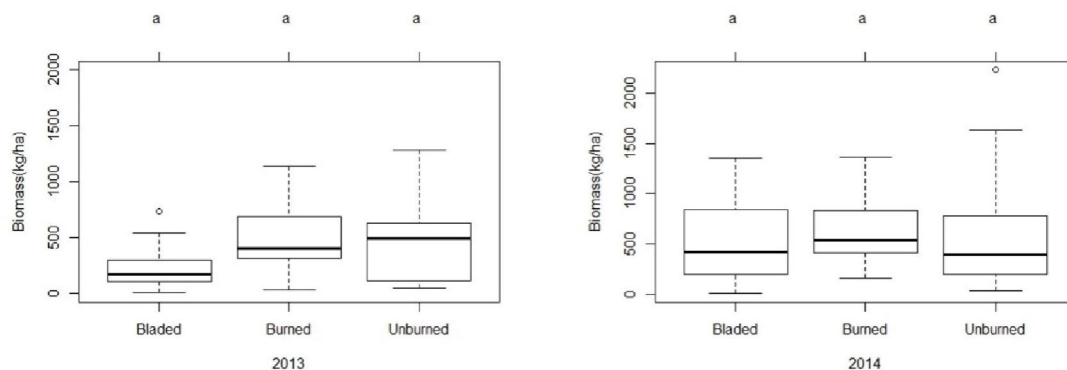


Figure 2.6. Perennial forb production of year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

One year after the fire, bulldozed fireline impact on perennial forb production was the greatest but not significant except HW. On RBS site perennial forb production was greatest on bladed and followed by burned and unburned. However treatment effects were not consistent across all the study sites. Construction of fireline removed perennial forbs on most of our study sites while burning did not change perennial forb production. Two years after the fire, perennial forb production on fireline was the highest on RBE and RBS, rest stayed lower or equal relative to burned. Although, none of them were statistically significant (figure 2.6).

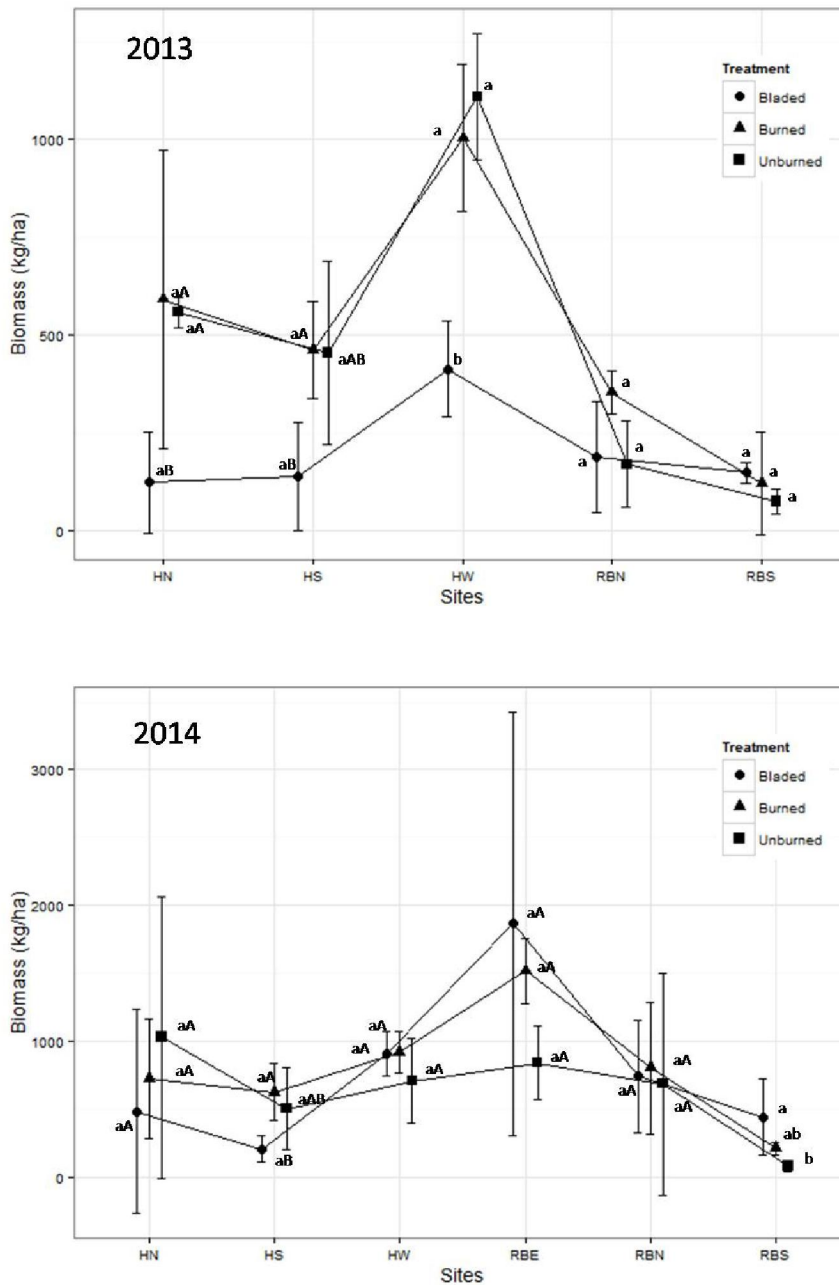


Figure 2.7. Perennial forb production by site in 2013 (above) and 2014 (below). Different lower case letters indicate significant difference at $p \leq 0.05$ and different upper case letters indicate difference at $p \leq 0.1$ within sites

Annual Grass. Pairwise comparison results from Tukey Honest Significant

Difference (HSD) test performed on ANOVA model of annual grass production detected

significant higher biomass on bladed relative to unburned (p -value=0.00001) and burned (p -value=0.00001); and significant higher in year 2014 relative to 2013 (p -value=0.03). It failed to detect significant difference between unburned and burned (p -value=0.7). Results from treatment comparison within a year also showed difference only in bladed treatment (figure 2.7).

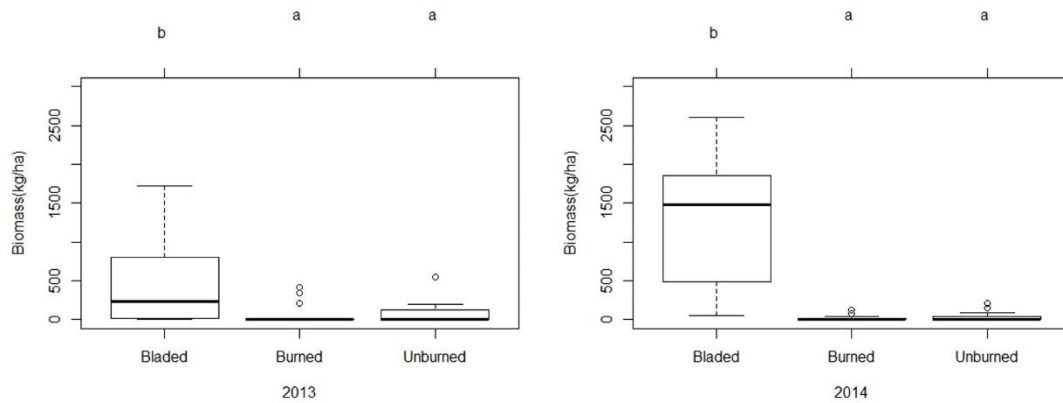


Figure 2.8. Annual grass production of year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

One year after the fire, annual grass production increased relative to unburned and burned but was dependent on the site. By the second year post fire, annual grass production showed a statistically significant increase in response to bladed treatment across all sites (figure 2.8).

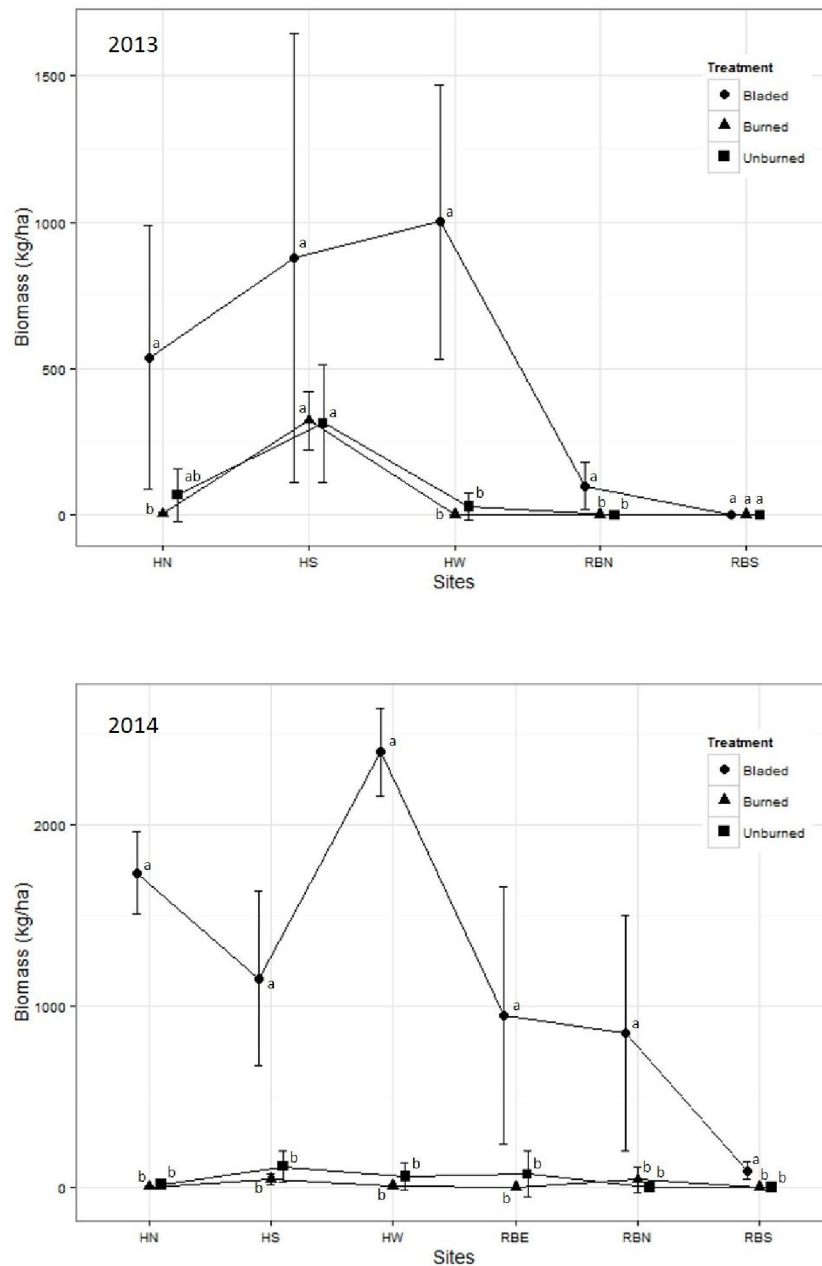


Figure 2.9. Annual grass production by site in 2013 (above) and 2014 (below). Different letters indicate significant difference ($p \leq 0.05$) within sites

Annual Forb. Pairwise comparison results from Tukey Honest Significant

Difference (HSD) test performed on ANOVA model of annual forb production indicated

that annual forb production was significantly greater in bladed treatment than that of

unburned ($p\text{-value}<0.0001$) and burned ($p\text{-value}<0.0001$). No difference was detected between years (figure 2.9).

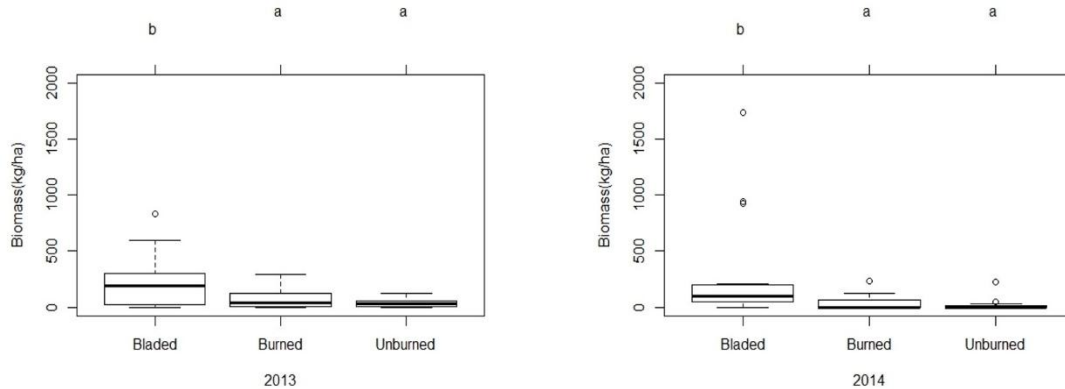


Figure 2.10. Annual forb production of year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

Annual forb production of bladed treatment on each site was higher than that of on unburned and burned but not all were statistically significant. Within site variation in annual forbs on bladed treatment was high. Annual forb production by site did not change much in 2014 except for bladed treatment of RBN which doubled the production. Within site variation declined which suggested that annual forbs were distributed more evenly across the replicates in 2014 (figure 2.10).

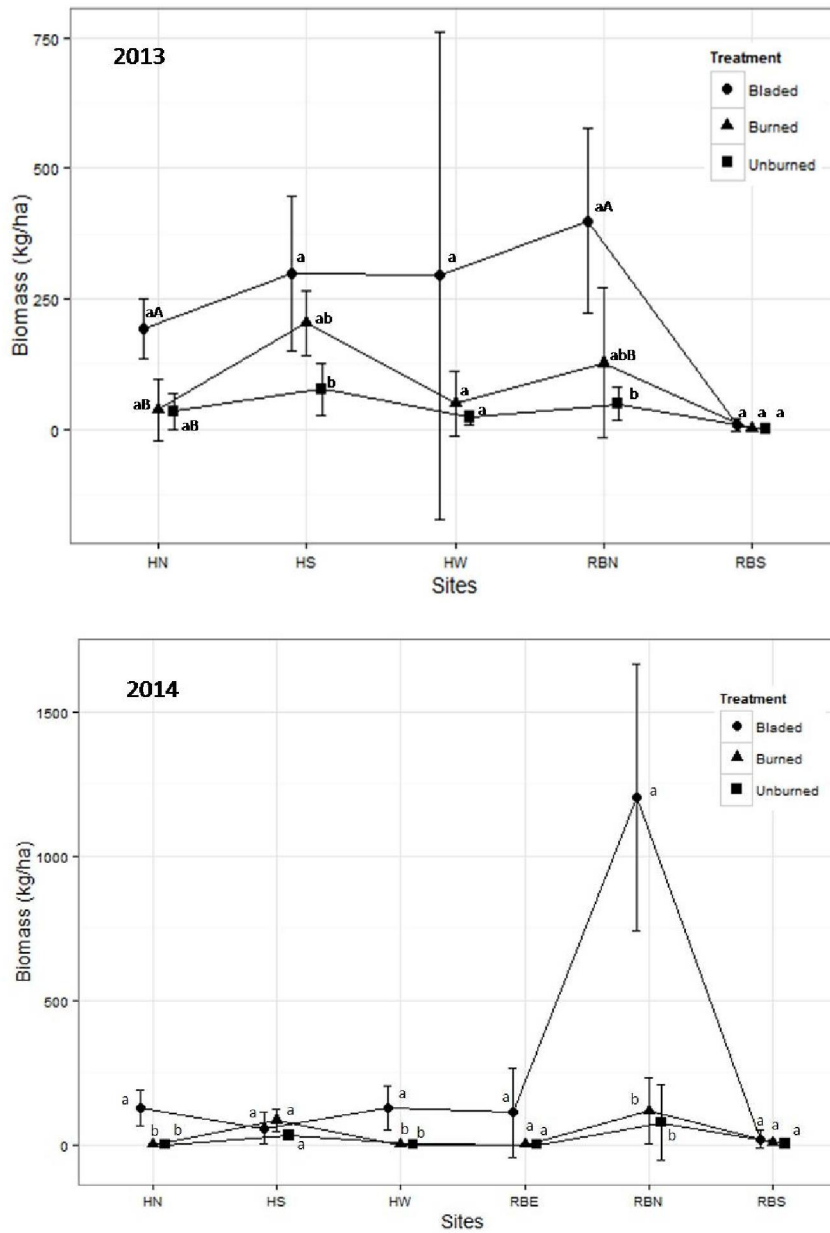


Figure 2.11. Annual forb production by site in 2013 (above) and 2014 (below). Different lower case letters indicate significant difference at $p \leq 0.05$ and different upper case letters indicate difference at $p \leq 0.1$ within sites

Shrub. Pairwise comparison results from Tukey Honest Significant Difference

(HSD) test performed on ANOVA model of shrub production detected significant higher biomass on unburned relative to bladed (p -value=0.02), but failed to detect difference

between burned and bladed (p -value=0.92), and unburned and bladed (p -value=0.07). No difference was detected between years (p -value=0.56). Treatment effects within years did not differ (figure 2.11).

