RESILIENCE OF SAGEBRUSH STEPPE PLANTS NINE YEARS AFTER FIRE AND SOIL DISTURBANCE IN SOUTHWEST MONTANA

by

Isaac Todhunter Newell

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TABLE OF CONTENTS

LIST OF TABLES

Table Page

LIST OF FIGURES

Figure Page

ABSTRACT

Sagebrush steppe in the western United States faces an increasing risk of degradation and overall habitat loss from land use and climate change. One important result of these drivers is the spread of the nonnative annual grass, *Bromus tectorum*. In many of these grasslands, *B. tectorum* has altered fire regimes through a positive feedback loop in which increased *B. tectorum* leads to more severe and frequent fires, and consequently increased *B. tectorum* cover. This ultimately displaces native flora and fauna, moving grasslands away from historic structure and function. However, in the northern portion of the sagebrush steppe, native communities were recently found to be resilient to fire 3 years post-fire, but not to soil damage. I revisited the study site (Red Bluff, Montana, US) to evaluate how this plant community has responded to fire and soil damage 9 years after disturbance. I sampled the same three treatments: an area burned by the 2012 Red Bluff fire, an adjacent unburned area, and a bulldozed firebreak between the two. In each treatment, 28 x 1m^2-plots were sampled along three transects running parallel to the fire break. Percentage cover of each species, litter, bare ground, rock, and manure were recorded and analyzed for differences in richness, ground cover, and *Bromus tectorum* cover, as well as alpha- and beta-diversities, using generalized linear models. Nearly a decade after disturbance, fire and soil damage still significantly affected species richness ($p = 0.05$) and species composition ($p = 0.02$). Both the unburned and burned treatments had a greater cover of native species than did the bulldozed plots ($p < 0.01$). *Bromus tectorum* was the most abundant species in the bulldozed treatment, with 186.7% greater relative abundance than unburned treatment, and 107.6% greater than the burned treatment ($p < 0.01$). This shows that while the plant community is resilient and resistant to nonnative invasion after fire, it is much less so after soil disturbance. This may have implications on future applications of fire breaks and the way they are managed after fire, as well as other mechanical disturbances to soil structure.

1. INTRODUCTION

1.1 Global change and disturbance

Global change is altering the world's ecosystems and is projected to continue doing so at increasing rates. Of the terrestrial ecosystems (arctic, boreal, grassland, savanna, Mediterranean, desert, tropical, and temperate), grasslands are expected to experience among the highest levels of change. These changes in grasslands are predicted to be primarily driven by climate and land-use change (Sala et al., 2000; Smith et al., 2015). Disturbance, defined here as 'any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment' (White & Pickett, 1985) is a part of every ecosystem in some capacity, whether that be naturogenic disturbances such as wildfire, heavy wind events or draught, or anthropogenic disturbances such as logging, mechanical soil disturbance, or a combination thereof. However, both land use and climate change have the ability to alter the frequency and types of disturbance that occur within a plant community beyond what would be historically expected (Hobbs et al., 2007; Schäfer et al., 2019; Pecl et al., 2017). Other drivers such as nitrogen deposition, biotic exchange, and atmospheric carbon dioxide levels are important factors of global change as well, though likely to have lesser effects than land use and global change on grasslands (Sala et al., 2000).

Ecosystem resilience, or the capacity of an ecosystem to regain its fundamental structure, processes, and function after disturbance or stress, is often linked with disturbance (Hirota et al., 2011; Chambers et al., 2014; Seidl et al., 2016). Many ecosystems have co-evolved with and can respond resiliently to their respective disturbance regimes, exhibiting various stable states and successional stages (Turner et al., 1998; Grime, 1977). However, as disturbance regimes are being altered, so are plant community's levels of resilience (Turner et al., 2003). Partnered with resilience is resistance: the capacity of ecosystems to retain their fundamental structure when exposed to stresses or disturbances (i.e., resist nonnative species invasions) (Folke et al., 2004; Chambers et al., 2014). Resistance is a function of abiotic and biotic attributes and is increasingly important in grassland systems (D'Antionio & Thomsen, 2004). Resilience and resistance often interact when faced with a changing disturbance regime; a reduction in resilience via changes in disturbance regime may also result in longer recovery time and decreased resistance, leading to a more rapid shift to a novel plant community (Oliver et al., 2015).

