Vegetation responses to thinning and prescribed fire restoration treatments in the Southern Interior Rocky Mountain Trench.

Prepared by: Morodoluwa Akin-Fajiye, Jillian Caissie and Lauchlan H. Fraser

Department of Natural Resource Science, Thompson Rivers University.

November 2020

SUMMARY

Many North American forests have been shaped by fire, and fire suppression or exclusion over the last century has had consequences for forest structure and on understorey and forage species. Reintroduction of fire as a restoration tool into these forests has been touted as a means to restoring historical ecological conditions. However, thinning either by removing commercially marketable trees, or slashing of forest undergrowth has been suggested as an alternative to fire.

We conducted an analysis of the response of understory and overstorey vegetation to thinning and prescribed burns in the Southern Interior Rocky Mountain Trench in British Columbia. Using data from up to ten sites to which different thinning and prescribed fire treatments were applied, we analyzed the response of litter, bare soil, grasses, forbs, and exotic species in the understorey. We also tested the change in forage yield over time as a result of a combination of grazing exclosures and thinning/burning. Finally, we analyzed the impact of thinning and burning on the regeneration of new trees, and on the relationship between overstorey and understory vegetation.

Exotic species increased over time in the pooled dataset, regardless of treatment. Within Redstreak, exotic species increased over time but the rate of increase was not affected by treatments. It is likely that increase in bare mineral soil combined with burning provided more resources, leading to increases in native grasses, but no corresponding increases in exotic species. In Miller Road, where native vegetation declined after sixteen years, this decline was greater in sites that were not burned. Results from both sites indicate that thinning is unlikely to be as effective as burning in improving native vegetation, added effect of increased exotic species. Pooled results from the other eight sites showed that these treatments maintained native grasses at a sustained level compared to controls where grasses declined.

Pooled across all sites, overstorey was consistently negatively related to understorey species, confirming that a dense canopy can inhibit understorey development. Exclosures had no effect on pinegrass yield, but increased bunchgrass yield in thinned sites.

While some of the available evidence indicates that burning and thinning are unlikely to be surrogates for each other, it appears vegetation response to management action is complex and may depend on other factors such as site characteristics.

Table of Contents

1.0 INTRODUCTION 4

2.2 UNDERSTOREY VEGETATION SAMPLING 8

2.3 FORAGE SAMPLING 9

2.0 DATA ANALYSIS AND RESULTS 9

3.1 STAND STRUCTURE/OVERSTOREY 9

3.1.1 Data analysis 9

3.1.2 Results 10

3.2 UNDERSTOREY VEGETATION 13

3.2.1 Data analysis 13

3.2.2 Results 14

3.3 FORAGE 19

3.3.1 Data analysis 19

3.3.2 Results 20

4.0 DISCUSSION 22

4.1 How does thinning and/or burning affect tree regeneration and the relationship between understorey and overstorey vegetation? 22

4.2 What effect does thinning and burning have on understorey vegetation? 23

5.0 CONCLUSIONS AND RECOMMENDATIONS 26

1.0 INTRODUCTION

Historically, many terrestrial North American ecosystems have been shaped by fire, and continue to require intermittent fire to maintain native species diversity and ecosystem structure and function (Switzer et al. 2012, Ryan et al. 2013, Steel et al. 2015). The use of fire by many indigenous nations modify ecosystems and clear vegetation undergrowth is well documented (Abrams and Nowacki 2008, Ryan et al. 2013). However, fire exclusion and suppression became widespread in parts of Western North America, due to European immigration leading to resettlement of indigenous communities, climate change, preservation of commercial forests and increased cattle grazing leading to reduction in fire frequency and intensity (Veblen et al. 2000, Cumming 2011, Power et al. 2013, Ryan et al. 2013).

Fire suppression endeavors have led to changes in forest structure. Pre-fire suppression ecosystems are usually composed of early successional, shade intolerant trees, in which canopies are single-layered, and trees are widely spaced. In addition, the understorey is sparse, and highly diverse, with bunchgrass present (Keane et al. 2002, Keeling et al. 2006). Fire suppression and exclusion in forest-grassland mosaic ecosystems had led to domination of conifer stands in Western North America (Tveten 1997, Gallant et al. 2003). Sustained fire suppression has also reduced native species diversity, with attendant increases in exotic species (Macdougall et al. 2013). Years of effective fire suppression may build up forest undergrowth and fuels resulting in catastrophic wildfires and danger to urban areas (Parisien et al. 2020).

Thinning and prescribed fire are restoration tools that have been used to manage fire-maintained ecosystems. Structural restoration involves using mechanical methods such as slashing and logging to restore forest structure and composition to their historical conditions before fire is reintroduced in to the system, while process based or functional restoration involves simply re-introducing fire or other ecological processes (Stephenson 1999). Thinning and prescribed burning can mediate the interaction between understorey and overstorey vegetation in forest ecosystems. Mechanical thinning, by removing small and medium sized trees can alter forest structure, change the availability and distribution of canopy fuels, and increase soil resources (Switzer et al. 2012, Ryan et al. 2013). In addition, thinning can be economically viable when commercially important trees are selected from removal, while allowing other trees to remain to contribute to stand structure. Introducing fire to fire-supressed ecosystems using techniques mimicking historical fire patterns can restore overstory and understory forest structure and increase native species diversity. Prescribed fire can remove ground fuels and reduce tree density by killing small trees, reduce frequency of crown fires, and may restore historical fire regimes (Brockway and Lewis 1997, Schoennagel et al. 2004)

In combination, both processes can introduce or change disturbance regimes, releasing resources for establishment or spread of native species in the understorey (Gundale et al. 2005, Dodson et al. 2008). They can also trigger the germination of seeds in the soil and canopy seed bank by reducing litter accumulation and allowing seeds to access resources at the soil surface (Haase 1986, Goubitz et al. 2004). By altering overstorey and understorey vegetation, thinning and prescribed burning can increase the amount and heterogeneity of light, nutrients and water available to understorey plants and mature trees. However, the increased nutrient availability and open ground surface provided by both thinning and prescribed fire may also favor the introduction of exotic or invasive species (Potts and Stephens 2009, Symstad et al. 2014). Clearly, the use of both thinning and burning in ecosystem restoration can be complementary, however, it remains uncertain from the literature if both actions are necessary to achieve restoration goals. A meta-analysis by Fulé et al. (2012) showed that combined thinning and burning treatments performed better than thinning but worse than prescribed burns in reducing surface fuels, but better than both treatments in reducing stand density. In Dodson et al. (2008), total species richness of understorey vegetation in the Cascades was higher in combined thin/burn treatments than in thin only and burn only treatments, although this was accompanied by an increase in the number of exotic species.

Studies have reported positive, neutral or negative effects of grazing on plant diversity and productivity (Bakker 1985, Milchunas and Lauenroth 1993, Stohlgren et al. 1999, Lu et al. 2017, Hao and He 2019). Grazing is expected to reduce cover and yield of desirable native species due to consumption, resulting in an increase in bare soil, and vegetation patchiness (Adler et al. 2001, Molino and Sabatier 2001, Roxburgh et al. 2004). Grazing can also stimulate grassland productivity (Semmartin and Oesterheld 1996). Altesor et al. (2005) showed that in Uruguayan Campos, simulating grazing in sites that had been ungrazed for nine years increased productivity compared to ungrazed treatments, and areas that had been continuously grazed for 25 years. High intensity grazing can also increase exotic species richness and cover due to the disturbance and gap creation by native species loss, leading to replacement by fast growing exotics with characteristics such as short life histories and high seed production that enable them to thrive in highly disturbed environments (Pettit et al. 1995, Souther et al. 2019).

The Rocky Mountain Trench Ecosystem Restoration Program (Trench ER) was set up in the East Kootenay region of south-eastern British Columbia to restore a landscape that supports a mix of native trees, native plants and animals, while providing forage resources to wild and domestic grazers (Bond et al. 2013). Historically, the grasslands and open forests in this region have been maintained by frequent low-intensity fires occurring every fifteen years, on the average (Bond et al. 2013). More recent suppression of these fires have given rise to forest ingrowth and encroachment of forest into open grasslands, reducing forage for domestic and wild grazers (Bond et al. 2013). Ecological restoration in this system followed a three-phase process. In phase one: ingrown forest stands were thinned to between 20 and 70% of their basal area. The site conditions and stand history dictate what kind of tree removal was carried out (harvesting, spacing or slashing). In phase two, prescribed burning was used in some sites to kill small trees, while in phase three, sites were allowed to rest so that regenerated stems could grow (Page and Machmer 2003).

2.1 SAMPLING METHODOLOGY

Overall, we obtained data from a total of 11 sites (Table 1). All 11 sites contained understorey data sampled with different sampling methods. Forage data was available for eight sites, while tree data was available for five sites.

Table 1: Table showing all the sites in this included in this study and the types of data obtained from the sites. Stars indicate sites in which vegetation sampling was not in permanent quadrats. Numbers indicate paired treatment sites.

	Treatment		**BEC zone**	**Understorey data**	**Forage data**	**Tree data**
Site	**Thin**	**Burn**				
Gina Lake	Logged: Feb 2002, Slashed: 2006, 2010	Fall 2011	Ppdh2	Yes	Yes	Yes
Hawke Road	Logged: December 2005 Slashed: 2005	No burn	IDFxk	Yes	Yes	Yes
Hofert Control[1]	No thin	No burn	IDFxk	Yes	Yes	Yes
Hoodoo East[1]	Logged: December 2010	No burn	IDFxk	Yes	Yes	Yes
Redstreak (Treatments: Thin/Burn, Thin only, Control)	2003	2005	IDFxk	Yes	No	No
Miller Road (Treatments: Burn only, Thin, Thin/Burn, Control)	1997	1998	Ppdh2	Yes	No	No
Wolf Creek[2]	Logged: 2000 Slashed: 2002	2004	Ppdh2	Yes	Yes	No
Sheep Creek North[2]	Logged: 1999 Slashed: 2002	No burn	Idfdm2	Yes	Yes	No
Lewis Creek*	Slashed: 2009	2012	IDFdm2	Yes	Yes	No
Rocks*	Slashed: 2004	2006	IDFdm2	Yes	Yes	No
Stoddart North*	Logged: Feb 2002 Slashed: 2015	No burn	IDFxk	Yes	No	Yes

2.1 STAND STRUCTURE/OVERSTOREY SAMPLING

In each treatment and control site, a plot centre was randomly located and permanently marked. Plot centers ranged from five to fifteen per site. In each plot centre, nested fixed radius plots were established to sample different tree layers:

a. a 1.78m radius within which 4G/Germinants were sampled;
b. a 3.99m radius within which 2P (7.5 – 12.5 cm dbh), 3S 3S/Sapling (≥1.3 m height and <7.5 cm dbh), and 4R/Regeneration (<1.3 m height) trees were sampled
c. a 5.64 m radius within which 4R/Regeneration (<1.3 m height) and 3S/Sapling (≥1.3 m height and <7.5 cm dbh) trees were sampled.

d. a 11.28 m radius within which 1M/Mature (12.5 – 30 cm dbh) trees were sampled; and
e. a 25.23 m radius within which 1D/Dominant (>30 cm dbh) trees were sampled.