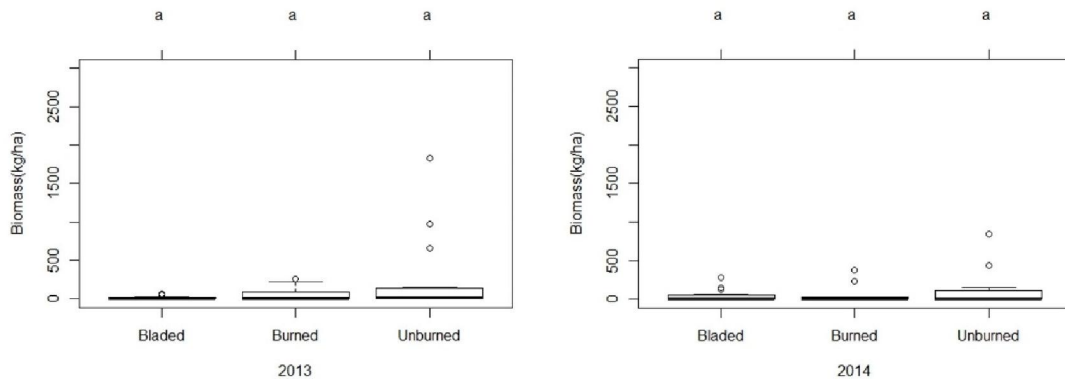


Figure 2.12. Shrub production of year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

Although all shrubs were removed with bulldozer and fire, biomass did not significantly differ at all. There was significant loss of shrub on HN and HW sites, but within site variation was high and likely masked any differences from burned and bladed. By the second year after the fire, sites that had shrub prior to fire showed some recovery but still less than unburned sites. RBS and RBE sites did not have shrub biomass at all both before and after the fire (figure 2.12).

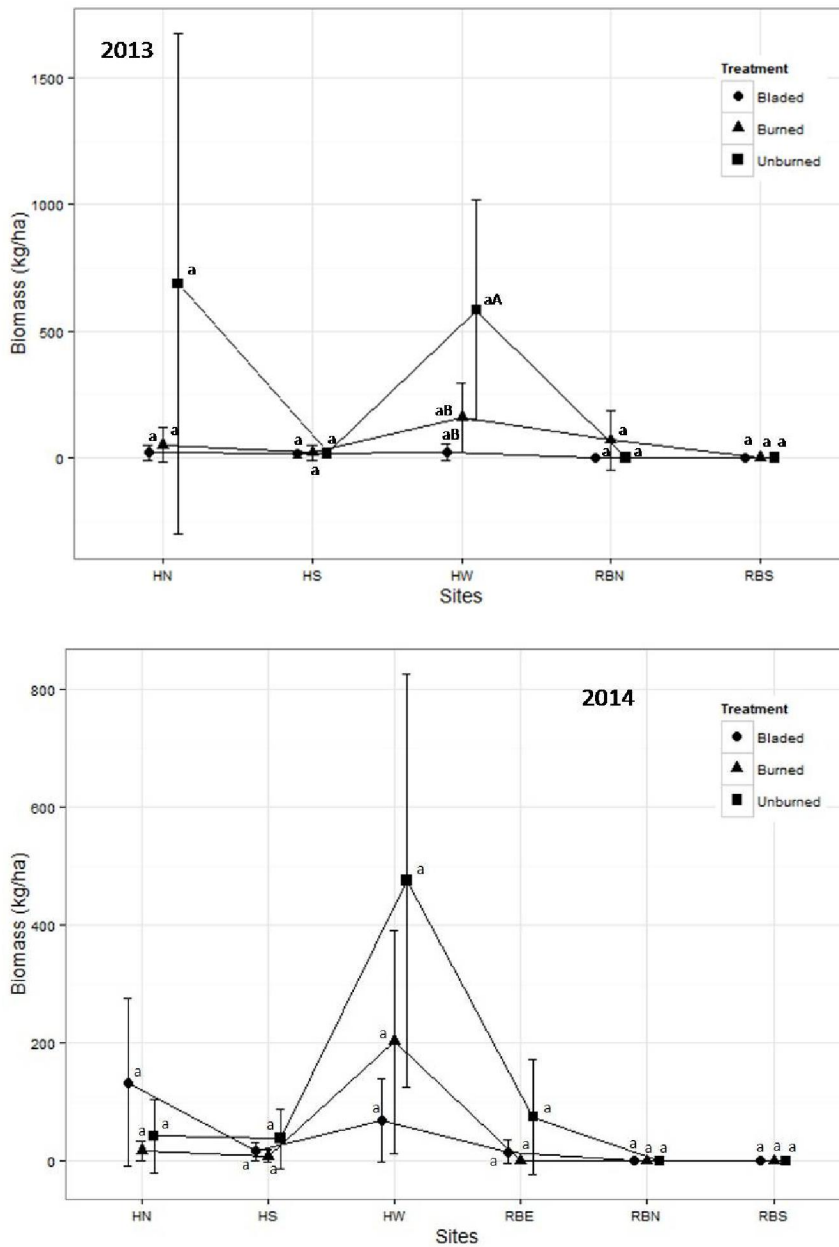


Figure 2.13. Shrub production by site in 2013 (above) and 2014 (below). Different lower case letters indicate significant difference at $p \leq 0.05$ and upper case letters indicate difference at $p \leq 0.1$ within sites

Litter. Pairwise comparison results from Tukey Honest Significant Difference (HSD) test performed on ANOVA model of litter production detected significant lower biomass on bladed treatment relative to burned (p -value=0.00001) and unburned (p -

value=0.000001). It failed to detect significant difference between year 2013 and 2014 (p-value=0.1). Fire had greater impact on litter biomass relative to unburned and bladed treatment one year after the fire, but by the second year, the burned treatment biomass was greater than bladed but not statistically significant (figure 2.13).

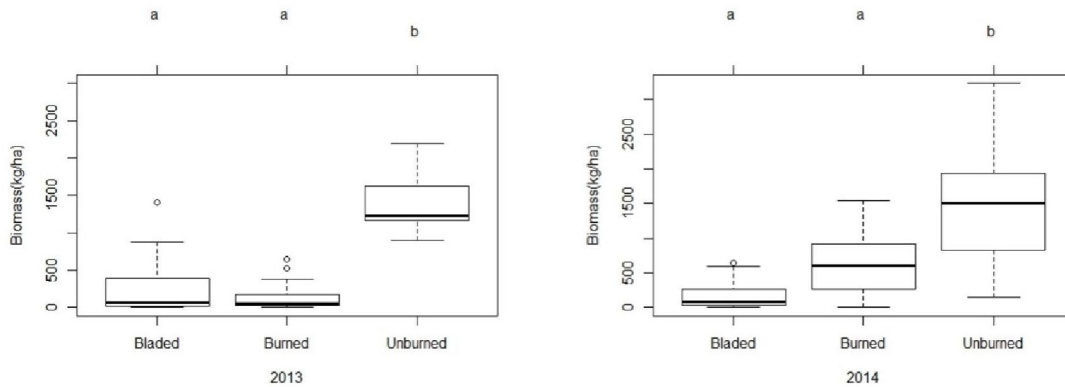


Figure 2.14. Litter production of year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

One year after the fire, the burning impact on litter production was the greatest and significant except RBN relative to unburned. Litter production was always lowest on the bladed treatment on all study sites except one. However treatment effects were not consistent across all the study sites. Impacts of bulldozed fireline on litter production was significant two years after the fire especially on Havre sites (figure 2.14)

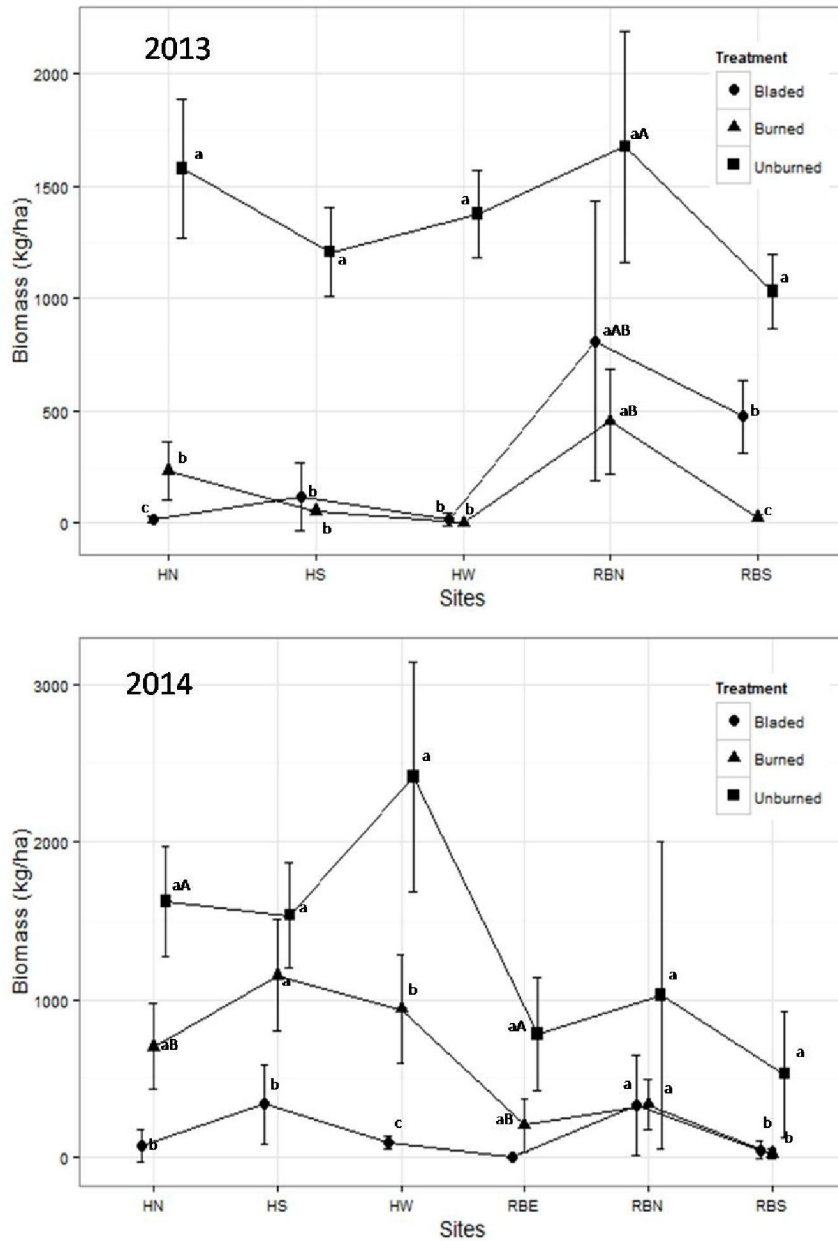


Figure 2.15. Litter production by site in 2013 (above) and 2014 (below). Different lower case letters indicate significant difference at $p \leq 0.05$ and upper case letters indicate difference at $p \leq 0.1$ within sites

Total Biomass. Pairwise comparison results from Tukey Honest Significant

Difference (HSD) test performed on ANOVA model of total biomass detected

significantly higher biomass on unburned treatment relative to burned (p -value=0.00003)

and bladed (p -value=0.00001). It also detected significant higher production in year 2014 relative to year 2013 (p -value=0.000001). It failed to detect significant difference between burned and bladed treatment (p -value=0.13), and year 2013 and 2014 (p -value=0.1). Comparison of treatment effects within years indicated the only difference between treatments was between bladed and unburned in 2013. Simply, loss of perennial plants have been replaced by exotic annuals (figure 2.15).

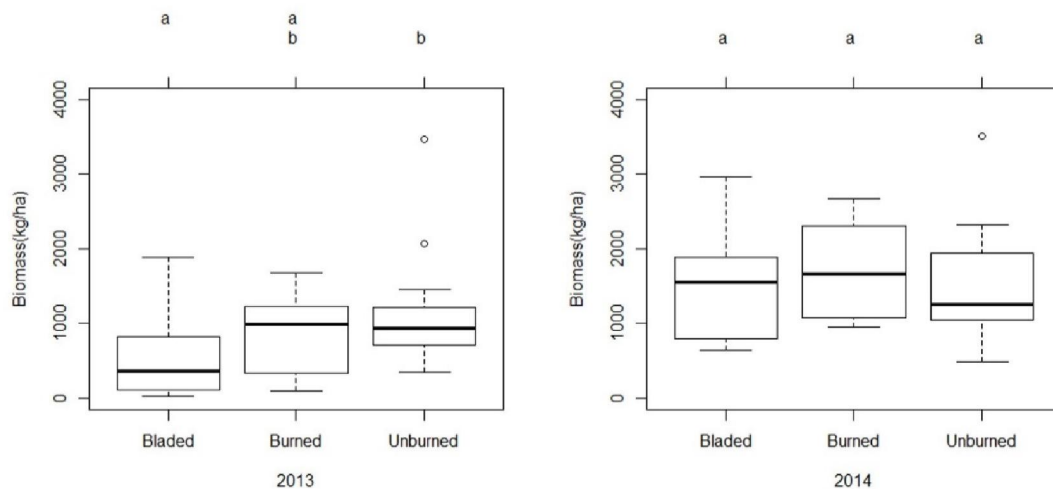


Figure 2.16. Total biomass year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

One year after the fire, the unburned treatment was significantly higher relative to the bladed treatment on 3 out of 5 sites. By the second year, only HW site had significantly higher biomass on unburned treatment relative to burned and bladed (figure 2.16). Annual grass production on the bladed treatment made up for the loss of perennial grasses. In 2014, perennial grass production on the unburned site did not change, while annual grass production on bladed treatment increased by up to two fold on some sites compare to year 2013.

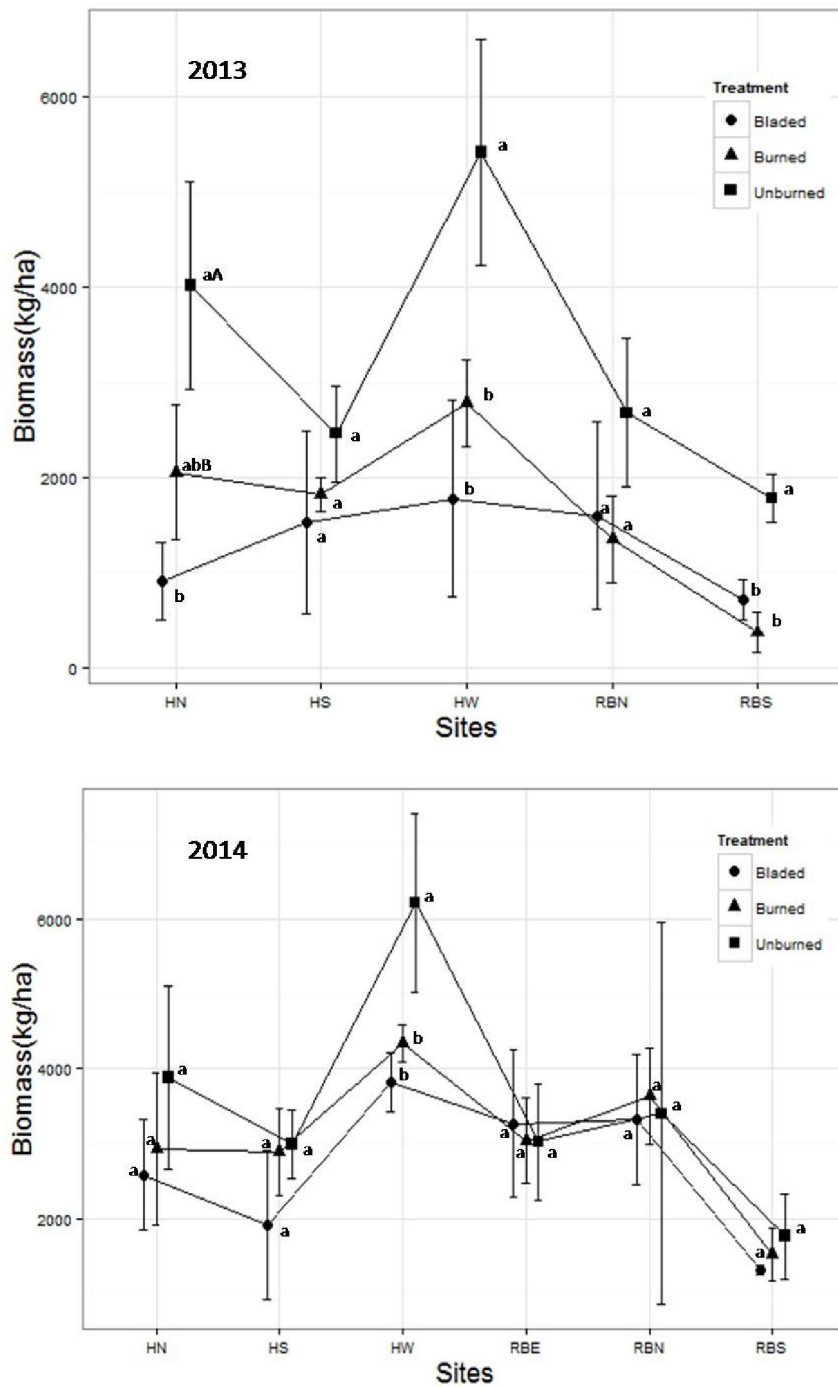


Figure 2.17. Total biomass by site 2013 (above) and 2014 (below). Different lower case letters indicate significant difference at $p \leq 0.05$ and upper case letters indicate difference at $p \leq 0.1$ letters within sites

Percent Cover

Perennial Grass. Perennial grass cover on bladed sites was significantly lower relative to burned ($p\text{-value}=0.0001$) and unburned sites ($p\text{-value}=0.0001$), while perennial grass cover was similar on burned and unburned ($p\text{-value}=0.99$) in both 2013 and 2014 (Figure 2.17).

