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VEGETATION RESPONSE TO POST-WILDFIRE SEEDING: BULL CANYON WILDFIRE

Prepared by:
Reg Newman, Rick Tucker, and Percy Folkard

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Sustainability of Forest and Range Resources Through Science and Stewardship

MINISTRY OF FORESTS, LANDS AND NATURAL RESOURCE OPERATIONS
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Management of forest and range resources is a complex process that often involves the balancing of ecological, social, and economic considerations. This evaluation report represents one facet of this process. Based on monitoring data and analysis, the authors offer the following recommendations to those who develop and implement forest and range management policy, plans, and practices.

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EXECUTIVE SUMMARY

Wildfires have the potential effect of inhibiting understorey plant communities and can therefore reduce livestock and wildlife grazing opportunities, as severe burns can damage or remove forage species and create opportunities for colonization by nuisance weeds and invasive plant species. Post-wildfire aerial seeding of agronomic species is sometimes used in BC to suppress weed establishment and replace or enhance forage production. The level of burn severity is the main factor used to determine if seeding would be effective at meeting these objectives and potentially influencing livestock distribution. This study examined the forage plant community response and level of weed invasion to test the appropriateness of seeding into high and moderate severity vegetation burn areas and moderate and low severity soil burn areas.

A seed mix of equal parts of orchardgrass, Italian ryegrass, and white clover was aerially applied at 10 kg/ha onto a mixed Douglas-fir, lodgepole pine forest that burned in the summer of 2010. Seeding took place in October 2010. Cover and biomass was sampled on two moderate soil burn severity sites and one low soil burn severity site.

The seeding treatment had variable effects on understorey species richness, but tended to increase species richness overall. The seeding treatment reduced the cover of several native species; however, it is too early to determine whether these will be important effects in the long term.

There was some evidence of slight positive weed suppression due to the seeding; however, the study sites were never under great threat from problematic weeds so the overall weed control benefit was negligible. Seeding for weed control will be more practical on sites that are at greater risk of invasion by problematic weeds.

There was evidence that seeding resulted in improved grazing distribution of feral horses in burned forest areas and hence reduced use of an overused wet meadow complex; however, it is not clear whether the fire alone would have resulted in the same benefits without the seeding treatment.

White clover established poorly on all sites, averaging only 0.34% cover, possibly due to the timing of the fall seeding. Italian ryegrass performed as expected, establishing quickly to moderate abundance in the first year after seeding, then declining rapidly in the second year. Orchardgrass established well and contributed substantial and palatable forage for four years. Overall, the seeded sites did not produce more forage than the unseeded sites. This was likely due to the low to moderate soil burn severity at our sites resulting in low mortality of native forage plants. The study provides evidence that seeding for the purpose of forage replacement or forage enhancement is unnecessary and ineffective at post-wildfire sites with moderate or lower soil burn severity ratings in the IDFdk4 and similar Douglas-fir – pinegrass range types. We recommend that future post-wildfire seeding should only be considered on high severity vegetation burn sites and high severity soil burn sites. This decision should be supported by a proper assessment and mapping of vegetation/soil burn severity coupled with an understanding of the invasive species threat in the area.

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1.0 INTRODUCTION

About 2000 wildfires occur in British Columbia every year, affecting an average of 115 464 ha annually. Many of these wildfires occur on lands tenured for livestock grazing in the BC Interior and can therefore have important effects on the forage resource. Moderate to severe intensity wildfires will reduce the current season's forage biomass, alter plant species composition, and may result in long-term loss of desirable forage plants. Loss of understorey vegetation can result in increased susceptibility to invasive plants. On the beneficial side, wildfires can increase cattle access to areas previously unavailable due to excessive windfall or high tree density. Forage may also be increased due to partial or total removal of the tree canopy by fire, which will allow more light to reach the understorey.

Broad-scale seeding of burned areas is a practice that can be used to remedy some of the undesirable effects of wildfire on the forage resource. Post-wildfire aerial seeding of agronomic species is sometimes used in BC for the objectives of suppressing weed establishment, replacing or enhancing forage production, and in taking advantage of an opportunity to improve livestock distribution (BC Ministry of Forests, Range and Natural Resource Operations 2015c, Dobb and Burton 2013).

This report outlines the results of a study that tested the effectiveness of post-wildfire agronomic forage seeding at improving forage production and suppressing weed establishment in forested rangelands burned by the Bull Canyon wildfire near Alexis Creek, BC. The severity of a wildfire is an assessment of the fire effects on trees and understorey vegetation, and can also assess effects on soil (e.g., Robichaud et al. 2000). The intensity of the fire at the study sites was low to moderate on average and allowed an examination of seeding effects at burn severities nearing the lower threshold at which seeding would be considered by range managers.

1.1 *Seeding for Weed Suppression*

Post-wildfire seeding can reduce the available seedbed for invasive plant establishment by achieving a quick cover of competing vegetation. Native vegetation recolonizes a severely burned area slowly (~4 years) depending on the distance to seed sources and dispersal rates, whereas broad-scale seeding usually provides good cover within 1-2 years (Anderson and Brooks 1975, Goodwin et al. 2002, Dyrness 1967, McClure 1958, Wilson 1949). Positive weed suppression using post-wildfire seeding has been demonstrated in many studies (e.g., Newman 2007, Ketcheson 1989, McClure 1958, Pettit 1968, Wilson 1949). On the 2003 Strawberry Hills wildfire[1] near Kamloops, BC, cheatgrass (*Bromus tectorum*) frequency was reduced by 41% on areas seeded to agronomic species. The short-lived agronomic species Italian ryegrass (*Lolium multiflorum*) achieved rapid establishment and high cover within 1 – 2 years post-fire and was thought to be the major factor in reducing subsequent cheatgrass infestations (Newman 2007). This technique was particularly effective against invasive species that colonize by wind-blown seed. Seeding in itself carries the risk of introducing invasive plants, other weeds or unexpected agronomic species due to contamination of the seed mix. Using the highest quality seed mix available or conducting independent tests on the seed mix may reduce this possibility.

1.2 *Seeding for Forage Replacement and Forage Enhancement*

Forage replacement by post-wildfire seeding may be an option in cases where severe intensity wildfire results in mortality of large areas of understorey vegetation. For example, pinegrass (*Calamagrostis rubescens*), an important forage species of forested range, can be killed by wildfires that consume the duff (FH) layers of the forest floor. Although pinegrass is not often eliminated completely, it may require many years to recover to pre-wildfire levels in this situation. Post-wildfire seeding of agronomic species can be used to quickly re-establish forage to pre-wildfire levels.

Forage enhancement by post-wildfire seeding may be an option when wildfire results in thinning of the understorey and where the existing forage species do not meet the forage objectives for quantity and quality in the management unit. Post-wildfire seeding of a short-lived agronomic species such as orchardgrass will result in an understorey of native forage species interspersed with productive and nutritious seeded plants. The replacement of native forage species with agronomic forage species in forestry clearcuts can more than double forage production (Youwe et al. 1991, McLean and Clark 1980). There is

[1] The Strawberry Hills wildfire started August 1, 2003 near Kamloops BC and burned 6 000 ha. Selected areas were seeded in May 2004 leaving equal unseeded areas. Post-wildfire vegetation monitoring occurred from 2004 through 2007. The area is in the Interior Douglas-fir very dry hot (IDFxh1) biogeoclimatic subzone and was characterized by scattered mature Douglas-fir and dense immature ponderosa pine (*Pinus ponderosa*), with pinegrass (*Calamagrostis rubescens*) in the understorey, and bluebunch wheatgrass (*Pseudoroegneria spicata*) and rough fescue (*Festuca campestris*) in the openings (Newman 2007).

also the potential for increased livestock weight gains on improved forage species, particularly later in the growing season, which could supplement the low nutrient profile of late-season pinegrass (McLean 1969).

1.3 *Other Objectives of Seeding*

Post-wildfire seeding using palatable agronomic species can be used to alter the distribution of livestock. Livestock have distinct preferences for some forage species. Strategically locating areas of palatable forage has been recommended as a method of attracting cattle away from overused preferred areas such as riparian zones (Fraser 2007, Gillen et al. 1984). Hence, broad-scale post-wildfire seeding of palatable forage species may have the effect of attracting livestock away from overused areas into newly created forage areas.

The use of agronomic species for erosion control and weed suppression is a conventional practice on fireguards, fire access roads and trails, stream crossings, staging areas, and sumps (Hope et al. 2015). However, broad-scale post-wildfire seeding for erosion control on areas undisturbed by fire-suppression activities is generally considered to be ineffective (Robichaud et al. 2000). As well, establishment in burned areas often differs little from that of non-burned areas, further suggesting that post-wildfire seeding for erosion control is unnecessary (Stark et al. 2006).

1.4 *Study Objectives*

This study was initiated to determine whether current post-wildfire seeding practices meet forage production replacement and weed suppression objectives. The primary objective of the study was to determine if post-fire forage production is affected by an aerially broadcast agronomic seed mix on high and moderate severity vegetation burn areas. The development of native plant species was also monitored to determine the effects of seeding agronomic species.

Secondary objectives included:

1. Determining if post-fire invasive plant establishment is affected by an aerially broadcast agronomic seed mix on high and moderate severity vegetation burn areas.
2. Determining if post-fire native plant community development is affected by an aerially broadcast agronomic seed mix on high and moderate severity vegetation burn areas.
3. Determining if seeding affects ungulate distribution.
4. Assessing the performance of three agronomic forage species in post-wildfire application.

2.0 BACKGROUND AND METHODOLOGY

2.1 *Study Area*

The study area was located in the Haines Creek Range Administrative Unit close to a complex of wet meadows and grasslands locally known as the “Big Open.” The three sites sampled were located about 12 km southwest of Alexis Creek, BC (52° 0′37.44″N 123°24′43.33″W). The sites occur in the Interior Douglas-fir dry cool biogeoclimatic zone and subzone (IDFdk4) (Steen and Coupé 1997). Elevation and slope at the sites average 1040 m and 0 - 5 %, respectively.