Within each plot center, all trees within each were counted and identified to the species level. The diameter at breast height was also measured.

2.2 UNDERSTOREY VEGETATION SAMPLING

The experimental design for understory vegetation sampling varied across sites. Two sites, Redstreak and Miller Road had three experimental units applied to plots within sites. In Redstreak, the treatment units were: thin/burn, thin/not burned, and a control that was neither burned nor thinned. In Miller Road, the treatment units were: burned/not harvested, harvested/not burned, harvested/burned and a control that was neither burned nor harvested. Other sites used in this analysis did not have any controls or differences in treatments within sites (see below).

Sampling methodology also varied across sites. Our analysis of vegetation data is based only on data collected using the Daubenmire sampling method to maintain consistency. In most sites (all sites except Miller Road), between 12 and 20 Daubenmire quadrats were established every 5m along a transect within each plot (Multiple Daubenmires per plot). Within each Daubenmire frame, the percentage cover of individual plant species was reported. From this data, the total species richness of plant functional groups like exotic plants and grasses was calculated. In Lewis Creek, Rocks and Stoddart North, plot locations were not permanent from year to year. The complete sampling protocol can be found in Machmer et al. (2001) and (Greene and Harris 2015). In Miller Road, ten permanent 50 meter transects were established in a stratified random fashion within each treatment along which herbaceous vegetation was sampled in ten 20cm X 50cm Daubenmire frames. Therefore, 100 plots were sampled in Miller Road every time period randomly located along the transects. These plots were nested within 1m X 2m plots in which trees were sampled (Tree data for Miller Road was not available for this report). See Ross (2011) for full details of study design for Miller Road. Hofert Control did not have any treatments within sites.

2.3 FORAGE SAMPLING

Four 1 m x 0.5 m quadrats were randomly placed in each of the 25.23 m radius plots (4 quadrats per plot). The quadrats were placed such that they do not overlap Daubenmire frame locations. Herbaceous vegetation in each quadrat was clipped to ground level in mid-July, after peak growth is reached. The quadrat positions were then changed in order to capture full growth in the subsequent years. Kinnikinick (*Arctostaphylos uva-ursi*) was not clipped, as it is not of direct interest for ecosystem restoration, due to its unpalatability to livestock (Crane 1991). Four 1m x 2m cages were established adjacent to each of the quadrats, and a 1 m x 0.5 m area within these cages was clipped to ground level. While we obtained forage data from seven sites, not all sites had data from caged and uncaged plots sampled in the same year (Greene and Harris 2015).

Clipped samples were separated by functional group: bunchgrasses (rough fescue, bluebunch wheatgrass, Idaho fescue, stipa spp.), pinegrass, other grasses and exotic plants. Samples were stored in a paper bag, air-dried for 48 hours, then oven-dried at 70°C to constant mass. Dried samples were weighed, and converted from g/0.5m^2 to kg/ha by multiplying by 20.

2.0 DATA ANALYSIS AND RESULTS

3.1 STAND STRUCTURE/OVERSTOREY

3.1.1 Data analysis

We extracted overstorey/tree data from five sites: Gina Lake, Hawke Road, Hoodoo East and North Stoddart and Hofert Control (Table 2). All of these sites excluding Hofert Control were thinned, while Gina Lake was also burned. Hofert Control did not have any treatments. We counted the number of trees of different layers and species within each plot in each site, and calculated total tree number per hectare using appropriate conversions based on plot sizes within which trees were measured. We then pooled the data and conducted a global analysis to test the overall influence of thinning and time on the number of saplings and germinants (i.e. 4R and 4G trees), and the number of trees in layers other than 4R and 4S.

We tested the change in tree number over time within site using a Kruskal Wallis test. We then calculated the average of all the vegetation measures for exotic species, forbs, grasses, soil and litter per plot across all quadrats. We pooled this data across sites, and used a linear regression to analyse the relationship between mean vegetation cover and richness, and total number of trees per hectare. We also conducted within site analyses to determine the possible impact thinning and/or burning may have on the relationship between understory and overstorey.

Table 2: Sites for which overstorey data was available

	Treatment		**BEC zone**
Site	**Thin**	**Burn**	
Gina Lake	Logged: Feb 2002, Slashed: 2006, 2010	Fall 2011	Ppdh2
Hawke Road	Logged: December 2005 Slashed: 2005	No burn	IDFxk
Hoodoo East	Logged: December 2010	No burn	IDFxk
Stoddart North	Logged: Feb 2002 Slashed: 2002/2015	No burn	IDFxk
Hofert Control	No thin	No burn	

3.1.2 Results

We found that the number of old and new trees (4R and 4G) was not significantly affected by treatment or time (Table 3, Figure 1A and B). The number of old trees significantly decreased over time (Table 3, Figure 1B).

Table 3: The effects of burning, thinning and time on the new trees (samplings and germinants), and on old trees across all sites.

	Old trees			New Trees		
Site	df	Chi square	P value	df	Chi square	P value
Treatment	2	1.784	0.410	1	0.014	0.907
Year from baseline	1	**37.417**	**<0.001**	1	2.131	0.144
Treatment * Year from baseline	2	0.282	0.869	1	0.005	0.945

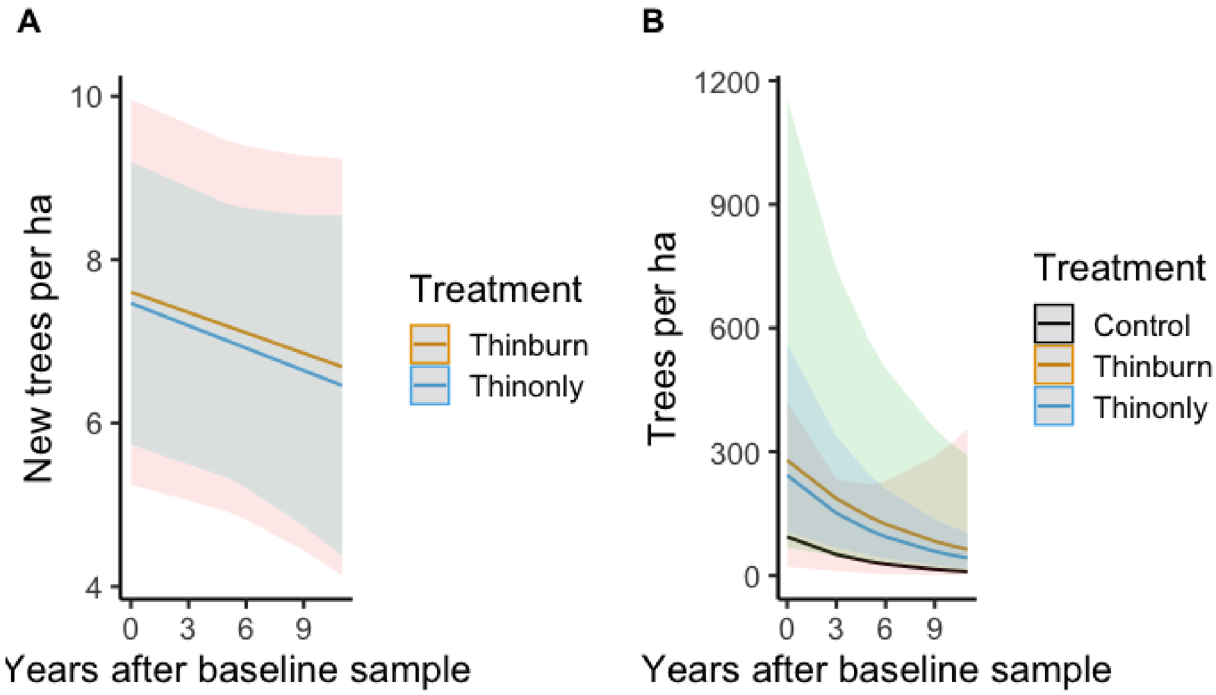

Figure 1: Response of new trees and old tress to burning and thinning over time. Plots show means and 95% confidence intervals for each treatment.

Figure 2: Plots in Gina Lake after burning in 2011 (left), and Hoodoo East after logging in 2010.

We did not detect a significant concave down relationship between understorey and overstorey vegetation, therefore we defaulted to testing only linear relationships. Pooled understorey grass richness across sites and years decreased as the number of trees increased, while grass cover was unaffected by number of trees (Grass richness: F value = -12.019, P-value < 0.001, Grass cover: F value = 2.923, P-value = 0.090, Figure 3). The interaction between treatment and number of trees was significant for the grass richness, but not cover.