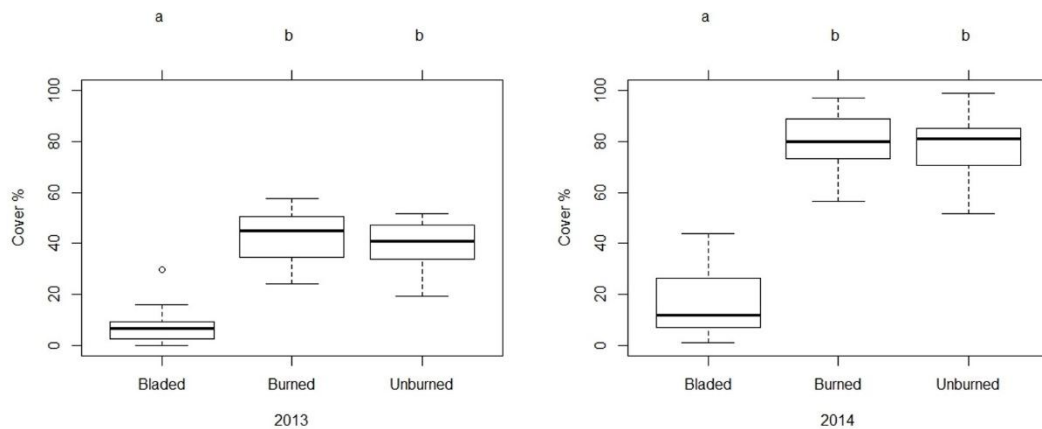


Figure 2.18. Perennial grass cover year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

Although perennial grass cover decreased significantly on bulldozed fireline across all sites and both years, the effects of burning were not consistent across all sites. Construction of fireline substantially reduced perennial grass on all study sites while burning did not change perennial grass cover relative to unburned across all study sites. Two years after the fire, perennial grass production on fireline remained low, while burned sites were equal or higher than unburned (Figure 2.18).

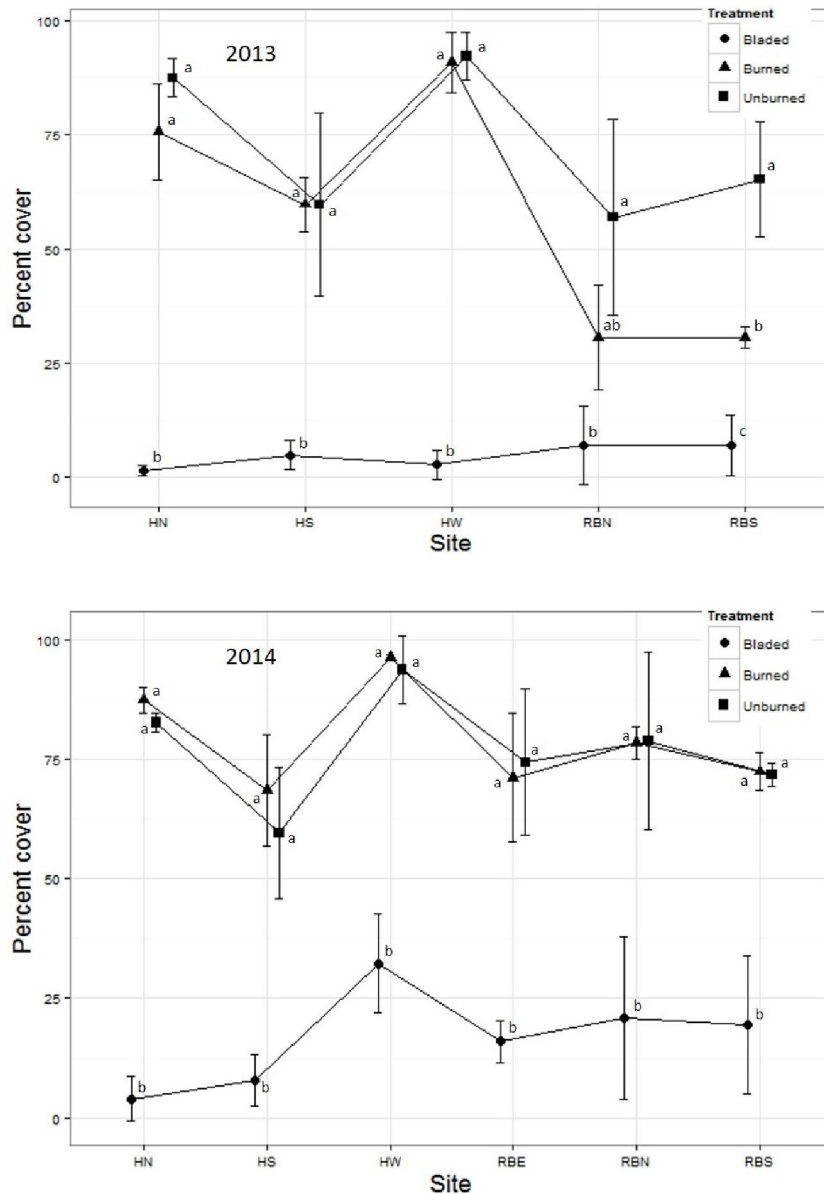


Figure 2.19. Perennial grass cover by site in 2013 (above) and 2014 (below). Different letters indicate significant difference ($p \leq 0.05$) within sites

Perennial Forb. Pairwise comparison results from Tukey Honest Significant Difference (HSD) test performed on ANOVA model of perennial forb cover indicated that the bladed treatment was significantly lower relative to the burned treatment (p -value=0.005); and lower but not significant relative to the unburned treatment (p -

value=0.47); the unburned treatment was lower but not significant relative to the burned treatment (p-value=0.11); and perennial forb cover was significantly higher in year 2014 relative to year 2013 (p-value=0.03). Treatments differed significantly in 2013, while treatments did not differ from each other in 2014 (figure 2.19).

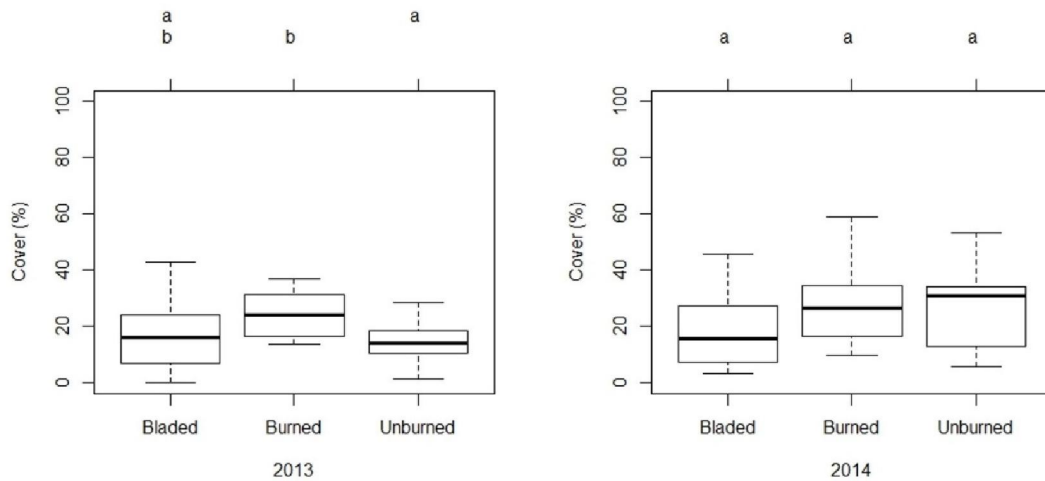


Figure 2.20. Perennial forb cover of year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

One year after the fire, bulldozed fireline impact on perennial forb cover was the greatest and significant except RBN and RBS. Perennial forb cover was always lowest on bladed treatment across all study sites except for one. However treatment effects were not consistent across all the study sites. Construction of fireline substantially reduced perennial forbs on most of our study sites while burning did not change perennial forb production. Two years after the fire, perennial forb production on fireline was similar relative to burned and unburned on all sites except two. On HN and HW, it stayed the lowest however on at the HN site were the differences statistically significant (figure 2.20).

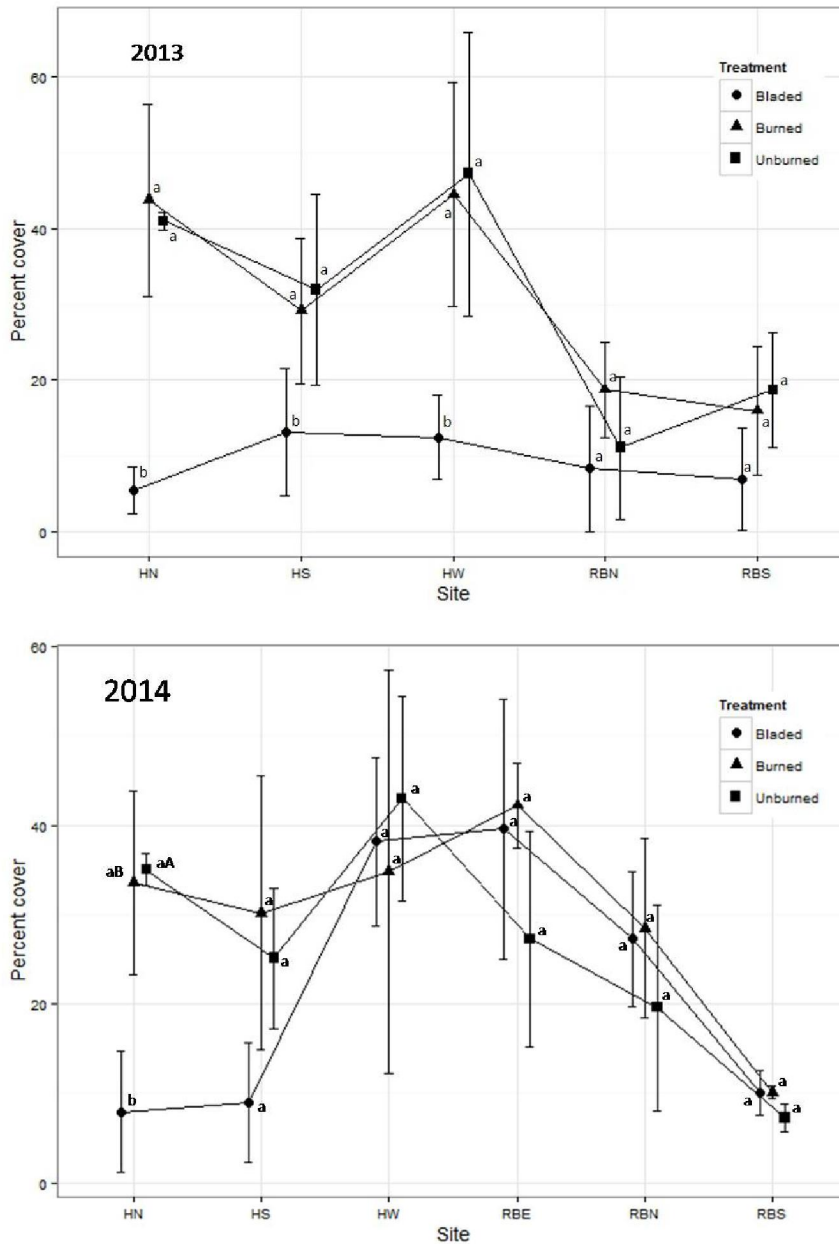


Figure 2.21. Perennial forb cover by site in 2013 (above) and 2014 (below). Different lower case letters indicate significant difference at $p \leq 0.05$ and upper case letters indicate difference at $p \leq 0.1$ within sites

Annual Grass. Pairwise comparison results from Tukey Honest Significant Difference (HSD) test performed on ANOVA model of annual grass cover detected significantly higher cover on bladed relative to unburned ($p\text{-value}=0.00001$) and burned ($p\text{-value}=0.00001$), and it failed to detect difference between unburned and burned ($p\text{-value}=0.31$) and year 2014 relative to year 2013 ($p = 0.17$). Results from treatment comparison within a year were consistent with the Tukey HSD test (figure 2.21).

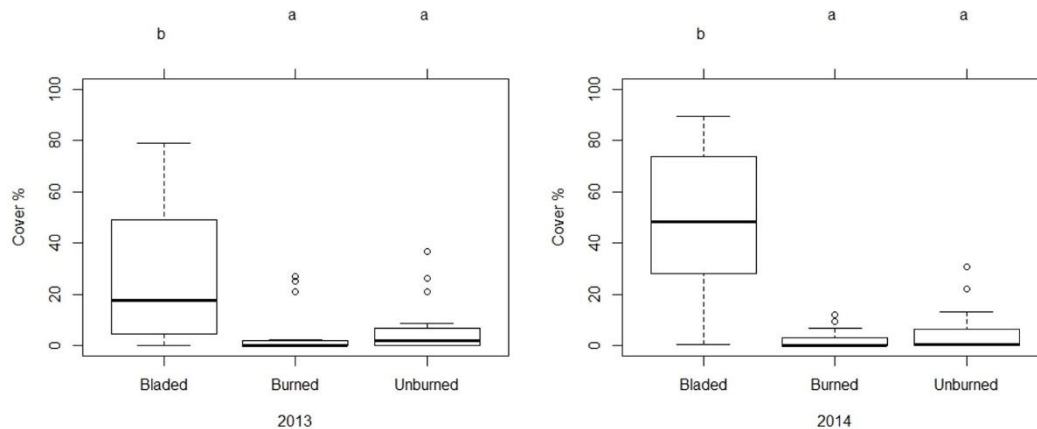


Figure 2.22. Annual grass cover of year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

One year after the fire, annual grass cover on the bladed treatment increased relative to unburned and burned but was not uniform across sites. Sites in Redbluff did not have much annual grass in 2013, however by the second year after the fire, annual grass production showed a statistically significant increase in response to bulldozed fireline across all sites (figure 2.22).

