Fuel Bed Response to Vegetation Treatments in Juniper and Cheatgrass Invaded Sagebrush Steppe; and Comparisons of Fuel Load Data Collected at Two Spatial Scales

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AUTHORIZATION TO SUBMIT THESIS

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Abstract

This study was conducted in conjunction with the Sagebrush Steppe Treatment Evaluation Project (SageSTEP) and is designed to determine the impact of vegetation treatments on several fuel variables two years post treatment in the sagebrush steppe ecosystem. Nineteen locations that characterize common juniper woodland and sagebrush steppe sites in the Intermountain West (Fig 1) were chosen and divided into juniper woodland sites and cheatgrass sites. Juniper woodland sites were treated with mechanical and prescribed fires while cheatgrass sites had mechanical, prescribed fire, and chemical treatments. ANOVA was used to analyze the fuel variables to determine if there was any effect on fuels two years post treatment. Prescribed fire increased herbaceous biomass and effectively reduced shrub biomass and down woody debris, but was not as effective in woodlands with higher juniper densities and drier open sagebrush steppe. Tebuthiuron had a minimal effect on the sagebrush steppe, but may not have fully impacted the sagebrush at the time of the post-treatment data collection. Mechanical treatments were effective in preserving the shrub biomass in the woodlands and reducing shrub biomass in cheatgrass sites, and increased down woody debris in both sites.

Introduction

The sagebrush steppe is the dominant vegetation type of the Intermountain West. Its historical extent covered 63 million ha throughout 11 states making it the most expansive vegetation type in the western United States (West 1983a, 1983b, Knick et al. 2003). Over the last 150 years this ecosystem has been greatly impacted by land use, invasive species, environmental changes, and alterations in the fire regime (West 1999, Miller and Rose 1999, Tausch 1999a). The cumulative impact of all these factors have fragmented and degraded this ecosystem. Today the sagebrush steppe is considered one of the most endangered ecosystems in the United States (Noss et al. 1995, Knick et al. 2003).

Two leading factors in the sagebrush steppe's decline are invasive species such as cheatgrass (*Bromus tectorum*) and the expansion of juniper woodlands (*Juniperus* spp.) (Pellant 1996, Miller and Tausch 2001, Miller et al. 2005, Welch 2005). Both of these factors may impact the continuity and availability of fuel in the ecosystem. Fuel is defined as the live and dead biomass that can contribute to the spread, intensity, and severity of a fire (Anderson 1982, Rothermel 1983, Burgan and Rothermel 1984). A change in fuel abundance may impact the ecosystem for years. Fire rate of spread, potential for crown fire, fire residence time, and fire severity are potentially affected by changes in vegetation (Brown et al. 1994, Miller and Urban 1999, Schoennagel et al. 2004). In the sagebrush steppe, most fires are considered moderate to high severity fires due to sagebrush susceptibility to fire. The high severity can be represented by the removal of sagebrush and fire residence time which could heat the soil, destroy the seed bank, and kill the resprouting perennial plants. Down woody debris (DWD) combustion requires more time to occur than combustion of herbaceous fuels and will increase the fire's residence time, thus increases in 100-hr DWD and 1000-hr DWD can indicate a potential increase in fire severity.

Cheatgrass impacts the sagebrush steppe at lower elevations, increasing fuel continuity and fire frequency which reduces native vegetation and can result in dominance by annual grasses (Klemmedson and Smith 1964, Pellant 1996). Juniper woodlands impact sagebrush at higher elevations, increasing in density until the sagebrush steppe vegetation is significantly reduced (Blackburn and Tueller 1970, Miller and Tausch 2001). The reduced understory in juniper woodlands decreases the fire frequency in the ecosystem, however there is an increase in larger fuels that may facilitate high severity crown fires (Miller et al. 2005, Keane et al. 2008). If cheatgrass is present in the understory of the juniper woodland, a crown fire may result in an annual grassland dominated ecosystem (Tausch 1999b). In addition to the changes in fire frequency and severity, impacts of vegetation conversion include significantly reducing available forage for wild and domestic animals (DiTomaso 2000, Kitchen and McArthur 2007, Tausch and Hood 2007), increasing soil erosion (Petersen and Stringham 2008, Pierson et al. 2011), redistributing soil nutrients (Klemmedson and Tiedemann 2000, Evans et al. 2001), and negatively affecting wildlife habitat (Miller et al. 2005, Welch 2005). These transitions may be particularly detrimental for sagebrush steppe obligates such as the sage grouse (*Centrocercus* spp.) and pygmy rabbit (*Brachylagus idahoensis*) (Wisdom et al. 2000).

Land managers recognize the negative impacts of cheatgrass and juniper woodland expansion on sagebrush steppe ecosystems. Many strategies exist to restore and rehabilitate the sagebrush steppe and to prevent a vegetation transition. Popular strategies include prescribed fire, mechanical treatments such as mowing or chainsaw use, and chemical applications, with the purpose of removing juniper, reducing sagebrush densities, and stimulating herbaceous vegetation growth (Olson and Whitson 2002, Wroblesky and Kauffman 2003, Stevens and Monson 2004, Ellsworth and Kauffman 2010). These treatments alter the vegetation structure and change abundance of fuels which has a direct effect on the fire behavior of the ecosystem.

Shrub biomass, DWD, and herbaceous biomass are key fuel bed strata. Total shrub biomass is divided into two categories; 1-hr (twigs 0-0.63 cm in diameter), and 10-hr (branches 0.63-2.54 cm; Frandsen 1983). DWD is categorized into size classes defined by the time it takes for fuel moisture to become equivalent to ambient relative humidity; 10-hr DWD are small branches (0.6-2.5 cm diameter), 100-hr DWD are medium branches (2.5-7.6 cm diameter), and 1000-hr DWD are large branches and tree trunks (>7.6 cm diameter; Fosberg 1970). The purpose of this study is to explore the effect of mechanical, chemical, and prescribed fire treatments on the fuel bed two years after treatments have been implemented and the impact to fire severity. The questions that this study will explore are: (1) What is the direct effect of the vegetation treatment on the fuel load of each fuel variable? (2) Does vegetation treatment reduce the fuel load in the ecosystem?

Sagebrush Steppe

The sagebrush steppe is a semi-arid ecosystem consisting of sagebrush (*Artemisia* spp.) as the dominant overstory, an herbaceous understory dominated by bunchgrass, and sparsely vegetated interspaces between sagebrush plants that is dominated by bare ground or cryptogams. This ecosystem is home to more than 300 wildlife species, many of which are highly dependent on the sagebrush steppe (Mac et al. 1998, Wisdom et al. 2000). The sagebrush steppe is often referred to as a cold desert, receiving the majority of its moisture in the winter in the form of snow. There are two climatic regions in the sagebrush steppe, a mesic higher elevation northern extent and a dryer lower elevation southern extent. The northern extent covers 44.8 million ha and may be dominated by mountain big sagebrush (*A. tridentata* spp. *vaseyana*) and contains greater floristic diversity, higher density of sagebrush, and has a more rapid recovery following disturbance (West 1983b, West and Young 2000). The dryer southern extent covers 17.9 million ha and is often dominated by Wyoming big sagebrush (*A. tridentata* spp. *wyomingensis*) (West 1983a).

Fire has historically been the primary mechanism for disturbance in the sagebrush steppe (Welch 2005). The fires were stand replacing with mean fire return intervals that varied between the more productive northern region and the arid southern region. Estimations for fire return intervals have a wide range and have been based on indirect measurements such as fire scars from nearby trees. The most productive northern sites are estimated to have a mean fire interval of 35-80 years (Keane et al. 2008) while the dryer southern region has a fire return interval of 100–200+ years (Keane et al. 2008).

Historically, fires created mosaics of burned and unburned islands across the ecosystem. This was caused by the flaming front fragmenting over barren interspaces and due to the variability in sagebrush density, loading of fine fuels, fuel moisture, topography, and wind (Ralphs and Busby 1979, Sapsis and Kauffman 1991). Sagebrush is poorly adapted to fire and will experience stand replacement where flame contact has occurred, relying on regeneration from the seed bank or surviving unburned individuals to restore the sagebrush population (Sapsis and Kauffman 1991). Seed dispersal from unburned patches is wind driven and limited to a relatively short distance from the parent plant. The seed bank is not long lived, with the rate of recovery dependent on seed establishment within the first two to three years post disturbance (Johnson and Payne 1968, Chambers 2000, Ziegenhagen and Miller 2009). Recovery to pre-fire shrub cover is dependent on the dominant sagebrush sub-species in the ecosystem. Recovery may take up to 15-35 years in mountain big sagebrush communities, but may take over 100 years for Wyoming big sagebrush communities (Harniss and Murray 1973, Bunting et al.1987, Nelle et al. 2000, Cooper et al. 2007, Lesica et al. 2007, Ziegenhagen and Miller 2009). Perennial bunchgrasses and forbs, on the other hand, are quite resistant to the medium and high severity fires and will re-sprout quickly following a fire (Ellsworth and Kauffman 2010). Thus fires have been important in maintaining a high degree of diversity in the sagebrush steppe.

Sagebrush-Cheatgrass Dominance

Cheatgrass was introduced in the late 1800's from Eurasia in contaminated agricultural seed. By the 1930's it had spread across the sagebrush steppe to become a dominant species on disturbed rangeland (Klemmedson and Smith 1964, Mack and Pyke 1983, Mack 1981). Cheatgrass dominance is attributed to its tolerance of grazing, high fecundity, fall germination, superior soil water exploitation, and rapid autumn and spring growth (Hulbert 1955, Young et al. 1987, Melgoza et al. 1990, Pellant 1996). Cheatgrass reaches maturity and desiccates earlier than native vegetation resulting in an extension of the fire season (Mack and Pyke 1983). Currently it is estimated that 50-60% of the sagebrush steppe either has cheatgrass in the understory or has been replaced by a non-native annual grassland (Knick et al. 2003).

As cheatgrass increases in density it fills the otherwise sparsely vegetated interspaces between sagebrush, creating a more uniform fuel continuum that increases the potential for fire in the ecosystem (Pellant 1996). When a fire occurs, the flaming front is more continuous, significantly increasing the extent of the fire while reducing the mosaic of burned and unburned vegetation expected in the sagebrush steppe (Welch 2005, Keene et al. 2008). Sagebrush is largely removed from areas burned under any intensity, with recovery dependent on the existing seed bank. This recovery occurs in pulses based on sagebrush recruitment and maturation. The first pulse is the flush of shrubs germinating from the seedbank, and the second occurs after those initial shrubs reach sexual maturity (Ziegenhagen and Miller 2009). However, the release of competition allows cheatgrass to aggressively recruit in the newly disturbed site (Knapp 1996). The post-fire vegetation may then be dominated by cheatgrass which subsequently increases the abundance of fine flashy fuels on the ecosystem, increasing the probability of another fire. This self-perpetuating fire cycle can be less than ten years, much shorter than the natural fire regime and preventing any possible sagebrush recovery (Klemmedson and Smith 1964), resulting in a non-native annual grassland.

The transition to an annual grassland redistributes soil nutrients on the ecosystem. Without cheatgrass, nutrients such as nitrogen and organic carbon accumulate beneath sagebrush, creating a resource island that can persist many years after the sagebrush is removed (Halvorson et al. 1997). Nutrient cycling is limiting, with plant available nutrients being rapidly sequestered among plants and soil microbes (Norton et al. 2004). As cheatgrass increases and sagebrush is removed, total nitrogen increases in the A horizon and nitrogen immobilization increases (Bolton et al. 1990, Halvorson et al. 1997, Evans et al. 2001). Deep soil nitrogen is incorporated in cheatgrass and redeposited on the soil surface as litter (Sperry et al. 2006). However, this does not represent an increase in plant available nitrogen. Cheatgrass creates more litter than native vegetation and its litter has a higher C:N ratio than native vegetation (Belnap and Phillips 2001). Decomposition of cheatgrass litter takes longer than the decomposition of native vegetation and increases nitrogen limitations to microbial activity (Evans et al. 2001). Despite the accumulation of total nitrogen, long-term cheatgrass dominance represents an overall loss of nitrogen from the ecosystem. Physical destruction of the litter through UV radiation, high temperatures, wind, and the increased return interval of fire volatilizes nitrogen and removes it from the ecosystem (Halvorson et al. 1997, Evans et al. 2001, Sperry et al. 2006).