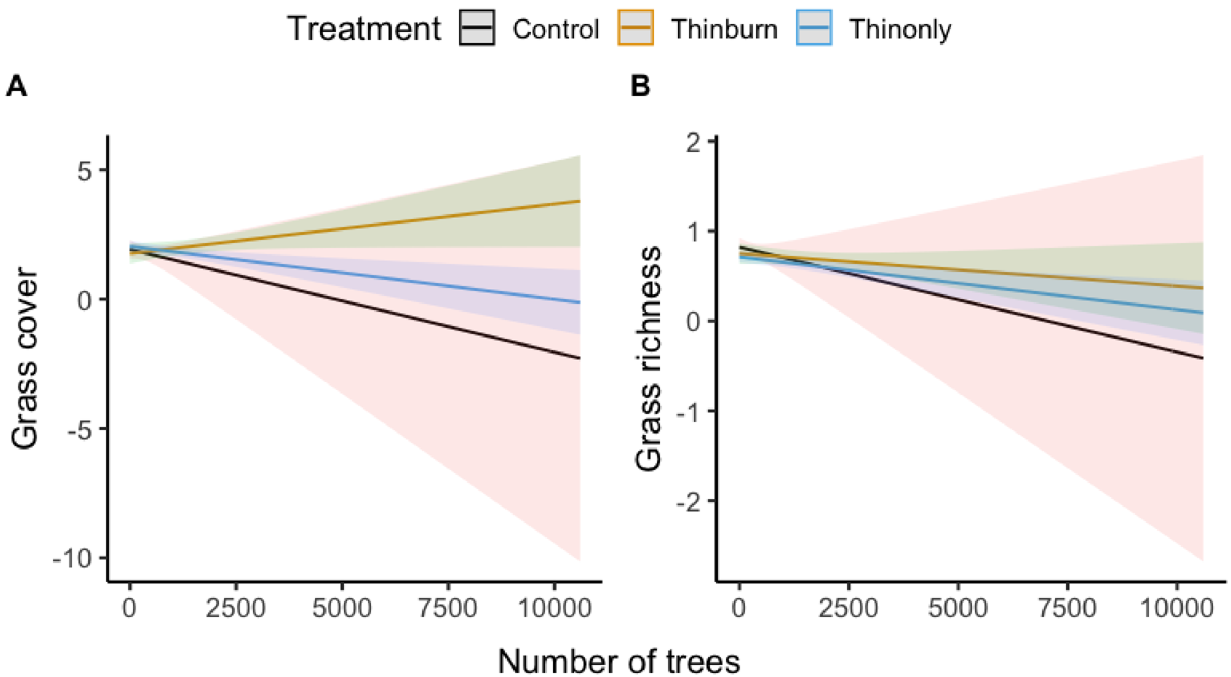

Figure 3: Relationship between grasses and overstorey trees pooled across all sites and years for which data was available. Plots show means and 95% confidence intervals for each treatment.

Exotic plants decreased significantly as the number of trees increased (Exotic richness: F value = 6.472, P-value = 0.012, Grass cover: F value = 5.556, P-value = 0.020, Figure 4).

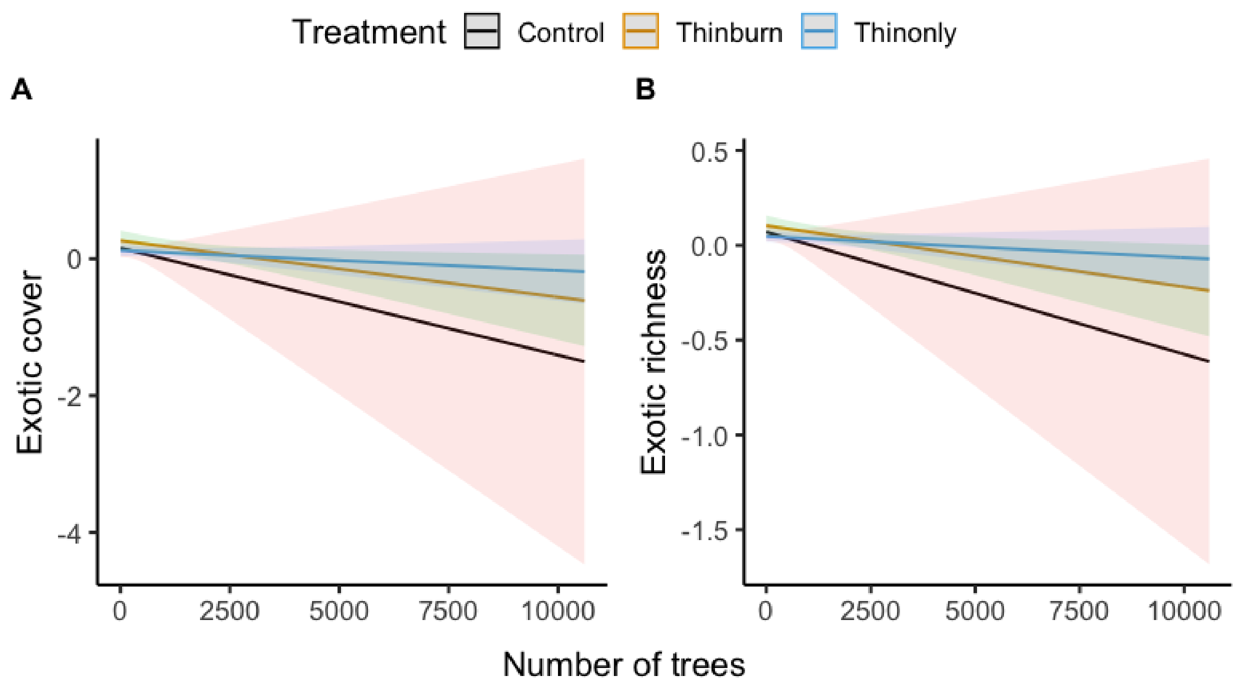

Figure 4: Relationship between exotic species and overstorey trees pooled across all sites and years for which data was available. Plots show means and 95% confidence intervals for each treatment.

3.2 UNDERSTOREY VEGETATION

3.2.1 Data analysis

Our analysis of the vegetation data depended on the experimental design and sampling methodology within each site. We first extracted a list of 13 noxious, problematic and exotic species (Table A1) from the pooled data, using the field guide of noxious and invasive species by the Invasive Species Council of British Columbia (Ralph et al. 2014). This allowed us to eliminate exotic but agronomic species like wheatgrass and alfalfa. We used this list to calculate total exotic species richness and cover per quadrat.

In Redstreak, which had multiple blocks which each treatment unit, we used a plot nested within block as the random factor. We used the glmer function in lme4 with family = "Poisson", to model richness counts (Bates et al. 2007), and the 'lmer' function with family = 'Gaussian' to model exotic and grass cover. We then used Tukey post hoc tests to determine which years were significantly different using the "emmeans" package (Lenth et al. 2018). Using the Poisson distribution for richness analysis ensured that the residuals of our models did not have to conform to the assumptions of normality. In some cases, the residual of the plant cover data was non-normal. We improved the residuals by taking the natural logarithm of grass and exotic cover after adding 1 to avoid taking the logarithm of 0.

For the other sites (apart from Redstreak and Miller Road), we used the plot level richness and cover measurements for analysis. For sites with single quadrat Daubenmires per plot, we used measurements as taken within plots for analysis. In Wolf Creek and Sheep Creek North, plot level data, averaged across the 20 Daubenmires measured in each plot was available for analysis. For the other sites, we calculated the plot level average across all quadrats for each year. We pooled this data across all sites and used the site treatment as a factor in our analysis. Each site was either thinned/not burned, thinned/burned, and a control that was not thinned or burned. The year in which sampling was conducted across sites varied with year, therefore for each sampling year, we calculated amount of time that had passed between the sampling year and the year the first/baseline sample was collected. Thus, baseline samples are noted as Year 0, with subsequent samples noted in relation to the first sample. We analysed this data using the glmer function in lme4 with family = “Gaussian”, to model the relationship between vegetation measures and time. (Bates et al. 2007), after log-transforming non-normal data.

3.2.2 Results

There were two sites with multiple treatment units and adequate controls, in which the treatments were sampled at the same time intervals: Redstreak and Miller Road.

There are nine sites in which thinning and/or burning was applied across the sites without any controls. Multiple Daubenmires (12 -15) per plot were sampled in six of these sites over time, while the remaining two (Sheep Creek North and Wolf Creek) had only one Daubenmire per plot. Note that in three of the multiple Daubenmire sites (Lewis Creek, Rocks, and North Stoddart), location of plots within sites varied from year to year.

Across all sites, we found that total species and grasses alone were influenced by the interaction of treatment and time, but total richness was not influenced by time alone (Figure 5, Table 4). Selected exotic species were not influenced by treatments, but increased over time.

Table 4: Table showing the response of understorey vegetation over time across eleven sites with different management treatments in the Southern Interior Rocky Mountain Trench.

		Total cover		**Grass Cover**		**Exotic Cover**	
Site	df	Chi square	P value	Chi square	P value	Chi square	P value
Treatment	3	1.338	0.720	0.714	0.870	0.183	0.980
Year from baseline (Linear)	1	**34.800**	**<0.001**	**32.040**	**<0.001**	**3.896**	**0.048**
Treatment * Year from baseline	3	**44.191**	**<0.001**	**28.827**	**<0.001**	6.78	0.079
		Total Richness		**Grass Richness**		**Exotic Richness**	
Site	df	Chi square	P value	Chi square	P value	Chi square	P value
Treatment	3	1.080	0.782	0.355	0.949	0.517	0.915
Year from baseline (Linear)	1	0.166	0.683	**39.356**	**<0.001**	**11.575**	**<0.001**
Treatment * Year from baseline	3	**48.060**	**<0.001**	**65.626**	**<0.001**	2.993	0.397