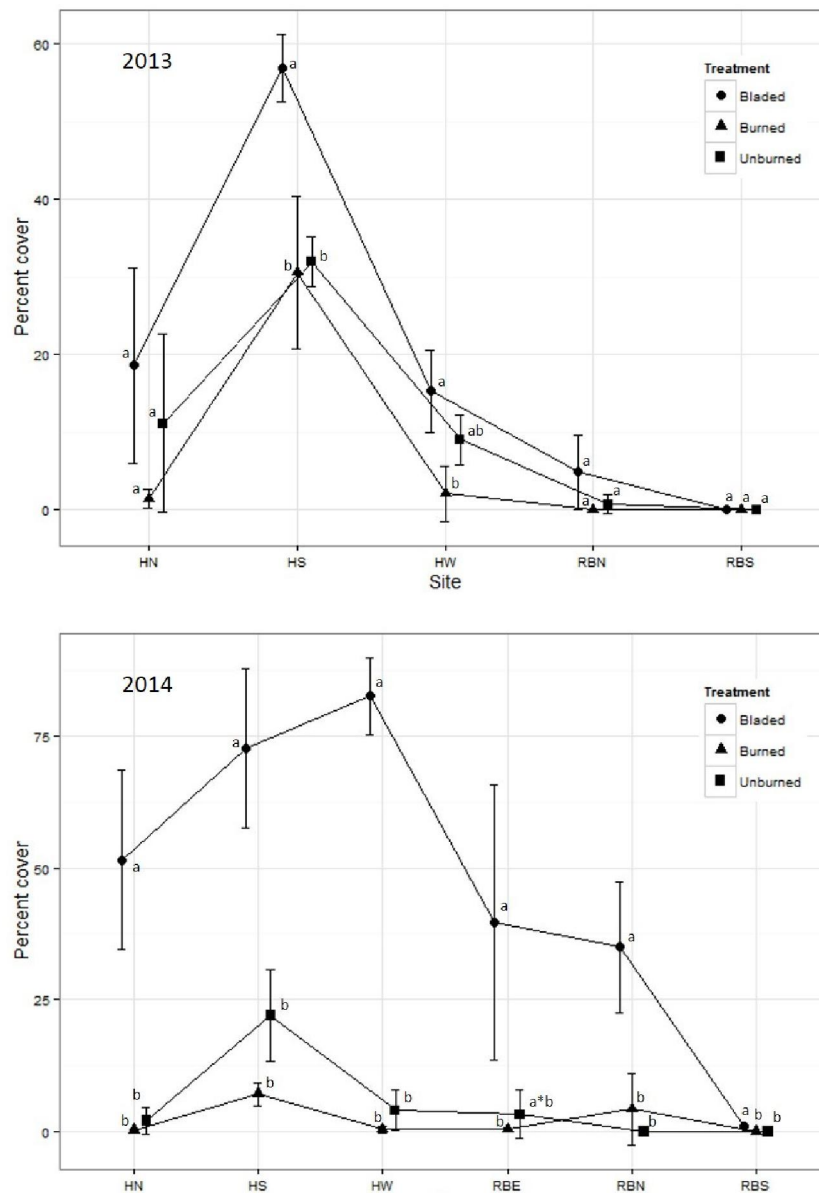


Figure 2.23. Annual grass cover by site in 2013 (above) and 2014 (below). Different letters indicate significant difference ($p \leq 0.05$) within sites

Annual Forb. Pairwise comparison results from Tukey Honest Significant

Difference (HSD) test performed on ANOVA model of annual forb cover indicated that annual forb cover was significantly greater in blade relative to unburned (p -

value=0.0001) and burned (p-value=0.0001). No difference was detected between unburned and burned and year 2013 and 2014 (figure 2.23).

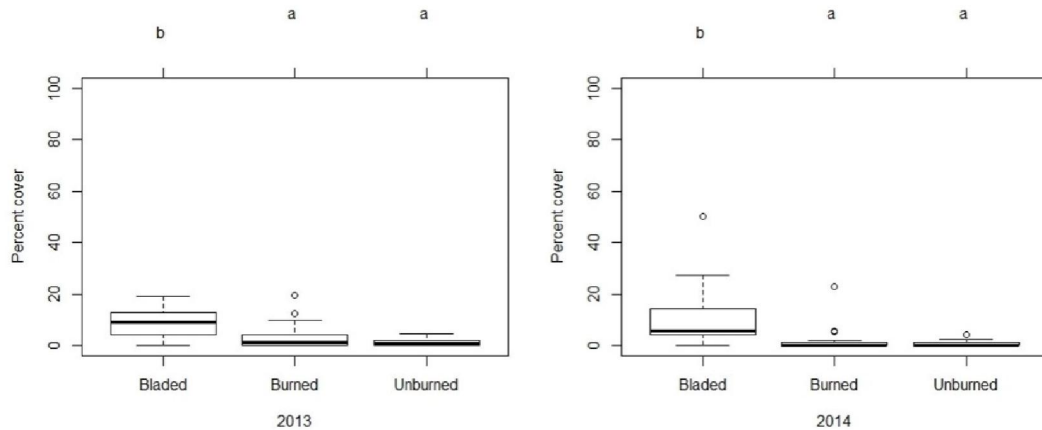


Figure 2.24. Annual forb cover of year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

Annual forb cover of bladed treatment on HN, RBN and RBS was higher than that of on unburned and burned but not statistically significant ($p=0.12$) while the annual forb cover in burned treatments on HS and HW was higher relative to unburned and bladed but not statistically significant ($p=0.24$). Within site variation of annual forb on bladed was high depending on sites. By the second year, annual forb cover on bladed treatment was significantly different on HN, HW, and RBN relative to unburned and burned. Within site variation declined in year two indicating that annual forbs were more evenly distributed across the replicates (figure 2.24).

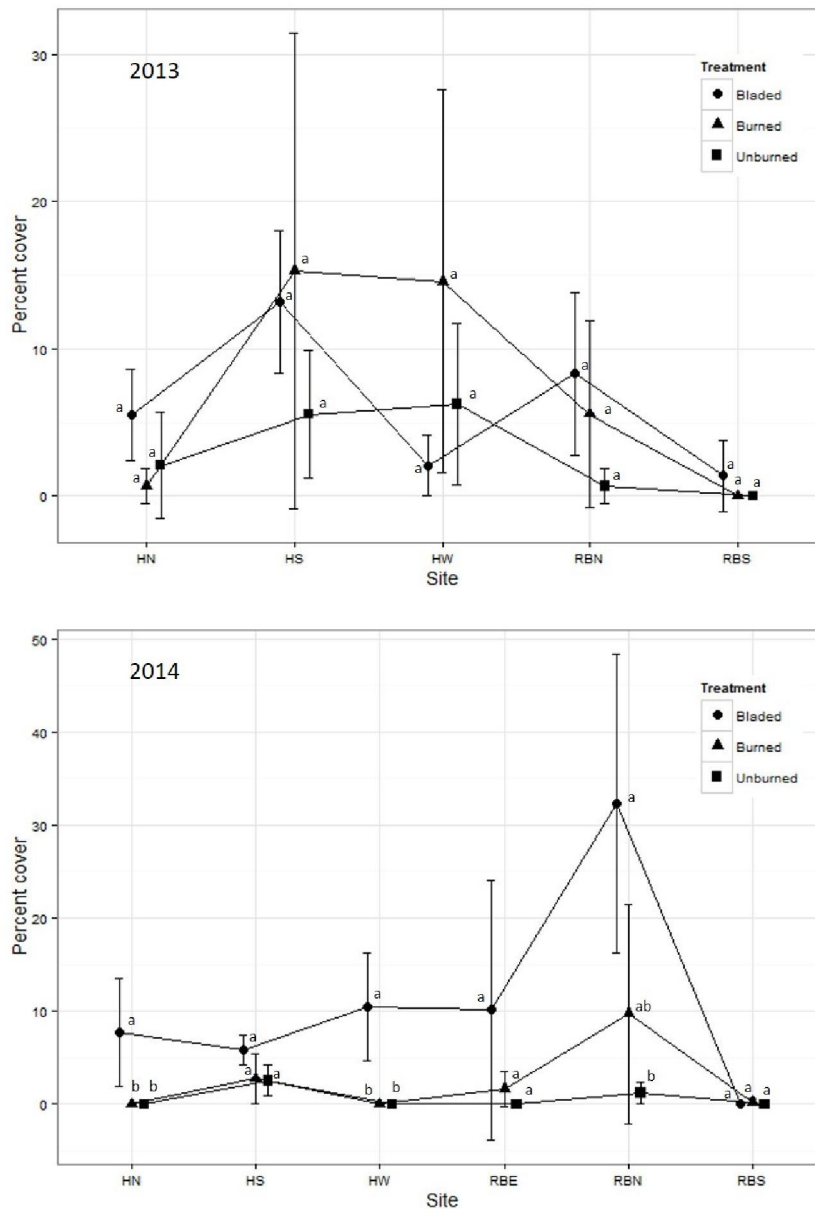


Figure 2.25. Annual forb cover by site in 2013 (above) and 2014 (below). Different letters indicate significant difference ($p \leq 0.05$) within sites

Shrub. Pairwise comparison results from Tukey Honest Significant Difference (HSD) test performed on ANOVA model of shrub cover detected significantly higher cover on unburned relative to bladed (p -value=0.002) and burned (p -value=0.02) treatments but failed to detect a difference between burned and bladed (p -value=0.69),

and year 2013 and year 2014 (p-value=0.71). Treatment effects within years did not differ (figure 2.25).

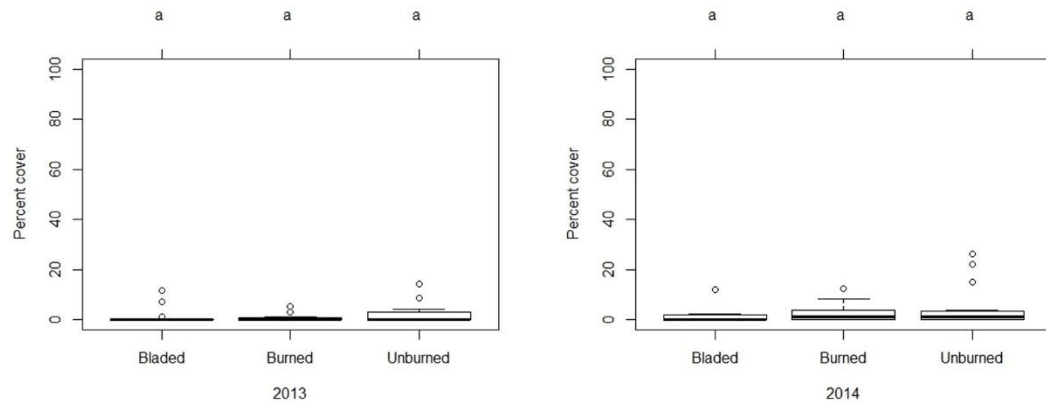


Figure 2.26. Shrub cover of year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

Although all shrubs were removed with bulldozer and fire, cover did not significantly differ at all except HW in 2013. There was substantial loss of shrubs on HN and HW sites, but within site variation did not make them differ from burned and bladed. By the second year after the fire, sites that had shrubs prior to the fire showed some recovery but still less than unburned sites. RBS and RBE sites did not have shrub cover at all both before and after the fire (figure 2.26).

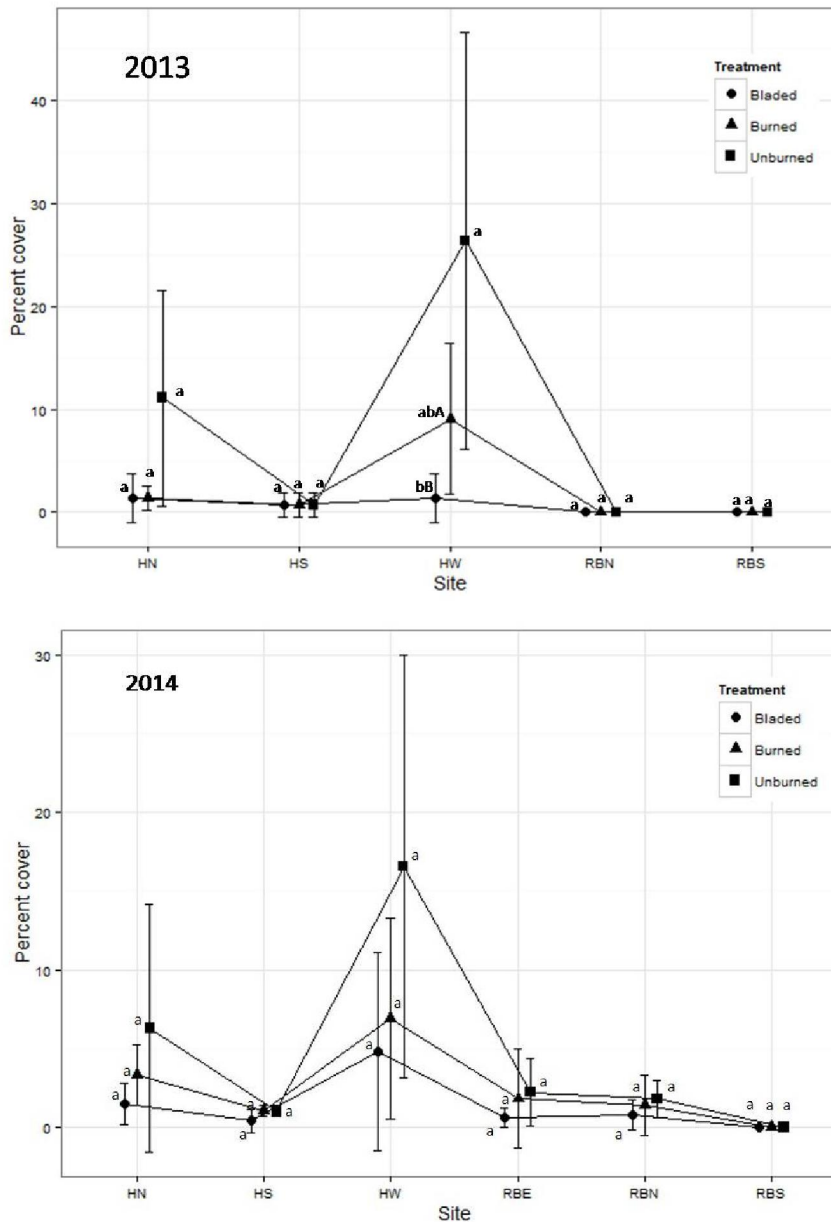


Figure 2.27. Shrub cover by site in 2013 (above) and 2014 (below). Different lower case letters indicate significant difference at $p \leq 0.05$ and upper case letters difference at $p \leq 0.1$ within sites

Litter. Pairwise comparison results from Tukey Honest Significant Difference (HSD) test performed on ANOVA model of litter cover detected significantly higher cover on unburned relative to burned (p -value=0.001) and bladed (p -value=0.01); and in

year 2014 relative to 2013 (p -value=0.02). It failed to detect difference between burned and bladed (p -value=0.7). It failed to detect significant difference between year 2013 and 2014 (p -value=0.1). Fire had greater impact on litter cover relative to unburned and bladed sites one year after the fire, but second year, litter cover on burned site was significantly greater than bladed (figure 2.27).