Cheatgrass dominance results in degraded soil organic matter (OM) cycles (Norton et al. 2004). The OM cycle beneath sagebrush is a highly refined complex involving a large assemblage of microbial species, while cheatgrass OM cycle is comparatively simplistic (Belnap and Phillips 2001, Norton et al. 2004). The breakdown of OM decreases soil surface roughness and increase penetration resistance, decreasing water infiltration rates (Boxell and Drohan 2009). This decrease in water infiltration rate has the potential to increase water runoff and soil erosion during episodic summer rain storms (Pierson et al. 2011). Potential runoff vulnerability is mitigated by the increased density of cheatgrass in an ecosystem; however, the accelerated return interval of fire may create a 3-10 year cycle where the ecosystem has an enhanced vulnerability to accelerated erosion and runoff (Boxell and Drohan 2009, Pierson et al. 2011).

Juniper Woodland Encroachment

Juniper woodlands characterize a vast area across the Intermountain West, covering 17 million ha (West 1984). These woodlands are represented by a dominant juniper species which vary from north to south depending on elevation, climatic, edaphic, and topographic features (Miller et al. 2005). With the exception of western juniper (*Juniperus occidentalis*), they are often associated with a pinyon pine (*Pinus* spp.) species. Historically juniper woodlands have occurred as sparsely populated savannas, on isolated rocky outcrops, or in relatively poor soil (Miller and Rose 1999, Miller and Tausch 2001). Over the past 130 years there has been an increase in tree density within its historic extent, and an expansion of juniper woodlands into adjacent vegetation types (Miller and Wigand 1994), resulting in a 9-fold increase in population (Miller et al. 2005). The cause for this expansion is confounded due to several factors, both natural and anthropomorphic, occurring roughly simultaneously. The factors include the climatic changes following the end of the Little Ice Age, reduced competition from grasses due to extensive grazing, reduced fire occurrence due to active fire suppression and the reduced fine fuel loading and continuity through livestock grazing, increased atmospheric CO_2 which enhances juniper growth, and habitat fragmentation due to agriculture and human expansion (Miller and Rose 1999, Floyd et al. 2004, Miller et al. 2005).

The encroachment process of juniper woodland into sagebrush steppe has been defined into three phases. Phase 1 has an open, actively expanding juniper canopy of ≤10% with an intact shrub layer, Phase 2 has an actively expanding juniper cover between 10-30% and a thinning shrub layer, and Phase 3 has a nearly stabilized juniper cover >30% with ≥75% shrub mortality (Miller et al. 2005). As encroachment progresses, there is a dramatic transition in vegetation structure and natural processes. Increased juniper cover decreases sagebrush steppe vegetation and species richness (Blackburn and Tueller 1970, Bunting et al. 1999, Robert and Jones 2000, Roland et al. 2011), creating large, sparsely vegetated interspaces. These interspaces have less structure than normal sagebrush steppe vegetation and promote an increase in water runoff and soil erosion (Peterson and Stringham 2008). Nitrogen and carbon cycles, normally associated with nutrient islands beneath sagebrush, become nutrient islands beneath junipers. These nutrients become tied up in the slowly decaying juniper litter, reducing the turnover rate of nutrients and restricting available nutrients to other plants (Klemmedson and Tiedemann 2000, Roberts and Jones 2000, Bates et al. 2002, 2007). The reduction of sagebrush steppe vegetation and increase in large fuel increases fire severity when fires occur (Blackburn and Tueller 1970, Floyd et al. 2004, Miller et al. 2005).

Land restoration efforts have tried to reverse these changes in vegetation structure through a wide range of treatments which have included mechanical juniper removal and prescribed fire. Treatments are often broad-scale with different degrees of effectiveness. Mechanical treatments are more effective in removing larger trees (Schwilk et al. 2009), especially in Phase 3 woodlands where finer ground fuels may be limiting. Prescribed fire has been shown to be more effective at reducing total surface fuel loads (Schwild et al. 2009). Both methods have shown promise in restoring the sagebrush steppe, however, restoration efforts may be confounded by the presence of invasive species such as cheatgrass. Treatments often increase the abundance of invasive species. Progression to Phase 3 is associated with a reduction in sagebrush steppe vegetation and its seed banks, which reduces the sagebrush vegetation response of treated areas and can facilitate invasive species expansion. In many areas the risk of invasive species expansion is greater than the potential gains from restoration and treatments are not recommended (Miller et al. 2005). Treatments can also alter the fuel load of a site, increasing down and dead woody material or reducing fine fuels depending on method chosen and if any post-treatment rehabilitation was performed (Agee and Skinner 2005).

This study focused on quantifying changes in fuel load two years after treatment implementation. It is hypothesized that the finer fuels, live herbaceous, will increase after all treatments types. Shrub biomass will decrease in prescribed fire plots, in plots that were chemically treated with tebuthiuron, and in plots treated with a mechanical mower, but will increase in the juniper network. DWD will increase in mechanically treated sagebrush and juniper stands.

Methods

This study was conducted in conjunction with the Sagebrush Steppe Treatment Evaluation Project (SageSTEP; McIver et al. 2010). SageSTEP was designed to monitor long-term changes to the ecosystem from different treatment methods in the sagebrush steppe communities of the Intermountain West. This study is conducted at 19 sites that characterize common juniper woodland and sagebrush steppe ecosystems located in the Intermountain West (Fig 1). The sites are separated into two networks, cheatgrass and woodland. The cheatgrass network encompasses sagebrush steppe that has been invaded by cheatgrass and is divided into two regions; a more productive region referred to as SageWest, and a more arid region referred to as SageEast. The woodland network includes sites that have been affected by the expansion of juniper woodland into sagebrush steppe and is divided into three regions based on the type of juniper present; western juniper, pinyon-juniper (*J. osteosperma and Pinus monophylla*), and Utah juniper (*J. osteosperma*).

Each of the 19 sites used a randomized design to create a permanent core plot per treatment. Core plot size varied at each site and ranged from 20–80 ha. Core plots had permanent subplots established within them. Exact number of subplots per core plot varied between 14 and 24 and was dependent on the core plot's size. Each subplot was established along a systematic grid with a minimum distance of 50-m between the subplot centers. The subplots were 30 x 33-m with seven transects running parallel to the 33-m length (Fig 2).

Treatments of the cheatgrass network include a control, prescribed fire, chemical application of tebuthiuron, and a mechanical treatment. The prescribed fires were conducted in the fall and were designed to be low severity and to blacken 100% of the core plot. Tebuthiuron was applied aerially from a fixed wing aircraft that distributed 1.69kg/ha at a uniform rate across each site. The application was designed to decrease sagebrush abundance by 50%. The mechanical treatment was a rotary mower set at a height of 30.48-cm with the goal of removing 50% of the sagebrush cover. Half the plots within each treatment in the cheatgrass network also had the chemical herbicide Plateau®, from the BASF chemical company, added in the late fall after application of the other treatments. Plateau was designed to reduce cheatgrass density and was applied with a hand sprayer at a rate of 0.42kg/ha. Cheatgrass sites were divided into two groups based on the average sagebrush canopy cover of the region. SageWest: Group 1 < 18.7%, Group 2 > 18.7%. SageEast: Group 1 was < 24.6%, Group 2 was > 24.6%. (Stebleton and Bunting 2009). Treatments in the woodland network included a control, prescribed fire designed to blacken 100% of the core plot, and mechanically using a chainsaw to clear cutting all juniper (and pinyon if present) taller than 0.5-m and leaving them where they fell. The core plots were classified based on encroachment phase previously described.

Common measurement protocols were used across all sites, refer to Table 1 for specific methods and transects used for each reported variable (Bourne and Bunting 2011). Statistical software SAS 9.2 from the SAS institute was used for all statistical analysis. Fuel variables analyzed included live herbaceous, standing dead herbaceous,

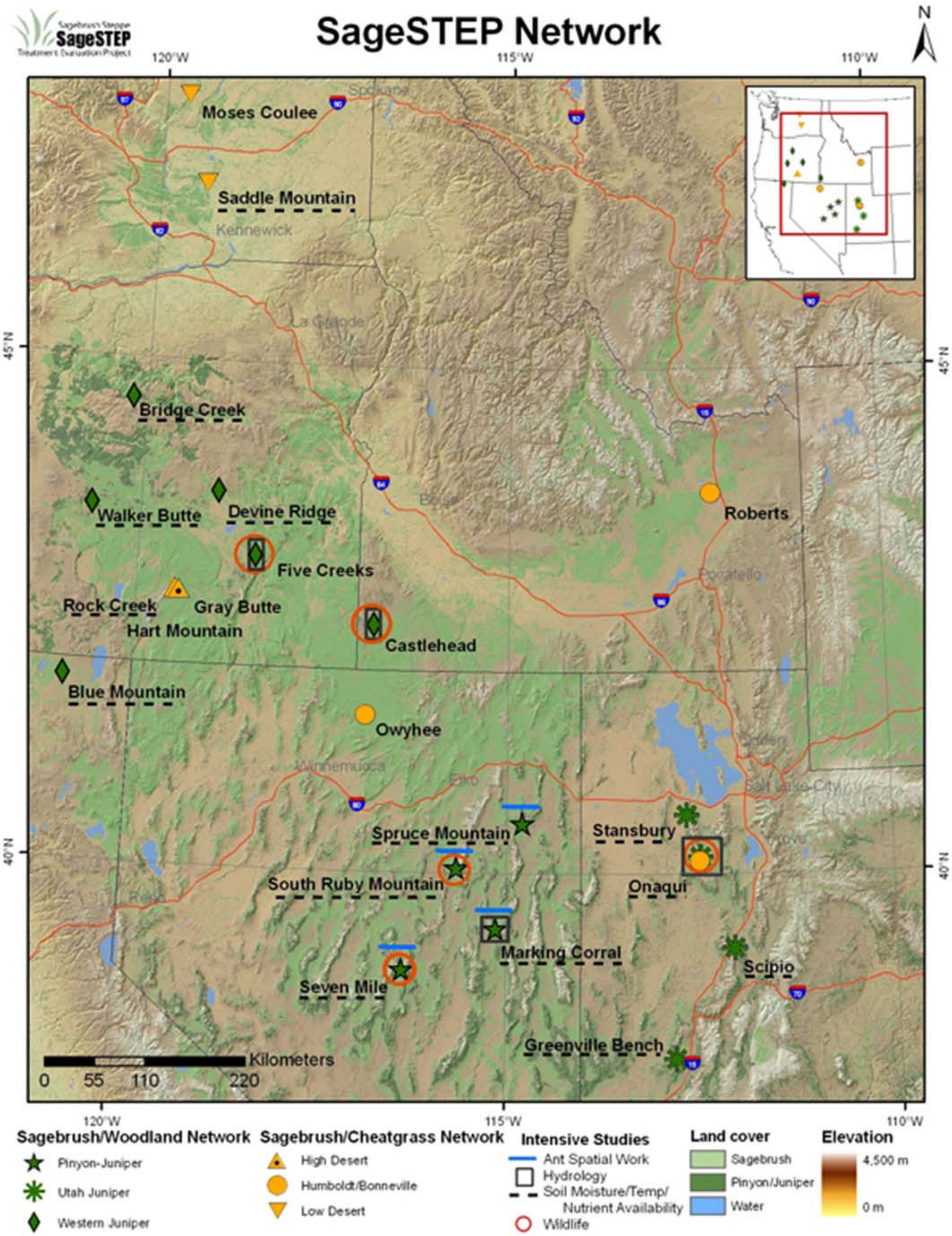

Figure 1: SageSTEP Network sagebrush steppe sites (McIver et al. 2010).

total shrub biomass, 10-hr DWD, 100-hr DWD, 1000-hr solid DWD, and 1000-hr rotten DWD. All shrub biomass data was specific to *A. tridentata*. In addition to shrub total biomass, two size classes were created to represent shrub 1-hr and shrub 10-hr size classes of biomass. The shrub data collection protocol changed after 2006. Sites with pre-treatement data collected during 2006 had their shrub data excluded from this analysis. These sites included Bridge Creek, Walker Butte, Onaqui, and Marking Corral (Fig 1). Each variable was derived from Ottmar's fuel stratum (Ottmar et al. 2007). Descriptive variables for all subplots included: region name, site name, subplot number, sampling year, UTM coordinates at zero corner, percent slope, aspect, macro-topography (ridgetop, sideslope, terrace, or bottom), micro-topography (flat, convex, or concave), and vegetation phase. Each site was grouped together by region within each network (Appendix A). Six sites were excluded from the study due to incomplete data: Five Creeks, Spruce Mountain, Castlehead, South Ruby, and the fire treatment from Moses Coulee.