Mean annual precipitation modelled by ClimateBC is 395 mm with 52% occurring during the growing period (Centre for Forest Conservation Genetics 2016). Bimodal peaks in precipitation occur in December (42 mm) as snow and in June (48 mm) as rain. Highest average daily maximum temperatures occur in July (22 °C); coldest average daily minimum temperatures prevail in January (-15.5 °C) (see Figure 1).

Soils are predominantly dystric brunisols. Before the fire, the area was a mosaic of uneven-aged forests dominated by 20- to 30-year-old lodgepole pine (*Pinus contorta var. latifolia*) and Douglas-fir (*Pseudotsuga menziesii*) with scattered older Douglas-fir (~270 years old). The stand history includes a severe mountain pine beetle infestation that peaked in 1983 (Alfaro et al. 2010). The pre-wildfire understorey plant community was characterized by pinegrass (*Calamagrostis rubescens*) and heart-leaved arnica (*Arnica cordifolia*).

The invasive plant, scentless chamomile (*Matricaria perforata*), occurs along roadsides near the sites. Other *Forest and Range Practices Act* (FRPA) Invasive Plant Regulation listed species found in the area include leafy spurge (*Euphorbia esula*), spotted knapweed (*Centaurea biebersteinii*), hoary alyssum (*Berteroa incana*), and common tansy (*Tanacetum vulgare*) (BC Ministry of Forests, Range and Natural Resource Operations 2015b).

There were no cattle in the area during the study period; however, feral horses were observed during aerial surveys from 1991 - 2009. A 2007 survey counted 11 horses near the “Big Open” wet meadow complex (Fraser 2007). The feral horse population in the Haines Creek Unit overall was as high as 442 during 2008, declining to 313 in 2009 (Hamilton 2009).

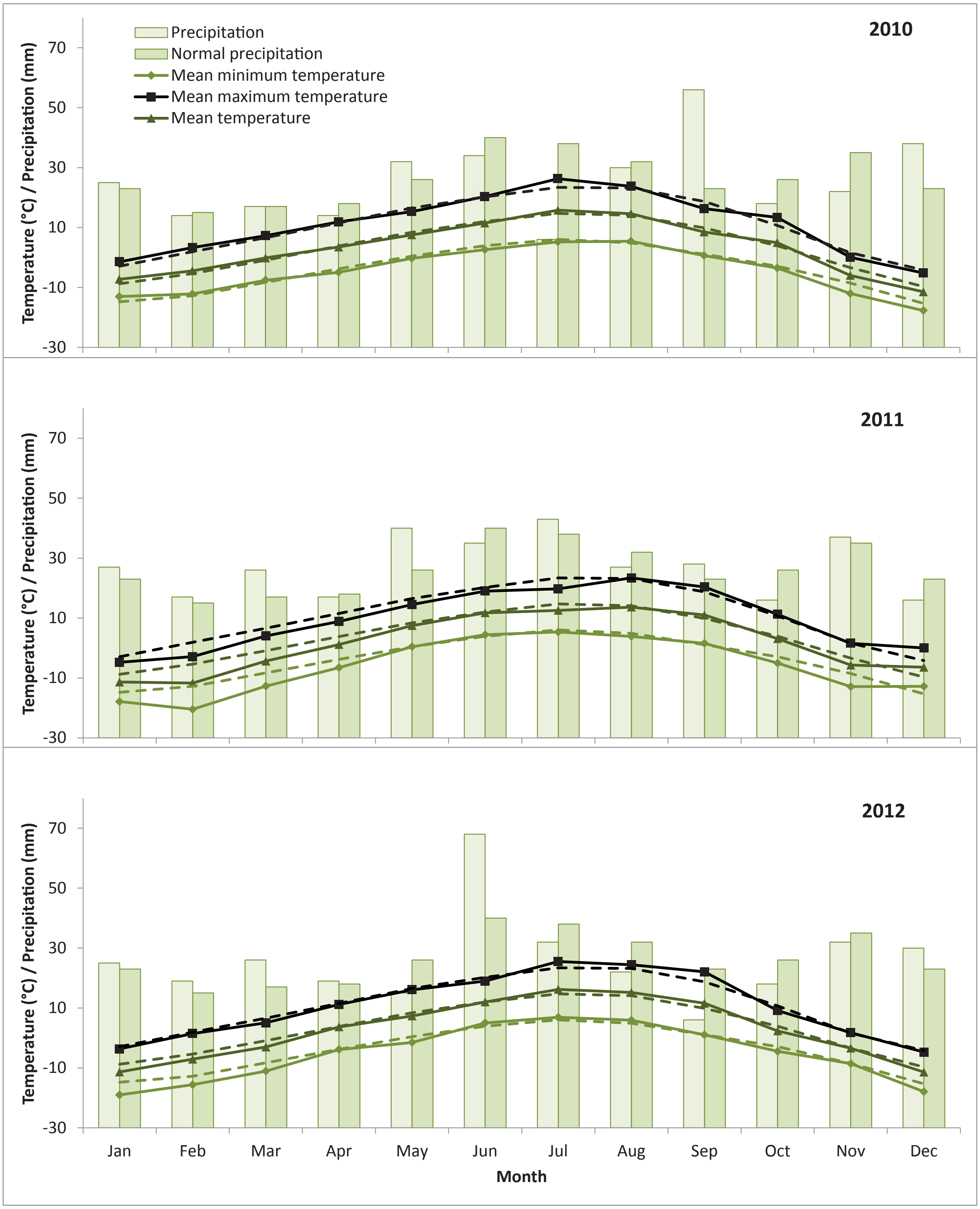
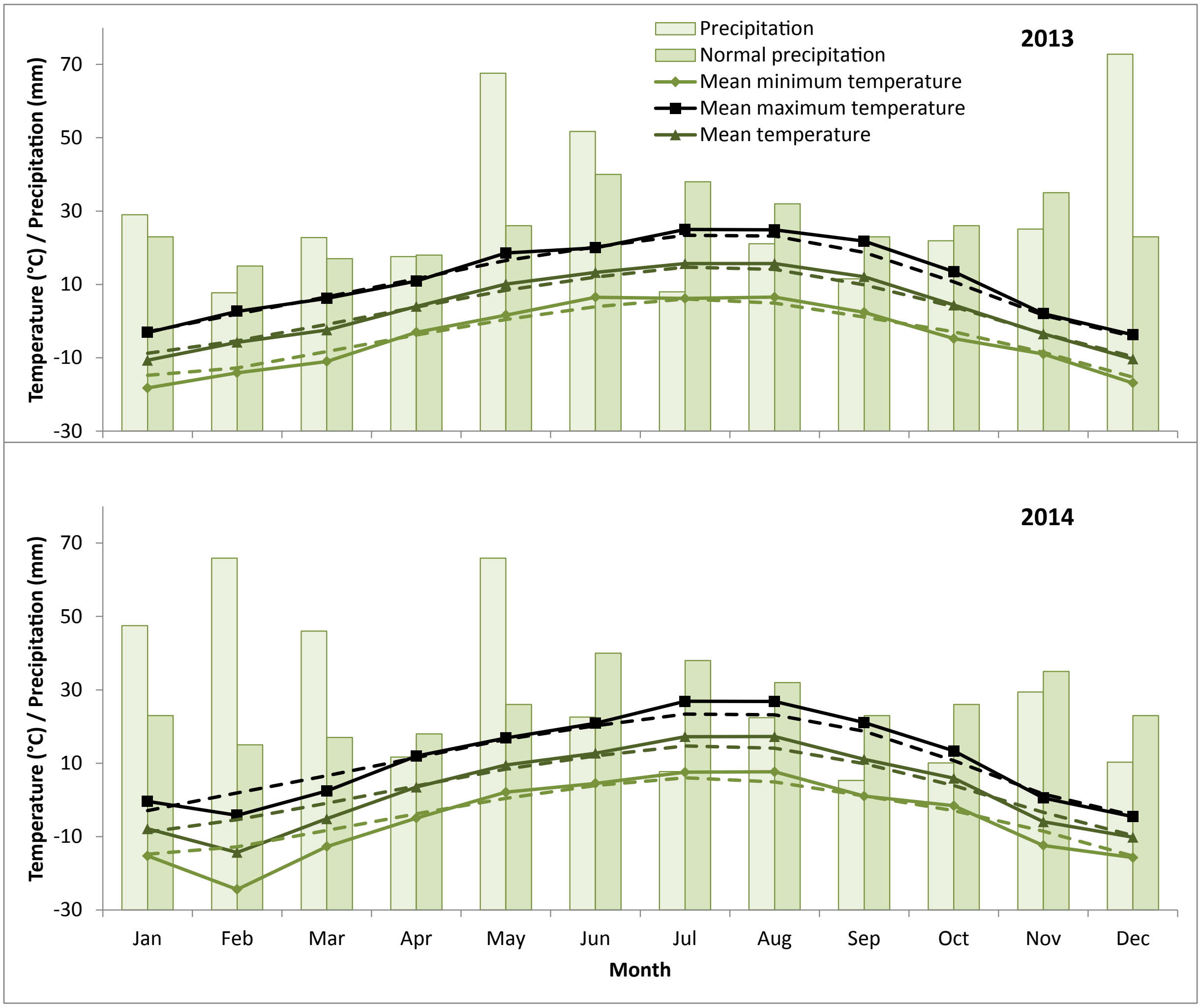


Figure 1. Mean monthly temperatures and total monthly precipitation for 2010 through 2014 (solid lines) from Puntzi Mountain (909 m elevation; 50 km from Bull Canyon wildfire study site) (Environment Canada 2016) and mean monthly temperature normals and total monthly precipitation normals for 1981–2010 (dashed lines) modelled using ClimateBC (Centre for Forest Conservation Genetics 2016)

Figure 1. (continued)



2.2 *Wildfire and Seeding Treatments*

The Bull Canyon complex of fires started on July 28, 2010 from a lightning strike on the south side of the Chilcotin River near the rim of Bull Canyon. The fire spread rapidly due to dry and windy conditions, reaching 35 000 ha in size (BC Ministry of Forests, Lands and Natural Resource Operations 2015a). Following the fire, a decision was made to seed some burned areas to take advantage of an opportunity to increase the quantity and quality of forage in order to improve livestock distribution and address potential weed concerns. The area was aerially seeded by fixed-wing aircraft in October 2010 at 10 kg/ha using a mix[2] of Italian ryegrass (*Lolium multiflorum*), orchardgrass (*Dactylis glomerata*), and white clover (*Trifolium repens*) (see Table 1). A centrally located strip (7 km x 100 m) following a northwest bearing was left unseeded for use as a control (see Figure 2). Growing season precipitation during the first year of germination and establishment (2011) was very close to normal (see Figure 1). Growing season precipitation in 2012 was also close to normal except that June precipitation was notably above normal. May precipitation was above normal and July was below normal for 2013 and 2014, but otherwise close to normal (see Figure 1). The growing season mean temperature was cooler in 2011 by 0.5 °C compared to normal, but warmer by 0.6, 1.5 and 1.6 °C for 2012, 2013 and 2014, mostly due to differences from July through September (see Figure 1).