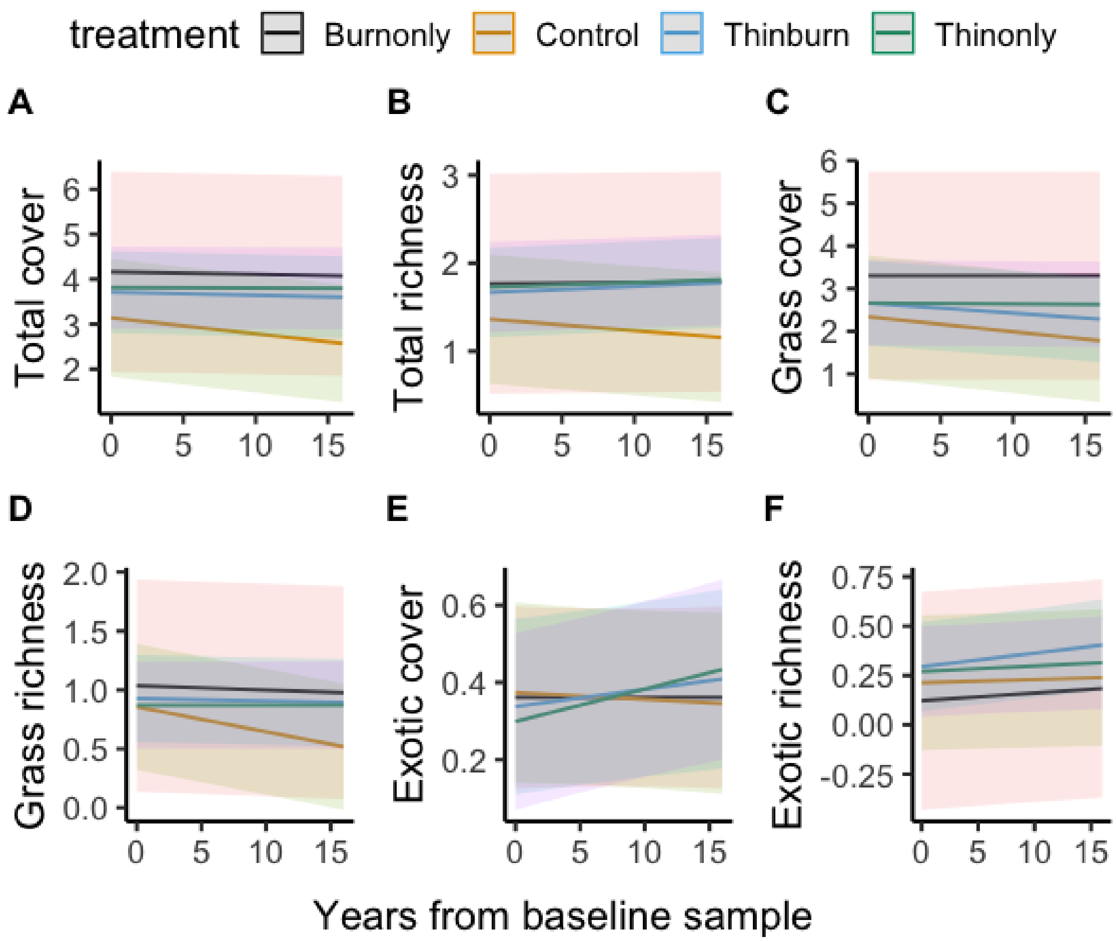

Figure 5: The response of understorey vegetation to different management treatments in the Southern Interior Rocky Mountain Trench. Plots show means and 95% confidence intervals for each treatment.

Redstreak

Redstreak was baseline-sampled in 2004 (Year 0) with further sampling conducted in 2005 (Year 1), 2006 (Year 2), and 2009 (Year 5). Logging in Redstreak was conducted in 2003, while burning was done in Spring 2005.

We observed a significant effect of year and treatment on grasses (Figures 6 and 7), however there were no differences in exotic species between treatments (Figure 7). Treatments units that were thinned only, and thinned and burned had higher initial grass cover and richness compared to the control unit. We therefore examined changes in grass cover and richness over time within treatment units using post hoc tests. In the control unit, there were no significant changes in grass cover and richness over a time period of five years. On the other hand, both burned and unburned units displayed statistically significant changes in grass cover and richness as shown by post-hoc tests (Figure 6). In the unburned unit, Year 2 grass cover was significantly higher compared to the baseline and Year 1 sample, while grass richness was statistically similar over time (Figure 6). In the burned treatment unit, grass cover increased every time period, while grass richness was only higher in the Year 5 sample compared to Year 0, 1 and 2 samples.

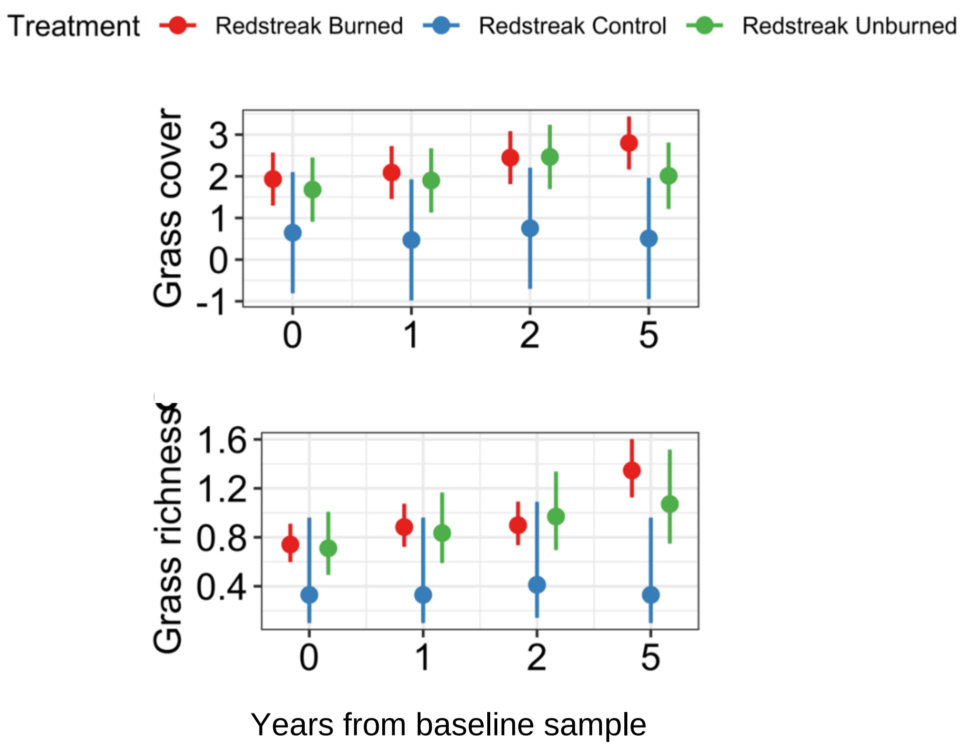

Figure 6: Response of grass species sampled treatments in Redstreak. Figures show means and 95% Confidence Interval.

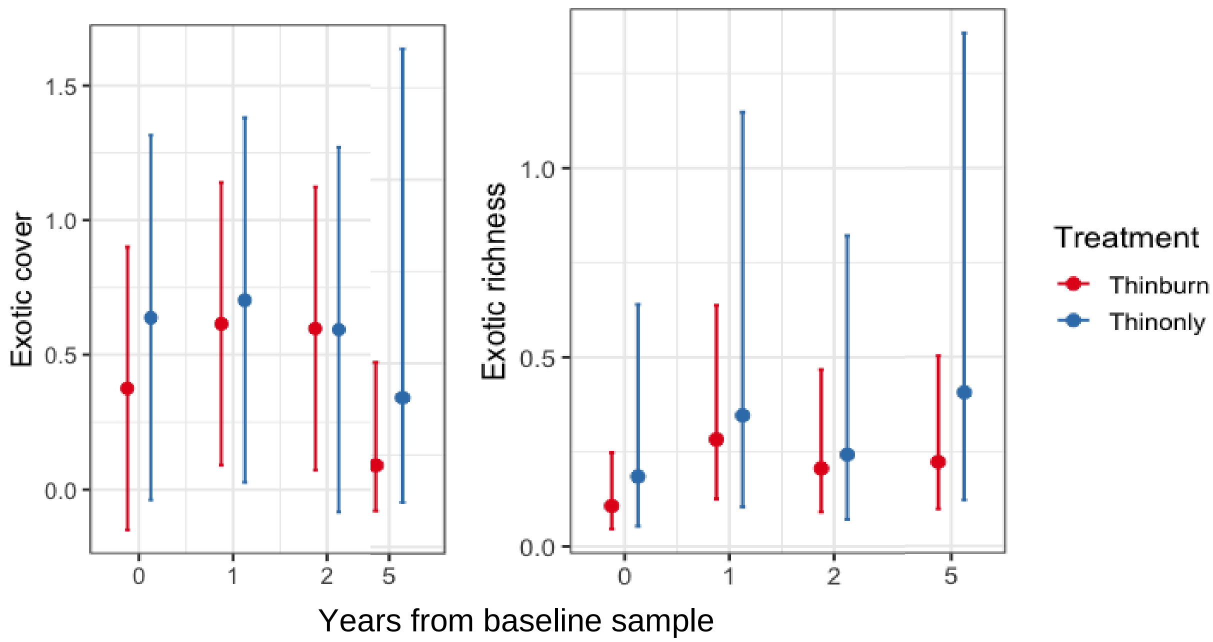

Figure 7: Response of exotic species in Redstreak. Figures show means and 95% Confidence Interval.

Miller Road

Miller Road was baseline sampled in 1996 with follow-up sampling conducted 16 years later in 2012. Prescribed burn was applied to treatment units in 1998, and logging was conducted in harvest plots in 1997. Unlike Redstreak, only grass cover and richness changed after 16 years, while grass and exotic species richness did not respond to the treatments applied.

Post hoc tests showed that, grass cover and richness did not change significantly after 16 years in burnonly treatment units (Figure 8). In control and thinburn treatments however, grass cover and richness significantly decreased over time. Exotic species data was not adequate for testing in Miller Road.