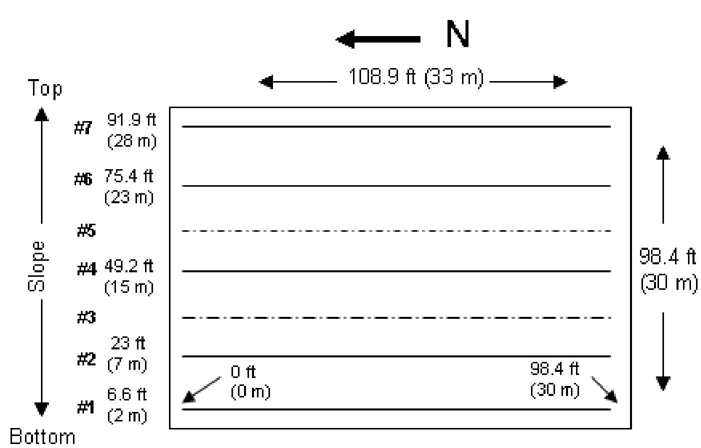

Figure 2. Subplot and transect layout. Solid lines signify vegetation transects; dotted lines denote fuels transects (Bourne and Bunting 2011)

Table 1: Sampling Methods used for each of the reported variables by fuel stratum (Ottmar et al. 2007). Transect number refers to the corresponding number in Figure 2. Table compiled by Bourne and Bunting (2011).

Stratum	Variable(s)	Method	Transect(s) #
Shrubs	Cover	Line point intercept (Bonham 1989)	1, 2, 4, 6, 7
	Height	Nested circular frame (Bonham 1989)	4
	Density	Belt transect (Krebs 1999, Saltzer 1994)	2, 3, 6
		Nested Circular frame (Bonham 1989)	4
	Loading and Bulk Density	Harvest (Pechanec and Pickford 1937)	NA
		Nested circular frame (Bonham 1989)	4
Nonwoody fuels	Cover	Line point intercept (Bonham 1989)	1, 2, 4, 6, 7
	Height	50 cm X 50 cm quadrat (Bonham 1989)	3 in 2006; 5 in 2007
	Loading and Bulk Density	Harvest (Pechanec and Pickford 1937; Riser 1984)	3 in 2006; 5 in 2007
		50 cm X 50 cm quadrat (Bonham 1989)	3 in 2006; 5 in 2007
Woody Fuels	10-hour loading	Planar intercept (Brown et al. 1982)	2, 4, 6
	100-hour loading	Planar intercept (Brown et al. 1982)	2, 4, 6
	1000-hour Sound and Rotten loading	Planar intercept (Brown et al. 1982)	1, 2, 4, 6, 7

Each variable was tested for normal distribution with a univariate student's t test to determine whether parametric or non-parametric analysis would be more appropriate. A randomized block factorial ANOVA was run on normally distributed variables within each group in each region of the cheatgrass network and within each phase in each region of the woodland network. Non-normal variables were analyzed with a Wilcoxen two-sample non-parametric one-way ANOVA. Variables subject to multiple tests within a group or phase per region had a Bonferoni correction applied, the correction value depending on the number of tests conducted on that variable, to reduce the probability of a type 1 error.

Results and Discussion

The univariate student's t test determined that several variables had a non-normal distribution throughout both the cheatgrass and woodland networks (Appendix B-H). These fuel variables were largely represented by the 1000-hr Solid and Rotten DWD fuel. The sagebrush steppe does not have an abundance of 1000-hr DWD, though the woodland network may see an increase as the site progresses to a Phase 3 woodland. The transect length of 150-m, despite being the longest for any fuel variable measured, may not be long enough to record the amount of 1000-hr DWD fuel needed for a normal distribution.

Cheatgrass Network: Prescribed Fire

SageWest Group 1 prescribed fire live herbaceous biomass increased from 158 – 830 kg/ha ($p<0.0001$; Table 2), total shrub biomass decreased from 3920 – 36 kg/ha ($p<0.0001$), shrub 1-hr decreased from 1403 – 19 kg/ha ($p<0.0001$), shrub 10-hr decreased from 1573 – 17 kg/ha ($p<0.0001$), and resulted in a decrease in DWD 10-hr (1255 - 40 kg/ha; $p<0.0001$), 100-hr (1973 – 290 kg/ha; $p<0.0001$), and 1000-hr solid (417 – 0 kg/ha; $p<0.0001$). SageWest Group 2 prescribed fire live herbaceous biomass increased from 115 – 598 kg/ha ($p=0.0004$). Fuel loads decreased for total shrub biomass (4213 – 63 kg/ha; $p<0.0001$), shrub 1-hr (1619 – 34 kg/ha; $p<0.0001$), shrub 10-hr (1713 – 25 kg/ha; $p<0.0001$), 10-hr DWD (945 – 474 kg/ha; $p<0.0001$), 100-hr DWD (1417 – 302 kg/ha; $p<0.0001$), and 1000-hr solid DWD (4019 – 0 kg/ha; $p<0.0001$). SageWest

Table 2: SageWest mean fuel variables pre and post-treatment (kg/ha) and two-year post-treatment percent change. Group 1: Shrub Cover < 18.7%. Group 2: Sagebrush Cover > 18.7%. **=Significant change at p<0.05.

SageWest (kg/ha)													
		Control			Prescribed Fire			Tebuthiuron			Mechanical		
		Pre	Post	Change (%)	Pre	Post	Change (%)	Pre	Post	Change (%)	Pre	Post	Change (%)
Live Herbaceous Biomass	Group 1	234	330	41%	158	830	426%**	260	372	43%	219	522	139%**
	Group 2	207	319	54%	115	598	420%**	102	152	49%	133	468	251%**
Live Herbaceous Biomass (Plateau)	Group 1	214	262	22%	192	220	15%	254	255	0%	282	367	30%
	Group 2	215	262	22%	111	169	52%	118	108	-8%	148	347	135%**
Total shrub biomass	Group 1	3134	3947	26%	3920	36	-99%**	3366	3199	-5%	3474	854	-75%**
	Group 2	3363	4175	24%	4213	63	-99%**	5409	3474	-36%**	2850	538	-81%**
Shrub 1-hr	Group 1	1227	1481	21%	1403	19	-99%**	1200	1125	-6%	1184	486	-59%**
	Group 2	1264	1592	26%	1619	34	-98%**	2044	1355	-34%**	1091	289	-73%**
Shrub 10-hr	Group 1	1233	1545	25%	1573	17	-99%**	1397	1248	-11%	1341	560	-58%**
	Group 2	1289	1583	23%	1713	25	-99%**	2085	1329	-36%**	1167	327	-72%**
10-hr DWD	Group 1	913	832	-9%	1255	406	-68%**	799	935	17%	828	1177	42%**
	Group 2	699	756	8%	945	475	-50%**	505	859	70%**	477	1575	230%**
100-hr DWD	Group 1	1259	1001	-21%	1973	290	-85%**	1387	1230	-11%	1154	1644	42%**
	Group 2	1170	1387	19%	1417	302	-79%**	637	894	40%	836	1917	129%**
1000-hr Solid DWD	Group 1	565	166	-71%	417	0	-100%**	338	183	-46%	743	222	-70%**
	Group 2	1177	243	-79%**	4019	0	-100%**	349	157	-55%	478	355	-26%

Group 1 and Group 2 had a similar response to the prescribed fire. Herbaceous biomass increased, shrub biomass decreased, and DWD decreased. Over the region there was a 5-fold increase in herbaceous biomass. The removal of competing shrubs increased available nutrients, soil moisture, and exposure to sunlight, which, when combined with the flush of nutrients from combusted material, can increase herbaceous production (Blaisdell 1953, Young and Evans 1978, Ralphs and Busby 1979, Wambolt et al. 1999, Wrobleski and Kauffman 2003, Ellsworth and Kauffman 2010). Plots treated with the chemical compound Plateau had no herbaceous response to the fire, suggesting that the increase in herbaceous biomass in untreated plots was in the form of cheatgrass.

There was a greater than 98% decrease in total shrub biomass, shrub 1-hr, and shrub 10-hr in response to the prescribed fire in both SageWest groups. A decrease in shrub biomass is expected as sagebrush is particularly vulnerable to fire and is readily consumed (Sapsis and Kauffman 1991). A return of mountain big sagebrush to pre-treatment biomass may take several decades (Harniss and Murray 1973, Bunting et al.1987, Ziegenhagen and Miller 2009). Though young mountain big sagebrush shrubs have been known to produce a sizable seed crop two to three years after germination (Daubenmire 1975, Young et al. 1989), studies show that sagebrush densities and cover may take 15-35 years to return to pre-burn cover (Harniss and Murray 1973, Bunting et al.1987, Pieper and Wittie 1990, Nelle et al. 2000, Cooper et al. 2007, Lesica et al. 2007, Ziegenhagen and Miller 2009). This recovery may take much longer where Wyoming big sagebrush is the dominant sagebrush.

SageWest Group 1 fires consumed 68% of the 10-hr DWD fuel and 85% of the 100-hr DWD fuel. SageWest Group 2 fires consumed 50% of the 10-hr fuel and 79% of the 100-hr DWD fuel. Fires consumed 100% of the available 1000-hr solid DWD in both groups, which will decrease the potential fire severity of the ecosystem. The decrease in DWD was due to a flaming front that must have been relatively uniform in both groups, which provided a fairly continuous burn. DWD may increase in the future depending on shrub combustion. If shrub skeletons persist, then eventually they will decompose and break apart, adding to the surface fuels (Harmon et al. 1986, Passovoy and Fule 2006).

SageEast Group 1 prescribed fires increased herbaceous biomass (82 – 407 kg/ha; p=0.0122), had no effect on sagebrush biomass, and decreased 10-hr DWD (701 – 396 kg/ha; p=0.0208; Table 3). SageEast Group 1 had a 4.8-fold increase in herbaceous biomass which may be associated with the sudden availability of resources provided by the input of nutrients from combustion (Bunting 1984, Wambolt and Payne 1986, Wrobleski and Kauffman 2003). However, there was an increase in herbaceous biomass in the control (p=0.0005), suggesting that the increase in herbaceous biomass recorded in the prescribed fire treatment may be due to climatic influences as opposed to treatment influences. Shrub biomass was unaffected by the prescribed fire treatment. The surviving shrub component suggests an incomplete or unsuccessful application of the prescribed fire despite the reduction in 10-hr DWD. Heterogeneous fuels and a wide variety of fuel moisture and weather across the network at the time of the fire may have proved problematic in effectively implementing the treatments.

Table 3: SageEast mean fuel variables pre and post-treatment (kg/ha) and two-year post-treatment percent change. Group 1: Shrub Cover < 24.6%. Group 2: Shrub Cover < 24.6%. **=Significant change at p<0.05.