1.2 Fire

Fire is an important disturbance globally, promoting growth and biodiversity in many ecosystems. Plant communities that have evolved in the presence of periodic fire are often resilient to, or reliant on fire (He et al., 2019, Heim et al., 2021). This has been demonstrated in sagebrush steppe plant communities that have co-evolved with periodic fire disturbance being able to return to a pre-fire state after a burn (Lavin et al., 2013; Seipel et al., 2018). However, fire regimes are being significantly altered, especially in the Western United States (Brown et al., 2004). Climate change, fire suppression, and altered species compositions, among other factors, are all leading to divergences from historic fire regimes (Brooks et al., 2004). Fires on average are becoming more frequent, more severe, and are often resulting in novel post-fire plant communities, further leading to altered disturbance regimes (Millspaugh et al., 2000). For example, in sagebrush steppe fire is estimated to have occurred at a 30–80-year interval pre-Euro-American colonization. This has shifted to burns occurring as frequently as every 3-5 years, disallowing the establishment of traditional plant communities in many areas (Slaton and Stone, 2013; Whisenant, 1990; Burkhart and Tisdale, 1976).

1.3 Sagebrush Steppe

One of the most important types of grassland is in the western US is sagebrush steppe. Steppes are unforested grass and shrublands in semi-arid climates (25-50 cm annual rainfall) found between the tropics and polar regions, and sagebrush steppe is dominated by one of many sagebrush (*Artemisia*) species, typically *Artemisia tridentada* (West, 1983). Sagebrush steppe is often referred to as a "threatened" or "endangered ecosystem" due to losses of total area and changes of composition (Reisner et al., 2013; Pyke et al., 2015). Like grasslands as a whole, climate and land-use changes are the primary drivers for this (Sala et al., 2000). Overgrazing by livestock is a major factor in altering sagebrush steppe. Removal of native perennial grass biomass, creation of bare and disturbed soils, and spread of nonnative seeds are all results of overgrazing in this community (Pyke et al., 2015; Mack and Thompson, 1982). This in turn can alter fire regime, as well as drive pinyon and juniper expansion into sagebrush habitat (Romme et al., 2009; Miller and Rose, 1999; Slaton and Stone, 2013).

1.4 *Bromus tectorum*

Within sagebrush steppe, the nonnative species of greatest concern is *Bromus tectorum L,* commonly called cheatgrass. *Bromus tectorum* is a winter annual grass native to Europe, northern Africa, and Southwest Asia. Originally brought to the Americas in the mid-1800s, *B. tectorum* has now invaded plant communities across much of the US and Canada (Sheley, 1999; Kostivkovsky & Young, 2000). It has been estimated that up to 40% of sagebrush habitat is at risk of *B. tectorum* invasion (Suring et al., 2005). In Montana, *B. tectorum* is listed as a Priority 3 regulated plant by the Montana Department of Agriculture, due to its outcompeting of native vegetation and its relationship to fire (Pokorny & Mangold, 2020). Controlling this species is an ongoing challenge for land managers both in Montana and across the US (Morrow & Stahlman, 1984).

Bromus tectorum's early-season growth and subsequent early senescence greatly increase fine fuel loads in early-to-mid summer, which has been linked to earlier fires, as well as increased frequency and severity of fire (Knapp, 1996; Reisner et al., 2013; Whisenant, 1990). In some areas of the Western US, this altered fire regime then primes the plant community for further *B. tectorum* invasion, creating a positive feedback loop in which *B. tectorum* can increase its dominance (Figure 1) (Taylor et al., 2014; Young and Evans, 1979). However, this positive feedback loop is not consistent across the sagebrush biome and is not well documented in Montana and Northern Idaho in areas without heavy grazing pressure. In 2013, Lavin et al., (2013) found that fire by itself represents only a relatively weak force in changing the post-fire species composition of sagebrush steppe, including the invasion of *B. tectorum*. In areas with grazing pressure, this pattern has been seen more often (Young and Evans, 1979; Mack, 2010). In southwest Montana, it was found that fire had a small positive effect on *B. tectorum* cover 3 years after fire in a grazed community, though the positive effect of fire was much smaller than that of soil disturbance in the same area (Seipel et al., 2018). Soil disturbance often has been found to have a strong positive relationship with subsequent *B. tectorum* increases*,* though is generally less discussed than fire as a driver of the invasion (Mack, 2010; Lavin et al., 2013; Leffler et al., 2016; Speziale et al., 2018). However, most studies on *B. tectorum* invasion focus on the first few years after disturbance (both fire and soil disturbance), and so a gap in information for longer-term resilience and resistance to *B. tectorum* exists for the northern sagebrush steppe.