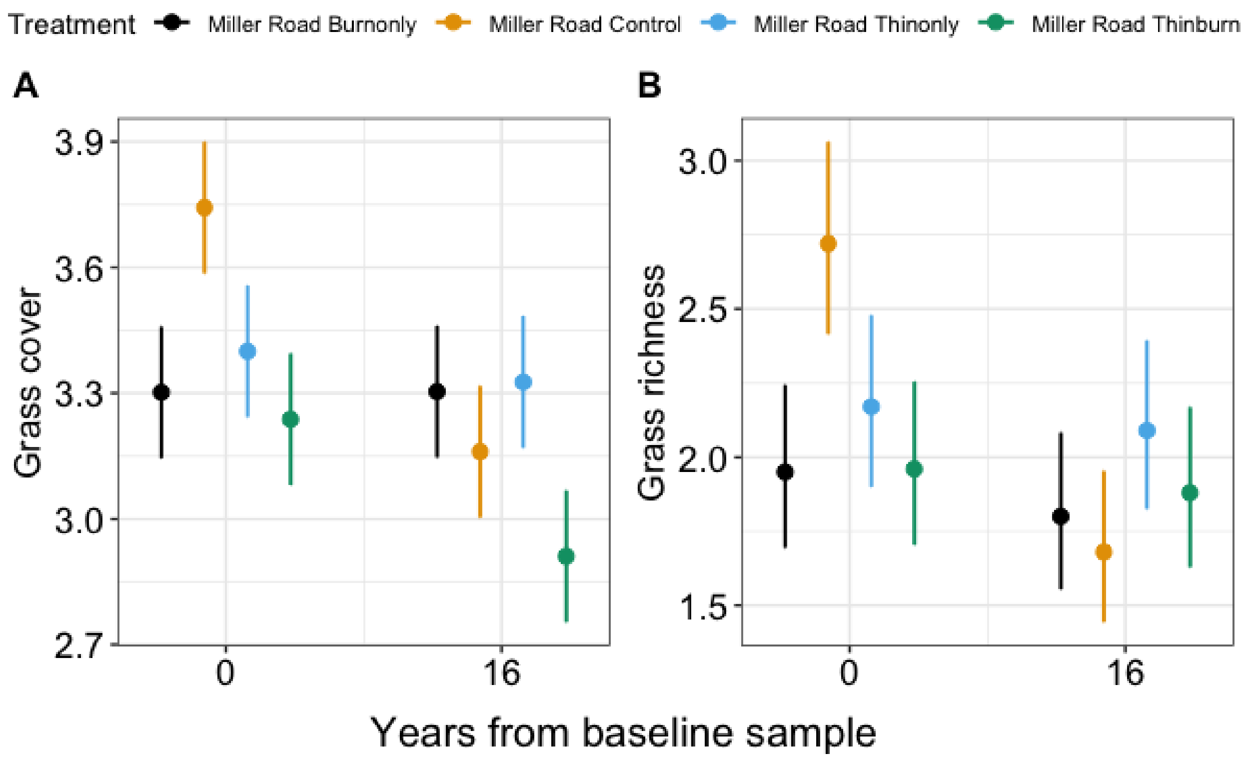

Figure 8: Response of grasses to the treatments in Miller Road. Figures show means and 95% Confidence Interval.

3.3 FORAGE

3.3.1 Data analysis

We pooled the data from all the different sites across years and tested the effect of management, i.e. burning, thinning and control, and the effects of enclosure, on grasses, forbs and exotic species yield. We eliminated Stoddart North from further analysis because data was available for only one sampling period (Table 6). We also removed from analysis years in which pinegrass and other grasses were combined into a single sample. For bunchgrass, pinegrass and other grass yield, we calculated the difference between caged and uncaged yield for each plot using the log response ratio: Log (Caged/Uncaged), LRR. We placed the caged values as a numerator, because we expected the yield in caged plots to be higher than uncaged plots, thus producing a positive, value. We added the number 1 to both sites to prevent errors due to dividing by zero. We included number of years from the baseline sample, in which baseline samples were denoted as collected at Year 0, and subsequent sampling years within each site were denoted as number of years after the baseline sample.

Table 5: Summary of available forage data for caged and uncaged plots.

	Enclosure			
Site	**Uncaged**	**Caged**	**Thin**	**Burn**
Hawke Road	2005, 2008, 2011, 2016	2008, 2016	Yes	No
Hofert Control	2006, 2007, 2008, 2009, 2010, 2011, 2015	2008, 2009, 2010, 2011, 2015	No	No
Hoodoo East	2010, 2015	2015	Yes	No
Stoddart North	2006	No	Yes	No
Rocks	2002, 2009, 2015	2002, 2007, 2009, 2015	Yes	Yes
Lewis Creek	2008, 2012	No	Yes	Yes
Gina Lake	2002, 2011	2002	Yes	Yes
Wolf Creek	1999, 2000, 2001, 2004, 2005, 2006	2003, 2004, 2005, 2006	Yes	Yes
Sheep Creek North	2000, 2001, 2004, 2005, 2006	2000, 2004, 2005, 2006	Yes	No

3.3.2 Results

Time significantly affected the difference in bunchgrass and pinegrass yield between caged and uncaged plots (Table 6, Figure 9). Log response ratio of other grasses did not differ according to any factor.

Table 6: Table showing yield of bunchgrasses, pinegrass, other grasses and exotic species over time across nine sites with different management treatments in the Southern Interior Rocky Mountain Trench.

		Bunchgrass		**Pinegrass**		**Other grasses**	
	df	Chi square	P value	Chi square	P value	Chi square	P value
Treatment	2	0.264	0.877	5.681	0.058	3.042	0.219
Year from baseline	1	**3.862**	**0.049**	**9.519**	**0.002**	2.187	0.139
Treatment * Year from baseline	2	**7.067**	**0.029**	5.902	0.052	3.880	0.144

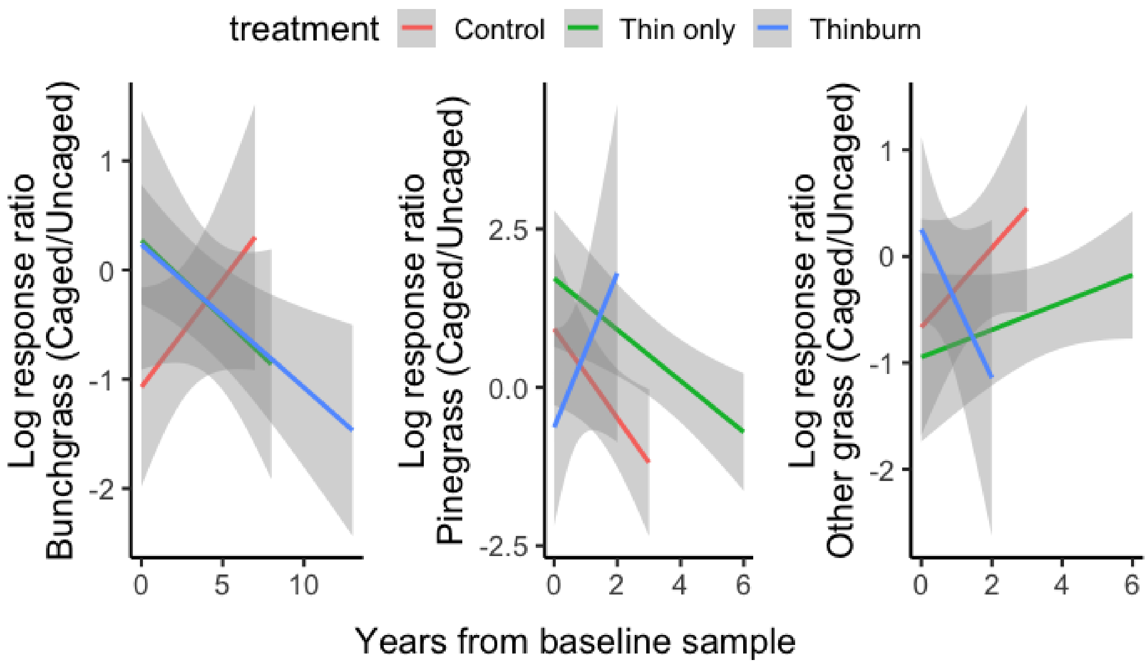

Figure 9: The effects of time and enclosure on bunchgrass and forb yield. Plots show responses and 95% confidence intervals.

Figure 10: Forage exclusion cages in Rocks site in 2007 (left) and 2009 (right)

4.0 DISCUSSION

4.1 How does thinning and/or burning affect tree regeneration and the relationship between understorey and overstorey vegetation?

Partial harvests, such as thinning and burning overstorey and mid-storey vegetation provide an opportunity for establishment of a new set of trees, through increased space and resources provided by thinning, and nutrient release and seed stimulation of serotinous cones that occurs after a burn (Bartuszevige and Kennedy 2009, Puettmann et al. 2013). We did not see an increase in natural regeneration of seedlings after thinning or burning in the sites in this study. The effects of prescribed burns on tree regeneration can be influenced by elevation, competing vegetation, available moisture, herbivory or severity of burns (Goto et al. 1996, Johnstone and Chapin 2006, Crotteau et al. 2013). Further, Stevens-Rumann and Morgan (2019) noted declining tree regeneration densities can happen due to changing climatic conditions and wildfires. Long term additional data from more sites and documentation of environmental and site characteristics will likely make it easier to highlight the relative importance of these factors on tree regeneration.

Stem density exhibited a strong effect on understory species richness and cover across different functional groups. Grasses and exotic species decreased with increasing tree density indicating that the ease of establishment and success of understorey plants declined with increasing tree density. Stem density is an important forest parameter that has direct effects on understory species diversity, although the nature of the relationship can range from a concave or convex to inversely linear (Thomas et al. 1999, Olson and Kabrick 2014, Ahmad et al. 2018). Bunchgrasses in this region been shown to have a light requirement, pinegrass prefers shady environment (Newman and DeMaere 2002, Page and Bork 2003). Therefore, further studies can identify the impact of thinning on the proportion of bunchgrasses or pinegrass that comprise the understory. The inability to detect a concave down relationship across the four sites for which overstorey and understorey vegetation data was present indicates that the best conditions for vegetation growth is when there’s no canopy present. Within site, thinning and/or burning in Gina Lake and Hawke Road modified the relationship between overstorey and understory from a significant inverse relationship to one that was not significant, i.e. flat. Thinning or burning releases understorey vegetation from the effects of dense canopy cover and provides better conditions for survival and establishment (Puettmann et al. 2013).