SageEast (kg/ha)													
		Control			Prescribed Fire			Tebuthiuron			Mechanical		
		Pre	Post	Change (%)	Pre	Post	Change (%)	Pre	Post	Change (%)	Pre	Post	Change (%)
Live Herbaceous Biomass	Group 1	51	161	217%**	82	407	396%**	94	201	113%**	75	404	442%**
	Group 2	105	66	-37%	85	77	-10%	89	61	-32%	79	92	17%
Live Herbaceous Biomass (Plateau)	Group 1	51	89	73%	74	97	31%	73	104	43%	78	155	98%
	Group 2	79	55	-30%**	75	48	-36%	103	94	-8%	82	66	-20%**
Total shrub biomass	Group 1	3635	4246	17%	4063	4434	9%	5043	4236	-16%	4047	1656	-59%**
	Group 2	4853	7132	47%**	4749	2854	-40%**	6961	7148	3%	4820	1665	-65%**
Shrub 1-hr	Group 1	1406	2722	94%**	2152	2632	22%	2058	2098	2%	1881	970	-48%**
	Group 2	2287	3649	60%**	2803	1565	-44%**	2609	2851	9%	1877	958	-49%**
Shrub 10-hr	Group 1	1621	1827	13%	1744	1832	5%	2274	1898	-17%	1774	712	-60%**
	Group 2	2449	3546	45%**	2530	1422	-44%**	3470	3568	3%	2360	762	-68%**
10-hr DWD	Group 1	430	489	14%	504	353	-30%**	591	689	17%**	561	687	23%
	Group 2	670	798	19%**	701	396	-43%**	796	732	-8%	634	942	49%**
100-hr DWD	Group 1	821	753	-8%	801	518	-35%	1114	997	-10%	928	1194	29%
	Group 2	1277	1376	8%	1310	737	-44%**	1880	1313	-30%**	1341	1868	39%**
1000-hr Solid DWD	Group 1	69	235	244%	66	125	91%	93	242	159%**	107	193	80%
	Group 2	106	393	272%**	149	296	98%**	146	423	189%**	102	248	144%

Prescribed fires on the SageEast Group 2 decreased total shrub biomass (4749 – 2854 kg; p=0.0494) and shrub 1-hr (2803 – 1565 kg/ha; p=0.0072), shrub 10-hr (2530 – 1422 kg/ha; p=0.0284), 10-hr DWD (701 – 396 kg/ha; p<0.0001), and 100-hr DWD (1310 – 737; p<0.0001). The reduction in fuel represented a 40% decrease in total shrub biomass and a 43% decrease in both 10-hr and 100-hr DWD. As the flaming front propagates across the plot, sagebrush will be eliminated (Sapsis and Kauffman 1991) and DWD will be consumed. The decrease indicates that the prescribed fire was more consistent and uniform in sites with higher shrub cover than that of SageEast Group 1. However, 40% consumption is relatively low considering the prescription was designed to blacken 100% of a plot. A 60% surviving shrub fuel strata may suggest that though the higher canopy cover of SageEast group 2 facilitated a more effective prescribed fire, the effectiveness may still have been impeded by climatic variability and fuel continuity of these arid sites. Other studies that implemented a prescribed burn found that the shrub fuel strata was nearly completely eliminated, was associated with an initial increase in herbaceous biomass, and could take up over 100 years for Wyoming big sagebrush to return to pre-treatment cover values (Ralphs and Busby 1979, Wambolt and Payne 1986, Bunting et al.1987, Wrobleski and Kauffman 2003, Cooper et al. 2007).

Cheatgrass Network: Tebuthiuron

Tebuthiuron is an herbicide widely used in controlling vegetation. It is spread in the form of dry pellets and is water soluble. Once dissolved, plants will uptake the chemical through the roots where it causes phytotoxisity, killing the plant. It is a broad-spectrum chemical, however effectiveness is dose dependent (Steinert and Stritzke 1977, Scifres and Mutz 1978). Sagebrush is sensitive to the chemical, allowing for relatively low doses to effectively reduce shrub densities without adversely affecting other vegetation (McDaniel and Balliette 1986).

Tebuthiuron had no effect on SageWest Group 1 and SageEast Group 2 fuels (Table 2 and 3). SageWest Group 2 tebuthiuron total shrub biomass decreased (5409 – 3474 kg/ha; $p=0.0445$), as did shrub 1-hr (2044 – 1355 kg/ha; $p=0.0245$), and shrub 10-hr (2085 – 1329 kg/ha; $p=0.0188$). There was a corresponding 70% increase in 10-hr DWD (505 – 859 kg/ha; $p=0.0006$). SageEast Group 1 herbaceous biomass doubled (94 – 201 kg/ha; $p=0.0043$) and there was a 17% increase in 10hr DWD (591 – 689 kg/ha; $p=0.0460$). SageWest Group 2 was the only group that responded appreciably to the tebuthiuron treatment. The reduction in shrub biomass is initiated by the application of the herbicide, and the corresponding 70% increase in 10-hr DWD occurs as the dead shrubs decompose and add to the DWD fuel strata. There was an increase in herbaceous biomass in SageEast Group 1. However, there was a greater increase in the control indicating that the increase in the herbaceous biomass may actually represent a decrease in biomass. The minimal response to tebuthiuron was inconsistent with the literature and our predictions. Many studies found that applying tebuthiuron caused sagebrush mortality within the first two years. This mortality increased available nutrients, space, and water, causing a rapid increase in herbaceous vegetation (Whitson and Alley 1984, McDaniel and Balliette 1986, Murray 1988, McDaniel et al. 2005). No response suggests that the chemical was either ineffective, or has not yet become apparent to the crews collecting the data, probably the latter. Tebuthiuron comes in the form of dry pellets and will not effect vegetation until dissolved. Precipitation is not common in the sagebrush steppe and is variable across all sites in the cheatgrass network. Thus it may take longer than expected for the effects of tebuthiuron to influence the vegetation on all sites. Long-term studies demonstrate the effectiveness of tebuthiuron to control sagebrush biomass (Whitson 1988, Johnson et al. 1996, Olson and Whitson 2002, McDaniel et al. 2005). These studies show long-term treatments lasting 12-30 years depending on the concentration of tebuthiuron used, though eventually all sites returned to sagebrush dominance.

Cheatgrass Network: Mechanical Treatment

Mechanical treatments uniformly mowed each plot to a height of 30.48 cm with the objective to reduce shrub cover and facilitate herbaceous vegetation growth (McIver et al. 2010). SageWest Group 1 herbaceous biomass had a 2.3-fold increase 19 - 522 kg/ha; $p=0.0001$; Table 2), and SageWest Group 2 herbaceous biomass had a 3.5-fold increase (133-468 kg/ha; $p<0.0001$). Herbaceous biomass increases when mechanical treatments remove the shrub component from the ecosystem (Hedrick et al. 1966, Wambolt and Payne 1986, Dahlgren et al. 2006). This increase is associated with the release of nutrient and water competition from the shrubs. A mechanical mow of 30.48 cm is specifically designed to maintain shrub density while increasing the availability of light and reducing demand by the sagebrush for nutrients and water, allowing for an increase in herbaceous biomass (McIver et al. 2010). Payton et al. (2011) found a similar increase in herbaceous biomass two years after mechanically mowing a mixed Wyoming and mountain big sagebrush site to a height of 20-cm.

SageWest Group 1 mechanical treatment plots recorded a decrease in total shrub biomass (3474 – 854 kg/ha; $p<0.0001$), shrub 1-hr (1184 – 486 kg/ha; $p<0.0001$), and shrub 10-hr (1341 – 560 kg/ha; $p<0.0001$). SageWest Group 2 had similar results with a decrease in total shrub biomass (2850 – 538 kg/ha; $p<0.0001$), shrub 1-hr (1091 – 289 kg/ha; $p<0.0001$), and shrub 10-hr (1167 – 327 kg/ha; $p<0.0001$). There is a paucity of research on the recovery of sagebrush steppe ecosystems after a mechanical mow treatment at a height of 30.48 cm, particularly with regard to mountain big sagebrush. Mountain big sagebrush responds relatively quickly to disturbance (West 1983b), can produce a sizable seed crop 2-3 years after germination (Daubenmire 1975, Young et al. 1989), and studies on chaining sagebrush removal indicate a recovery time of 5-20 years (Tausch and Tueller 1977, Skousen et al. 1989). Davies et al (2009) studied Wyoming big sagebrush response to mechanical mowing at a height of 20 cm and found that Wyoming big sagebrush recovered from the treatment in 10-20 years. A more rapid response can be expected from mountain big sagebrush, since Wyoming big sagebrush is slower to recover from disturbance (West 1983a).

DWD increased by 42% for both 10-hr (828 – 1177 kg/ha; $p=0.0082$) and 100-hr DWD (1154 – 1644 kg/ha; $p=0.0351$) in SageWest Group 1. In SageWest Group 2, the increase was substantially greater, with a 230% increase in 10-hr DWD (477 -1575 kg/ha; $p<0.0001$) and a 129% increase in 100-hr DWD (836 – 1917 kg/ha; $p<0.0001$). The increase in surface fuel loading was caused by the mechanical treatment's conversion of 50% of the existing shrub cover to DWD. Greater increases were recorded in Group 2 because of the greater sagebrush cover represented at those sites. This increase in DWD may increase fuel continuity and can increase the potential for a higher severity fire in the short term.

SageEast Group 1 herbaceous biomass had a 5-fold increase (75-404 kg/ha; $p<0.0109$; Table 3). However, this result may not represent a mechanical response due to a 3-fold increase in herbaceous biomass in the control. The variability in herbaceous biomass response may be due to climatic factors across sites, or due to the multi-year temporal nature inherent in the data collection process. Studies that mechanically mowed Wyoming big sagebrush to a height of 20 cm found no response from perennial grasses in six years of testing, but recorded an increase in annual grasses (Davies et al. 2009, 2011, Hess and Beck 2012). However, mechanical treatments have been recorded to increase herbaceous biomass (Hedrick et al. 1966, Wambolt and Payne 1986, Dahlgren et al. 2006). Herbaceous response may vary with climatic conditions across the whole cheatgrass network.

Mechanical treatments in SageEast Group 1 resulted in a 59% decrease in total shrub biomass (4047 – 156 kg/ha; $p<0.0001$), a 48% decrease in shrub 1-hr (1881 – 970 kg/ha; $p=0.0008$), and a 60% decrease in shrub 10-hr (1774 – 712 kg/ha; <0.0001). The mechanical treatment in SageEast Group 2 resulted in a 65% decrease in total shrub biomass (4820 – 1665 kg/ha; $p<0.0001$), a 49% decrease in shrub 1-hr (1877 – 958 kg/ha; $p=0.0132$), a 68% decrease in shrub 10-hr (2360 – 762 kg/ha; $p<0.0001$), and had a corresponding 49% increase in 10-hr DWD (634 – 942 kg/ha; $p<0.0001$) and a 39% increase in 100-hr DWD (1341 – 1868 kg/ha; $p=0.0053$). The mechanical treatment effectively reduced shrub biomass in both SageEast Group 1 and 2 for all shrub size categories. The treatment converted shrub biomass into DWD, adding to the surface loading of the fuel bed. This increase was only recorded in SageEast Group 2, suggesting that mechanical treatments at sites with low sagebrush cover may not add appreciably to the DWD at a site. The increase in DWD at SageEast Group 2 sites may persist in the ecosystem for an extended length of time due to the slow decomposition rates at these low elevation arid sites. The DWD can potentially remain in the ecosystem, adding to fuel continuity and potential fire severity, until after the site reverts back to pre-treatment sagebrush cover levels. Studies on Wyoming big sagebrush mechanically mowed to a height of 20 cm indicate that shrub cover recovery to pre-treatment levels may take 10-20 years (Davies et al. 2009, 2011).