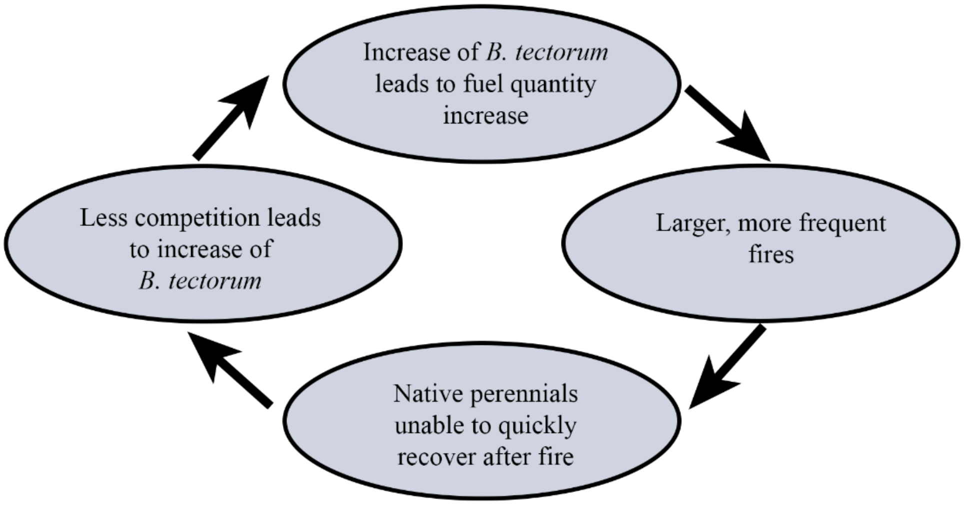

Figure 1. *Bromus tectorum* feedback loop with fire.

1.5 Goals

The goal of my study was to determine how the plant biodiversity within a sagebrush steppe has responded to fire and mechanical disturbance that occurred nearly a decade ago. I ask the specific questions, how does fire and mechanical soil disturbance affect plant: (1) species richness (2) total, dominant and *Bromus tectorum* abundance, (3) alpha diversity and species evenness, as well as (4) beta-diversity.

2. METHODS:

2.1 Study site:

I conducted research 14 to 18 June, 2021, at the Montana State University Red Bluff Research Ranch (45.593°N, 111.628°W) in southwestern Montana, USA. On 7 July 2012, a low-to-moderate severity fire burned a large section of the Red Bluff Research Ranch (646 hectares) (Becker et al., 2012). A bulldozed fire break was implemented at the site, which successfully stopped the fire's progress. Since 2012, the burned and bulldozed areas have been passively revegetated in the presence of light livestock grazing.

Mean annual temperature at Red Bluff is 6.4°C, and mean annual precipitation is 40 cm (PRISM Climate Group, 2004). The vegetation type at Red Bluff is classified as *Festuca idahoensis– Pseudoroegneria spicata, Artemisia tridentata–Pseudoroegneria spicata* or *Artemisia tridentata–Festuca idahoensis*, depending on *A. tridentata* cover (Mueggler & Stewart 1979). Both *Pseudoroegneria spicata* and *Festuca idahoensis* are perennial bunchgrasses. Soils are of the Nuley Rock Outcrop complex and are sandy loam from 0–10 cm and sandy clay loam from 10–30 cm. During data collection, Red Bluff and surrounding areas were in a period of drought (NOAA, 2021).

2.2 Sampling Design:

Data were collected at four blocks delineated in a previous study at the Red Bluff Research Ranch (Seipel et al., 2018). Block locations were selected for having a shallow east-facing slope and open steppe vegetation. Within each block, three parallel 50-m transects were randomly placed, one each in the burned area, unburned area, and the firebreak (n =12 transects). Firebreak transects were placed in the middle of the firebreak and the unburned and burned transects were placed 10 m away from the edge of the firebreak in their respective treatment areas. 1m x 1m quadrats were placed every 8 m along each transect (n =7 per transect, n =84 total). In each quadrat, I recorded the ocular cover of each species present, as well as the percent cover of bare ground, rock, litter, and manure. I performed all sampling during the third week of June to align temporally with previous studies at the site.