4.2 What effect does thinning and burning have on understorey vegetation?

Response of grass and exotic plants to thinning and burning was highly variable across all the sites. In Redstreak, grass cover and richness showed significant increases one and four years after burning in burned sites, which was not observed in control plots, while grass cover only increased one year after burning in thinned sites, suggesting that fire was crucial in restoring grasses in the understorey that are important for forage. In Miller Road, burnonly sites did not exhibit a significant change in grass cover and richness 16 years after burning, while thinned (but not burned) sites in Miller Road significantly decreased in grass cover and richness over this time period. In a similar study in this site, Ross (2012) demonstrated an overall decrease in grass cover, when data from burn/thin treatments was included, in addition to the treatments included in this study for Miller Road.

In the pooled analysis, we also observed an overall decline of understorey native grasses in the control sites pooled in this study, while grasses did not change over time in the other three treatments. This contrasts with studies that show that burning increases grass richness and cover due to increase in resources (Fynn et al. 2004, Morgan et al. 2015). The differences between overall results, and the results from Redstreak and Miller Road may be driven by pooling of data from areas with different environmental conditions and differences in burn intensity and severity. Other studies have noted the complex nature of understory vegetation responses to thinning and burning, and it is likely that the responses to these treatments is species specific (Abella and Springer 2015, Willms et al. 2017). More data with within site treatments and appropriate controls should clarify these vegetation responses, in addition to testing the responses of ecologically important plants individually.

Understorey species can decline after thinning due damage resulting from tree cutting operations, or burying of plants by slash (Metlen et al. 2004). Such piled slash can have long term negative effects lasting up to 15 years (Munger and Westveld 1931, Abella and Springer 2015). The differences in the results obtained between Redstreak and Miller Road, may be due to the increase in the amount of fuel obtained by thinning before prescribed burns (Wayman and North 2007). Importantly, these different results may also imply that fire and thinning are not surrogates for each other, and that burning and thinning may both be needed to obtain increase in native grasses (Willms et al. 2017). Thinning resulted in increases in understorey native and exotic vegetation over time across nine sites. Ross (2012) indicated that in Miller Road, increasing forest cover may lead to increases in pinegrass, accompanied by decreases in bunchgrass. It should be noted that since the data reported here for Miller Road were collected

15 years after logging and 8 years after slashing, the effects of these treatments in the short term for which there is no available data may be different from the long-term effects reported here.

In the pooled analysis, exotics species richness also increased regardless of treatment, while exotic species cover increased in thinonly and thin/burn sites. Generally, this suggests that the community in thinned and burned treatments gradually shifts towards having a higher proportion of exotic species. This mirrors results from a previous study, Newman and Hamilton (2018) in which exotic species increased in Redstreak following two prescribed fires, although they did not find long term increases in Miller Road. Similarly, in Bull River, another monitoring site within this program, Page (2010) found that invasive cheatgrass increased after a cool burn. In a systematic review of western North American conifer forests, Abella and Springer (2015) discovered that the highest number of exotic species occurred in treatments with cutting and prescribed fire compared to either treatment alone. Increasing opportunities for propagule transport into forests, combined with increased disturbance frequency, increased resource availability and increased intensity of tree removal are all factors that can play a role in increasing exotic species in thin-burn sites (Dodson and Fiedler 2006, Collins et al. 2007, Abella and Springer 2015). A meta-analysis of 32 studies from North American forests indicated that thinning alone can result in increased number of exotic species, compared to burns alone in the short term (Willms et al. 2017). Both fire and thinning can increase bare soil, remove litter and increase the amount of light that gets to the ground surface (Bartuszevige and Kennedy 2009). The additional influence of fire in stimulating seed germination by increasing soil temperatures may make it more beneficial for both native and exotic species. Thus, it is possible that thinning can create enough disturbance to induce exotic species success, but the response of exotic species to thinning is not likely to be as strong as it is to thin/burn, and is likely dependent on functional traits available in the species pool, the severity of the thinning treatments, and climatic factors (Willms et al. 2017).

4.3 How do thinning and prescribed fire influence differences in forage yield between caged and uncaged plots?

Grazing exclusion is generally regarded as an effective strategy to restore degraded grasslands (Medina-Roldán et al. 2012, Wang et al. 2019). The use of enclosures to rehabilitate overgrazed rangelands is common in many parts of the world (Verdoodt et al. 2010, Hao and He 2019, Xu et al. 2020). We expected the log response ratio: log(Caged/Uncaged) to be greater than one, but we did not find that caged consistently improved yield over time. Instead, cages decreased yield of bunchgrass over time in cages in thinonly and thinburn plots, but increased yield of pinegrass in thinburn plots. This suggests that grazing may be beneficial for bunchgrass yield in the long term. Studies from Californian ecosystems have found that grazing, when applied in short durations in the early spring can improve bunchgrass reproduction and native species cover and richness by remove exotic species inflorescences (Safford and Harrison 2001, Gelbard 2003, Davies et al. 2005). Also, when grazing is applied in the dormant season, it can remove thatch and allow more light to reach the ground (Menke 1992, Gelbard 2003). Bunchgrass has been shown to increase in vegetative growth and seedling production following wildfires (Marty et al. 2005, Ellsworth and Kauffman 2010), but we did not find any relationship between management and bunchgrass yield in this study. Forbs also increased in uncaged, grazed plots. Forbs are thought to comprise less than 20% of forage consumption by cattle (Clambey et al. 1986, Souther et al. 2019). The forb response in this study is likely due to lower consumption of forbs by grazers compared to grasses, coupled with greater resource availability due to grass consumption by grazers.

Pinegrass is reported to maintain abundance under partial understorey shade in this region and can decrease in response to thinning (Page et al. 2005, Ducherer 2006, Harrod et al. 2009). Page et al. (2005) reported decreases in pinegrass yield in a ponderosa pine zone after thinning. One explanation is that an overly dense canopy may also be detrimental to pinegrass success. In Northwestern USA, pinegrass production can increase following overstorey removal (McConnell and Smith 1965, Youngblood et al. 2006, Dodson et al. 2007). Yield of grasses apart from pinegrass and bunchgrass, did not respond to any of the treatments.

5.0 CONCLUSIONS AND RECOMMENDATIONS

1. Establishment of new trees was unaffected by thinning and burning treatments. However, grasses and exotics species clearly increase after tree removal. Removal of overstorey can be used to stimulate understorey grass growth, but exotics will be present and have to be controlled.
2. Both burning and thinning can improve richness and cover of native grasses, but exotic species may also increase. Any management actions applied should therefore consider the trade-offs in the increase of both grasses and exotic species.
3. It is difficult to reach conclusions on the impact of thinning and burning in many sites due to lack of controls. Assessing the impact of management is confounded by time between when treatments and sampling, time interval between thinning and burning, differences in sampling methods or ecological differences between sites. Control sites can be established in untouched areas around treatment locations with similar ecological characteristics. This will provide future data that can be compared to treatment sites.
4. Cages appear to depress bunchgrass yield, but does not affect yield of other grasses. Burning and thinning do not have an effect on grass yield.

LITERATURE CITED

Abella, S. R., and J. D. Springer. 2015. Effects of tree cutting and fire on understory vegetation in mixed conifer forests. Forest Ecology and Management **335**:281-299.

Abrams, M. D., and G. J. Nowacki. 2008. Native Americans as active and passive promoters of mast and fruit trees in the eastern USA. The Holocene **18**:1123-1137.

Adler, P., D. Raff, and W. Lauenroth. 2001. The effect of grazing on the spatial heterogeneity of vegetation. Oecologia **128**:465-479.

Ahmad, B., Y. Wang, J. Hao, Y. Liu, E. Bohnett, and K. Zhang. 2018. Optimizing stand structure for trade-offs between overstory timber production and understory plant diversity: A case-study of a larch plantation in northwest China. Land Degradation & Development **29**:2998-3008.

Altesor, A., M. Oesterheld, E. Leoni, F. Lezama, and C. Rodríguez. 2005. Effect of grazing on community structure and productivity of a Uruguayan grassland. Plant Ecology **179**:83-91.

Bakker, J. 1985. The impact of grazing on plant communities, plant populations and soil conditions on salt marshes. Vegetatio **62**:391-398.

Bartuszevige, A. M., and P. L. Kennedy. 2009. Synthesis of knowledge on the effects of fire and thinning treatments on understory vegetation in US dry forests.

Bates, D., D. Sarkar, M. D. Bates, and L. Matrix. 2007. The lme4 package. R package version **2**:74.

Bond, S., Mike Gall, Don Gayton, Randy Harris, Brad Munroe, Rob Neil, Hillary Page, Wendy Rockafellow, and S. Witbeck. 2013. Blue Print for Action 2013: Progress and Learnings 1997 - 2013.

Brockway, D. G., and C. E. Lewis. 1997. Long-term effects of dormant-season prescribed fire on plant community diversity, structure and productivity in a longleaf pine wiregrass ecosystem. Forest Ecology and Management **96**:167-183.

Clambey, G. K., D. R. Kirby, and M. F. Pessin. 1986. Forage Preferences Exhibited by Cattle on a Short Duration Grazing System. Farm Research; 43: 6; May/Jun 1986.

Collins, B. M., J. J. Moghaddas, and S. L. Stephens. 2007. Initial changes in forest structure and understory plant communities following fuel reduction activities in a Sierra Nevada mixed conifer forest. Forest Ecology and Management **239**:102-111.

Crane, M. 1991. Arctostaphylos uva-ursi. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory.

Crotteau, J. S., J. M. Varner III, and M. W. Ritchie. 2013. Post-fire regeneration across a fire severity gradient in the southern Cascades. Forest Ecology and Management **287**:103-112.

Cumming, S. 2011. Effective fire suppression in boreal forests. Canadian Journal of Forest Research **35**:772-786.

Davies, K. F., P. Chesson, S. Harrison, B. D. Inouye, B. A. Melbourne, and K. J. Rice. 2005. Spatial heterogeneity explains the scale dependence of the native–exotic diversity relationship. Ecology **86**:1602-1610.

Dodson, E. K., and C. E. Fiedler. 2006. Impacts of restoration treatments on alien plant invasion in Pinus ponderosa forests, Montana, USA. Journal of Applied Ecology **43**:887-897.