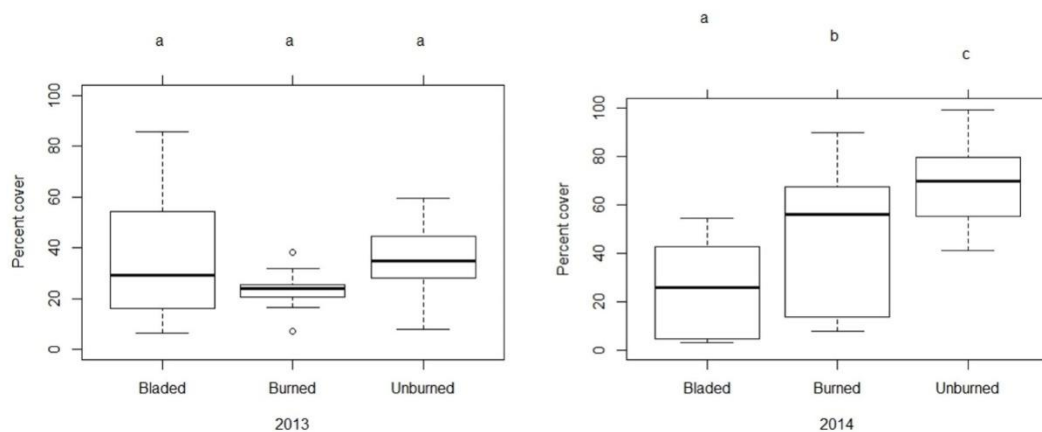


Figure 2.28. Litter cover of year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

One year after the fire, the burning treatment had the greatest impact on litter cover at the Redbluff sites while the bladed treatments had the lowest litter cover at the Havre sites. By the second year after the fire, the first year pattern remained but with less difference between burned and unburned treatments, while bladed sites showed still significant lower litter cover (figure 2.28)

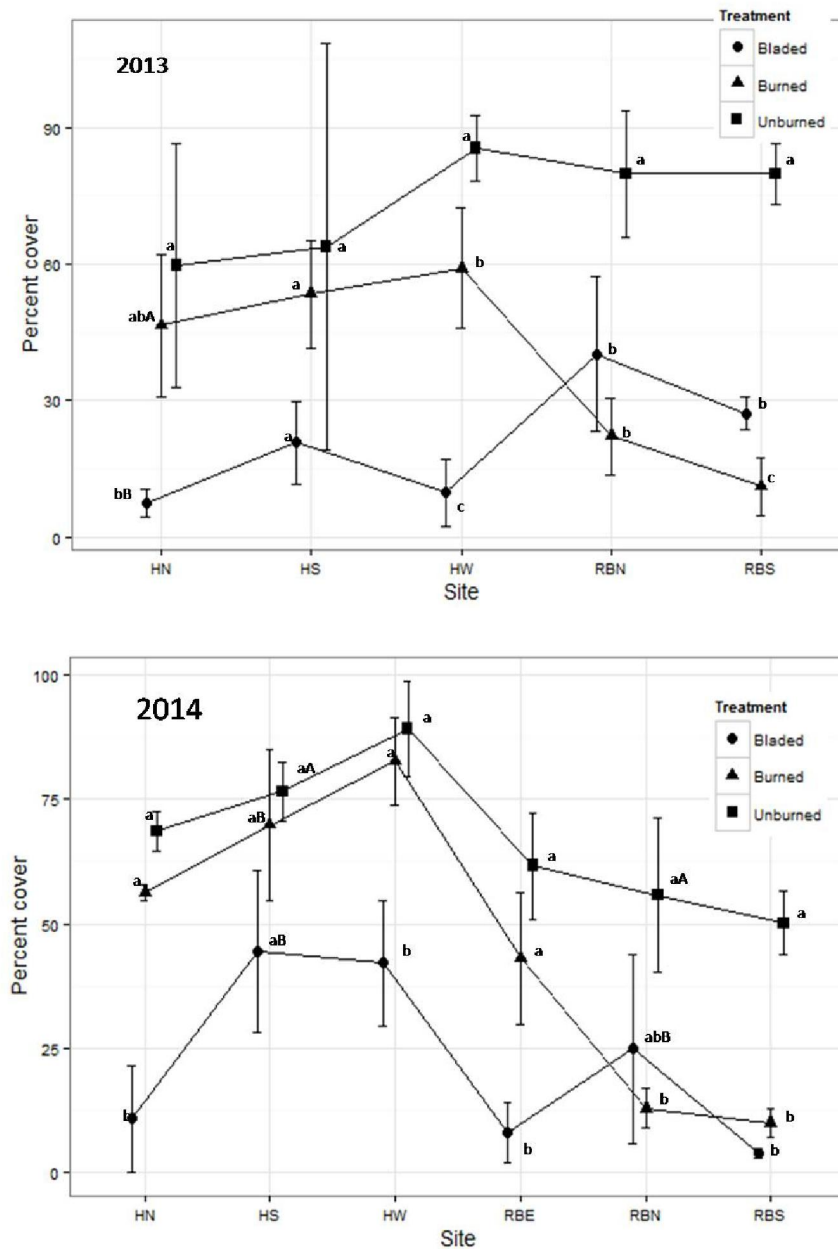


Figure 2.29. Litter cover by site in 2013 (above) and 2014 (below). Different lower case letters indicate significant difference at $p \leq 0.05$ and upper case letters difference at $p \leq 0.1$ within sites.

Total Cover. Pairwise comparison results from Tukey Honest Significant

Difference (HSD) test performed on ANOVA model of total cover detected significantly

higher cover on unburned relative to burned (p-value=0.002) and bladed (p-

value=0.00001); significantly higher cover on burned relative to bladed (p-value=0.000001) and year 2014 relative to 2013 (p-value=0.02). The same pattern was observed between treatments within year 2013, however, total cover on the unburned treatment was significantly higher than bladed treatment in 2014 (figure 2.29).

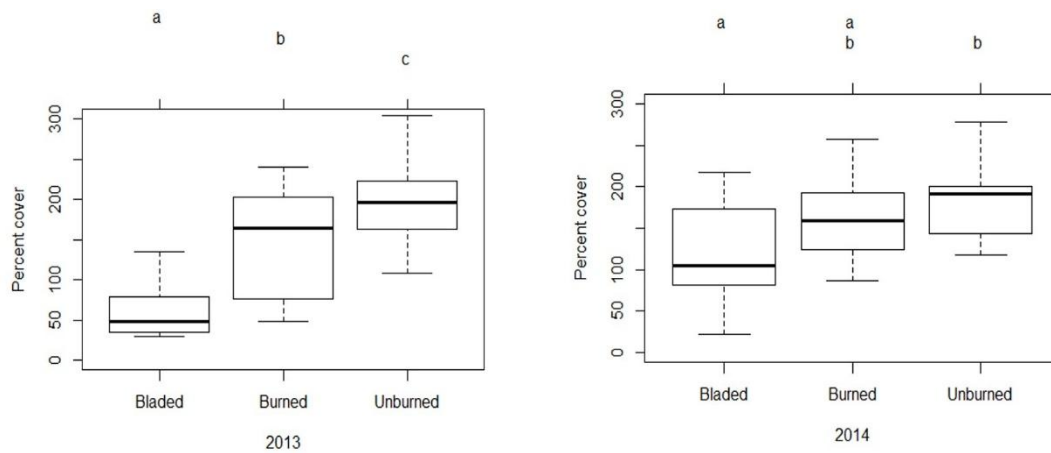


Figure 2.30. Treatment effect on total cover of year 2013 (left) and 2014 (right). Different letters indicate significant difference ($p=0.05$) between treatments within a year

First year after the fire, total cover on unburned treatment was significantly higher relative to bladed on all 5 sites. By the second year, only two sites had statistically significant lower cover on bladed treatment relative to unburned (figure 2.30). Annual grass cover on the bladed treatment made up for the loss of perennial vegetation. In year 2014, perennial grass cover on unburned sites remained stable, while annual grass cover on the bladed treatment increased by up to 50% on some sites compared to year 2013.

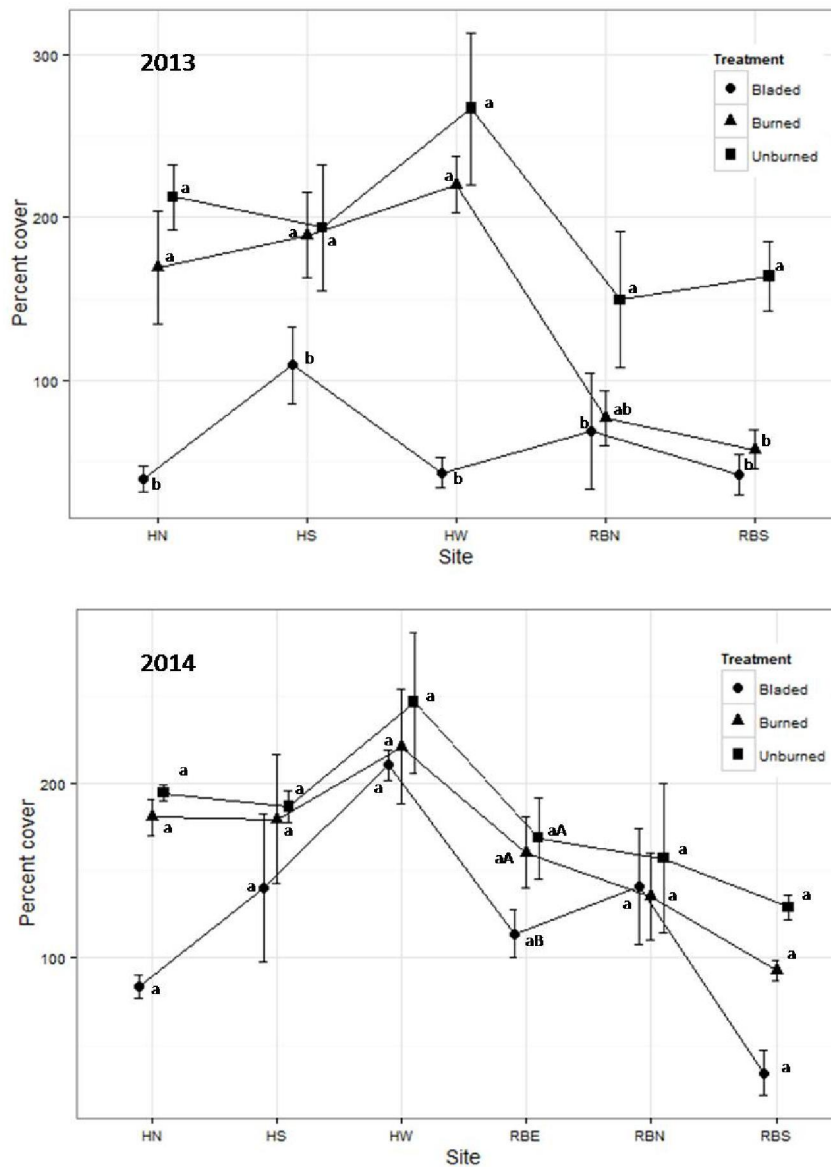


Figure 2.31. Total cover by site in 2013 (above) and 2014 (below). Different lower case letters indicate significant difference at $p \leq 0.05$ and upper case letters indicate difference at $p \leq 0.1$ within sites

Discussion

Perennial Grass

Results were consistent with our original hypothesis that bladed treatment would cause a decrease in perennial grass abundance and initiate non-native annual grass invasion. By the second year after the fire, none of the sites had shown regrowth of dominant perennial grass in the bladed treatment. In bladed treatments, vegetation communities come back from the propagules in the seed bank, or dispersed seed from adjacent undisturbed sites (O'Connor 1991). In our sites most of the dominant perennial grass species are long-lived, obligate seed reproduction (i.e., an absence of stoloniferous or rhizomatous growth), and with short longevity seeds (only 2-3 year) (O'Connor 1991). Unfortunately many perennial grasses display weak dispersal patterns (Marlette et al. 1986; O'Connor 1991), implying that the only option would be regrow from propagules in the seed bank or tillers from existing plants that have been barely scraped by bulldozer blade. Perennial grass seeds tend to concentrate in the near surface soil horizons and litter layer (Strickler et al. 1976). Iverson et al. (1982) found that 94% seeds were in top 7.5cm soil in unmined prairie (Iverson 1982). Thus the perennial grass seed bank on the bladed treatment might be depleted during the construction process of fireline. But there were some exceptions on bladed treatments where increase of perennial grass biomass was observed. This increase was mostly due to sampling at the Redbluff site. Regrowth from the live tussocks that had been removed and placed on the edge of fireline by the blade occurred on RBS site. While at Havre, the increase was because of regrowth of Kentucky bluegrass from residual rhizomes buried in part of the fireline. The burned treatment

showed greater production of biomass relative to the bladed, and similar or lower relative to unburned treatment which could be the result of low intensity fall burning (Wright 1982).

Perennial Forb

The bladed treatment did not have a significant effect on perennial forb biomass or cover overall. With the exception of the HW site, our results were consistent with our original null hypothesis that perennial forb production and cover would be the same across all treatments. Most of the forbs that were found on bladed treatment such as silky lupine (*lupinus sericeus* Purch), golden banner (*Thermopsis montana* Nutt. Var. *montana*) in Havre sites and plain locoweed (*Oxytropis campestris*) and purple locoweed (*Oxytropis lambertii*) were all deep rooted cryptophyte perennials (USDA 2012) which means they regrow from points well below the ground (Tracy et al. 1997).

Annual Grass

The bladed treatment appeared to trigger an exotic annual grass invasion. The first year after the fire, the bulldozed fireline in HN, HW and HS sites showed increase of biomass and cover of a mix of exotic annual species including cheatgrass (*Bromus tectorum*) and Japanese brome (*Bromus japonicus*), while the bulldozed fireline in RBN showed increase of cover and biomass of cheatgrass only. By the second year, all three bulldozed firelines in Redbluff sites appeared to be invaded by cheatgrass. Both cheatgrass and Japanese brome are well known for their highly invasive and competitive ability (D'Antonio 1992; Brooks et al. 2004). The main mechanism that makes these two species invasive are their high reproductive capacity because of high seed production and

long seed germination period from fall through winter to early spring and high germination rate (Baskin 1981; Mack et al. 1983; Karl et al. 1999). In addition they have vigorous winter root growth, which allows cheatgrass to utilize soil water efficiently by the time native plants become stressed by low water potentials in the spring and early summer (Harris 1967; Whisenant 1990). Both species reproduce solely from seeds with long longevity up to 11 years (Baskin 1981; Mack et al. 1983) and disperse efficiently either by animal or wind (Young et al. 1997; Cheplick 1998). Varied presence of annual grass across the sites and years could be explained by either previously invaded sites or dispersed seed to intact sites. Our data supports concepts that sites that had exotic annual grasses before the fire has shown invasion on bulldozed firelines by the first year after the fire. Both annual grasses existed in all three sites in Havre and only cheatgrass existed on the RBN site before the fire. While RBS site did not have annual grass biomass or cover on unburned site. RBE site also did not have any annual grasses, unfortunately we do not have data to show this in 2013. Therefore, RBS and RBE sites were invaded by cheatgrass two years after the fire.

Annual Forb

Increase of annual forb biomass and cover on burned and bladed treatment was consistent with other similar study results that annual forb composition in sites recently exposed to soil disturbance were higher relative to unburned or undisturbed (Benson et al. 1995; Merriam et al. 2006; Moroney 2013). Variation between sites related to magnitude of disturbance occurring on or adjacent sites. Our data clearly suggests that greater biomass and cover of annual forb on RBN site which had higher grazing impact as it was

near sheep camp. Merriam et al (2006) also found similar result that burned site that had been previously grazed had higher non-native annual forb in southern California. Year effect was apparent on both biomass and cover. By the second year annual forb biomass and cover decreased on both burned and bladed treatments with some exceptions of decreased biomass and increased cover on bladed treatment of HW and increased biomass and cover on bladed treatment of RBN site. Decrease of annual forb biomass and cover is related with increased perennial grass on burned and increased annual grass on bladed treatments.

Shrub

In general bulldozed fireline construction did not have significant impact on shrub biomass and cover. Shrub composition was smallest in RBS site 0%, highest in HW site less than 10% of relative cover, mostly snowberry (*Symphoricarpo salbus*) followed by HN site, where it was mostly creeping juniper (*Juniperus horizontalis*) and HS site where only shrub was wild prairie rose (*Rosa arkansana*). Although all of those shrubs have shown recovery on burned and bladed treatment, size was still small compared to that of unburned sites. However we do see an interesting effect of bladed treatment that perennial shrub production was higher but not statistically significant in RBE site by the second year after the fire. Shrub production increased mostly because of newly growing fringed sagewort (*Artemisia frigida*). Removal of dense clubmoss (*Selaginelladensa*) which impedes the establishment of other plants as it forms dense mat-forming cover (Lacey et al. 1995) gave opportunity to grow fringed sagewort. This result was consistent with the result of clubmoss chiseling study done by Lacey et al (1995).

Conclusion

Changes in vegetation functional groups associated with the construction of bulldozed firelines were significant in comparison to the unburned treatment. Loss of desired perennial grass species on bladed treatment was the most detrimental consequence. We do not know whether these perennial grasses would recover on bladed areas where the soil and vegetation attributes are completely different than prior to the disturbance. The competitive annual exotic, cheatgrass has shown an increasing trend on bladed sites. Whether cheatgrass will keep increasing or will be outcompeted by the perennial grass remains unknown. The consequence of perennial grass loss will be the altered nutrient cycle on bladed treatment. Because supply rate of nitrogen (N), growth limiting soil nutrient, depends largely on N mineralization which partly regulated by the supply of above and belowground litter (Jenny 1980). Litter decomposition rate depending on the quality, quantity, and timing of inputs (Berendse et al. 1989; Melillo et al. 1982; Seastedt 1988). Thus, the nature of the vegetation in a system can have a large effect on that system's N supply rate, and N supply rates can in turn strongly influence vegetation composition. This feedback could create alternative stable states for the vegetation-soil system under abiotically identical conditions (Vitousek 1982; Pastor et al. 1986; DeAngelis et al. 1989). In contrast, consequences of cheatgrass invasion can be worse than loss of perennial grass. Fine fuels that cheatgrass contributes to the ecosystem promote fire spread, and can change fire regime into grass-fire cycles that provide positive feedbacks for its continued invasion (D'Antonio 1992; Brooks et al. 2004).