[2] Seed mix regionally known as BCMF RB #5 mix.

Table 1. Composition of seed mix used on the Bull Canyon wildfire.

Forage species	Proportion by weight (%)	Proportion by kernel count (%)	Seeds per m² (at 10 kg/ha rate)
Italian ryegrass	55	28	276
Orchardgrass	30	45	432
white clover	15	27	264

2.3 Site Selection

The vegetation burn severity of the forest was visually estimated from low-level aerial photographs (see Figure 2) taken within two months post-wildfire. Two sites were selected in high severity vegetation burn areas using definitions similar to those provided by Hope et al. (2015). High severity vegetation burn conditions occur when canopy trees are blackened and dead, the needles are consumed, and the understorey is burned (see Table 2). A third site was selected in a moderate severity vegetation burn area. Moderate severity vegetation burn conditions occur when trees are burned and dead, but scorched needles remain on canopy trees, and the understorey is burned and blackened. Sites were located near the unseeded control strip to allow for comparison of the seeded and unseeded treatments. Sites were also selected based on availability of paired (homogenous) areas within the seeded and adjacent unseeded strip.

Soil burn severity is a more sensitive indicator of damage to underground plant structures and propagules than vegetation burn severity. Upon field inspection, both sites with high severity vegetation burn conditions were found to have moderate severity soil burn conditions and the site with a moderate severity vegetation burn class was found to have a low severity soil burn condition (see Table 3).

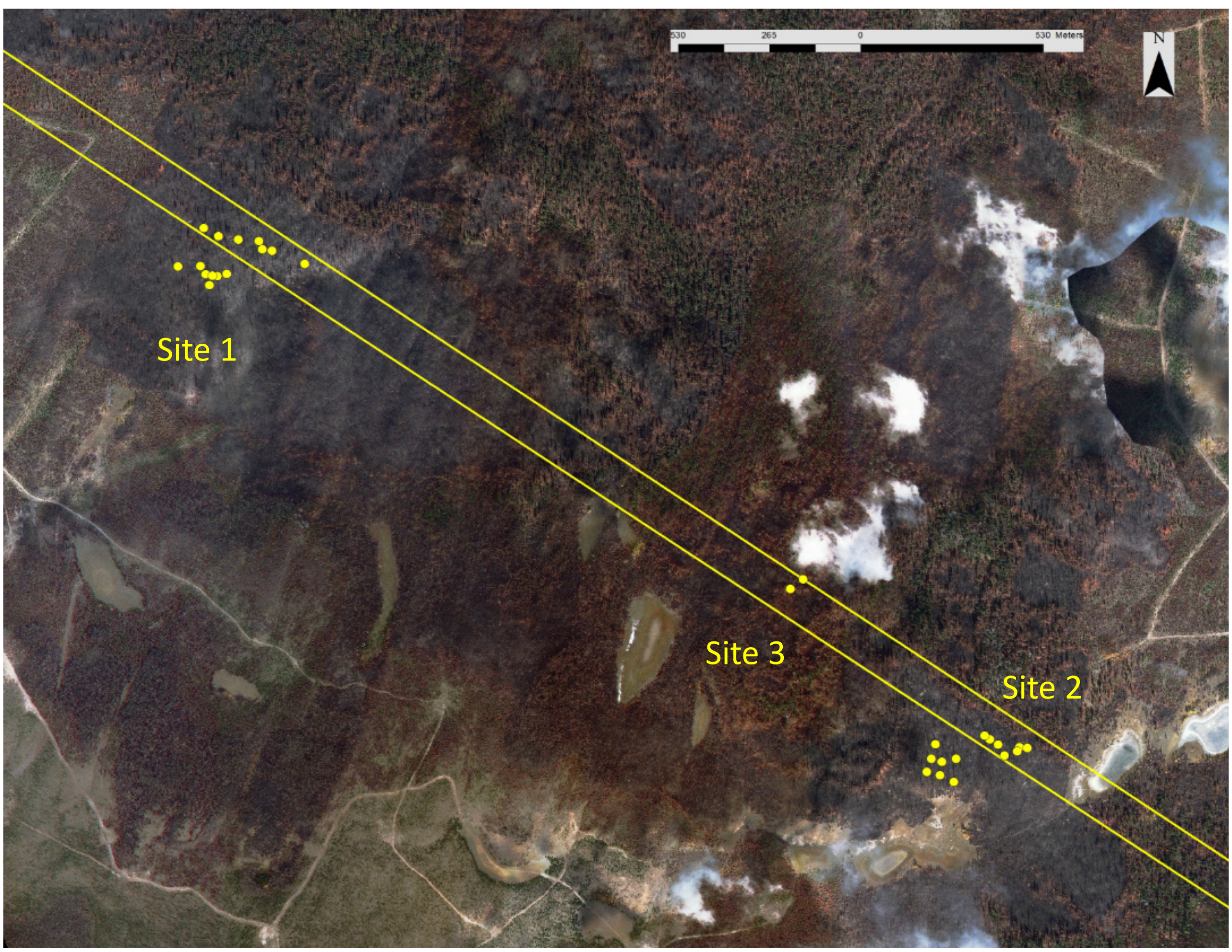


Figure 2. Location of sites and macro-plot clusters within the Bull Canyon fire in relation to the unseeded strip (between yellow lines).

Table 2. Vegetation burn severity classification for a coniferous forest, and its relationship to Burned Area Reflectance Classification (BARC) and soil burn severity classes (Hope et al. 2015).

Vegetation burn severity class	Definition	BARC class	Typical soil burn severity class
High (black)	Canopy trees blackened (charred) and dead, needles consumed, understorey burned	High	H or M
Moderate (brown or red)	Trees burned and dead, scorched needles remain on canopy trees, understorey burned and blackened	Moderate	M or H, can be L
Low (green)	Canopy unburned, trunks partially burned, understorey lightly or patchily burned	Low	L, can be M or H
Unburned	Vegetation in natural unburned state	Unburned	

Table 3. Soil burn severity classification based on post-fire appearance, and forest floor and soil properties (Hope et al. 2015).

Soil and forest floor factors	Soil burn severity		
	High	Moderate	Low
Litter	Consumed	Consumed	Scorched, charred, patchily consumed
Duff (FH layers)	Consumed in most locations	Deep char, may be consumed	Intact, surface char
Woody debris – small	Consumed	Partly–completely consumed	Partly consumed, charred
Woody debris – logs	Many consumed, others deeply charred	Charred	Charred
Ash colour (if still present)	Fine, white or grey	Greyish or blackened	Black
Mineral soil exposure (may still be covered with loose ash or charcoal)	>40%	<40%	Little
Mineral soil	Altered structure, porosity, etc.; often grey or reddish around burned large fuel; often strongly water repellent	Unchanged; water repellency is slight or patchy	Unchanged
Depth to live roots or rhizomes (in mineral soil)	>5 mm	0-5 mm	0 mm

2.4 Macro-plot Establishment

At Sites 1 and 2, for both unseeded and seeded areas, seven macro-plot positions were randomly generated using the Hawth's Analysis Tools extension for ESRI's ArcGIS 10.2. A macro-plot consists of a 1 x 1 m forage cage location and an associated 7-m transect for vegetation composition measurements. At Site 3, five 10-m transects were systematically laid out along a 40-m baseline. Each transect and clipped plot location at Site 3 was considered the equivalent of a macro-plot established for Sites 1 and 2. The transects were permanently marked using hooked metal pins driven into the ground.

2.5 Measurements

2.5.1 Forage biomass

Portable cages (1x1x1 m) constructed of meshed wire panels were used to exclude grazing by domestic and wild ungulates at Sites 1 and 2. No cages were used at Site 3. Living above-ground biomass was estimated from the peak annual standing crop of grasses and forbs. One-half m2 areas were clipped to ground level in July of 2012, 2013 and 2014 and the plant litter was separated from the current year's growth of forbs and grasses. Cages were moved to new locations after clipping. Living plant material was sorted by species groups (native species and seeded agronomic plant species), stored in paper bags, and air-dried to minimize decomposition. Samples were oven-dried at 70°C to a constant weight and weighed to the nearest 0.1 g.

2.5.2 Canopy cover

The cover of all vascular plant species was estimated using the canopy coverage method in 20 cm x 50 cm frames (Daubenmire 1959). The cover of bare soil, litter and ash was also estimated. For Sites 1 and 2, seven plots were systematically taken every 1 m along the transects for a total of 49 plots per treatment unit. For Site 3, 10 plots were systematically taken every 1 m along five 10-m transects for a total of 50 plots per treatment unit. Plots were sampled on Aug 16 in 2011 and then in the second week in July for 2012, 2013 and 2014. The plot positions remained the same throughout the project.