2.3 Data Analysis:

Plant species richness, total plant species abundance, vegetative cover, litter cover, bare ground cover, *Bromus tectorum* cover, Simpson's and Shannon alpha diversity, and Morisita Horn and Bray-Curtis beta-diversity were response variables and fire treatment (burned, bulldozed, and unburned) was the fixed effect, and block the random effect in all models. I assessed all response variables for normality; *B. tectorum* cover was transformed via logarithm to achieve normality.

I evaluated species richness using a generalized linear model (GLM) with a Poisson distribution. I evaluated total cover, vegetative cover, litter cover, bare ground cover, and *B. tectorum* cover each with a GLM with a Gaussian distribution. Differences were assessed using analysis of variance (ANOVA) and Tukey's pairwise comparison. Rank abundance, relative abundance, and top-five most abundant species were found for each treatment using R package BiodiversityR.

I evaluated Alpha diversity with inverse Simpson's (1/D) and Shannon diversity and evenness with Inverse Simpson's and Shannon evenness, in all cases a GLM with a Gaussian distribution. I assessed differences using ANOVA and Tukey's pairwise comparison for both alpha diversity indices and evenness indices. Inverse Simpson's evenness was found by dividing Inverse Simpson's diversity by total number of species (richness). I assessed Shannon evenness by dividing Shannon diversity by the logarithm of the species richness. Simpson's and Shannon indices were selected as measures of alpha-diversity because they are regarded as the most reliable indices of this metric (Magurran, 2004).

I assessed beta diversity using the Bray-Curtis and Morisita-Horn indices and differences assessed using an Adonis multivariate analysis of variance test. I performed a principal coordinate analysis on the Bray-Curtis beta diversity. Bray-Curtis Morisita-Horn indices were selected as measures for beta-diversity because they are regarded as the most reliable beta-diversity indices (Magurran, 2004). All analyses were performed on RStudio Version 1.3.959 (Oksanen et al., 2019), with vegan, lmerTest, and Biodiversity R packages.

3. RESULTS:

3.1 Species Richness

The total number of species found at the site was 61, with 42 in the burned treatment, 44 in the unburned treatment, and 46 in the bulldozed treatment. Mean species richness varied among treatments ($p = 0.05$). The burned treatment (8.7 species/m^2, SD = 3.35) contained 22% more species per m^2 than the unburned area (7.1 species/m^2, SD =3.18; $p = 0.04$) (Figure 2) when compared with Tukey pairwise comparison. Mean species richness did not vary between the burned and bulldozed treatments (7.2 species/m^2, SD = 4.09; $p = 0.39$), or between the unburned and bulldozed treatments ($p > 0.99$) (Figure 2).

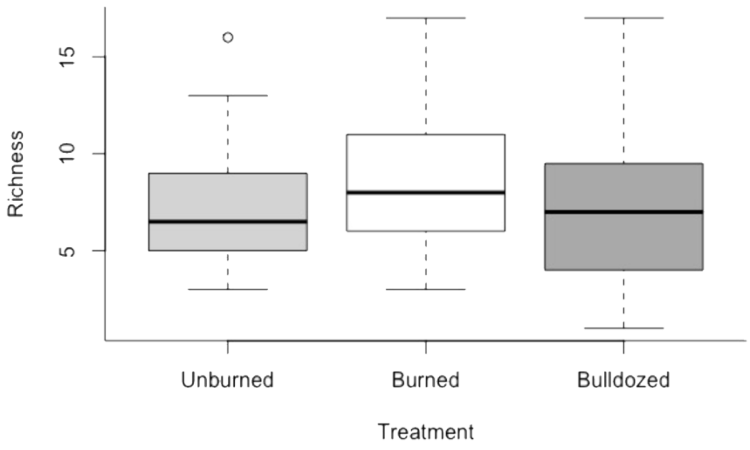

Figure 2. Mean plant species richness among treatments nine years post-disturbance at Red Bluff Research Ranch, Montana.