A question arises that what could have been lost if we did not put the fireline? On the burned treatment, vegetation changes associated with burning were not significant compare to unburned and may even be beneficial to ecosystem. Desired perennial grass abundance was the same as in unburned sites, with some exceptions of higher abundance and undisturbed soil. In terms of rangeland condition, nothing is lost.

CHAPTER 3

SHORT-TERM SOIL RESPONSE TO BULLDOZED FIRELINE

Contribution of Authors and Co-Authors

Manuscripts in Chapters 2, 3

Author: Samdanjigmed Tulganyam

Contributions: Conceived and implemented the study design. Collected and analyzed data. Wrote first draft of the manuscript.

Co-Author: Craig A. Carr

Contributions: Helped conceive and implement the study design. Provided field expertise and funding. Provided feedback on drafts of the manuscript.

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SHORT-TERM SOIL RESPONSE TO BULLDOZED FIRELINE IN NORTHERN GREAT PLAINS GRASSLANDS

The following work is currently in progress to be submitted for publication

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Abstract

Bladed fire line construction is a fire suppression technique that limits fire spread by altering fuel continuity through vegetation removal and mineral soil exposure. Anecdotal evidence suggests bladed fire lines can cause long-term changes in soil and vegetation properties; however, relatively little research has been performed to corroborate these changes. In this study we compared soil properties among burned, unburned, and bulldozed fire line conditions (treatments) at three locations in north-central Montana and three locations in southwest Montana. Depth of the soil “A” horizon, bulk density, aggregate stability, runoff and sediment were quantified and comparisons made with Tukey HSD pairwise comparison using Analysis of Variance (ANOVA) model ($\alpha = 0.05$). Thinner “A” horizon, higher bulk density, weaker aggregate stability, higher runoff and higher sediment yield were observed on firelines relative to both unburned burned treatment type. Responses from the burned treatment were similar to unburned locations with the exception of higher runoff and sediment yields. These data support the contention that bladed fire lines could cause long-term ecological change associated with altered soil properties, and hydrologic function with more soil loss.

Introduction

Increasing number of literatures indicate that every fire suppression activity results in damage to ecosystems in certain extent. Generally, all the attempts to weigh the costs of fire suppression against potential losses due to fire and fire suppression activity the ecological cost is often not acknowledged, despite the fact that adverse effects from suppression activities may be substantial and persistent and, in some instances, may exceed impacts attributable to the fires themselves (Mohr 1989; Pyne et al. 1996). The use of bulldozer is an efficient and faster way to construct a fireline when fire intensity is high and heavy fuel load must be removed (Phillips et al, 1984). Bulldozed fireline refers to strips of exposed bare ground where top and organic matter rich “A” horizon is displaced (National Wildfire Coordinating Group 2012). According to National Wildfire Coordinating Group (2012) bulldozed firelines are limited on unstable soil and steep terrain $\leq 60\%$ slope due to its destructive nature.

Ecological impacts associated with bulldozed firelines persist long term and promote non-native vegetation invasion (D’Antonio et al. 1999; Young et.al 2002; Merriam 2009,) which is one of the unintended consequence of constructing bulldozed firelines (Merriam et al 2006; Moroney et al 2013).

Another consequence of bulldozed fireline is exposed bare ground, possibly with disrupted soil properties. Removal of vegetation cover and litter can decrease surface water storage and result in decreased infiltration rates, thereby increasing surface runoff (Robichaud and Waldrop 1994). Bulldozed fireline that equals to 1% of burned area contributed 20% of total erosion in the Appalachian region (Christie et al. 2013). Christie

et al. (2013) also found that as fireline slopes increased, potential erosion rates also increased. Flat slope fireline (0-5%) contributed 2%, intermediate slope (6-29%) contributed 33%, and steep (>30%) contributed 65% of total run from fireline (Christie, Aust et al. 2013). However a majority of the research regarding firelines has concentrated on vegetation impacts, very few study have addressed soil erosion. In addition fireline impacts on grassland have not been done except in California shrubland, all the studies have been done in forest ecosystem.

Our study examined the short-term ecological impacts associated with bulldozed fireline in Northern Great Plains grasslands. More specifically, this study focused on determining the soil response to bulldozed fireline in comparison with burned and unburned treatments. To understand the ecological impacts we investigated soil bulk density, aggregate stability, A horizon depth, and run off and soils erosion rates. We hypothesized that the bulldozed fireline would result in a loss of top soil depth, increased soil bulk density, decreased aggregate stability, and higher bare ground cover with more runoff and sediment yield from relative to burned and unburned treatments.

Methods

Site Description

We investigated two firelines in two different locations, Redbluff Agricultural Research and Teaching Ranch, 35 miles from Bozeman and the Thackarey Ranch, 15 miles south of Havre, MT (figure3.1). Both locations had grassland fires in August 2012 which were suppressed by constructing bulldozed firelines.

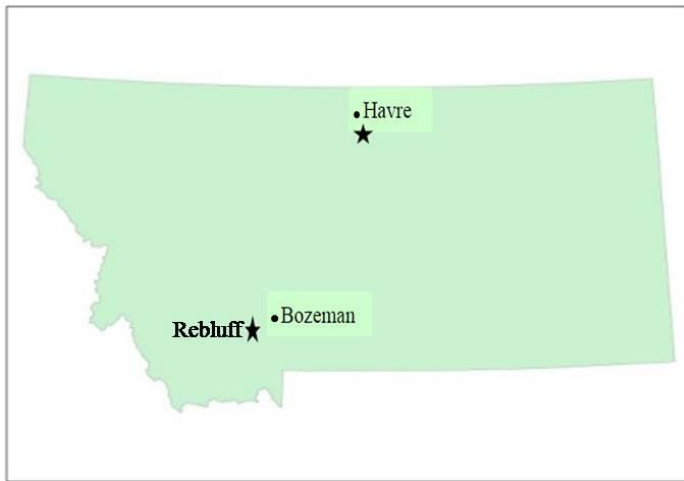


Figure 3.1. Study site locations

The Red Bluff Ranch, a 10,803 acre property, is located near Norris in Madison County, Montana, along the west side of the Madison River. Most of this land is rangeland, with limited hay meadows along the valley bottoms. Elevations range from 4,600 feet to 6,200 feet above sea level.

Soils mapped in the study area include Crago Gravelly loam (USDA NRCS 2011). Soils of these associations are Loamy skeletal, carbonatic, frigid Aridic Calciustepts. The soil surface is often very cobbly, stony, or gravelly, medium textured, has moderately slow water intake rates, low water holding capacity, rapid runoff, and are well drained (USDA NRCS 2003).

The current dominant plant community is Idaho fescue (*Festuca idahoensis*) with bluebunch wheatgrass (*Elymus spicatum*) and fringed sagebrush (*Artemisia frigida*). Long term temperature and precipitation data were obtained from the Norris weather station (45.4855° N, 111.632° W, 4915ft)(US Climate Data 2014) located approximately 15 km from the study site. The 30-year (1981-2010) mean annual temperature was 10.2°C ranging from a low of 4.2°C and a high of 16.2°C. Mean yearly precipitation was 42.9

cm, and has ranged between 30.4 cm and 52.7 cm over the last 4 years. However, for the first 11 months of 2014, total precipitation was 52.7 cm, while for the same period in 2013, total precipitation was 45.4 cm. Much of the precipitation comes as snow from November to April with local thunderstorms accounting for scattered summer precipitation.

The Thackeray Ranch is located about 17 miles south of Havre in Hill County, Montana, in the Bear's Paw Mountains and comprises approximately 4,000 acres of foothill rangeland. Elevation ranges 3080ft to 6060ft and slope ranges between 25%-70%. Soils mapped in the study area include the Whitelash-Perma-Rock outcrop association (USDA NRCS 2011). Soils of these associations are loamy skeletal, mixed, superactive Typic Haploborolls. Parent materials are weathered from igneous rocks, and soils are very deep and associated with rocky outcrops. The soil surface is often cobbly loam or gravelly loam, medium textured, has moderately slow water intake rates, low water holding capacity, rapid runoff, and are well drained (USDA NRCS 2003).

The current dominant plant community is rough fescue, prairie junegrass, Sandberg bluegrass, Idaho fescue, needleandthread, thickspike or western wheatgrass, fringed sagewort, and increaser forbs such as arrowleaf balsamroot and goldenpea. There are remnant amounts of some of the late-seral species such as bluebunch wheatgrass, and green/Columbia needlegrass present. In some situations, non-native grasses such as Kentucky bluegrass may also occur, sometimes comprising up to about 50 percent of the vegetation..

Long term temperature and precipitation data were obtained from the Havre weather station (48°54' N, 109°76' W, 789 m); (US Climate Data 2014) located approximately 25 km from the study site. The 30-year (1981-2010) mean annual

temperature was 6.36°C ranging from a low of -0.9°C and a high of 13.6°C. Mean yearly precipitation was 28.34 cm and has ranged between 27.7 cm and 48.6 cm over the last 4 years. For the first 11 months of 2014, total precipitation was 27.7cm, while for the same period in 2013, total precipitation was 45.4 cm. Much of the precipitation comes as snow from November to April with local thunderstorms accounting for scattered summer precipitation. In August 2012, a wildfire started from an isolated lightning strike. Wildfire was suppressed with construction of 3.2 km long bulldozed fireline along the perimeter of the fire in Thackarey.

Site Selection

We limited our selection to portions of fire-line within a common ecological site to ensure that we could replicate sampling under the same site conditions. According to Natural Resource Conservation Service (NRCS), an ecological site is described as “a distinctive kind of land with specific soil and physical characteristics that differs from other kinds of land in its ability to produce distinctive kinds and amounts of vegetation, and in its ability to respond similarly to management actions and natural disturbances” (USDA NRCS 2003). We selected three ecological sites at each location. In Havre, our ecological sites were:

- 1). Havre North site (hereafter refer to HN): loamy steep ecological site on the north facing slope with 25% grade, the “A” horizon depth of 21-25cm. Dominant grasses were rough fescue (*Festuca campestris*), bluebunch wheatgrass (*Elymus spicatus*), prairie junegrass (*Koeleria macrantha*) needle and thread (*Heterostipa comata*), Idaho fescue (*Festuca idahoensis*), kentucky bluegrass (*Poa pratensis*). Dominant forbs were

goatsbeard (*Tragopogon* species), silky lupine, chickweed (*Stellaria species*), prairie goldenbean (*Thermopsis rhombifolia*), western yarrow (*Achillea millefolium*) and some creeping juniper (*Juniperus horizontalis*).

2). Havre West site (hereafter refer to HW): clayey ecological site on the northwest facing slope with 18-20% grade, the “A” horizon depth of 35-40cm. Dominant grasses were kentucky bluegrass (*Poa praterensis*), rough fescue (*Festuca campestris*), bluebunch wheatgrass (*Elymus spicatus*), prairie junegrass (*Koeleria macrantha*) needle and thread (*Heterostipa comata*). Dominant forbs were goatsbeard (*Tragopogon* species), silky lupine, chickweed (*Stellaria species*), prairie goldenbean (*Thermopsis rhombifolia*), western yarrow (*Achillea millefolium*) and some snowberry (*Symphoricarpos albus*).

3). Havre South site (hereafter refer to HS): shallow clayey ecological site on the north facing slope with 22% grade, the “A” horizon depth of 15-20cm. Dominant grasses were idaho fescue (*Festuca idahoensis*), bluebunch wheatgrass (*Elymus spicatus*), rough fescue (*Festuca campestris*), prairie junegrass (*Koeleria macrantha*) needle and thread (*Heterostipa comata*), kentucky bluegrass (*Poa praterensis*). Dominant forbs were goatsbeard (*Tragopogon* species), silky lupine, chickweed (*Stellaria species*), prairie goldenbean (*Thermopsis rhombifolia*), arrowleaf balsamroot (*Balsamorhiza sagittata*), western yarrow (*Achillea millefolium*) and some prairie wild rose (*Rosa arkansana*).

We established 3 replicate plots within each ecological site and each treatment (unburned, bladed and burned) for a total of 54 plots in two locations. Plots were 50-150 meters apart away from each other to insure independence between replicates. Plot sizes were 16.7m x 3m in Redbluff and 16m x 4m in Havre. Plots on unburned and burned

treatments were established 2 to 10 meter apart from fireline and parallel to it (figure 3.2). Plot widths were dictated by the fireline width .All soil data was collected from all 3 sites both in 2013 and 2014 from Redbluff.

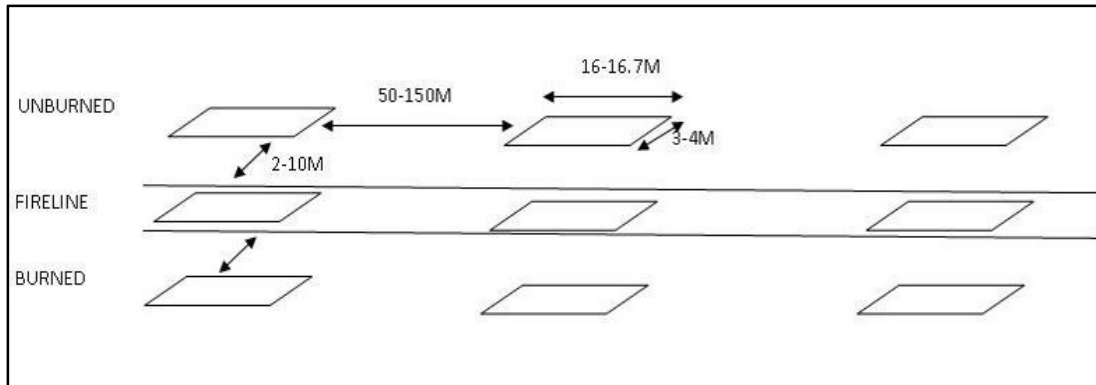


Figure 3.2. Study design for an ecological site

Soil Assessment

Soil Bulk Density. Soil bulk density was measured to evaluate soil compaction that could have been caused by heavy duty bulldozer trampling and/or removal of organic material rich top soil. To collect bulk density samples, we used 6cm diameter metal tube pounded to a depth of 10 cm. Four cores were removed from each transect and aggregated into one sample bag. A total of 12 cores were collected per plot. Samples were placed in plastic bags and stored frozen to minimize microbial activities. In the laboratory, all samples were weighted and sieved with 2mm mesh and bulk density was calculated based on dry weights of >2mm particles.

Aggregate Stability. In summer 2013, soil aggregate stability was tested with soil stability kit (Herrick, Whitford et al. 2001). Soil aggregate stability is widely recognized

as a key indicator of soil quality and rangeland health. A soil aggregate stability kit was employed because it is inexpensive, rapid and is as sensitive as laboratory analysis (Herrick, Whitford et al. 2001). We collected 6 samples per transect, for a total of 18 soil aggregates per plot from the soil surface. Soil samples were rated on a scale from one to six based on a combination of ocular observations of slaking during the first 5 min following immersion in distilled water, and the percent remaining on a 1.5-mm sieve after five dipping cycles at the end of the 5-min period.

Rainfall Simulation. In 2014, simulated rainfall was applied at a rate of 6.4 cm h⁻¹ using a known pressure and flow, which represented the 100-year return period storm intensity for the area (NOAA 2011). Plots were not pre-wet prior to simulation. A single nozzle small-plot rainfall simulator was used to simulate rainfall (Spaeth 1995). This simulator design was chosen due to its lightweight, small frame size and adaptability to steep, rugged terrain (Wilcox et al. 1986). The simulator consisted of a tripod frame with a nozzle head, the rainfall plot frame, collection trough, and a water supply system (Appendix A). The plot frame was constructed with three 10 cm tall walls designed to reduce subsurface lateral flow and a collection trough for collecting surface runoff and sediment. The simulator nozzle was centered over each simulation frame at a height of 2.4 m to allow the raindrops gain terminal velocity. Runoff and sediment were collected within six 5-minute intervals during the half hour rainfall period (Spaeth 1995). At the end of each 5-minute interval, all runoff and sediment was scraped from the collection trough and collected in a bucket. Runoff (with sediment) volume was measured and recorded for each time interval. Half liter runoff subsample was taken for sediment

analysis and stored in numbered 0.5liter Nalgene® bottles. Each 0.5liter bottle was allowed to settle in the laboratory, the excess water was filtered out with Whatman® filter. The sediments on the filter were stored in paper trays and placed in a microwave oven at defrost setting until all excess water was evaporated. The paper trays were weighed and subtracted from the final weight to provide a mass of sediment content in each bottle per time interval. Total runoff was calculated as the sum of runoff collected during each time interval. Total sediment yield was calculated similarly.