2.6 *Experimental Design*

The experiment has one treatment factor (Treat) with two levels (unseeded control, 10 kg/ha seeding rate) and is repeated at three sites (Block). Sites were sub-sampled seven times at Sites 1 and 2 and five times at Site 3. The analysis was run separately for each year using a randomized complete block design (RCB) with sub-sampling. ANOVA (SAS 1988) was used to test for forage biomass and plant cover differences due to the treatment.

3.0 RESULTS AND DISCUSSION

3.1 *Plant Species Composition*

3.1.1 Seeded agronomic species

All three seeded species (Italian ryegrass, orchardgrass, white clover) established on the three sites to varying degrees. There was no evidence that severity of soil burn condition affected establishment of seeded species. In particular, establishment at Site 3, which had low soil burn conditions, did not differ much from establishment on Sites 1 and 2, which had moderate soil burn conditions (see Table 3 and Figure 3).

White clover

White clover established initially only at trace levels (0.33% cover), decreased to even lower levels by year 2 following the fire, and was mostly non-existent by year 3 (see Figure 3). There was some recovery of white clover at Site 1 in year 4, perhaps due to later germination of hard seed. Overall, the lack of initial cover establishment and subsequent loss by year 4 represents a failure for this species at this site. This species is expected to establish well in environments with annual precipitation greater than 400 mm (Dobb and Burton 2013) which means that the Bull Canyon fire site, at 395 mm, is only marginally suitable. There is evidence that white clover does not establish well when aerially seeded in November (before snowfall) or onto snow in BC (Brooke and Holl 1988). The poor establishment of white clover sown in fall/winter compared to spring is thought to be related to loss of viability following exposure to rapid freeze/thaw cycles (Brooke 1984). It is possible that our October seeding date was the reason for poor establishment at the Bull Canyon fire sites, although the borderline annual precipitation may have also contributed to poor performance. A similar weak catch of October-seeded white clover was documented on the Dog Creek fire[3] (see Appendix Figure A-1).

Italian ryegrass

Italian ryegrass initially achieved a consistent cover of just over 9% on all sites in year 1, then declined sharply on Sites 1 and 3 by year 2. By year 3, Italian ryegrass was reduced to trace levels on all sites (see Figure 3). This trend represents a typical lifespan for Italian ryegrass which is expected to persistent for only 1 to 2 years (Dobb and Burton 2013). For example, on the Dog Creek fire, Italian ryegrass achieved 9% cover one year after seeding then declined to trace levels (see Appendix Figure A-1). Newman (2007) reported 17% cover of Italian ryegrass one year following a May aerial seeding at 1.6 kg/ha onto a wildfire site near Kamloops BC. In that study, cover declined to trace levels by three years after seeding.

Orchardgrass

Orchardgrass initially established well at all sites, averaging 20% cover in year 1. Cover remained about the same in year 2 on all sites, but varied in behaviour by site in year 3. Orchardgrass increased on Site 2 in year 3, to reach the site maximum for the study period. Orchardgrass

[3] The Dog Creek fire, located 10 km north of the community of Dog Creek, burned 7 495 ha of forest starting on July 28, 2010. Burned areas were seeded with a mix of orchardgrass, Italian ryegrass, and white clover at ~10 – 15 kg/ha in Oct 2011. Post-wildfire vegetation monitoring was conducted during 2013 and 2014. The area is in the Interior Douglas-fir dry cool biogeoclimatic subzone and was dominated by 40-year old Douglas-fir with pinegrass understory.

cover decreased in year 3 on Site 3 and dropped below 10% by year 4. Cover remained about the same on Site 1 throughout the study period (see Figure 3). Orchardgrass was the only seeded species that retained good cover for the entire study period (see Table 3), although based on the trend in Figure 3, it has reached the maximum abundance on all sites and is now declining. Orchardgrass was also the most successful seeded species on the Dog Creek fire, achieving 30% cover after three years of steady increases (see Appendix Figure A-1).

Table 4. Significance of ANOVA tests (prob.>F) and mean cover values (%) for seeded agronomic plant species at the three sites at Bull Canyon wildfire for the first four years following the fire (2011 – 2014).

				Site					
	Significance			1		2		3	
Year	Block	Treat	Block*Treat	Seed	Unseeded	Seed	Unseeded	Seed	Unseeded
Italian ryegrass									
2011	0.405	0.002	0.937	9.80	0.00	9.49	0.61	8.45	0.00
2012	0.500	0.343	0.075	0.10	0.00	6.61	0.00	0.90	0.00
2013	0.500	0.378	0.306	0.05	0.00	0.97	0.00	0.05	0.00
2014	0.524	0.247	0.363	0.26	0.00	0.36	0.05	0.00	0.05
Orchardgrass									
2011	0.505	0.041	0.114	28.21	0.00	16.17	0.10	16.15	0.00
2012	0.496	0.014	0.703	22.96	0.00	15.48	0.00	19.90	0.90
2013	0.484	0.054	0.235	22.91	0.00	27.96	1.28	10.80	0.80
2014	0.586	0.063	0.229	21.84	0.00	24.95	0.00	9.55	1.25
White clover									
2011	0.500	0.008	0.968	0.31	0.00	0.31	0.00	0.40	0.00
2012	0.500	0.210	0.371	0.05	0.00	0.00	0.00	0.10	0.00
2013	-	-	-	0.00	0.00	0.00	0.00	0.00	0.00
2014	0.500	0.436	0.440	0.31	0.00	0.00	0.00	0.00	0.00

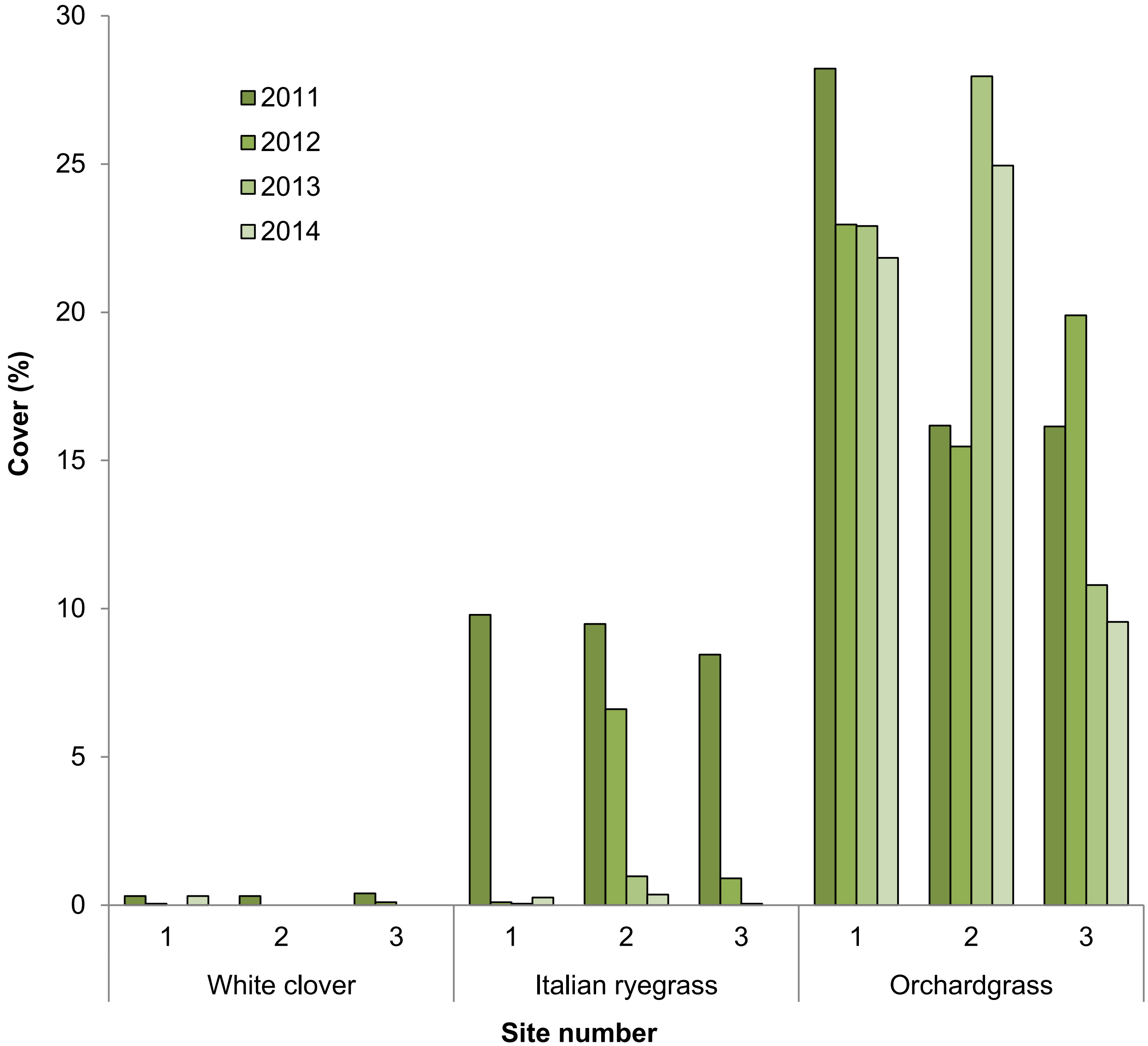


Figure 3. Seeded species cover at each site for seeded areas only for the first four years following the fire (2011 – 2014).

3.1.2 Invasive plants and other weeds

Of those species listed under the FRPA Invasive Plant Regulation and present in the surrounding area, none were found in the three burned sites sampled during the study. Other opportunistic weedy species (non-agronomic exotic plant species) were not abundant during the first three growing seasons following the wildfire with the exception of dandelion (*Taraxacum officinale*) at Site 2.