3.2 Community composition

Vegetative cover in the unburned area (40%) was the lowest among the three treatments. This was 13.4% less than the burned ($p < 0.01$), and 7.2% less than the bulldozed treatment ($p = 0.026$) (Figure 3). There was weak evidence that the burned area had a mean vegetative cover 6.2% greater than the bulldozed treatment ($p = 0.054$) (Figure 3). Total ground cover (including bare ground, litter, and manure) did not differ among the three treatment levels ($p = 0.31$). Assessing bare ground and litter individually found that bare ground did not differ by disturbance ($p = 0.21$), but litter cover did ($p < 0.01$). The highest mean litter cover was found in the unburned treatment (43.1%), which had 57% more litter than the burned plots (27.5%) ($p < 0.01$) and 23.8% more than the bulldozed treatment (34.8%) ($p = 0.01$). No difference in litter cover was found between the burned and bulldozed treatment ($p = 0.09$) (Figure 4).

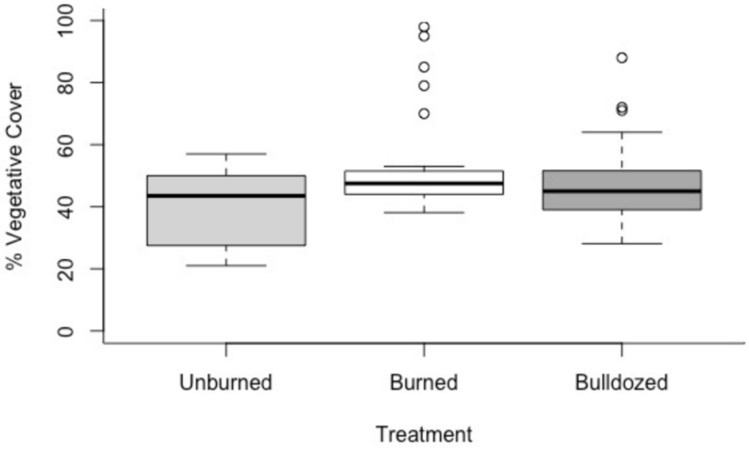

Figure 3. Mean vegetative cover (%) among treatments nine years post-disturbance at Red Bluff Research Ranch, Montana

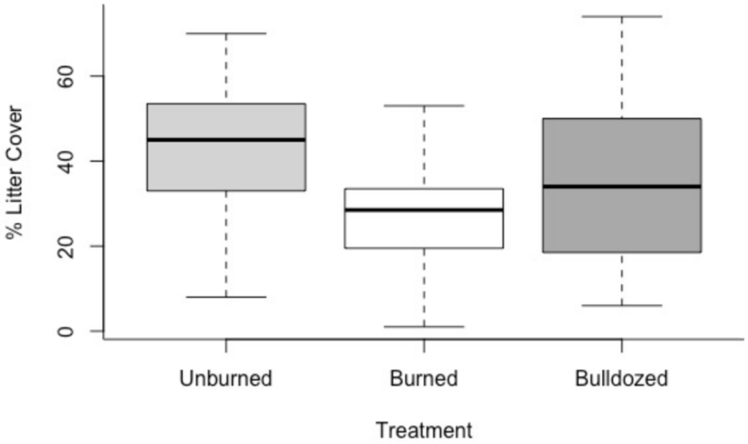

Figure 4. Mean litter cover (%) among treatments nine years post-disturbance at Red Bluff Research Ranch, Montana.

3.3 Individual species cover

The bulldozed treatment (14.28%) had a mean *B. tectorum* cover 238.8% greater than the unburned treatment (4.21%) ($p = 0.03$). There was weak evidence that the burned treatment (7.72%) had a mean *B. tectorum* cover 83.13% greater than the unburned treatment ($p = 0.055$). No significance was found between the burned and bulldozed treatments ($p = 0.57$) (Figure 5). The bulldozed area did however have significantly greater relative abundance than the burned area (107.6%) ($p < 0.01$) (Table 1).