Dodson, E. K., K. L. Metlen, and C. E. Fiedler. 2007. Common and uncommon understory species differentially respond to restoration treatments in ponderosa pine/Douglas-fir forests, Montana. Restoration Ecology **15**:696-708.

Dodson, E. K., D. W. Peterson, and R. J. Harrod. 2008. Understory vegetation response to thinning and burning restoration treatments in dry conifer forests of the eastern Cascades, USA. Forest Ecology and Management **255**:3130-3140.

Ducherer, K. L. 2006. Effects of burning and thinning on species composition and forage production in British Columbia grasslands.

Ellsworth, L. M., and J. B. Kauffman. 2010. Native bunchgrass response to prescribed fire in ungrazed mountain big sagebrush ecosystems. Fire Ecology **6**:86-96.

Fulé, P. Z., J. E. Crouse, J. P. Roccaforte, and E. L. Kalies. 2012. Do thinning and/or burning treatments in western USA ponderosa or Jeffrey pine-dominated forests help restore natural fire behavior? Forest Ecology and Management **269**:68-81.

Fynn, R., C. Morris, and T. Edwards. 2004. Effect of burning on grass and forb diversity in a long-term grassland experiment. Applied Vegetation Science **7**:1-10.

Gallant, A. L., A. J. Hansen, J. S. Councilman, D. K. Monte, and D. W. Betz. 2003. Vegetation dynamics under fire exclusion and logging in a Rocky Mountain watershed, 1856–1996. Ecological Applications **13**:385-403.

Gelbard, J. 2003. Potential Positive and Negative Effects of Livestock Grazing on Valley and Foothill Plant Communities of the Interior California Floristic Province.

Goto, Y., T. Yoshitake, M. Okano, and K. Shimada. 1996. Seedling regeneration and vegetative resprouting after fires in Pinus densiflora forests. Vegetatio **122**:157-165.

Goubitz, S., R. Nathan, R. Roitemberg, A. Shmida, and G. Ne’eman. 2004. Canopy seed bank structure in relation to: Fire, tree size and density. Plant Ecology **173**:191-201.

Greene, G., and B. J. Harris. 2015. BC Ecosystem Restoration Program - Intensive Monitoring Protocols.

Gundale, M. J., T. H. DeLuca, C. E. Fiedler, P. W. Ramsey, M. G. Harrington, and J. E. Gannon. 2005. Restoration treatments in a Montana ponderosa pine forest: effects on soil physical, chemical and biological properties. Forest Ecology and Management **213**:25-38.

Haase, S. M. 1986. Effect of prescribed burning on soil moisture and germination of southwestern ponderosa pine seed on basaltic soils. Rocky Mountain Forest and Range Experiment Station.

Hao, Y., and Z. He. 2019. Effects of grazing patterns on grassland biomass and soil environments in China: A meta-analysis. PloS one **14**:e0215223-e0215223.

Harrod, R., R. Fonda, and M. McGrath. 2009. Vegetation Response to Thinning and Burning in a Ponderosa Pine Forest, Washington. Northwest Science **82**:141-150.

Johnstone, J. F., and F. S. Chapin. 2006. Effects of soil burn severity on post-fire tree recruitment in boreal forest. Ecosystems **9**:14-31.

Keane, R. E., K. C. Ryan, T. T. Veblen, C. D. Allen, J. Logan, and B. Hawkes. 2002. Cascading Effects of Fire Exclusion in Rocky Mountain Ecosystems: A Literature Review.

Keeling, E. G., A. Sala, and T. H. DeLuca. 2006. Effects of fire exclusion on forest structure and composition in unlogged ponderosa pine/Douglas-fir forests. Forest Ecology and Management **237**:418-428.

Lenth, R., H. Singmann, and J. Love. 2018. Emmeans: Estimated marginal means, aka least-squares means. R package version **1**.

Lu, X., K. C. Kelsey, Y. Yan, J. Sun, X. Wang, G. Cheng, and J. C. Neff. 2017. Effects of grazing on ecosystem structure and function of alpine grasslands in Qinghai–Tibetan Plateau: a synthesis. Ecosphere **8**:e01656.

Macdougall, A., K. McCann, G. Gellner, and R. Turkington. 2013. Diversity loss with persistent human disturbance increases vulnerability to ecosystem collapse. Nature **494**:86-89.

Machmer, M., H. Page, and C. Steeger. 2001. An effectiveness monitoring plan for NDT4 ecosystem restoration in the East Kootenay Trench. Prepared for.

Marty, J. T., S. K. Collinge, and K. J. Rice. 2005. Responses of a remnant California native bunchgrass population to grazing, burning and climatic variation. Plant Ecology **181**:101-112.

McConnell, B. R., and J. G. Smith. 1965. Understory response three years after thinning pine. Rangeland Ecology & Management/Journal of Range Management Archives **18**:129-132.

Medina-Roldán, E., J. Paz-Ferreiro, and R. D. Bardgett. 2012. Grazing exclusion affects soil and plant communities, but has no impact on soil carbon storage in an upland grassland. Agriculture, ecosystems & environment **149**:118-123.

Menke, J. W. 1992. Grazing and fire management for native perennial grass restoration in California grasslands. Fremontia **20**:22-25.

Metlen, K., C. Fiedler, and A. Youngblood. 2004. Understory response to fuel reduction treatments in the Blue Mountains of Northeastern Oregon. Northwest Science **78**:175-185.

Milchunas, D. G., and W. K. Lauenroth. 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments: Ecological Archives M063-001. Ecological monographs **63**:327-366.

Molino, J.-F., and D. Sabatier. 2001. Tree diversity in tropical rain forests: a validation of the intermediate disturbance hypothesis. Science **294**:1702-1704.

Morgan, P., M. Moy, C. A. Droske, S. A. Lewis, L. B. Lentile, P. R. Robichaud, A. T. Hudak, and C. J. Williams. 2015. Vegetation response to burn severity, native grass seeding, and salvage logging. Fire Ecology **11**:31-58.

Munger, T. T., and R. H. Westveld. 1931. Slash disposal in the western yellow pine forests of Oregon and Washington. US Dept. of Agriculture.

Newman, R., and C. DeMaere. 2002. Monitoring restoration of fire-maintained ecosystems in the Invermere Forest District. Ministry of Forests, Research Branch.

Newman, R., and E. Hamilton. 2018. Burning Questions: Effects of restoration burning on selected PP and IDF zone ecosystems in the Southern Interior Rocky Mountain Trench.

Olson, E. K., and J. M. Kabrick. 2014. Understory vegetation composition and abundance in relation to light, water, and nutrient supply gradients in upland oak woodlands. Pages 63-66 *in* In: Groninger, John W.; Holzmueller, Eric J.; Nielsen, Clayton K.; Dey, Daniel C., eds. Proceedings, 19th Central Hardwood Forest Conference; 2014 March 10-12; Carbondale, IL. General Technical Report NRS-P-142. Newtown Square, PA: US Department of Agriculture, Forest Service, Northern Research Station: 63-66.

Page, H. 2010. Monitoring ecosystem restoration treatments at North Stoddart Creek and Bull River.

Page, H., and E. W. Bork. 2003. Using Bunchgrass Plugs to Restore Degraded Rangeland. Pages 43 - 52 *in* ECOSYSTEM AT RISK: ANTELOPE BRUSH RESTORATION.

Page, H., and M. Machmer. 2003. Monitoring Ecosystem Restoration Treatments in the Rocky Mountian Trench: Site Establishment Report.

Page, H. N., E. Bork, and R. Newman. 2005. Understorey responses to mechanical restoration and drought within montane forests of British Columbia. Journal of Ecosystems and Management **6**.

Parisien, M.-A., Q. E. Barber, K. G. Hirsch, C. A. Stockdale, S. Erni, X. Wang, D. Arseneault, and S. A. Parks. 2020. Fire deficit increases wildfire risk for many communities in the Canadian boreal forest. Nature Communications **11**:1-9.

Pettit, N. E., R. H. Froend, and P. G. Ladd. 1995. Grazing in remnant woodland vegetation: changes in species composition and life form groups. Journal of Vegetation Science **6**:121-130.

Potts, J. B., and S. L. Stephens. 2009. Invasive and native plant responses to shrubland fuel reduction: comparing prescribed fire, mastication, and treatment season. Biological Conservation **142**:1657-1664.

Power, M., F. Mayle, P. Bartlein, J. Marlon, R. Anderson, H. Behling, K. Brown, C. Carcaillet, D. Colombaroli, and D. Gavin. 2013. Climatic control of the biomass-burning decline in the Americas after AD 1500. The Holocene **23**:3-13.

Puettmann, K. J., E. K. Dodson, A. Ares, and C. A. Berger. 2013. Short-term Responses of Overstory and Understory Vegetation to Thinning Treatments: A Tale of Two. Page 44 *in* Density Management in the 21st Century: West Side Story: Proceedings of the Density Management Workshop, 4-6 October 2011, Corvalllis, Oregon. US Department of Agriculture Forest Service, Parcific Northwest Research Station.

Ralph, D., V. Miller, C. Hougen, and J. Leekie. 2014. Field guide to noxious and other selected weeds of British Columbia. Invasive Species Council of Birtish Columbia.

Ross, T. J. 2011. Range inventory and ecosystem restoration monitoring methodology: 1996 to 2011. BC Ministry of Natural Resource Operations Rocky Mountain Forest District.

Ross, T. J. 2012. Plant Community Response Following Dry Forest Ecosystem Restoration. The Rocky Mountain Trench Natural Resources Society.

Roxburgh, S. H., K. Shea, and J. B. Wilson. 2004. The intermediate disturbance hypothesis: patch dynamics and mechanisms of species coexistence. Ecology **85**:359-371.

Ryan, K. C., E. E. Knapp, and J. M. Varner. 2013. Prescribed fire in North American forests and woodlands: history, current practice, and challenges. Frontiers in Ecology and the Environment **11**:e15-e24.

Safford, H. D., and S. P. Harrison. 2001. GRAZING AND SUBSTRATE INTERACT TO AFFECT NATIVE VS. EXOTIC DIVERSITY IN ROADSIDE GRASSLANDS. Ecological Applications **11**:1112-1122.