Data Analysis

If certain assumptions are met, a normal linear model can be used for examining evidence that at least one treatment has a different mean parameter than the others(Ramsey 2012). If we consider the three treatments burned, unburned and bladed as three groups, we can test the null hypothesis that the mean value for a certain parameter for the three treatments after accounting for site and year are equal versus the alternative hypothesis that at least one is different from the others. Under the null hypothesis, the mean value for the certain parameter for each treatment is the same and our equal-means and separate-means model should explain approximately the same amount of variation after accounting for site and year. The full model with separate means among treatments can be written as:

$$\mu\{Y_i/Treatment, Site, Year\}=\beta_0+\beta_1BU_i+\beta_2UB_i+\beta_3Site2_i+\beta_4Site3_i+\beta_5Site4_i+\beta_6Site5_i+\beta_7Site6_i+\beta_7Year2014_i$$

Where B_i and UB_i are indicator variables for observations in burned and unburned treatment and the $Site_i$ variables are indicators for observations in the different ecological

sites. Site 2 and 3 are located in Havre and site 4 and 5 is located in Redbluff. Note that this model is using bladed fireline as the reference treatment and site 1 as the reference for all sites as well as year 2013. The normal linear model requires three assumptions (Ramsey 2012):

1. Residuals are independent. This will be the assumption to be most carefully controlled. Our model accounts for sites and nesting of treatments within each site, but some spatial correlation may exist if sites closer together are more similar than sites farther apart. This may be present but it would also be difficult to quantify because of the small number of sites in the study. If positive correlation is present then the p-values of our analysis may be too small and the confidence intervals too narrow. However, since we have a small number of sites there is not much that can be done to check this assumption.

2. The residuals have constant variance. This can be checked by looking for patterns in the residuals versus a fitted values plot. If it is found that the residuals have a fanning shape, one remedy is to apply a log or square root transformation to the response variable which can help equalize the spread of the residuals.

3. The residuals are approximately normally distributed. This can be checked with a normal QQ plot. If we have a problem with our constant variance assumption then we will most likely have a problem assuming normality. A log transformation on the response variable can sometimes remedy both situations depending on the spread of the residuals. Plotting a normal QQ plot with randomly generated standard normal

observations with the same sample size as our data can provide a way to compare how far our residuals deviate from truly normal residuals.

Results

Depth of Soil “A” Horizon

Pairwise comparison results from Tukey Honest Significant Difference (HSD) test performed on ANOVA model of soil depth detected a significant loss of the A horizon on the bladed treatment relative to unburned (p-value=0.0002) and burned (p-value=0.0001). It failed to detect difference between unburned and burned (p-value=0.56) (figure 3.3).

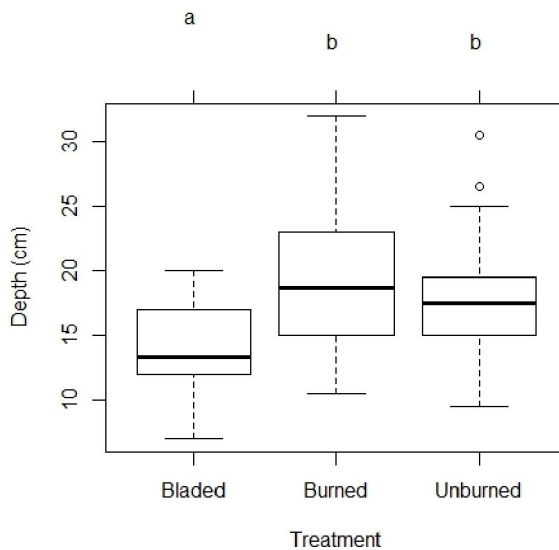


Figure 3.3. Treatment effects on soil “A” horizon depth. Different letters indicate significant difference (p=0.05) between treatments

Although the construction of a bulldozed fireline changed top soil depth, the treatment effects were not consistent across all sites. The “A” horizon depth was

significantly lower on bladed treatment in all Havre sites while bladed treatments in Redbluff sites were lower or equal but not statistically significant. Top soil depth on burned treatment was higher on HS and HW site, while the rest were similar or lower relative to unburned (figure 3.4).

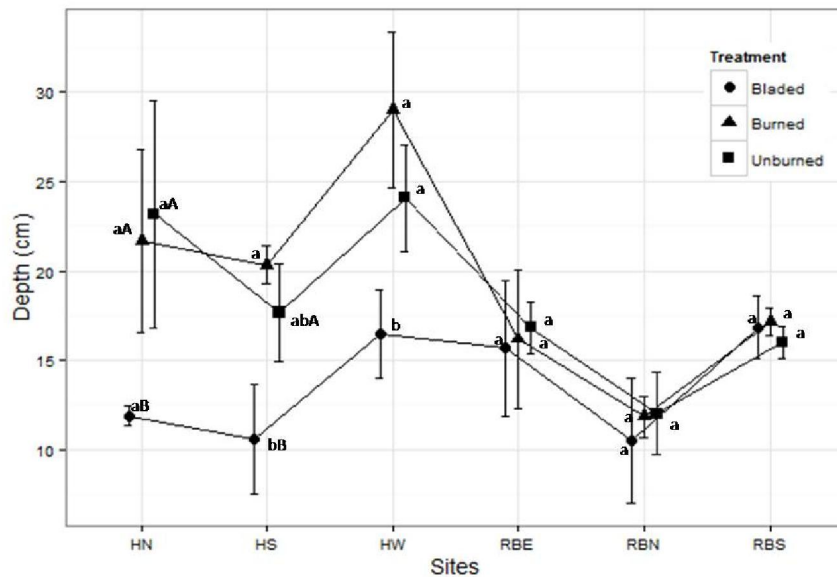


Figure 3.4. Treatment effects on soil “A” horizon depth by site. Different lower case letters indicate significant difference at $p \leq 0.05$ and upper case letters indicate difference at $p \leq 0.1$ within sites

Soil Bulk Density

Pairwise comparison results from Tukey Honest Significant Difference (HSD) test performed on ANOVA model of soil bulk density detected significantly higher bulk density on the bladed treatment relative to unburned (p -value=0.00001) and burned (p -value=0.0001). It failed to detect difference between unburned and burned (p -value=0.76) (figure 3.5).

Treatment effects were not consistent across the sites. Construction of bulldozed fireline definitely changed soil bulk density. Bulk density on bladed treatment was higher relative to burned and unburned across all sites but only significant HS and HW sites. Burning generated similar bulk density relative to unburned (figure.3.6).

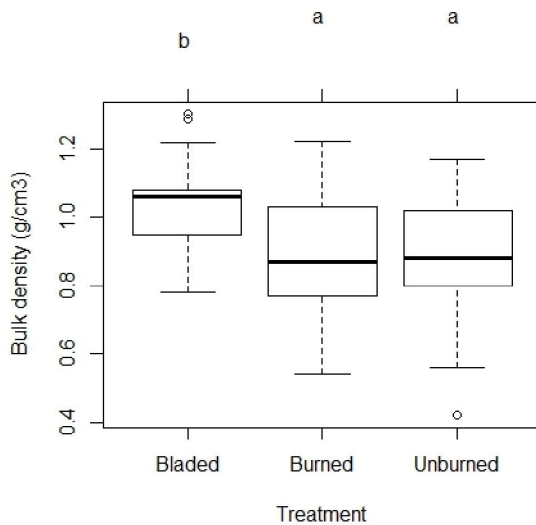


Figure 3.5. Treatment effects on soil bulk density. Different letters indicate significant difference ($p=0.05$) between treatments

Soil Aggregate Stability

Pairwise comparison results from Tukey Honest Significant Difference (HSD) test performed on ANOVA model of soil aggregate stability detected significantly lower aggregate stability scores on bladed treatments relative to unburned ($p\text{-value}=0.00001$) and burned ($p\text{-value}=0.00001$). It failed to detect significant difference between unburned and burned ($p\text{-value}=0.78$) (figure 3.7).

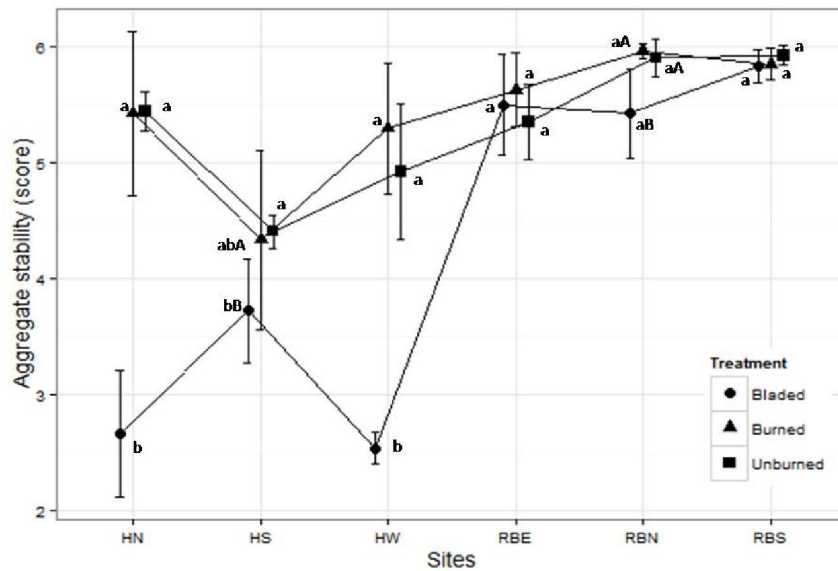


Figure 3.6. Treatment effects on bulk density by site. Different lower case letters indicate significant difference at $p \leq 0.05$ and upper case letters indicate difference at $p \leq 0.1$ within sites

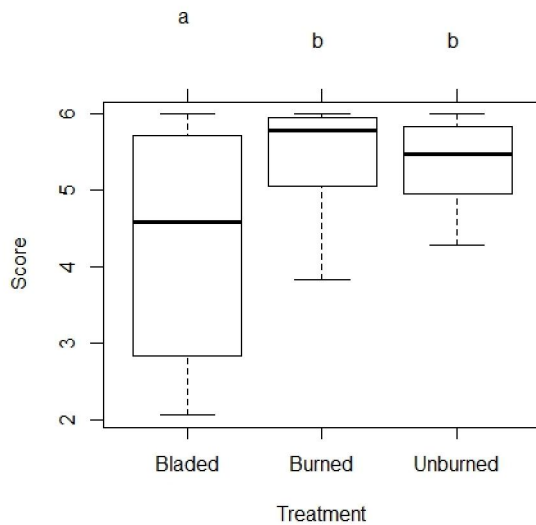


Figure 3.7. Treatment effects on soil aggregate stability. Different letters indicate significant difference ($p = 0.05$) between treatments

Construction of bulldozed fireline definitely changed soil aggregate stability.

Treatment effects were not consistent across the sites. Aggregate stability on bladed

treatment in Havre sites was significantly lower relative to burned and unburned.

Aggregate stability on bladed treatment in Redbluff sites was similar except RBN site relative to burned and unburned treatments. While burning generated stronger aggregates but not statistically significant on HW, RBE and RBN sites relative to unburned treatments (figure.3.8).

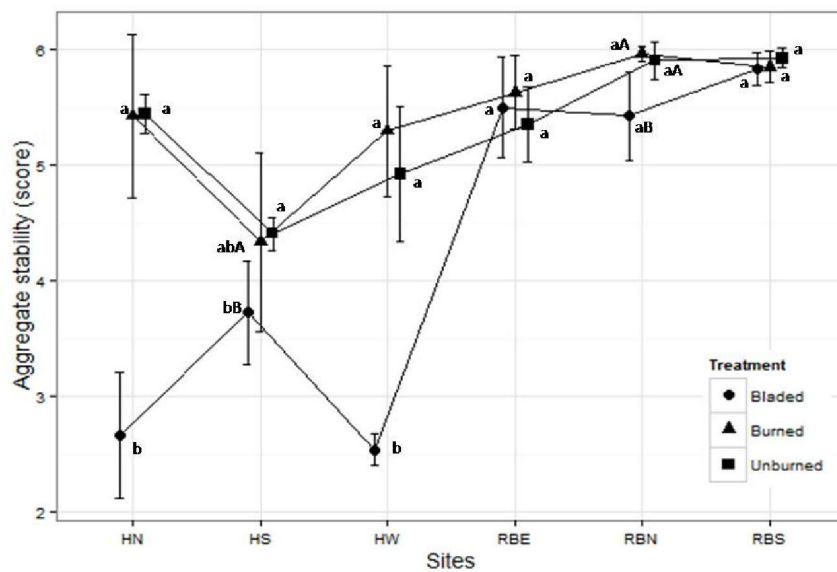


Figure 3.8. Treatment effects on soil aggregate stability by site. Different lower case letters indicate significant difference at $p \leq 0.05$ and upper case letters indicate difference at $p \leq 0.1$ within sites

Rainfall Simulation

Runoff. Pairwise comparison results from Tukey Honest Significant Difference (HSD) test performed on ANOVA model of cumulative runoff detected significantly higher cumulative runoff on bladed treatment relative to unburned (p -value=0.00001). It failed to detect significant difference on burned relative to unburned (p -value=0.06) and bladed treatments (p -value=0.058) (figure 3.9).

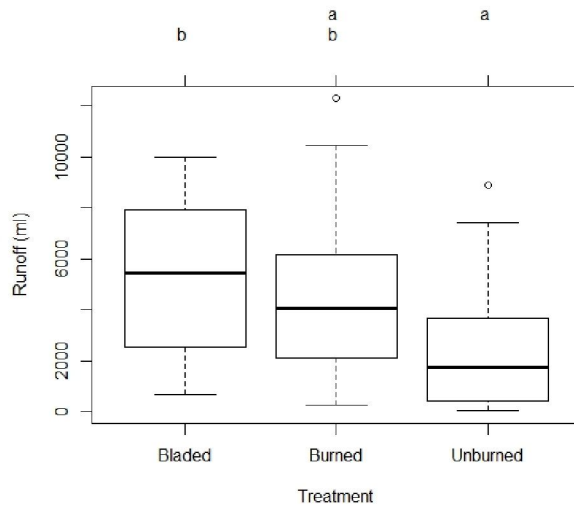


Figure 3.9. Treatment effect on cumulative runoff. Different letters indicate significant difference ($p=0.05$) between treatments

Cumulative runoff was highest on unburned followed by burned and unburned throughout the 30 minute rainfall simulation but none of them were significant from each other except at 5th minute (figure.3.10).

Sediment Yield. Pairwise comparison results from Tukey Honest Significant Difference (HSD) test performed on ANOVA model of cumulative sediment yield detected significant higher sediment yield on bladed treatment relative to unburned (p -value=0.00013) and burned (p -value=0.0001). It failed to detect significant difference between unburned and burned (p -value=0.5) (figure 3.11).