Woodland Network: Prescribed Fire

Prescribed fire in the western juniper region resulted in an increase in live herbaceous biomass in all phases ($p<0.0001$; Table 4). Phase 1 increased from 257 – 877 kg/ha, Phase 2 increased from 205 – 796 kg/ha, and Phase 3 increased from 130 – 541 kg/ha. The pinyon-juniper region had an increase in live herbaceous biomass in Phase 1 (122 – 578 kg/ha; $p=0.0037$) and Phase 2 (144 – 507 kg/ha; $p<0.0053$; Table 5). These pinyon-juniper results excluded Marking Corral due to lost data. The Utah juniper region had an increase in live herbaceous biomass across all three phases ($p<0.0200$; Table 6). Phase 1 increased from 204 - 759 kg/ha, Phase 2 increased from 141 – 377 kg/ha, and Phase 3 increased from 70 – 345 kg/ha. Over the woodland network, prescribed fire resulted in a 3-fold increase in herbaceous biomass in Phase 1 and 2, and a 4-fold increase in western juniper and Utah juniper Phase 3 sites. An increase in herbaceous

Table 4: Western juniper woodland mean fuel variables pre and post-treatment (kg/ha) and two-year post-treatment percent change. **=Significant change at $p<0.05$.

		Western Juniper (kg/ha)								
		Control			Prescribed Fire			Mechanical		
		Pre	Post	Change (%)	Pre	Post	Change (%)	Pre	Post	Change (%)
Live Herbaceous Biomass	Phase 1	289	352	22%	257	877	241%**	242	472	95%**
	Phase 2	197	225	14%	205	797	288%**	164	393	139%**
	Phase 3	125	114	-9%	130	541	315%**	82	318	288%**
Total shrub biomass	Phase 1	1630	1490	-9%	2546	132	-94%**	4495	5830	30%
	Phase 2	909	891	-2%	1970	147	-92%**	1498	1718	15%
	Phase 3	358	297	-17%	562	232	-59%	132	83	-37%
Shrub 1-hr	Phase 1	817	755	-8%	1223	59	-95%**	2335	2959	27%
	Phase 2	447	444	-1%	980	67	-93%**	778	929	19%
	Phase 3	157	132	-16%	263	130	-51%	73	50	-31%
Shrub 10-hr	Phase 1	667	607	-9%	1017	55	-94%**	1816	2294	26%
	Phase 2	381	366	-4%	801	62	-92%**	618	708	15%
	Phase 3	155	128	-17%	246	97	-60%	58	38	-35%
10-hr DWD	Phase 1	611	685	12%	784	323	-58%**	677	895	32%
	Phase 2	769	866	13%	703	446	-36%**	718	1218	69%**
	Phase 3	632	829	31%	527	581	10%	815	2070	153%**
100-hr DWD	Phase 1	1438	993	-31%	2088	893	-57%**	1243	1769	42%**
	Phase 2	1843	1450	-21%	1561	1225	-22%	1714	4013	134%**
	Phase 3	1278	965	-24%	1247	1680	35%	1259	6084	383%**
1000-hr Solid DWD	Phase 1	625	284	-55%	829	333	-59%**	583	2681	360%**
	Phase 2	1332	482	-64%	640	1964	206%	707	8842	1149%**
	Phase 3	611	897	47%	827	4491	443%	878	16098	1733%**

biomass is expected post fire. The removal of competition from shrubs and trees combined with the rapid release of nutrients into the ecosystem facilitates regeneration and growth (Everett and Ward 1984, Agee 1993, Wrobleski and Kauffman 2003, Rau et al. 2008). An increase in herbaceous biomass is expected to continue until available space and resources are expended (Tausch and Tueller 1977, Everett and Ward 1984, Bates et al. 2005).

The prescribed fire treatment in western juniper Phase 1 woodland resulted in a decrease in total shrub biomass (2546 – 132 kg/ha; p<0.0001), shrub 1-hr (1223 – 59 kg/ha; p<0.0001), and shrub 10-hr (1017 – 55 kg/ha; p<0.0001). In western juniper Phase 2 woodland there was a decrease in total shrub biomass (1970 – 147 kg/ha; p=0.0002), shrub 1-hr (980 – 67 kg/ha; 0.0002), and shrub 10-hr (801 – 62 kg/ha; 0.0002). Pinyon-juniper total shrub biomass (4158 - 475 kg/ha; p=0.0141), shrub 1-hr (1565 – 156 kg/ha; p0.0141), and shrub 10-hr (615 – 63 kg/ha; p=0.0139) all decreased in Phase 1. In Phase 2 pinyon-juniper there was a decrease in shrub 1-hr (808 – 64 kg/ha; p=0.0005) and shrub 10-hr (264 – 18 kg/ha; p=0.0009). Utah Juniper total shrub biomass decreased in Phase 1 (5693 – 120 kg/ha; p<0.0001), Phase 2 (3473 – 376 kg/ha; p<0.0001), and Phase 3 (819 – 112 kg/ha, p=0.0004); decreased in shrub 1-hr in Phase 1 (2005 – 38 kg/ha; p<0.0001), Phase 2 (1284 – 140 kg/ha; p<0.0001), and Phase 3 (264 – 73 kg/ha; 0.0081); and decreased in shrub 10-hr in Phase 1 (2144 – 45 kg/ha; <0.0001), Phase 2 (1336 – 144 kg/ha; p<0.0001), and Phase 3 (317 – 62 kg/ha; p=0.0007). Over the woodland network, prescribed fire resulted in a decrease of total shrub biomass by 92% in Phase 1 and 88% in Phase 2. The reduction of shrub biomass in Phase 3 was variable. Shrubs on western juniper and pinyon-juniper Phase 3 were unaffected by the

Table 5: Pinyon-juniper woodland mean fuel variables pre and post-treatment (kg/ha) and two-year post-treatment percent change. **=Significant change at $p<0.05$.

Pinyon-Juniper (kg/ha)										
		Control			Prescribed Fire			Mechanical		
		Pre	Post	Change (%)	Pre	Post	Change (%)	Pre	Post	Change (%)
Live Herbaceous Biomass	Phase 1	222	327	47%	122	578	373%**	171	298	75%**
	Phase 2	129	217	69%	144	507	252%**	79	214	172%**
	Phase 3	34	33	-4%	11	40	264%	23	122	421%**
Total shrub biomass	Phase 1	5279	5984	13%	4158	475	-88%**	5320	4372	-18%
	Phase 2	2739	2603	-5%	1726	168	-90%	4450	5420	22%
	Phase 3	702	732	4%	219	58	-73%	564	985	75%
Shrub 1-hr	Phase 1	1985	1817	-8%	1565	156	-90%**	2147	1421	-33%
	Phase 2	1139	736	-35%	808	64	-92%**	1430	1474	3%
	Phase 3	234	198	-15%	102	22	-78%	237	342	44%
Shrub 10-hr	Phase 1	866	892	3%	615	63	-89%**	847	592	-30%
	Phase 2	406	300	-26%	264	18	-93%**	643	738	15%
	Phase 3	83	79	-4%	28	5	-83%	76	113	48%
10-hr DWD	Phase 1	705	585	-17%	1391	790	-43%**	653	936	43%
	Phase 2	665	610	-8%	1081	671	-38%**	1013	1599	58%**
	Phase 3	559	644	15%	765	707	-8%	919	1778	94%**
100-hr DWD	Phase 1	1305	956	-27%	1999	1558	-22%	1639	2080	27%
	Phase 2	1026	713	-31%	1256	1448	15%	1770	3496	97%**
	Phase 3	746	607	-19%	576	1752	204%**	1450	3828	164%**
1000-hr Solid DWD	Phase 1	429	423	-1%	1042	350	-66%	617	1988	222%
	Phase 2	385	650	69%	620	459	-26%	1001	7331	632%**
	Phase 3	1957	772	-61%	403	2078	415%**	2145	10997	412%**

burn. In contrast, Utah juniper total shrub biomass was reduced by 91%. This variability across sites may be caused by the heterogeneous fuels and the differences in fire weather across each site at the time of the prescribed fire. The progression to a Phase 3 woodland reduces the herbaceous and shrub fuel variables on a site, which may result in a heterogeneous prescribed fire. Sagebrush biomass is expected to eventually return to pre-treatment levels, but may take up to 15-35 years in ecosystems dominated by mountain big sagebrush (Harniss and Murray 1973, Bunting et al.1987, Pieper and Wittie 1990, Nelle et al. 2000, Cooper et al. 2007, Lesica et al. 2007, Ziegenhagen and Miller 2009), and much longer where Wyoming sagebrush makes a greater percentage of shrub cover. The treatment was designed for 100% of the plots to be blackened, thus a surviving shrub component in Phase 3 woodlands indicates an inefficient prescribed fire. This is most likely due to the limited availability of fine fuels to carry the flaming front in a Phase 3 woodland and variability in wind and relative humidity between sites.

DWD in Western juniper prescribed fire sites decreased by 60% for all size categories in Phase 1, with 10-hr decreasing from 784 – 323 kg/ha ($p<0.0001$), 100-hr decreasing from 2087 – 893 kg/ha ($p=0.0001$), and 1000-hr solid decreasing from 829 – 333 kg/ha ($p=0.0163$). In Phase 2 western juniper, 10-hr DWD decreased by 37% (702 – 445 kg/ha; $p=0.0418$), but had a 3-fold increase in 1000-hr solid DWD (639 – 1964 kg/ha; $p=0.0437$). Pinyon-juniper Phase 1 and 2 10-hr DWD fuel decreased from 1391 – 790 kg/ha ($p=0.0211$) and 1081 – 671 kg/ha ($p=0.0071$), respectively. Pinyon-juniper Phase 3 had a 3-fold increase in 100-hr DWD (576 – 1752 kg/ha; $p=0.0003$) and a 5-fold increase in 1000-hr solid DWD (402 – 2078 kg/ha; $p=0.0221$). Utah Juniper Phase 1 DWD decreased for 10-hr (620 – 340 kg/ha, $p=0.0472$) and 1000-hr (1811 – 359 kg/ha;

p=0.0103) fuel. In Phase 3 there was an increase in 10-hr DWD (602 – 1065 kg/ha; p=0.0031) and an increase in 100-hr DWD (882 – 1812 kg/ha; p=0.0083).

Total fuel consumption roughly followed vegetation structure. Phase 1 and 2 had the highest fuel continuity, with a flaming front that is most likely to remain unbroken for each site, which is reflected in the greater than 80% shrub biomass consumption. Thus when DWD was consumed, it occurred in those phases with an intact herbaceous and shrub component. Phase 3 is known for a lack of fuel continuity, making it difficult to burn (Blackburn and Tueller 1970, Pieper and Wittie 1990, Miller and Taush 2001). Fire treatments are often not recommended for Phase 3 (Bates et al. 2000, Miller et al. 2005) and require more extreme fire conditions conducive to a crown fire (Bruner and Klebenow 1979, Erskine and Goodrich 1999, Huffman et al. 2009). Increases in DWD occurred largely in Phase 3 and was caused by partially consumed trees falling and adding to the surface fuels. This increase was not uniform across all regions in the woodland network, which may be a result of the heterogeneous nature of fuel and fire in a Phase 3 woodland. Eventually DWD will continue to increase in Phase 3 and Phase 2 as dead trees decompose, break apart, and fall, adding to the surface fuels (Harmon et al. 1986, Passovoy and Fule 2006, Clifford et al. 2008).

The potential for a high severity fire is likely decreased in Phase 1 and 2 after the prescribed fire treatment. Though there was a sizable increase in the herbaceous biomass, it has not yet compensated for the amount of sagebrush biomass consumed. This difference will decrease in the future as herbaceous biomass continues to increase into the open spaces and as shrubs recover from the treatment (Taush and Tueller 1977,

Table 6: Utah juniper woodland mean fuel variables pre and post-treatment (kg/ha) and two-year post-treatment percent change. **=Significant change at $p<0.05$.