Schoennagel, T., T. T. Veblen, and W. H. Romme. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience **54**:661-676.

Semmartin, M., and M. Oesterheld. 1996. Effect of Grazing Pattern on Primary Productivity. Oikos **75**:431-436.

Souther, S., M. Loeser, T. E. Crews, and T. Sisk. 2019. Complex response of vegetation to grazing suggests need for coordinated, landscape-level approaches to grazing management. Global Ecology and Conservation **20**:e00770.

Steel, Z. L., H. D. Safford, and J. H. Viers. 2015. The fire frequency-severity relationship and the legacy of fire suppression in California forests. Ecosphere **6**:art8.

Stephenson, N. L. 1999. Reference conditions for giant sequoia forest restoration: structure, process, and precision. Ecological Applications **9**:1253-1265.

Stevens-Rumann, C. S., and P. Morgan. 2019. Tree regeneration following wildfires in the western US: a review. Fire Ecology **15**:15.

Stohlgren, T. J., L. D. Schell, and B. V. Heuvel. 1999. How Grazing and Soil Quality Affect Native and Exotic Plant Diversity in Rocky Mountain Grasslands. Ecological Applications **9**:45-64.

Switzer, J. M., G. D. Hope, S. J. Grayston, and C. E. Prescott. 2012. Changes in soil chemical and biological properties after thinning and prescribed fire for ecosystem restoration in a Rocky Mountain Douglas-fir forest. Forest Ecology and Management **275**:1-13.

Symstad, A., W. Newton, and D. Swanson. 2014. Strategies for preventing invasive plant outbreaks after prescribed fire in ponderosa pine forest. Forest Ecology and Management **324**:81-88.

Thomas, S. C., C. B. Halpern, D. A. Falk, D. A. Liguori, and K. A. Austin. 1999. Plant diversity in managed forests: Understory responses to thinning and fertilization. Ecological Applications **9**:864-879.

Tveten, R. 1997. Fire effects on prairie vegetation Fort Lewis, Washington. Dunn and Ewing:123-130.

Veblen, T. T., T. Kitzberger, and J. Donnegan. 2000. Climatic and human influences on fire regimes in ponderosa pine forests in the Colorado Front Range. Ecological Applications **10**:1178-1195.

Verdoodt, A., S. M. Mureithi, and E. Van Ranst. 2010. Impacts of management and enclosure age on recovery of the herbaceous rangeland vegetation in semi-arid Kenya. Journal of Arid Environments **74**:1066-1073.

Wang, S., J. Fan, Y. Li, and L. Huang. 2019. Effects of grazing exclusion on biomass growth and species diversity among various grassland types of the Tibetan Plateau. Sustainability **11**:1705.

Wayman, R. B., and M. North. 2007. Initial response of a mixed-conifer understory plant community to burning and thinning restoration treatments. Forest Ecology and Management **239**:32-44.

Willms, J., A. Bartuszevige, D. W. Schwilk, and P. L. Kennedy. 2017. The effects of thinning and burning on understory vegetation in North America: A meta-analysis. Forest Ecology and Management **392**:184-194.

Xu, L., Y. Nie, B. Chen, X. Xin, G. Yang, D. Xu, and L. Ye. 2020. Effects of Fence Enclosure on Vegetation Community Characteristics and Productivity of a Degraded Temperate Meadow Steppe in Northern China. Applied Sciences **10**:2952.

Youngblood, A., K. L. Metlen, and K. Coe. 2006. Changes in stand structure and composition after restoration treatments in low elevation dry forests of northeastern Oregon. Forest Ecology and Management **234**:143-163.

Table A1: List of exotic and problematic species used in this study.

smooth brome (BROMINE)
cheatgrass (BROMTEC)
Canada thistle (CIRSARV)
bull thistle (CIRSVUL)
field bindweed (CONVARV)
quackgrass (ELYMREP)
leafy spurge (EUPHESU)
common St. John's-wort (HYPEPER)
summer-cypress (KOCHSCO)
Oxeye daisy (LEUCVUL)
perennial sow-thistle (SONCARV)
Sonchus sp (SONCHUS)
yellow salsify (TRAGDUB)
great mullein (VERBTHA)

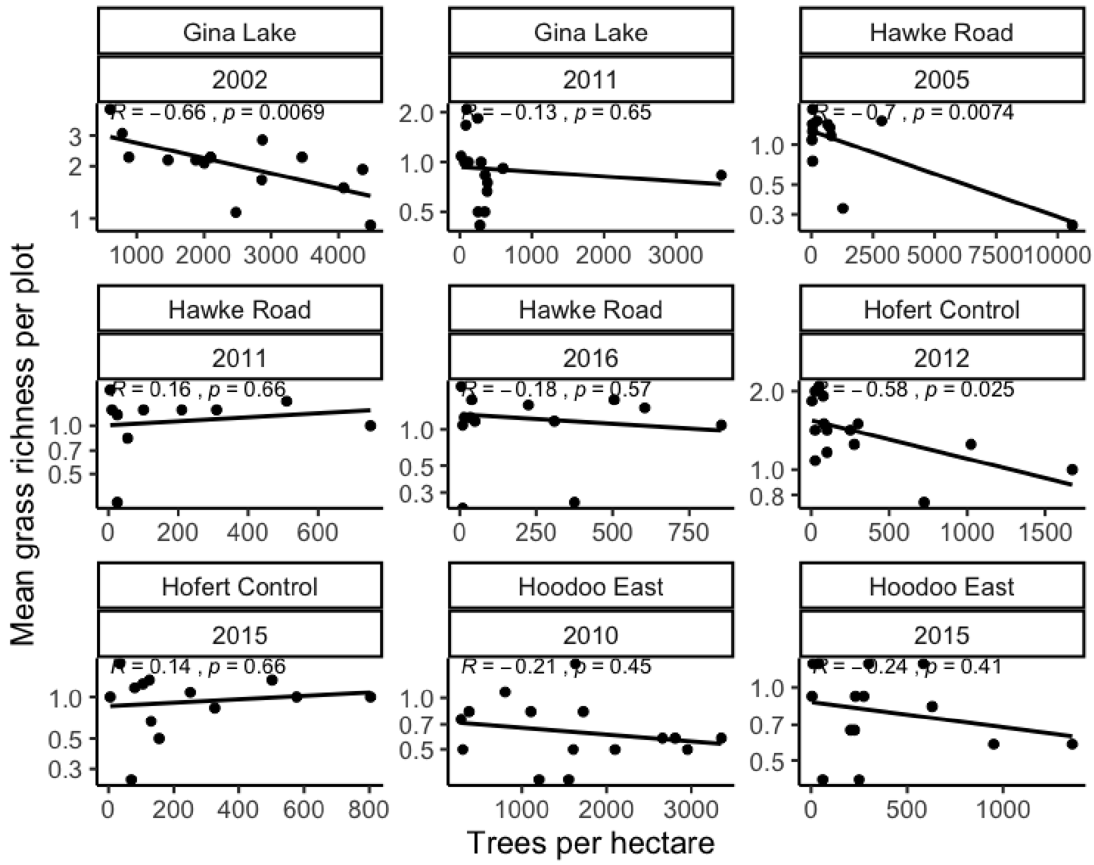

Figure A1: Relationship between grass richness and overstorey trees for each site-year combination for which data was available.

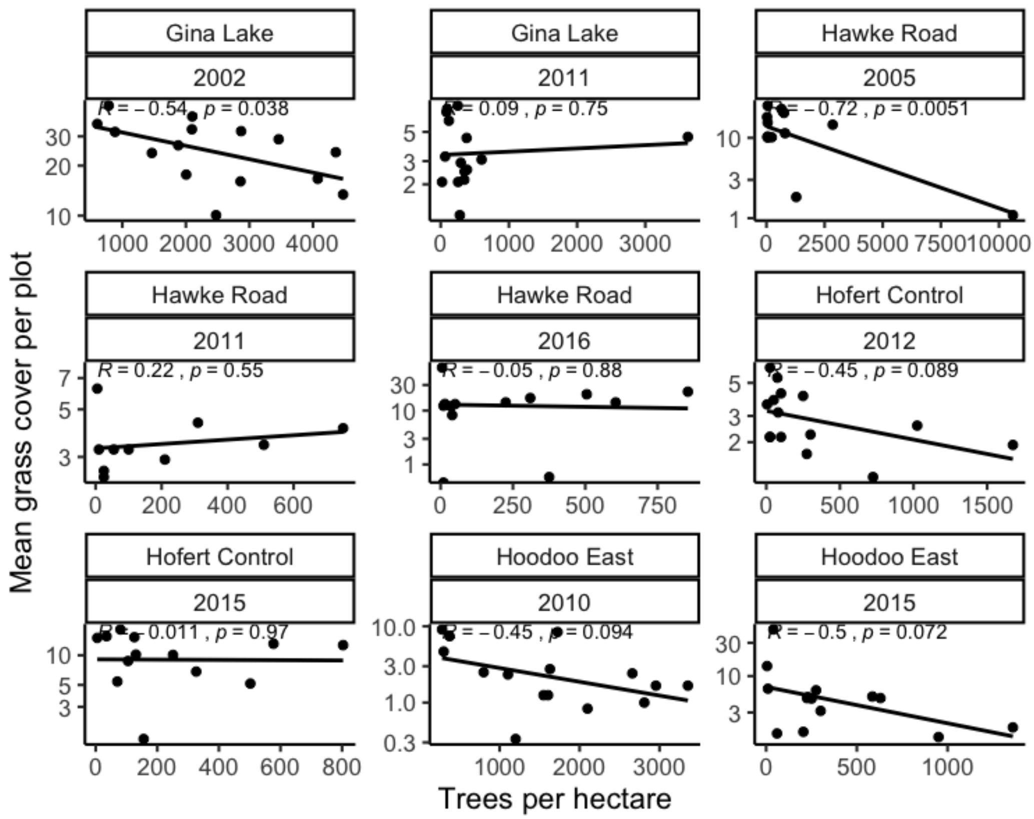

Figure A2: Relationship between grass cover and overstorey trees for each site-year combination for which data was available.