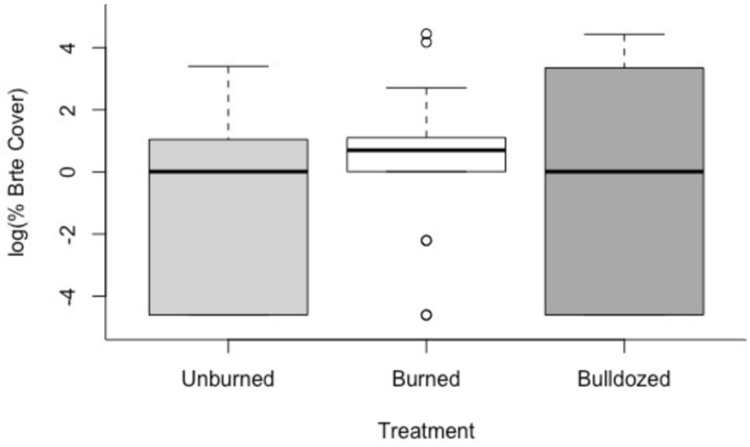

Figure 5. Logarithm of mean *Bromus tectorum* cover (%) among treatments nine years post-disturbance at Red Bluff Research Ranch, Montana.

The shape of the rank abundance curves shows that each treatment was dominated by three to four species. The unburned treatment's top species were most disproportionately dominant, with one primary (*Festuca idahoensis*) and two secondary species (*Stipa comata*, *Bromus tectorum*). The two disturbed treatments had smoother rank abundance curves, with the burned area having the smoothest and most even species abundance (Figure 6) (Table 1). *Bromus tectorum* and *Festuca idahoensis* were in each treatment's top three dominant species, and *Pseudoroegneria spicata* and *Stipa comata* were in each treatment's top four species. *Festuca idahoensis* had the highest overall mean cover and *Bromus tectorum* the second. *Festuca idahoensis* in the unburned treatment had the highest relative abundance of any plant species in any treatment (Table 1).

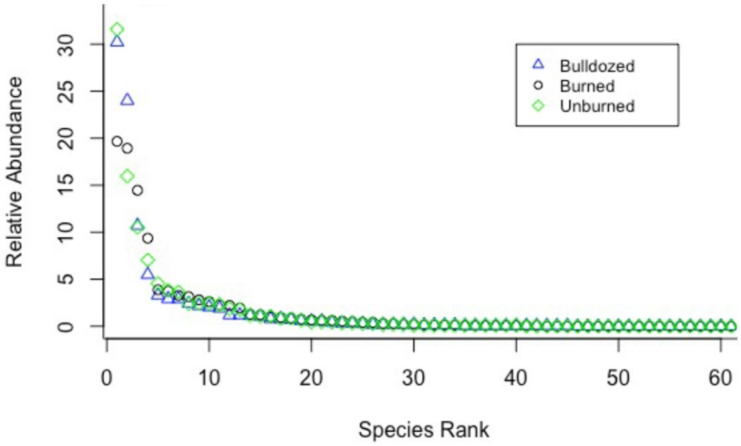

Figure 6. Relative species abundance curves for three treatments nine years post-disturbance at Red Bluff, MT.

Table 1. Five most dominant plant species in each of three treatments nine years post-fire at Red Bluff Research Ranch, Montana

	Treatment		
Rank	Unburned	Burned	Bulldozed
1	*Festuca idahoensis*	*Stipa comata*	*Bromus tectorum*
	31.60%	19.70%	30.10%
2	*Stipa comata*	*Festuca idahoensis*	*Festuca idahoensis*
	16.00%	18.90%	23.90%
3	*Bromus tectorum*	*Bromus tectorum*	*Pseudoroegneria spicata*
	10.50%	14.50%	10.70%
4	*Pseudoroegneria spicata*	*Pseudoroegneria spicata*	*Stipa Comata*
	7.10%	9.40%	5.50%
5	*Selaginella densa*	*Taraxacum officinale*	*Agropyron smithii*
	4.60%	3.90%	3.30%

3.4 Alpha diversity, evenness, and beta diversity

Alpha diversity, a function of species richness and evenness, did not differ among the treatments for either inverse Simpson's diversity index ($p = 0.41$) or Shannon alpha diversity index ($p = 0.30$). Nor was there a treatment difference for Simpson's ($p = 0.15$), nor Shannon's evenness ($p = 0.08$).