Utah Juniper (kg/ha)										
		Control			Prescribed Fire			Mechanical		
		Pre	Post	Change (%)	Pre	Post	Change (%)	Pre	Post	Change (%)
Live Herbaceous Biomass	Phase 1	193	247	28%	204	759	272%**	121	236	95%**
	Phase 2	140	155	11%	141	377	166%**	126	357	182%**
	Phase 3	66	63	-4%	70	345	396%**	84	442	425%**
Total shrub biomass	Phase 1	6507	6148	-6%	5693	120	-97%**	3975	5073	28%
	Phase 2	3947	2936	-26%	3473	376	-89%**	2306	3357	46%
	Phase 3	1045	584	-44%	819	112	-86%**	859	778	-9%
Shrub 1-hr	Phase 1	2344	2307	-2%	2005	38	-98%**	1836	1853	1%
	Phase 2	1462	1325	-9%	1284	140	-89%**	1121	1402	25%
	Phase 3	441	429	-3%	264	73	-72%	279	308	10%
Shrub 10-hr	Phase 1	2442	2422	-1%	2144	45	-97%**	1936	2190	13%
	Phase 2	1589	1421	-11%	1336	144	-89%**	1207	1588	32%
	Phase 3	485	347	-28%	317	62	-80%**	299	296	-1%
10-hr DWD	Phase 1	722	645	-11%	621	341	-45%**	622	866	39%
	Phase 2	622	639	3%	639	609	-5%	654	1332	103%**
	Phase 3	450	511	14%	603	1066	76%**	474	1880	296%**
100-hr DWD	Phase 1	987	1306	32%	1341	629	-53%	1480	2077	40%
	Phase 2	1242	1304	5%	1175	1213	3%	1288	4922	282%**
	Phase 3	971	1268	31%	882	1813	105%**	964	4056	320%**
1000-hr Solid DWD	Phase 1	696	510	-27%	1811	359	-80%**	762	2519	230%
	Phase 2	1630	992	-39%**	703	653	-7%	2408	5313	120%**
	Phase 3	755	1370	82%	1421	842	-41%	1689	9205	444%**

Everett and Ward 1984, Bates et al. 2005). In Phase 3, western juniper had an herbaceous biomass increase greater than the amount of shrub biomass consumed. This, combined with an increase in 1000-hr DWD in some Phase 3 sites, suggests a potential increase in fire severity.

Woodland Network: Mechanical Treatment

Mechanical treatments used chainsaws to remove all trees taller than 0.5-m, removing the crown fuels. Western juniper mechanical treatments resulted in an increase in live herbaceous biomass in all three phases ($p<0.0001$; Table 4). Phase 1 increased from 241 - 471 kg/ha, Phase 2 increase from 164 - 393 kg/ha, and Phase 3 increased from 81 - 318 kg/ha. Pinyon-Juniper treatments resulted in an increase in live herbaceous in Phase 1 (171-298; $p=0.0012$), Phase 2 (79 – 214 kg/ha; $p=0.0011$) and Phase 3 (23 – 122 kg/ha; $p=0.0241$; Table 5). Utah juniper mechanical treatments resulted in an increase in live herbaceous biomass for all three phases ($p<0.0500$; Table 6). Phase 1 increased from 120– 235 kg/ha, Phase 2 increased from 126 – 356 kg/ha, and Phase 3 increased from 69 – 345 kg/ha. Over the woodland network, herbaceous biomass increased in Phase 1, increased 2 to 3-fold in Phase 2, and increased 3 to 5-fold in Phase 3. An increase in Phase 1 herbaceous biomass was found across the network, indicating that even at juniper cover of less than 10%, removal of juniper may release enough resources for an herbaceous vegetation response (Bates et al. 2005, Miller et al. 2005). Similar studies were primarily conducted in Phase 2 and Phase 3 juniper woodlands and found that mechanical treatments increase soil nitrogen and water availability, leading to an initial flush of herbaceous biomass in the first two years post juniper treatment (Tausch and Tueller 1977, Bates et al. 1998, 2000, Brockway et al 2002, Bates et al. 2005).

Total shrub biomass was not affected by the treatment. It was expected that the shrub biomass would increase as the sagebrush would have benefited from the increase in soil nitrogen and water availability. However, two years may not have been adequate time for the shrubs to respond. Previous studies have shown that chaining treatments caused a vigorous shrub response within the first two years post treatment in mountain big sagebrush ecosystems (Tausch and Tueller 1977, Skousen et al. 1989). However, Bates et al. (2005) found minimal shrub response 13 years after a mechanical treatment. He cited a lower initial shrub density within his plots as a possible cause of this slower response. This would not be accurate in our study as Phase 2 still had a relatively intact shrub component. There was a mild indication of positive shrub growth in all phases; continued long-term study is needed to determine if the shrub variables will respond to the treatment.

Mechanical treatments in western juniper Phase 1 resulted in an increase in 100-hr DWD (1242 - 1768 kg/ha; p=0.0481) and a 4-fold increase in 1000-hr solid DWD (582 – 2681 kg/ha; p=0.0003). In western juniper Phase 2 there was an increased in 10-hr, 100-hr, and 1000-hr DWD from 718 – 1217 kg/ha (p=0.0016), 1713 – 4013kg/ha (p<0.0001), and 707 – 8841 kg/ha (p<0.0001), respectively. Western Juniper Phase 3 DWD increased in all size classes; 10-hr increased from 815 – 2069 kg/ha (0.0126), 100-hr increased from 1259 – 6083 kg/ha (p=<0.0001), and 1000-hr solid increased from 877 – 16,097 kg/ha (p=0.0002). Pinyon-juniper DWD increased in Phase 2 from 1013 – 1599 kg/ha for 10-hr (p=0.0234), 1770 – 3495 kg/ha for 100-hr (p<0.0001), and 1000 – 7330 kg/ha for 1000-hr solid fuels (p<0.0001). There was also a similar increase in Phase 3 DWD for 10-hr, 100-hr, and 1000-hr fuels from 918 – 1778 kg/ha (p=0.0202), 1449 – 3828 kg/ha (p<0.0001), and 2144 – 10,997 kg/ha (p=0.0064), respectively. Utah juniper Phase 2 DWD increased for 10-hr, 100-hr, and 1000-hr solid from 654 – 1331 kg/ha (p=0.0002), 1288 – 4922 kg/ha (p<0.0001), and 2407 – 5312 kg/ha (p=0.0258), respectively. Phase 3 had the largest DWD increased with 10-hr increasing from 473 – 1879 kg/ha (p<0.0001), 100-hr increasing from 963 – 4055 kg/ha (p<0.0001), and 1000-hr solid from 1689 – 9205 kg/ha (p<0.0001). The mechanical treatment essentially a converted fuels from the live tree canopy strata to the DWD strata. Thus there is a logical progression of treatment influence from the minimal impact recorded on Phase 1 DWD to a more pronounced impact in Phase 2 and Phase 3. Across the woodland network, Phase 2 10-hr DWD had a 1.6 to 2-fold increase, 100-hr had a 2 to 3.8-fold increase, and 1000-hr solid had a 2 to 7-fold increase. Phase 3 had a 2 to 4-fold increase in 10-hr DWD fuels, a 2.6 to 4.8-fold increase in 100-hr DWD fuels, and a 5 to 18-fold increase in 1000-hr solid DWD.

While the potential of a canopy fire has been dramatically reduced by the mechanical treatment, there is a corresponding increase to DWD surface fuels which can increase the potential for a high severity surface fire. DWD moisture content is less than live trees and the fuel is now layered on the surface, which can increase soil heating in the event of a fire. Herbaceous biomass increased in all phases, increasing the continuity of fine fuels. Similar studies suggest that herbaceous biomass will peak within the first five to ten years and that there will eventually be a shrub response (Tausch and Tueller

1977, Skousen et al. 1989, Bates et al. 2005) combining with the increase in DWD and adding to the potential fire severity for years to come. Thousand-hour DWD increase to the fuel bed was substantial. For example, western juniper pre-treatment had 877 kg/ha, but post-treatment it had over 16,000 kg/ha. Thousand-hour DWD fuels can remain in the ecosystem for decades. Decay rates in the sagebrush steppe are variable and slow (Harmon et al. 1986) and may be influenced more through abiotic factors than biotic factors (Waichler et al. 2001). As the 1000-hr fuel decomposes and becomes rotten, they have an increased risk of smoldering and soil heating (Passovoy and Fule 2006, Clifford et al. 2008) which may increase fires severity.

Conclusions

Vegetation treatments are evident on the ecosystem two years after implementation. The change in vegetation structure alters the fuel bed characteristics, potentially manipulating the fire behavior of the ecosystem.

Prescribed fire's effect in the cheatgrass network was variable. SageWest Group 1 and 2 had similar results, with herbaceous biomass increasing and total shrub biomass decreasing (Table 2). SageEast Group 1 was not greatly affected by the fire (Table 3). SageEast Group 2 had a decrease in total shrub biomass, but did not have an herbaceous response two years post-treatment. Prescribed fire's effect in the woodland network was similar within each phase. In Phase 3 the treatment's effectiveness seemed questionable, but the general trend was a decrease in total shrub biomass and an increase in 100-hr and 1000-hr DWD. In Phase 1 and Phase 2 the total shrub biomass decreased, herbaceous biomass increased, and DWD decreased for all size classes. The most consistent prescribed fire effects occurred in sites with encroaching juniper or sites in the most mesic ecosystems of the sagebrush steppe. These included Phase 1 and 2 of the woodland network and both SageWest groups within the cheatgrass network. Juniper woodland and these sagebrush ecosystems are characterized as more mesic with higher biodiversity and increased vegetation densities (West 1983b, Miller et al. 2005, Welch 2005). The abundant vegetation increases the fuel load of the ecosystem, particularly the herbaceous biomass, and may assist in perpetuating an unbroken flaming front which may make fire effects more consistent. Land managers need to understand the highly variable nature of a prescribed fire and the vegetation they are managing when considering the post-treatment implications to the fuel bed.

Tebuthiuron has not yet had a sizable effect on the sagebrush in both SageEast groups and SageWest Group 1. SageWest Group 2 had a decrease in all three shrub categories and a corresponding increase in 10-hr DWD, however there was no herbaceous biomass response. Studies on tebuthiuron indicate its effectiveness in causing sagebrush mortality and causing an increase in herbaceous biomass (Whitson and Alley 1984, Johnson et al. 1996, Olson and Whitson 2002, McDaniel et al. 2005). A lack of response from the study sites may indicate that two years is insufficient for a measurable effect. Tebuthiuron pellets require precipitation to dissolve into the soil and to be absorbed by the sagebrush. The lack of shrub biomass response to the application of chemical treatment after two years may be due to the variability of precipitation in the sagebrush steppe which could have delayed the chemical’s effectiveness. Continued study is needed to determine the fuel load effect of this chemical treatment.

The mechanical treatment on the cheatgrass network had a slightly varied effect between SageWest and SageEast. SageWest had an increase in herbaceous biomass, a decrease in shrub biomass, and an increase in DWD. Both SageEast groups had a decrease in shrub biomass, but only SageEast Group 2 had an increase in DWD. The mechanical treatment effectively reduced shrub biomass at all sites, but converted it to DWD which may have increased fuel continuity. As the ecosystem recovers from the treatment, the DWD will persist in the surface fuels which could increase fire severity. There's a paucity of long-term research on mechanically mowing at a set height capable of maintaining sagebrush density, but reports indicate a recovery time of 10 – 20 years for Wyoming big sagebrush (Davies et al. 2009, 2011). It could be surmised that this recovery time would be faster for mountain big sagebrush due to its rapid response to disturbance (West 1983b).

The mechanical treatment in the woodland network had a very uniform effect. Live herbaceous biomass increased in all regions for all phases and shrub biomass remained unaffected. The potential for crown fire was reduced to zero while there was a corresponding increase in all size classes for DWD surface fuels in Phase 2 and Phase 3. The increase in fuel load may affect the ecosystem for many years. Passovoy and Fule (2006) found that 1000-hr solid DWD in a ponderosa pine (*Pinus Ponderosa*) woodland became rotten in 27 years. Decomposition of 1000-hr DWD may take longer in a more arid ecosystem like the sagebrush steppe. Long-term studies of Phase 3 woodlands indicate that herbaceous biomass may take 5 years to colonize available space, that mountain big sagebrush may become dominant after 20 years, and that in less than 50 years the site may once again be dominated by juniper (Taushe and Tueller 1977, Everett and Ward 1984, Skousen et al 1989, Bates et al. 2005). The increase in DWD fuel will persist throughout this process, adding to the potential fire severity, and may even be present when re-application of the mechanical treatment becomes necessary. Thus mechanical treatment may best be used as a restoration strategy as opposed to a fuel mitigation strategy.