Dandelion was present at Site 2 during every year of the study, reaching about 2% cover by year 4. Site 2 was notably weedier than the other two sites. Site 2 also had low cover of annual hawksbeard (*Crepis tectorum*) and prickly lettuce (*Lactuca serriola*), and minor cover of three other weedy species (see Table 5). The seeding treatment reduced the cover of dandelion at all sites (P=0.092) in year 4, but this was a minor effect (0.41% vs. 0.80%).

*Table 5. Cover (%) of non-agronomic exotic invasive plant species at Bull Canyon Wildfire sites in year 4 following the fire (2014). **Significant seeding treatment difference at P<0.10.*

	Site					
	1		2		3	
Species	Seeded	Unseeded	Seeded	Unseeded	Seeded	Unseeded
Shepherd's purse	0.00	0.00	0.00	0.05	0.00	0.00
Lamb's-quarters	0.00	0.00	0.00	0.05	0.00	0.00
Annual hawksbeard	0.00	0.05	0.56	3.21	0.00	0.10
Prickly lettuce	0.00	0.00	0.00	1.79	0.00	0.00
Common plantain	0.00	0.00	0.05	0.00	0.00	0.00
Dandelion	0.00	0.20**	1.22	1.84**	0.00	0.35**

3.2 *Forage Production*

3.2.1 Site effects

Initial forage production was similar across all sites, despite differences in soil burn severity, and ranged from 1057 kg/ha to 1166 kg/ha. By the third growing season, forage production declined sharply to 712 kg/ha and remained at about the same level in year 4 (2014) (see Figure 4). This decline in production was not related to weather because total annual precipitation was uncorrelated and similar among the three years, ranging from 422 mm to 511 mm. The high initial forage production followed by a decline was likely due to short-term changes in availability of nutrients following wildfire. Ammonium and nitrate are reported to increase immediately after fire then return to pre-fire levels after one (Wan et al. 2001) to three years (Gundale et al. 2005). This pattern has been attributed to greater microbial mineralization of organic matter due to higher soil moisture and temperature following fire, combined with immediate release of organic nitrogen during the fire (Elliott and White 1986).

3.2.2 Seeding treatment effects

Agronomic species establishment

Italian ryegrass and orchardgrass established reasonably well in the first growing season (2011), achieving 9% and 20% cover by August of that year. Despite this, the seeding treatment did not result in increased total forage production at any site or year. Surviving and/or colonizing native plant species performed as well as seeded species in producing forage biomass. It is apparent that very few native forage plants were killed by the low to moderate soil burn severity conditions at the sites. Surviving native forage plants in the unseeded areas would have an early growth advantage over agronomic seedlings. Even though agronomic species like orchardgrass are generally more productive than pinegrass on an equal area basis, on the seeded areas the high survival of native plants meant that there was not sufficient growing space for a pure stand of agronomic plants. The seeded areas were characterized by a mix of native and agronomic plants with the agronomics never achieving greater than 35% of the total production.

Seeding slightly reduced native forage compared to unseeded areas in year 2 due to competition between agronomic and native plant species (see Table 5 and Figure 4). This effect did not carry into year 3. By the fourth growing season, agronomic species were almost absent from the plant community at Sites 1 and 3.

Forage production on the Dog Creek fire behaved in a similar way in that the unseeded areas produced the same biomass as the seeded areas (see Appendix Figure A-2). Agronomic plants formed a much greater proportion of the forage biomass on seeded areas (86 - 96%) at the Dog Creek fire.

Influence on livestock distribution

Although agronomic forage production did not increase total forage, it likely improved overall palatability. Palatability of orchardgrass (the main contributor to agronomic forage at the study sites) is reported to be high, while that of pinegrass (the main native forage species) is fair in the spring and becomes poor by autumn (Dobb and Burton 2013). A 2007 rangeland health evaluation of the Haines Creek Range Unit reported only light use of pinegrass in open cutblocks and in the understorey of live and dead pine forests, while the open meadows and wetlands received over-use from the combination of cattle and feral horse grazing (Fraser 2007). There was observational evidence that the Bull Canyon wildfire and our agronomic seeding treatment did increase feral horse use of burned forest areas and reduced grazing on the wet meadow complex (see Table 6). However, it was also reported for the adjacent Brittany Lakes Range Unit, that wildfire alone (without agronomic seeding), resulted in shifts from heavy feral horse use of the wet meadows to use of pinegrass stands in the burned forest (Mackenzie and Iverson 2005). Post-fire conditions increase the palatability of native forage species by increasing crude protein content and digestibility (e.g., DeByle et al. 1989, West and Hassan 1985).

Table 6. Significance of ANOVA tests (prob.>F) and mean production values (kg/ha) for agronomic, native, and total forage production at the three sites at Bull Canyon Wildfire for year 2 through year 4 following the fire (2012 – 2014).

	Significance			Site					
				1		2		3	
Year	Block	Treat	Treat*Block	Seeded	Unseeded	Seeded	Unseeded	Seeded	Unseeded
Agronomic									
2012	0.545	0.031	0.622	301	0	371	0	196	8
2013	0.500	0.082	0.496	173	0	256	0	73	0
2014	0.500	0.269	0.233	46	0	243	0	31	0
Native									
2012	0.841	0.079	0.900	596	1218	873	1090	784	1197
2013	0.604	0.308	0.208	508	635	473	1071	538	546
2014	0.507	0.130	0.566	461	749	557	912	717	757
Total									
2012	0.832	0.482	0.846	897	1218	1244	1090	980	1204
2013	0.329	0.625	0.495	681	635	729	1071	612	546
2014	0.183	0.209	0.686	508	749	799	912	748	757

Table 7. Observations supporting altered grazing distribution of feral horses at the Bull Canyon wildfire study area.

Date	Observation
July 2011	Heavy grazing use of wet meadow adjacent to Site 3.
	Grazed orchardgrass plants at Site 3. Horse manure.
July 2012	Light grazing use of wet meadow adjacent to Site 3.
	Grazed orchardgrass plants at all sites.
July 2013	Light grazing use of wet meadow adjacent to Site 3.
	Grazed orchardgrass plants at all sites. Occasional use of pinegrass.
July 2014	Light grazing use of wet meadow adjacent to Site 3.
	Grazed orchardgrass plants at all sites, but overall use was lower.

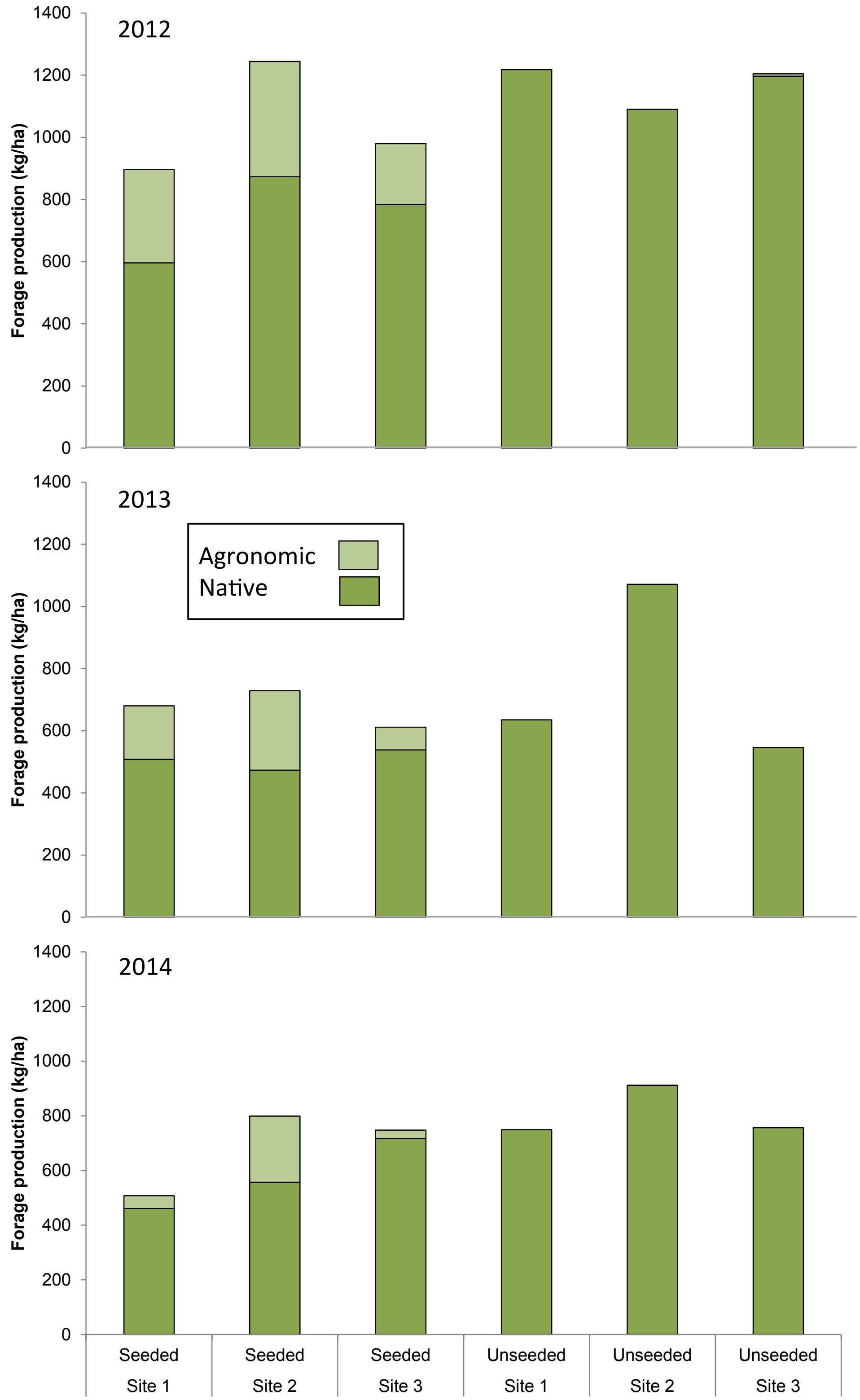


Figure 4. Forage production of agronomic (light green bars) and native (dark green bars) forage species for year 2 through year 4 following the fire (2012 – 2014) on seeded and unseeded areas at the Bull Canyon wildfire.