Beta diversity, which measures the differences in diversity among plots and treatments, showed that the three treatments are significantly dissimilar from one another using the Morisita-Horn Similarity Index ($p = 0.02$). The unburned area had the greatest similarity among its plots

(47.58%), while the burned area had the least similarity among its plots (31.37%), and the bulldozed plots had intermediate similarity among its plots (36.22%). Among treatments, the unburned was nearly identically similar to both the burned and bulldozed treatments (36.8% and 37.9%, respectively), while the burned and bulldozed treatments were the least similar (30.0%). The bulldozed area was 7.9% more similar to the unburned area than it was to the burned area. (Table 2) (Figure 7).

Table 2. Percent similarity between treatments and among plots within treatments (along downward diagonal) using the Morisita-Horn Index.

	Unburned	**Burned**	**Bulldozed**
Unburned	47.6	36.8	37.9
Burned		31.4	30.0
Bulldozed			36.2

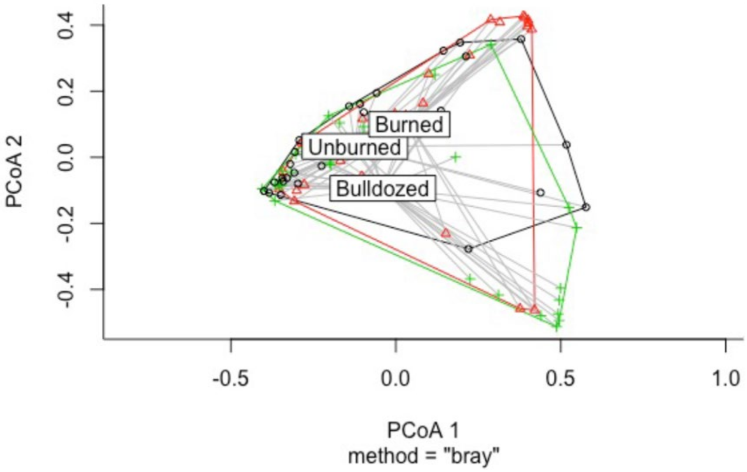

Figure 7. Bray Curtis Principal Coordinates Analysis of three treatment areas using Bray Curtis Similarity Index nine years post-disturbance at Red Bluff Research Ranch, Montana. (Unburned: ○; Burned: Δ; Bulldozed: +)

4. DISCUSSION:

Disturbance and disturbance type had significant effects on the sagebrush steppe plant communities nearly a decade after disturbance. The sagebrush steppe showed moderate resilience to fire; a disturbance that has historically occurred in this community. Plant species richness was found to be higher in the burned treatment than in the unburned treatment, though only by 1.6 species/m^2, and no different from the bulldozed area. The burned area also had the highest vegetative cover of the three treatment areas. Of the vegetative cover in the burned area, the two most dominant species were both native grasses, a key measure of resilience. This is not unexpected as other studies have also found that native grasses are resilient to fire in comparable ecosystems (Antos, et al., 1983; Link et al., 1990). The bulldozed area showed less resilience, having similar, but much more variable richness, and greater mean vegetative cover than the unburned area, much of which may be explained by increased *B. tectorum* cover. Seipel et al., (2018) found no difference in richness or vegetative cover at Red Bluff between the burned and unburned treatments 3 years after fire (Seipel et al., 2018). This may indicate that fire is a factor in stimulating the growth of a rich plant community in northern sagebrush steppe. The bulldozed area's richness, on the other hand, became more similar to the unburned area after 9 years when compared to 3 years after the fire, indicating that northern sagebrush steppe may have a weak delayed resilience to soil disturbance.

Neither bare ground, nor total percentage of the ground cover of the plant community showed any differences among treatments. Differences in bare ground cover were found at Red Bluff in the first 2 years after disturbance among the three treatments, and no difference 3 years after disturbance (Seipel et al., 2018). The lack of difference in bare ground found in this study may point towards the first two years being the period in which the bulk of plant establishment occurs. Litter cover, however, did vary in this study. There was significantly more litter in the unburned treatment than in the burned and bulldozed areas 9 years after disturbance. Although litter cover may act as a suppressant to nonnatives such as *B. tectorum*, it may also act as an increased fuel load, potentially leading to a higher severity burn in the future as has been demonstrated in other studies (Weiner et al., 2016; Brown et al., 1985). These results may indicate that *B. tectorum* increases in disturbed areas within the Montanan sagebrush biome do not always lead to litter buildup and subsequently do not increase fire frequency and severity via this mechanism. Litter cover results are unable to be compared to Seipel et al.'s (2018) litter cover as it was unreported.