Of the three vegetation treatments used, prescribed fire was the only treatment that removed fuel from the ecosystem. This was done through the combustion of the shrub and DWD fuel strata. However, prescribed fire was not effective at removing fuel biomass across the entire sagebrush steppe. In the arid SageEast region, where herbaceous biomass was at its lowest concentration, prescribed fire had minimal to no effect. In the woodland network Phase 3, prescribed fire even increased the surface fuel load by killing and felling trees. The other two treatments, chemical and mechanical, don't remove fuel from the ecosystem, but convert them to other fuel strata. Juniper woodland mechanical treatments significantly reduce the potential of a crown fire by removing the crown. However, the conversion of crown fuel to surface DWD can significantly increase the potential for a high severity surface fire and increase the potential of soil heating, which could threaten the remaining sagebrush steppe seed bank. In these treatments, fuel continuity is greater and the existing DWD has a lower moisture content than a live crown. A similar effect could be expected in the cheatgrass network where shrub biomass is converted into DWD. Though slower, chemical treatments also have a similar fuel conversion effect by killing shrubs and adding to the DWD as the dead shrubs decompose and break apart. If the management goals for vegetation treatments are to reduce fuel loads, then mechanical, chemical, and some prescribed fire treatments may not be suitable for those goals. For these vegetation treatments to be effective fuel reduction treatments, it may be necessary to add a secondary treatment to the ecosystem.

These vegetation treatments had a variety of effects on the structure of the fuel bed in the ecosystem. As land managers choose the best strategy to control favored species, they need to consider the long-term impact to the fuel load and the potential to fire behavior for years to come. Additional study is required to assess the long-term effect of these treatments on the fuel bed. Long-term monitoring is important in determining the continued effect of the treatments. Increases in herbaceous biomass, sagebrush recovery time, and decomposition rates of DWD can all benefit from continued study. How long these variables remain affected by the treatments can impact future land management decisions and will determine the economic value of the treatments, practicality of the treatments, and potential return interval for treatments.

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Appendix A: Name, county, state, and elevation for each SageSTEP site used in this study.

Woodland Network											
Western Juniper				Pinyon-Juniper				Utah Juniper			
Location	County	State	Elevation	Location	County	State	Elevation	Location	County	State	Elevation
Blue Mountain	Modoc	CA	1460-1590m	Marking Corral	White Pine	NV	2130-2260m	Greenville Bench	Beaver	UT	1770-1860m
Bridge Creek	Wheeler	OR	790-870m	Seven Mile	Eureka and Nye	NV	2190-2410m	Onaqui	Tooele	UT	1680-1890m
Divine Ridge	Harney	OR	1460-1590m					Sipio	Millard	UT	1680-1800m
Walker Butte	Lake	OR	1370-1430m					Stansberry	Tooele	UT	1710-1830m
Cheatgrass Network											
SageWest				SageEast							
Location	County	State	Elevation	Location	County	State	Elevation				
Rock Creek	Lake	OR	1510m	Onaqui	Tooele	UT	1650-1680m				
Grey Butte	Lake	OR	1500m	Owyhee	Elko	NV	1620-1650m				
Moses Coulee	Wenatchee	WA	520m	Roberts	Jefferson	ID	1460-1480m				
Saddle Mountain	Richland	WA	260-290m								

Appendix B: SageWest Group 1 fuel variables two-year post-treatment comparison p-values. Significance level at $\alpha = 95\%$ confidence interval as adjusted with a Bonferroni Correction per fuel variable. N = Sample size, CV = Coefficient of Variation, # = Non-parametrically calculated p-value, Clear = No change, Light Grey = Decrease, Dark Grey = Increase ($p < \alpha$).

SageWest Group 1													
	Control	N	CV	Burn	N	CV	Tebuthiuron	N	CV	Mechanical	N	CV	α
Live Herbaceous	0.0895	39	61	<.0001	24	52	0.0885	47	69	0.0001	42	63	0.050
Live Herbaceous (Plateau)	0.4261	42	81	0.5826	30	69	0.9871	44	73	0.2717	40	75	0.050
Dead Herbaceous	0.1557	39	69	0.0058	24	117	0.3492	47	78	0.0301	42	80	0.050
Dead Herbaceous (Plateau)	0.3071	42	144	0.4693	30	112	0.6829	44	67	0.7615	40	76	0.050
Total Shrub Biomass	0.2073	82	82	<0.0001	54	121	0.7238	75	61	<0.0001	82	106	0.050
Shrub 1-hr	0.2253	82	69	<0.0001	54	100	0.6332	75	57	<0.0001	82	90	0.050
Shrub 10-hr	0.1866	82	76	<0.0001	54	111	0.4182	74	58	<0.0001	82	91	0.050
10-hr DWD	0.4716	82	58	<0.0001	54	50	0.3313	91	77	0.0082	82	58	0.050
100-hr DWD	0.1746	82	76	<0.0001	54	74	0.4472	91	75	0.0351	82	74	0.050
1000-hr Solid DWD	0.0288	82	222	0.0001#	54	241	0.1491	91	196	0.0233	82	211	0.025

Appendix C: SageWest Group 2 fuel variables two-year post-treatment comparison p-values. Significance level at α = 95% confidence interval as adjusted with a Bonferroni Correction per fuel variable. N = Sample size, CV = Coefficient of Variation, # = Non-parametrically calculated p-value, Clear = No change, Light Grey = Decrease, Dark Grey = Increase ($p < \alpha$).

SageWest Group 2													
	Control	N	CV	Burn	N	CV	Tebuthiuron	N	CV	Mechanical	N	CV	α
Live Herbaceous	0.0750	32	65	0.0004	30	93	0.2317	26	81	<.0001	29	67	0.050
Live Herbaceous (Plateau)	0.3874	30	62	0.1272	24	64	0.8385	28	114	0.0057	32	77	0.050
Dead Herbaceous	0.0716	32	74	0.0143	30	131	0.6313	26	125	<.0001	29	53	0.050
Dead Herbaceous (Plateau)	0.0338	30	63	0.2452	24	76	0.4381	28	138	0.0577	32	92	0.050
Total Shrub Biomass	0.2396	61	71	<0.0001	54	113	0.0445	53	77	<0.0001	62	113	0.050
Shrub 1-hr	0.1867	61	67	<0.0001	54	99	0.0245	53	63	<0.0001	62	93	0.050
Shrub 10-hr	0.2389	61	67	<0.0001	54	105	0.0188	53	66	<0.0001	62	101	0.050
10-hr DWD	0.5372	62	49	<0.0001	54	55	0.0006	54	52	<.0001	62	61	0.050
100-hr DWD	0.2300	62	55	0.0001	54	87	0.2029	54	96	<.0001	62	74	0.050
1000-hr Solid DWD	<0.0001	62	124	<0.0001#	54	591	0.1903	54	211	0.5293	62	184	0.025

Appendix D: SageEast Group 1 fuel variables two-year post-treatment comparison p-values. Significance level at α = 95% confidence interval as adjusted with a Bonferroni Correction per fuel variable. N = Sample size, CV = Coefficient of Variation, # = Non-parametrically calculated p-value, Clear = No change, Light Grey = Decrease, Dark Grey = Increase ($p < \alpha$).

SageEast Group 1													
	Control	N	CV	Burn	N	CV	Tebuthiuron	N	CV	Mechanical	N	CV	α
Live Herbaceous	0.0005	36	78	0.0122	26	125	0.0043	48	83	0.0109	39	163	0.050
Live Herbaceous (Plateau)	0.1064	34	94	0.4210	34	97	0.0782	54	72	0.1157	42	133	0.050
Dead Herbaceous	<0.0001#	35	137	<0.0001#	26	237	<.0001#	48	141	0.0261	39	134	0.025
Dead Herbaceous (Plateau)	<0.0001#	34	159	<0.0001#	34	189	<0.0001#	54	122	<0.0001#	42	100	0.025
Total Shrub Biomass	0.3102	66	61	0.6141	52	62	0.1898	74	57	<.0001	63	64	0.050
Shrub 1-hr	0.0026	66	82	0.2249	52	59	0.8709	74	51	0.0008	63	71	0.050
Shrub 10-hr	0.4168	66	59	0.7906	52	66	0.1939	74	59	<.0001	63	64	0.050
10-hr DWD	0.3303	69	55	0.0208	60	58	0.0460	102	38	0.0776	81	51	0.050
100-hr DWD	0.6245	69	73	0.0628	60	88	0.3940	102	65	0.1160	81	71	0.050
1000-hr Solid DWD	0.1115	69	281	0.1122#	60	214	0.0005#	102	202	0.2525	81	225	0.025
1000-hr Rotten DWD	0.2365#	69	321	0.0455#	60	320	0.0650	102	217	0.0439	81	242	0.025

Appendix E: SageEast Group 1 fuel variables two-year post-treatment comparison p-values. Significance level at α = 95% confidence interval as adjusted with a Bonferroni Correction per fuel variable. N = Sample size, CV = Coefficient of Variation, # = Non-parametrically calculated p-value, Clear = No change, Light Grey = Decrease, Dark Grey = Increase ($p < \alpha$).

SageEast Group 2													
	Control	N	CV	Burn	N	CV	Tebuthiuron	N	CV	Mechanical	N	CV	α
Live Herbaceous	0.1638	38	99	0.7413	46	106	0.1876	26	71	0.5279	30	65	0.050
Live Herbaceous (Plateau)	0.0491	38	54	0.0808	38	74	0.8014	16	65	0.0345	28	62	0.050
Dead Herbaceous	0.8963	38	90	<.0001#	46	103	<0.0001#	26	121	0.0008#	30	117	0.025
Dead Herbaceous (Plateau)	0.7322	38	92	<0.0001#	38	103	0.0002#	16	98	0.0006#	28	100	0.025
Total Shrub Biomass	0.0064	30	35	0.0494	42	79	0.8821	22	41	<.0001	30	56	0.050
Shrub 1-hr	0.0005	30	32	0.0072	42	64	0.5904	22	38	0.0132	30	67	0.050
Shrub 10-hr	0.0058	30	34	0.0284	42	79	0.8742	22	41	<.0001	30	55	0.050
10-hr DWD	0.0143	76	30	<.0001	83	36	0.4072	42	32	<0.0001	62	30	0.050
100-hr DWD	0.5293	76	51	<.0001	83	54	0.0228	42	49	0.0053	62	45	0.050
1000-hr Solid DWD	0.0014	76	151	0.0497	83	152	0.0122	42	120	0.0835	62	187	0.050
1000-hr Rotten DWD	0.7126	76	264	0.0040	83	209	0.0740	42	158	0.0003	62	174	0.050

Appendix F: Western juniper woodland fuel variables two-year post-treatment comparison p-values. Significance level at α = 95% confidence interval as adjusted with a Bonferroni Correction per fuel variable. N = Sample size, CV = Coefficient of Variation, # = Non-parametrically calculated p-value, Clear = No change, Light Grey = Decrease, Dark Grey = Increase (p< α).