3.2.3 Native plant community development

Native species richness

The seeding treatment had no effect on species richness (P>0.10) measured four years after the wildfire. Species richness was similarly unaffected by a seeding treatment on the Strawberry Hills wildfire (Newman 2007).

A total of 49 vascular native plant species were recorded over all sites during the study period. Site 3 generally had the greatest species richness on both seeded and unseeded areas for the first four years following the fire (2011 – 2014) (see Figures 5 and 6). No doubt this was due to the low soil burn severity of this site compared to the other sites with moderate soil burn severity. The greater soil burn severity would result in greater mortality of perennial plants as well as increased seed losses from the soil seed bank. Site 1 had intermediate species richness, which remained largely unchanged over the study period (see Figure 5). Site 2 had low species richness initially, but this increased over time and surpassed Site 1 by year 3. The strong increase of species richness at Site 2 may be due to greater feral horse use and resulting increase in propagules in manure deposition. A study from a site located 39 km southeast of our study site reported species richness increases averaging 45% from the first to the second year following a 2003 wildfire[4] (MacKenzie and Iverson 2005).

Effects on native species

Nine of the most common species (each achieving a maximum cover greater than 5%) were selected for close examination. Pinegrass was the clear dominant in all years at Sites 1 and 3. Pinegrass increased over time at these sites, but only on the unseeded areas, achieving its highest values at 45% to 55% cover by 2014 (see Figure 6). Pinegrass levels were quite different on Site 2 where pinegrass was subdominant to prickly rose (*Rosa acicularis*), showy aster (*Aster conspicuus*), and timber milkvetch (*Astragalus miser*), especially in the last two years of the study. Pinegrass never exceeded 10% cover at Site 2 by the end of the study. The seeding treatment reduced the cover of pinegrass in the first and third year following the fire, but this effect primarily occurred on Site 1 (see Table 7 and Figure 6). A similar reduction occurred on the Dog Creek fire where pinegrass decreased in response to an increase in orchardgrass (see Appendix Figure A-1).

Fireweed, twinflower, wild strawberry, heart-leaved arnica, and kinnikinnick were most abundant or restricted to Site 3. This was probably due to the lower severity burn at Site 3. Kinnikinnick has been shown to be particularly sensitive to wildfire in other BC studies (Newman et al. 2012, MacKenzie and Iverson 2005). MacKenzie and Iverson (2005) found that kinnikinnick survival was linearly related to fire severity in Nuntsi Provincial Park. Newman et al. (2012) reported that kinnikinnick declined from 57% to 31% following an April prescribed fire in the East Kootenay region. Some of the apparent effects of the seeding treatment at Site 3 for this group of plants were minor and some were mixed. However, the seeding treatment did result in clear reductions of fireweed and twinflower cover in some years (see Table 7 and Figure 6). Heart-leaved arnica was present on seeded areas and absent on unseeded areas at Site 3, but this may have been due to pre-existing site differences.

Prickly rose, showy aster, and timber milkvetch were most abundant at Site 2 (P=0.011; see Table 7). Prickly rose was unaffected by the seeding treatment, increasing in cover on both treated and untreated areas over the period of the study. The trend of prickly rose cover (see Figure 6) and its known behaviour suggests that it is likely to continue to increase at Site 2. These results are consistent with that of MacKenzie and Iverson (2005) and suggest that prickly rose recovers quickly following wildfire in this region. The seeding treatment reduced showy aster, but increased timber milkvetch at Site 2 (see Table 7).

Natural regeneration of lodgepole pine and Douglas-fir was sparse. Only 7 lodgepole pine seedlings were sampled in unseeded and seeded plots, representing less than 0.05% cover. No Douglas-fir seedlings were sampled. As a result of the low cover of tree seedlings, it was not possible to properly determine the effect of the seeding treatment on natural regeneration. Heavy agronomic forage establishment (1031 kg/ha) was linked to reduced natural regeneration of lodgepole pine by Clark and McLean (1969). Light establishment (347 kg/ha) appeared to have no effect on natural regeneration. The maximum agronomic forage production of 370 kg/ha recorded at our sites suggests that the effect of seeding on natural regeneration will be negligible on the Bull Canyon wildfire.

[4] The Chilko Lake wildfire started on July 22, 2003 and burned 30 000 ha of dry forest, including 90% of Nuntsi Provincial Park. Post-wildfire vegetation monitoring was conducted during 2004 and 2005 within the park, which is located 39 km southeast of the Bull Canyon wild fire study site. The park is in the Sub-boreal Pine-Spruce very dry cold (SBPSxc) biogeoclimatic subzone and was dominated by lodgepole pine with kinnikinnick and pinegrass (MacKenzie and Iverson 2005).

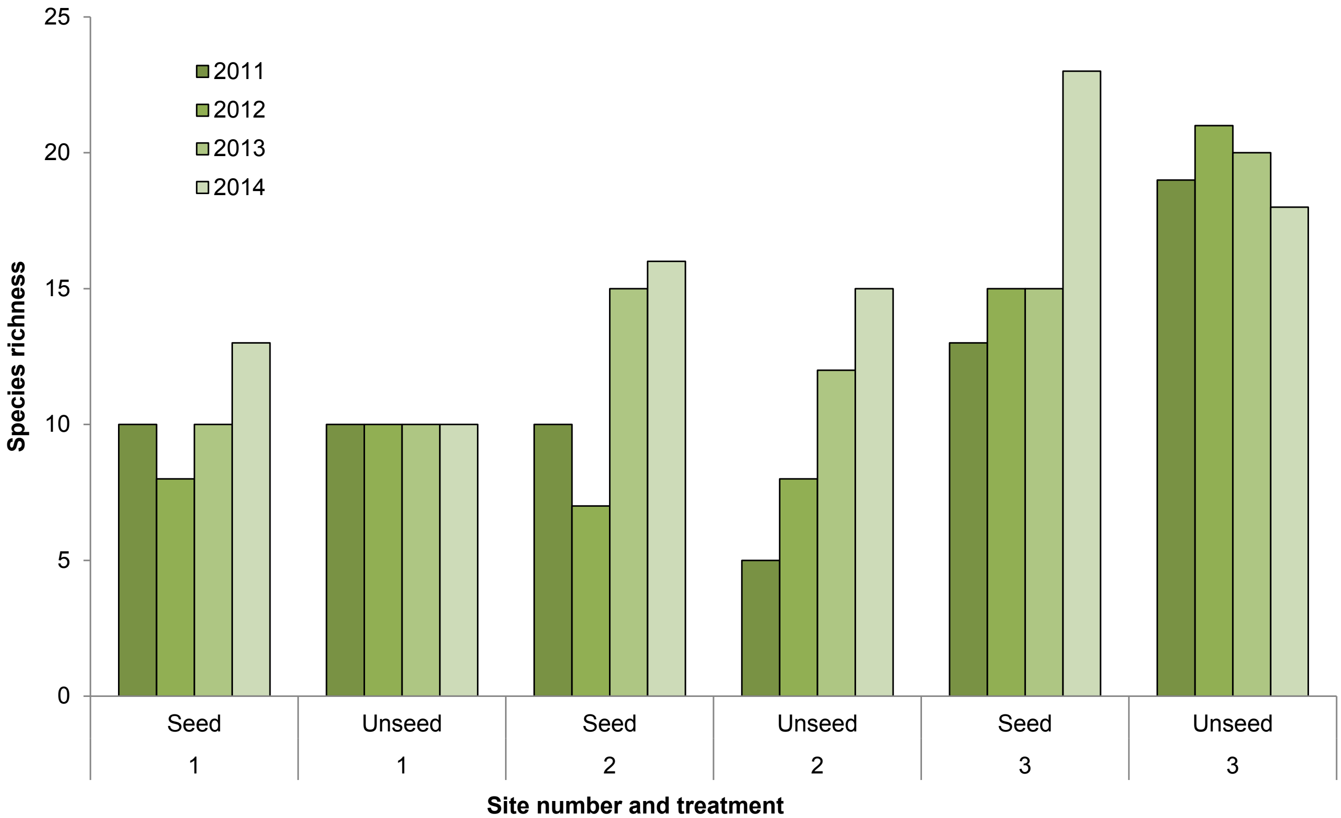


Figure 5. Vascular native species richness (n) for the three sites at Bull Canyon wildfire for the first four years following the fire (2011 – 2014).