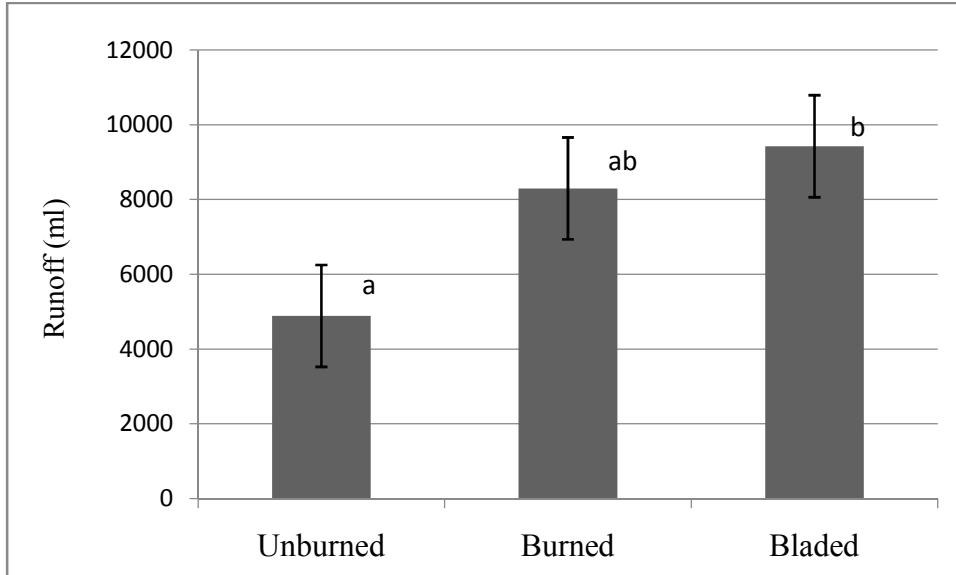


Figure 3.10. Treatment effect on cumulative runoff. Different letters indicate significant difference at $p \leq 0.05$

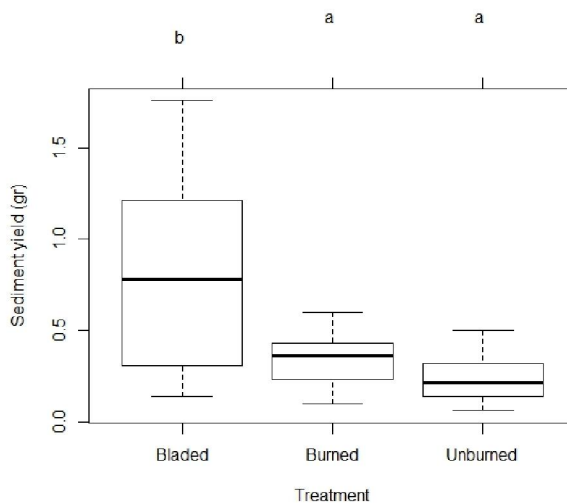


Figure 3.11. Treatment effect on cumulative sediment yield. Different letters indicate significant difference ($p=0.05$) between treatments

Treatment effects on cumulative sediment yield were consistent throughout 30minute of rainfall. Runoff from bladed treatment was highest followed by burned and unburned treatments (figure.3.12).

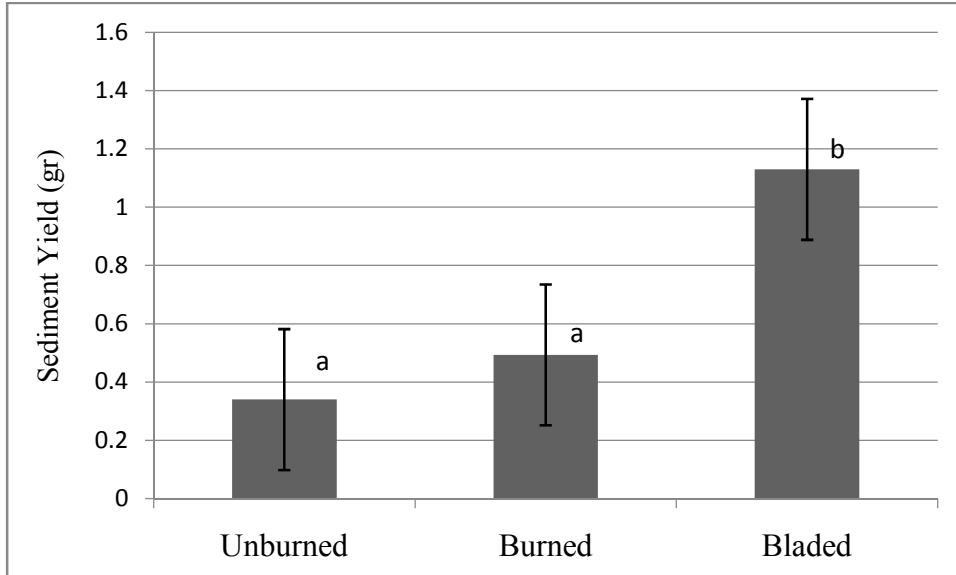


Figure 3.12. Treatment effect on cumulative sediment yield. Different letters indicate significant difference at $p \leq 0.05$ within sites

Discussion

Overall, the bladed treatment had a significant impact on the “A” horizon depth compared to burned and unburned. But depth of firelines was not the same across sites. Depth of the “A” horizon on bladed treatment was shallower in Havre sites, while it was almost the same compared to unburned treatment in Redbluff sites. It depended on slope, whether parallel or perpendicular to contour line, soil type, surface roughness and as well as a bulldozer operator’s skill. The fireline on RBS and RBE sites appeared to be built carefully as it just scraped plant cover and barely touched the soil surface. These two sites were less rocky and soil surface was less rugged and this likely allowed for more precision on fireline construction. Fireline construction was rougher in RBN site relative to RBS and RBE as boulders and rocky soil prevented the operator from maintaining a constant fireline depth. At Havre the firelines were constructed on steep slopes with

rocky surfaces and parallel to the contour line, which caused a deep excavation on the uphill side, a buried berm on downhill side and a moderate scrape in the middle. In some cases the uphill side was dug down to a depth of 50 cm.

Removal of the “A” horizon altered soil bulk density and aggregate stability. The deeper the fireline the higher the bulk density and the lower the aggregate stability. Because the A horizon is rich in organic matter it is less dense with highly aggregated particles (Chenu et al. 2000). Higher bulk density is an indication of less porosity, less water holding capacity and high compaction, which impedes plant root growth (Gomez et al. 2002). Whereas lower aggregate stability is an indication of weak aggregation of soil particles that would be disrupted easily by outside force, such as raindrop. Detached soil particles clog soil pores and reduce water infiltration rate into the soil, which eventually cause surface runoff (Shainberg et al. 1992).

Results from runoff data obtained from rainfall simulation experiment were consistent with our original hypotheses the bladed treatment was a significant factor in increasing the amount of surface runoff during our experiment. The burned treatment generated less runoff relative to bladed but higher relative to unburned although neither of them were significant. Our results were consistent with those found in similar studies indicating that plant cover is an important factor in dictating runoff because of higher infiltration rates associated with greater plant cover (Roundy et al. 1978; Seyfried 1991). Petersen (2004) found infiltration increased as a result of increased herbaceous cover and greater surface litter whereas bare soil was highly correlated with increased runoff.

Results from sediment yield data indicate that the bladed treatment was a significant factor in increasing sediment yield. On exposed dry soils, this reactive force of raindrop can entrain and transport soil particles (Bryan 2000). Since our data was collected from one site, these results cannot be conclusive. Increasing the sample size may have promoted a stronger relationship of the effects of plant foliar cover and slope on total runoff. The bladed treatment effect on runoff and sediment yield may have varied depending on the site because the bladed treatment on RBS site had the lowest slope and lowest percent cover relative to other sites. Steeper slope, greater annual grass and annual forb foliar cover in HW or HS site may also have generated a different response.

Conclusion

The results of this study clearly indicated that bulldozed fireline construction altered soil attributes bulk density, aggregate stability and reduced top soil horizon depth. Construction of bulldozed fireline created easily erodible soil with disrupted particles. Denser litter and higher canopy cover on unburned treatment provide barriers to overland flow by increasing the tortuosity of flow and increasing residence time of water on the soil surface, which enhances infiltration (Cline et al. 2010). As time goes, the burned sites will accumulate litter which will facilitate hydrologic cycle by increasing residence time of water on the soil surface. Slope of fireline is one of the important factors that dictates erosion potential (Christie et al. 2013), but we did not have enough sample to examine impacts of steeper .

CHAPTER 4

GENERAL CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Conclusions

Costs related to wildfire suppression has drawn increased attention of scientists, state and federal agencies. Most the time ecological costs of fire suppression activities are not acknowledged yet most of the wildland fire suppression techniques have adverse effect on ecosystem. One of the common and widely used, as well as the most destructive technique is bulldozed fireline. This technique is considered effective and cost efficient when there is a need of great amount of fuel removal especially in forest system, and when something valuable is at risk such as house and livestock. United States Forest Service has a regulation regarding the use of bulldozed fireline techniques and restricts its use on slopes less than 60% slopes, and requires rehabilitation action to reduce ecological impacts such as reseedling, mulching. However ranchers and land owners consider bulldozed fireline as an emergency response to wildland fire. As a result firelines are built without any pre-planning and without knowing the future consequences. The objectives of this study was to quantify the ecological cost of bulldozed fireline in comparison to burned and unburned sites and draw a management implications for ranchers and land owners. Vegetation cover and abundance were determined with biomass clipping and line point intercept method as separate functional groups across the bladed, burned, and unburned sites in two locations, north central and south west Montana for two years. Study area encompassed idaho fescue and bluebunch wheatgrass

dominated north facing slope in Redbluff and rough fescue and bluebunch dominated grassland in Havre. After quantifying vegetation responses, top soil horizon depth, bulk density, aggregate stability and runoff and sediment yield were determined to quantify soil response to bulldozed fireline. In general, burned site did not differ from unburned site particularly by the second year after the fire. In other words fire was not detrimental to the rangeland condition. Whereas bulldozed fireline altered primary ecological processes, particularly nutrient cycle and hydrologic functioning, by mechanically removing native plant species, creating exposed bare ground which was susceptible to soil erosion and invasion of competitive non-natives. Perennial plant and litter cover and abundance significantly decreased on bladed treatment relative to unburned and burned treatments while non-native annual species were significantly higher on bladed relative to unburned and burned treatments. Cheatgrass, in particular came to the bladed sites first due to its well dispersed seed and superior ability to use water, time and soil nutrients. All bladed sites were invaded by cheatgrass by the second year after the fire. Altered soil properties, higher bulk density, lower aggregate stability and thinner A horizon and more bare ground generated higher runoff and higher sediment yield from bladed treatment relative to unburned and burned treatment.

Management Implications

This research will aid land management prescriptions that target reduction of ecological impacts associated with wildland fire suppression activities, particularly bulldozed fireline. The results from this study suggest that bulldozed fireline triggers exotic non-native annual invasion and initiates soil erosion. Anecdotal evidence suggest

that these changes could exist for long term but our data will not allow us to evaluate the long-term nature of these changes. Now question arises that what did we save by suppressing the fire with the fireline? The answer depends on what was in danger. For this case, construction of bulldozed fireline saved pasture for winter grazing with a cost of increased abundance of cheatgrass and loss of perennial bunch grass.

Therefore, based on this study result, we would suggest ranchers and land owners that avoid constructing bulldozed fireline instead create pre-planned fireline system that is incorporated with land use management. Road system that goes through the ranch and along the fence line, ridge top with scarce plant cover, cliffs, rivers, and hiking trails can serve as a fireline with multiple sections. So that ranchers or land owners would not worry for losing all the forage that has been reserved for a certain season. However, when there is need of bulldozed fireline we would suggest following to reduce the ecological impacts:

- A. Minimize depth of bladed and from steep slopes and prominent rocky surface. Slope increases soil erosion potential and generates a substantial excavation on the uphill side. Prominent rocks cause unevenly scraped surface. .
- B. Re-seed the fireline right after putting the blade. For most desired perennial grasses that regrow from seed, seed bank likely gets depleted by exposure to ground surface or off-placement with berm. There could be a window of a one or more growing season depending on distance from closest cheatgrass invaded site for perennial grass establishment in intact sites before cheatgrass takes over.

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APPENDICES

APPENDIX A

PHOTOGRAPHY OF STUDY SITES



Figure 1. Photography of Redbluff



Figure 2. Photography of Redbluff

APPENDIX B

PHOTOGRAPHY OF RAINFALL SIMULATOR
AND SINGLE RING INFILTROMETER

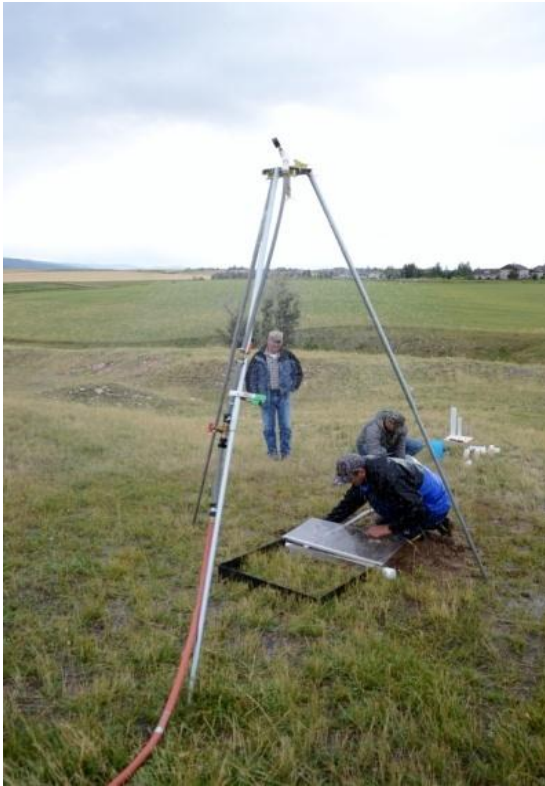


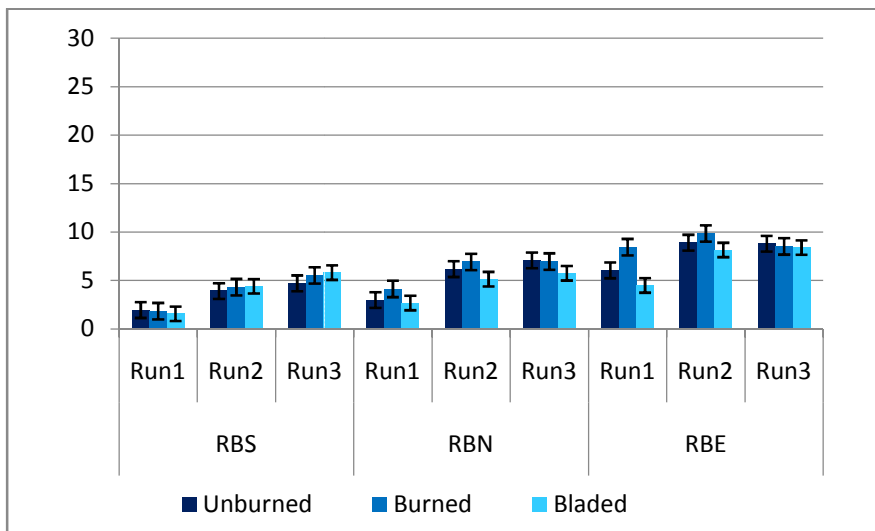
Figure 1. Photography of Single Nozzle Rainfall Simulator



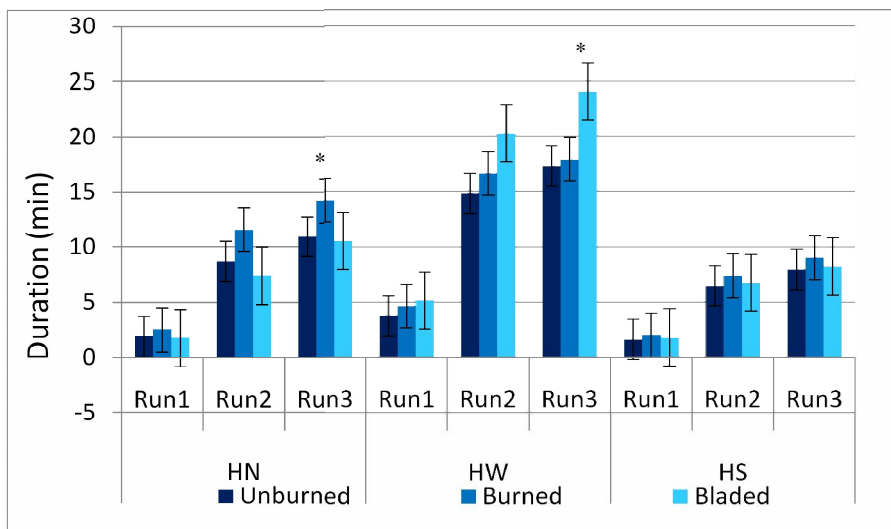
Figure 2. Photography of Single Ring Infiltrometer

APPENDIX C

INFILTRATION RATE MEASURE DATA



Graph 1. Infiltration rate measurement result of Redbluff sites



Graph 2. Infiltration rate measurement result of Havre sites