Plant community resistance to *B. tectorum* invasion was also affected by disturbance type. The burned area showed fair resistance to *B. tectorum,* having only slightly more cover than did the unburned area. Here, *B. tectorum* levels became more similar between the burned and unburned areas over time when compared to Seipel et al.'s (2018) results from 2015. Other studies have also reported increased *B. tectorum* cover shortly after fire (0-4 years) when compared to an adjacent unburned area, and a decrease to unburned levels 11-15 years after fire (Taylor et al., 2014). High native grass cover and climate are likely primary drivers of resistance at this site (Antos et al., 1983; Taylor et al., 2014; Chambers et al., 2007; Condon et al., 2011; Reisner et al., 2013).

The bulldozed area did not show strong resistance to *B. tectorum* invasion; *B. tectorum* relative abundance in the bulldozed area was nearly triple that of the unburned area, prohibiting the full establishment of native grasses. This follows the findings of Seipel et al., (2018), in which the firebreak at Red Bluff had a much greater *B. tectorum* cover and much lower native grass cover than in the unburned area (Seipel et al., 2018). Soil disturbance leading to increased annual grass cover (including *B. tectorum)* is a trend that has been found across the sagebrush biome (Lavin et al., 2013; Kotanen et al., 1997; Bradford and Lauenroth, 2006). The creation of novel, or semi-novel plant communities may have unexpected and negative implications on the biota, ecosystem processes, and ecosystem function of the immediate and surrounding ecosystems. However, while the bulldozed treatment did have a significantly different plant species composition, it is unclear whether this will persist as a novel community or if it will continue its trend of becoming more similar to the unburned treatment over time.

Beyond plant species richness and cover, beta diversity was found to differ. The species compositions of the burned and bulldozed treatments showed the least similarity when compared to other treatment pairings, while the bulldozed and unburned treatments had the greatest similarity. This lack of similarity between the bulldozed and burned plots demonstrates an important way in which a plant community can vary in its response to different disturbance types. The burned plots also had the least within-treatment similarity when compared to the other treatments, and the unburned the highest within-treatment similarity. This shows that in a sagebrush steppe plant community a lack of successionally resetting disturbance can create a stable, homogenous community, while a co-evolved disturbance such as fire can create a heterogeneous environment on small and large scales. Seipel et al., (2018) found that the compositions of the burned and unburned areas at the site had a higher level of similarity than did the burned and bulldozed, or the unburned and bulldozed. This change between among-treatment similarities from 3 years to 9 years after disturbance demonstrates that the effects of the fire continue and change for at least 9 years and may even be apparent beyond then.

Due to *B. tectorum*'s connection to increased fire occurrence and severity in many sagebrush communities, land managers have been warned to be very cautious of this species, especially in grazed areas (Chambers et al., 2014). However, the results of this study may show that not only is *B. tectorum* less of a concern after fire, but also that the fine fuel buildup that results from increased *B. tectorum* cover, is less of a concern in the northern reaches of the sagebrush steppe. The results of this study, along with those of Seipel et al., (2018) indicate that if *B. tectorum* is to be managed, the first 2 years may be critical as that is when is at its most abundant in both burned and unburned areas (Bradley et al., 2017; Seipel et al., 2018). Land managers should, however, be extremely cautious about implementing fire breaks or other practices that disrupt soil structure due to their positive effect on *B. tectorum,* and negative effect on native grasses. In Argentina, unpaved roads (somewhat analogous to a firebreak) act as spreading centers for *B. tectorum,* implying that the scope of soil disturbance extends beyond where the disturbance occurred (Speziale et al., 2018). These actions may have detrimental effects through the potential increase in fire risk, or perhaps more importantly, through the loss of native flora and ecosystem function (Linders et al., 2019). This is likely true for any severe disturbance that a plant community has not co-evolved with. With global change, novel disturbances may affect plant communities such that they are unable to resist loss of biodiversity or invasions of invasive species. These events have the potential to create novel communities that are unable to return to historic states, opening the door to negative effects on human health and industry, as well as flora and fauna health.

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