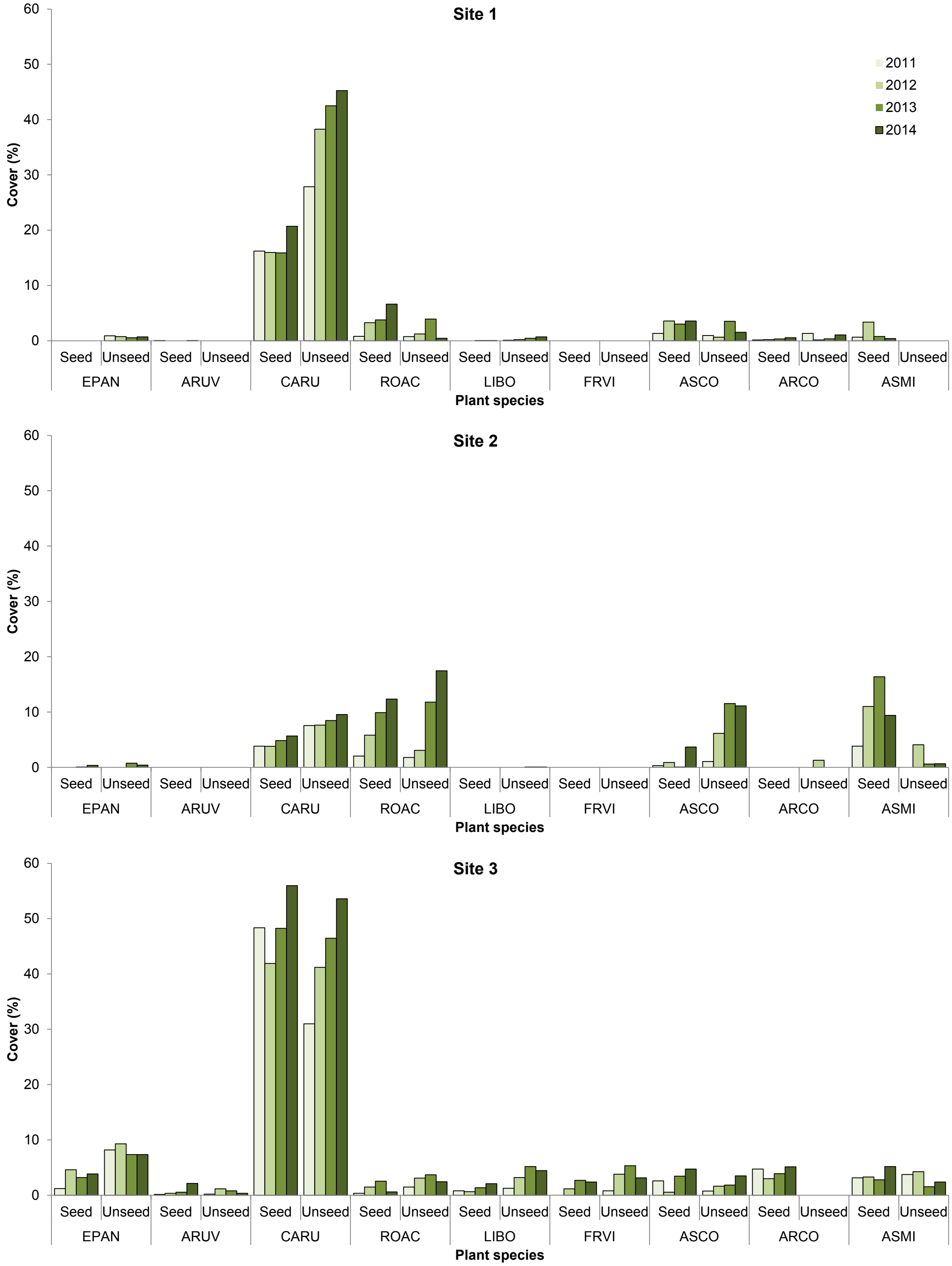


Figure 6. Cover of the nine common species (with >5% cover) at Sites 1-3 on seeded and unseeded areas in 2011, 2012, 2013 and 2014 (separate bars). Species codes: ARCO (Arnica cordifolia), ARUV (Arctostaphylos uva-ursi), ASCO (Aster conspicuus), ASMI (Astragalus miser), CARU (Calamagrostis rubescens), EPAN (Epilobium angustifolium), FRVI (Fragaria virginiana), LIBO (Linnaea borealis), ROAC (Rosa acicularis).

Table 8. Significance of ANOVA tests (prob.>F) and cover values (%) for common (> 5% cover) native plant species at the three sites at Bull Canyon Wildfire for the first four years following the fire (2011 – 2014).

	Significance			Site					
				1		2		3	
Year	Block	Treat	Block*Treat	Seed	Unseeded	Seed	Unseeded	Seed	Unseeded
Fireweed									
2011	0.351	0.332	0.001	0.00	0.92	0.00	0.00	1.20	8.20
2012	0.093	0.322	0.368	0.00	0.77	0.00	0.00	4.60	9.30
2013	0.113	0.244	0.109	0.00	0.56	0.05	0.77	3.20	7.35
2014	0.085	0.292	0.155	0.00	0.71	0.36	0.41	3.85	7.35
Kinnikinnick									
2011	0.071	0.992	0.613	0.05	0.00	0.00	0.00	0.15	0.20
2012	0.222	0.411	0.062	0.00	0.00	0.00	0.00	0.35	1.15
2013	0.033	0.401	0.741	0.00	0.00	0.00	0.00	0.55	0.80
2014	0.340	0.385	0.038	0.05	0.00	0.00	0.00	2.15	0.35
Pinegrass									
2011	0.157	0.943	0.054	16.22	27.86	3.83	7.55	48.35	31.00
2012	0.118	0.367	0.173	15.97	38.27	3.81	7.64	41.90	41.20
2013	0.128	0.396	0.048	15.89	42.50	4.85	8.47	48.25	46.45
2014	0.086	0.401	0.116	20.71	45.26	5.66	9.54	55.95	53.60
Prickly rose									
2011	0.240	0.568	0.719	0.82	0.77	2.04	1.79	0.35	1.50
2012	0.445	0.497	0.410	3.27	1.22	5.83	3.07	1.50	3.10
2013	0.011	0.185	0.942	3.78	3.93	9.90	11.79	2.55	3.70
2014	0.148	0.948	0.270	6.63	0.46	12.35	17.45	0.60	2.45

Table 8. (continued)

Year	Significance			Site					
				1		2		3	
	Block	Treat	Block*Treat	Seed	Unseeded	Seed	Unseeded	Seed	Unseeded
Twinflower									
2011	0.041	0.292	0.714	0.00	0.10	0.00	0.00	0.80	1.25
2012	0.299	0.365	0.004	0.00	0.20	0.00	0.00	0.65	3.20
2013	0.251	0.337	0.016	0.05	0.46	0.00	0.05	1.35	5.20
2014	0.102	0.260	0.513	0.05	0.71	0.00	0.05	2.10	4.45
Wild strawberry									
2011	0.215	0.401	0.413	0.00	0.00	0.00	0.00	0.25	0.80
2012	0.223	0.411	0.220	0.00	0.00	0.00	0.00	1.15	3.80
2013	0.098	0.401	0.448	0.00	0.00	0.00	0.00	2.70	5.35
2014	0.018	0.401	0.874	0.00	0.00	0.00	0.00	2.40	3.15
Showy aster									
2011	0.637	0.580	0.438	1.33	0.97	0.31	1.07	2.60	0.75
2012	0.768	0.685	0.101	3.57	0.66	0.89	6.14	0.55	1.65
2013	0.819	0.492	0.017	3.04	3.52	0.05	11.53	3.45	1.85
2014	0.523	0.705	0.139	3.57	1.53	3.67	11.12	4.75	3.50
Heart-leaved arnica									
2011	0.621	0.560	0.002	0.15	1.33	0.00	0.00	4.75	0.00
2012	0.710	0.675	0.056	0.20	0.15	0.00	1.29	3.00	0.00
2013	0.539	0.406	0.037	0.32	0.36	0.00	0.00	3.90	0.00
2014	0.580	0.462	0.001	0.56	1.07	0.00	0.00	5.15	0.00
Timber milkvetch									
2011	0.361	0.430	0.036	0.66	0.00	3.83	0.00	3.15	3.75
2012	0.277	0.289	0.393	3.37	0.00	11.01	4.07	3.30	4.25
2013	0.493	0.366	0.009	0.77	0.00	16.38	0.61	2.80	1.55
2014	0.423	0.268	0.017	0.41	0.00	9.39	0.66	5.20	2.40

4.0 CONCLUSIONS

White clover established poorly when aerially seeded onto the Bull Canyon wildfire possibly because of fall seeding. The poor establishment of fall-seeded white clover supports the recommendation of Brooke and Holl (1988) that spring seeding is preferable for this species. Italian ryegrass performed as expected, increasing quickly to moderate abundance in the first year after seeding then dropping by the second year. Italian ryegrass was therefore not a suitable choice for enhancing forage production at the site although it may have played some role in reducing weed invasion. Nonetheless, orchardgrass achieved twice as much cover as Italian ryegrass in the first year and would therefore be a more suitable species for suppressing pioneer/ruderal weeds. Orchardgrass established well and contributed substantial and palatable forage for four years. However, the seeded sites did not produce more forage than the unseeded sites. This was likely due to the low to moderate soil burn severity at our sites, resulting in low mortality of native pinegrass. Our study provides evidence that seeding for the purpose of forage replacement or forage enhancement is unnecessary and ineffective at wildfire sites with moderate or low soil burn severity in the IDFdk4 and similar Douglas-fir – pinegrass range types.

There was some evidence of slight positive common weed suppression due to the seeding; however, the site was not at threat from FRPA invasive plant species so the overall benefit was negligible. Seeding to suppress weeds will be more practical on sites that are at greater risk of invasion by listed and problematic species. In those cases, it would be preferable to use higher rates of seeding to improve the success of the weed control measure.

There was evidence that seeding resulted in improved grazing distribution of feral horses and hence reduced use of an overused wet meadow complex and increased use of the burned forest. However, it is not clear whether the fire alone would have resulted in the same benefits without the seeding treatment.

The seeding treatment had variable effects on species richness, but tended to increase species richness overall. The seeding treatment reduced the cover of several native species; however, it is too early to determine whether these will be important effects in the long term.

4.1 *Management Implications*

Much of the area that was seeded on the Bull Canyon wildfire near the study sites had a moderate vegetation burn severity index with only small portions of high vegetation burn severity index (see Figure 2). On-the-ground assessments showed that moderate vegetation burn severity translated into low soil burn severity and high vegetation burn severity translated into moderate soil burn severity. The results of this study show that seeding onto wildfires of these burn severity rankings did not yield benefits beyond what the native forage could produce. Therefore future post-wildfire seeding should only be considered on high severity vegetation burn sites and high severity soil burn sites. This decision should be supported by a proper assessment and mapping of vegetation/soil burn severity coupled with an understanding of the invasive species threat in the area.

Table 9. Summary of seeding treatment success against seeding objectives.

Seeding Objective	Success at meeting the objective
Forage replacement	Low. Not required. Native forage survived the fire.
Forage enhancement	Low. Did not increase forage production above native forage alone.
Weed control	Slight evidence of opportunistic weed suppression, but FRPA IP Reg species were not present at the study sites.
Improved grazing distribution	Distribution improved, but may have been similar following fire alone.

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APPENDIX

Table A-1. ***List of vascular plant species found at the Bull Canyon wildfire sites.***

Common name	Scientific name	Origin
alsike clover	Trifolium hybridum	exotic - agronomic
American dragonhead	Dracocephalum parviflorum	native
annual hawksbeard	Crepis tectorum	exotic - weed
common dandelion	Taraxacum officinale	exotic - weed
common harebell	Campanula rotundifolia	native
common plantain	Plantago major	exotic - weed
creamy peavine	Lathyrus ochroleucus	native
Douglas's knotweed	Polygonum douglasii	native
early blue violet	Viola adunca	native
field pussytoes	Antennaria neglecta	native
fireweed	Epilobium angustifolium	native
foxtail barley	Hordeum jubatum	native
graceful cinquefoil	Potentilla gracilis	native
heart-leaved arnica	Arnica cordifolia	native
Italian ryegrass	Lolium multiflorum	exotic - agronomic
Junegrass	Koeleria macrantha	native
Kentucky bluegrass	Poa pratensis	exotic - weed
kinnikinnick	Arctostaphylos uva-ursi	native
lamb's-quarters	Chenopodium album	exotic - weed
littlebells polemonium	Polemonium micranthum	native
lodgepole pine	Pinus contorta	native
northern bedstraw	Galium boreale	native
northern gentian	Gentianella amarella	native
Nuttall's alkaligrass	Puccinellia nuttalliana	native
orchardgrass	Dactylis glomerata	exotic - agronomic
pinegrass	Calamagrostis rubescens	native
prickly lettuce	Lactuca serriola	exotic - weed
prickly rose	Rosa acicularis	Native
Rocky Mountain butterweed	Senecio streptanthifolius	Native
rough-leaved ricegrass	Oryzopsis asperifolia	Native
Saskatoon	Amelanchier alnifolia	Native
shepherd's purse	Capsella bursa-pastoris	exotic - weed
short-beaked agoseris	Agoseris glauca	Native
showy aster	Aster conspicuus	Native
showy Jacob's-ladder	Polemonium pulcherrimum	Native
slender wheatgrass	Elymus trachycaulus	Native
soopolallie	Shepherdia canadensis	Native
spikelike goldenrod	Solidago spathulata	Native
timber milkvetch	Astragalus miser	Native
trembling aspen	Populus tremuloides	Native
twinflower	Linnaea borealis	Native
white clover	Trifolium repens	exotic - agronomic
wild strawberry	Fragaria virginiana	Native
willow	Salix sp.	Native
yarrow	Achillea millefolium	Native

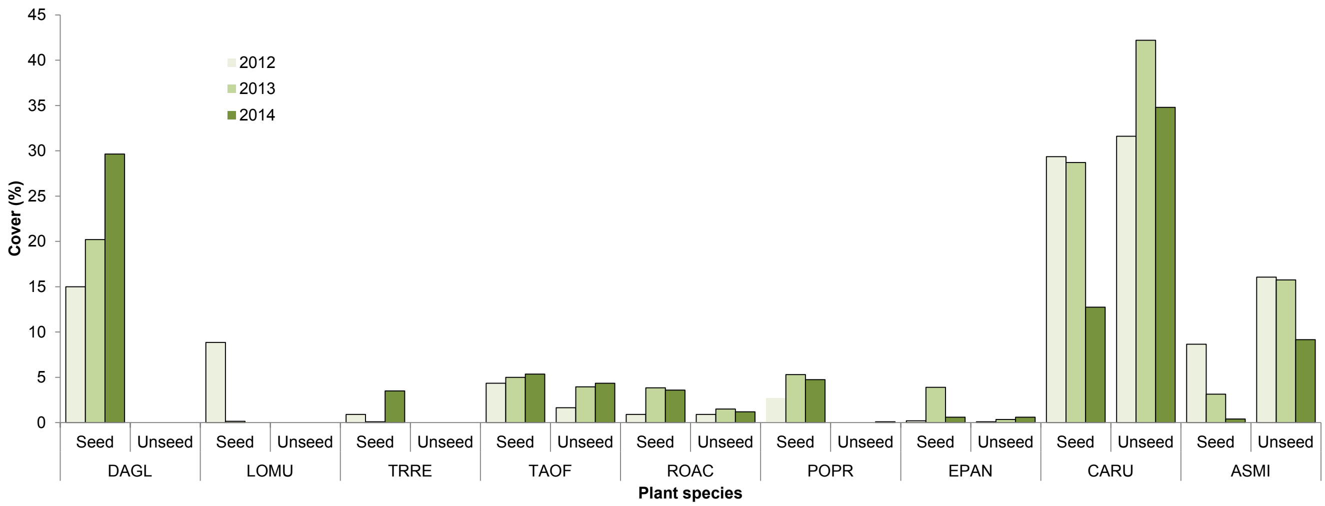

Figure A-1. ***Cover of seeded and native plant species at the Dog Creek Fire study site.***
Cover of seeded and native plant species (with >5% cover) at the Dog Creek Fire study site on seeded and unseeded areas for years 2 through 4 post-fire (2012, 2013, 2014). Species codes: ASMI (Astragalus miser), CARU (Calamagrostis rubescens), DAGL (Dactylis glomerata), EPAN (Epilobium angustifolium), LOMU (Lolium multiflorum), POPR (Poa pratensis), ROAC (Rosa acicularis), TAOF (Taraxacum officinale), TRRE (Trifolium repens).

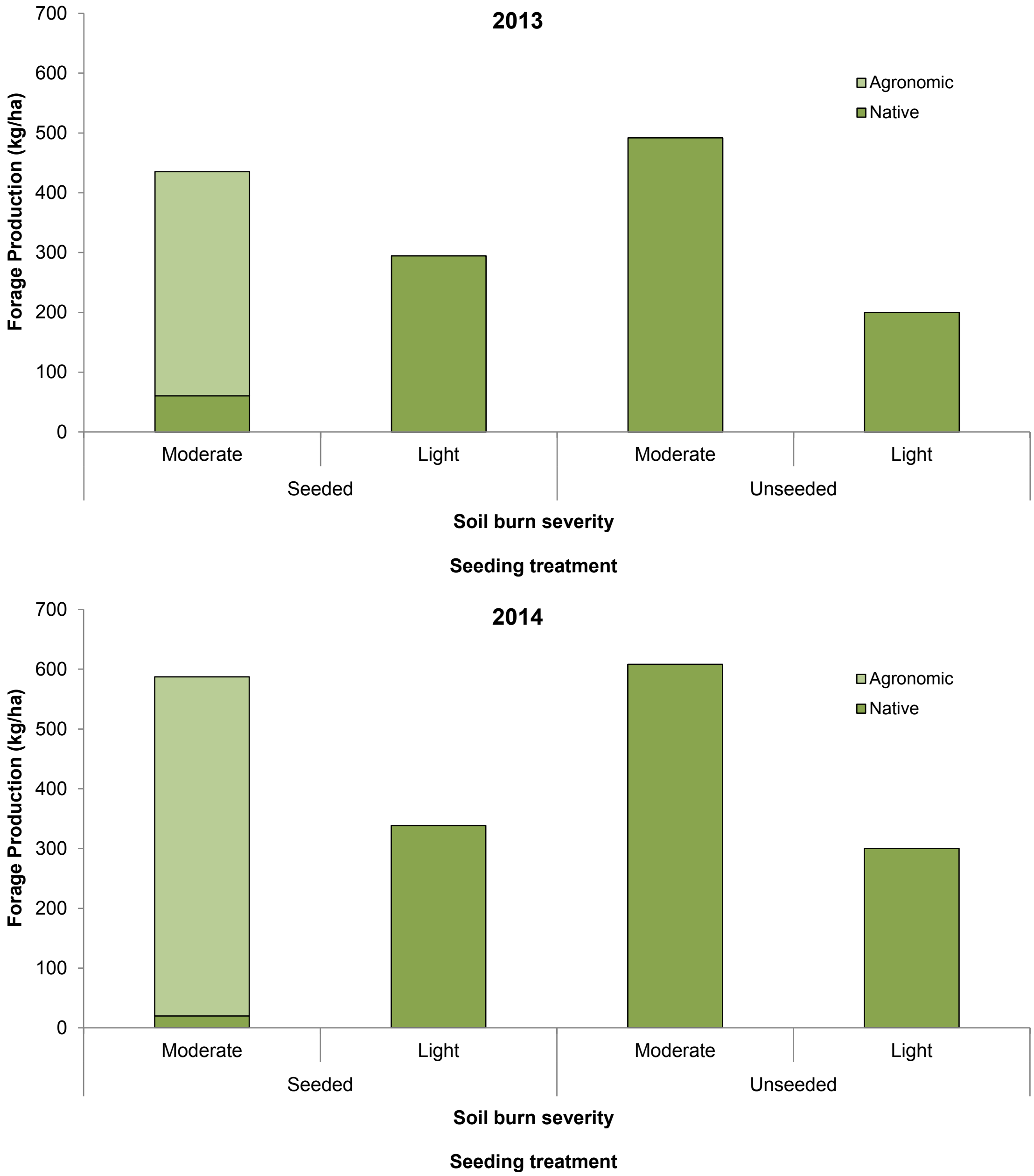

Figure A-2. ***Forage production of agronomic and native species at the Dog Creek fire study site.*** *Forage production of agronomic (light green bars) and native (dark green bars) forage species in 2013 and 2014 on seeded and unseeded areas at the Dog Creek fire study site.*