Western Juniper											
		Control			Burn			Mechanical			
		p-value	N	CV	p-value	N	CV	p-value	N	CV	α
Live Herbaceous	Phase 1	0.2049	46	51	<0.0001	60	57	<0.0001	53	43	0.050
	Phase 2	0.4553	56	66	<0.0001	37	56	<0.0001	47	56	
	Phase 3	0.6232	24	45	<0.0001	26	54	<0.0001	24	47	
Dead Herbaceous	Phase 1	0.6717	46	83	0.3812	60	86	0.0916	53	107	0.050
	Phase 2	0.6174	56	78	0.2132	37	116	0.6790	47	124	
	Phase 3	0.2122	24	88	0.6191	26	125	0.9972	24	107	
Total Shrub Biomass	Phase 1	0.7593	14	53	<0.0001#	32	96	0.4817	18	76	0.025
	Phase 2	0.9434	32	75	0.0002#	18	107	0.6687	31	88	
	Phase 3	0.6657	18	90	0.0549#	14	94	0.2526	14	71	
Shrub 1-hr	Phase 1	0.8092	14	59	<0.0001#	32	102	0.5102	18	74	0.025
	Phase 2	0.9793	32	76	0.0002#	18	113	0.5895	31	91	
	Phase 3	0.6679	18	84	0.1002#	14	81	0.4250	14	83	
Shrub 10-hr	Phase 1	0.7263	14	50	<0.0001#	32	90	0.4983	18	71	0.025
	Phase 2	0.8701	32	70	0.0002#	18	100	0.6561	31	84	
	Phase 3	0.6584	18	89	0.0459#	14	92	0.2764	14	70	
10-hr DWD	Phase 1	0.4152	46	47	<0.0001	60	46	0.1136	53	63	0.050
	Phase 2	0.4178	56	54	0.0418	37	64	0.0016	47	53	
	Phase 3	0.2291	24	54	0.7141	26	68	0.0126	24	78	
100-hr DWD	Phase 1	0.1461	46	84	0.0001	60	75	0.0481	53	63	0.050
	Phase 2	0.2056	55	70	0.3963	37	85	<0.0001	47	54	
	Phase 3	0.3113	24	66	0.5330	26	119	<0.0001	24	66	
1000-hr Solid DWD	Phase 1	0.1380	46	169	0.0163	60	134	0.0003	53	119	0.025
	Phase 2	0.0950	56	206	0.0437	37	150	<0.0001	47	91	
	Phase 3	0.5310	24	146	0.0529	26	172	0.0002#	24	53	
1000-hr Rotten DWD	Phase 1	0.1383#	46	433	0.0032#	60	388	0.0204#	53	410	0.025
	Phase 2	0.1237#	56	568	0.1684#	37	609	0.1562#	47	685	
	Phase 3	0.4763#	24	441	0.5000#	26	.	0.0419#	24	464	

Appendix G: Pinyon-juniper woodland fuel variables two-year post-treatment comparison p-values. Significance level at α = 95% confidence interval as adjusted with a Bonferroni Correction per fuel variable. N = Sample size, CV = Coefficient of Variation, # = Non-parametrically calculated p-value, Clear = No change, Light Grey = Decrease, Dark Grey = Increase (p< α).

Pinyon-Juniper											
		Control			Burn			Mechanical			
		p-value	N	CV	p-value	N	CV	p-value	N	CV	α
Live Herbaceous	Phase 1	0.1788	6	29	0.0037	10	51	0.0012	6	8	0.025
	Phase 2	0.1753	12	61	0.0053	18	73	0.0011	16	45	
	Phase 3	0.9081	15	67	NA	5	102	0.0097#	9	76	
Dead Herbaceous	Phase 1	0.1376#	6	46	0.2289#	13	96	0.0232	6	36	0.025
	Phase 2	0.0815	12	67	0.3859#	17	226	0.0055#	16	69	
	Phase 3	0.0092#	15	110	NA	4	67	0.0078#	9	52	
Total Shrub Biomass	Phase 1	0.6706	6	33	0.0141#	10	90	0.3227	6	21	0.025
	Phase 2	0.8673	12	52	0.0499	18	165	0.4789	16	54	
	Phase 3	0.4085#	15	93	0.1184	8	105	0.3121#	9	125	
Shrub 1-hr	Phase 1	0.4136#	6	36	0.0141#	10	84	0.0315	6	15	0.025
	Phase 2	0.2595	12	62	0.0005#	18	136	0.9081	16	52	
	Phase 3	0.7099	14	84	0.1184#	8	121	0.3121#	9	97	
Shrub 10-hr	Phase 1	0.9230	6	35	0.0139#	10	93	0.092#	6	34	0.025
	Phase 2	0.4382	12	65	0.0009#	18	168	0.6546	16	60	
	Phase 3	0.3216#	15	101	0.1184#	8	126	0.3561#	9	121	
10-hr DWD	Phase 1	0.4543	12	41	0.0211	16	42	0.1286	14	41	0.050
	Phase 2	0.6148	26	43	0.0071	46	56	0.0234	40	60	
	Phase 3	0.4676	25	48	0.7409	18	50	0.0202	14	45	
100-hr DWD	Phase 1	0.1874	12	38	0.4187	16	60	0.4113	14	52	0.050
	Phase 2	0.0919	26	52	0.3228	46	48	<0.0001	40	47	
	Phase 3	0.6269	25	104	0.0003	18	46	<0.0001	14	28	
1000-hr Solid DWD	Phase 1	0.2048#	12	201	0.0796#	16	143	0.1126#	14	136	0.025
	Phase 2	0.4018#	26	178	0.5033	46	150	<0.0001#	40	108	
	Phase 3	0.1214#	25	191	0.0221#	18	190	0.0064#	14	61	
1000-hr Rotten DWD	Phase 1	0.0838#	12	219	0.0422#	16	221	0.0109#	14	124	0.025
	Phase 2	0.006#	26	245	0.0064#	46	340	0.0711	40	267	
	Phase 3	0.0141#	25	209	0.0420#	18	228	0.0961#	14	181	

Appendix H: Utah juniper woodland fuel variables two-year post-treatment comparison p-values. Significance level at α = 95% confidence interval as adjusted with a Bonferroni Correction per fuel variable. N = Sample size, CV = Coefficient of Variation, # = Non-parametrically calculated p-value, Clear = No change, Light Grey = Decrease, Dark Grey = Increase (p< α).

Utah Juniper											
		Control			Burn			Mechanical			
		p-value	N	CV	p-value	N	CV	p-value	N	CV	α
Live Herbaceous	Phase 1	0.3193	38	75	0.0009	36	98	0.0469	24	75	0.050
	Phase 2	0.7192	46	96	0.0038	47	101	<0.0001	44	71	
	Phase 3	0.8991	40	104	0.0104	31	132	<0.0001	54	89	
Dead Herbaceous	Phase 1	0.3824	37	110	<0.0001#	29	130	0.2642	24	77	0.025
	Phase 2	0.5444	44	129	<0.0001#	44	122	0.6844	44	116	
	Phase 3	0.0056#	36	168	0.0010	31	157	0.0346	50	160	
Total Shrub Biomass	Phase 1	0.8251	26	65	<0.0001	32	81	0.4513	16	63	0.025
	Phase 2	0.4909	30	115	<0.0001	36	109	0.2494	32	89	
	Phase 3	0.1531	32	109	0.0004	20	86	0.8043	40	125	
Shrub 1-hr	Phase 1	0.9225	26	41	<0.0001	32	70	0.9795	16	67	0.025
	Phase 2	0.7668	30	90	<0.0001	36	77	0.4419	32	81	
	Phase 3	0.9410	32	103	0.0081	20	85	0.7669	40	105	
Shrub 10-hr	Phase 1	0.9674	26	49	<0.0001	32	68	0.7256	16	69	0.050
	Phase 2	0.7555	30	97	<0.0001	36	83	0.3489	32	81	
	Phase 3	0.3454	32	97	0.0007	20	74	0.9808	40	114	
10-hr DWD	Phase 1	0.5127	38	53	0.0472	36	83	0.1307	24	51	0.050
	Phase 2	0.8376	46	44	0.7391	47	49	0.0002	44	56	
	Phase 3	0.4027	40	48	0.0031	30	47	<0.0001	54	45	
100-hr DWD	Phase 1	0.1509	38	58	0.0740	36	115	0.2780	24	74	0.050
	Phase 2	0.7878	46	61	0.8591	47	61	<0.0001	44	71	
	Phase 3	0.2882	40	78	0.0083	30	67	<0.0001	54	80	
1000-hr Solid DWD	Phase 1	0.1311#	38	179	0.0103	36	142	0.0371#	24	129	0.025
	Phase 2	0.0109#	46	174	0.8242	47	113	0.0258	44	108	
	Phase 3	0.3130	40	179	0.1255#	31	245	<0.0001	54	93	
1000-hr Rotten DWD	Phase 1	0.1522#	38	324	0.1753#	36	601	0.1639#	24	490	0.025
	Phase 2	0.1383#	46	338	0.1387#	47	495	0.0189#	44	254	
	Phase 3	0.2786#	40	372	0.0787#	31	295	0.5000#	54	268	

Comparisons of Fuel Load Data Collected at Two Spatial Scales

Abstract

Sagebrush steppe is the largest vegetation type in the western United States that has for the last century been impacted by the invasions of cheatgrass and juniper. One of the challenges in modeling the spread and behavior of fire in these ecosystems in the reliance on models that assume homogenous and continuous fuels; characteristics not present in the steppe. This study, which was conducted in conjunction with the Sagebrush Steppe Treatment Evaluation Project (SageSTEP), is designed to evaluate the assumption of homogeneity by comparing variability of *in situ* fuel load data collected at a 30-m spatial scale with data collected at the 5- m scale. Three core plots within the SageSTEP program were selected, and three plots adjacent to those core plots were created. Fuel load data were collected at the 30-m scale within the core plot. Data at the 5-m spatial scale were collected in all plots. For the 5-m data, herbaceous biomass and down woody debris were destructively sampled in the adjacent plot and a regression calculation was used to estimate the fuel load within the core plot. A factorial ANOVA was used to compare the fuel load data of the 30-m with the 5-m data. The regression calculation was unable to yield R^2 values higher than 0.50 for herbaceous biomass or down woody debris. Shrub biomass regression equations' R^2 values were higher than 0.75. A comparison between the 30-m and 5-m spatial scales found means that were not equivalent, suggesting possible incompatibilities between the data collection methods or errors attributable to comparison across scales and the modifiable areal unit problem (MAUP).

Variances were larger at the 5-m scale at two of the sites, but this was not consistent across all sites tested. Compatibility between methods designed at different scales, MAUP, and the variability in the landscape all contribute to confounding comparisons between scales. Unless these phenomena can be accounted for, comparisons between scales may be imprecise, impractical, and should be avoided.

Introduction

The sagebrush steppe is the most expansive vegetation type in the western United States (West 1983a, b; Knick et al. 2003). It is a semi-arid ecosystem consisting of a dominant overstory of sagebrush (*Artemisia* spp.) with an herbaceous understory dominated by bunchgrasses. For over a century this ecosystem has undergone a dramatic vegetation transformation that has altered the historic fire regime (Miller and Rose 1999, Tausch 1999a). Two leading factors in the sagebrush steppe's decline are the invasive species cheatgrass (*Bromus tectorum*) and the expansion of juniper woodlands (*Juniperus* spp.) (Mack 1981, 1983, Pellant 1996, Tausch 1999b, Miller and Tausch 2001, Miller 2005, Welch 2005). Juniper woodlands expand in the more mesic northern regions of the sagebrush steppe, which are more productive and may be dominated by mountain big sagebrush. Cheatgrass heavily impacts the more arid lower elevations, which are dominated by Wyoming big sagebrush (Pellant 1996, Miller and Tausch 2001). These invasions alter the fuel load of the ecosystem. Historically the fires were moderate to high in severity with return intervals of 35-80 years in the more productive northern sites (Keane et al. 2008) and 100-200+ years in the more arid southern sites (Keane et al. 2008).

Cheatgrass increases fuel continuity throughout the interspaces between shrubs. The increase in fine fuels creates more homogonous burns with a return interval that can be less than ten years (Klemmedson and Smith 1964). In contrast, juniper woodlands can extend the return interval of fire (Miller et al. 2005). As juniper woodlands increase in density there is a reduction in understory vegetation which eventually replaces the sagebrush steppe (Blackburn and Tueller 1970, Bunting et al. 1999, Robert and Jones 2000, Rowland et al. 2011). This reduces the continuity of fuel so that while fires burn less frequently, however, when they do burn it will likely be a high severity crown fire (Miller et al. 2005, Keane et al. 2008). Changes in fire behavior can directly impact wildlife and wildlife habitats (Wisdom et al. 2000, Miller et al. 2005, Welch 2005), nutrient cycles (Klemmedson and Tiedemann 2000, Evans et al. 2001), hydrology (Petersen and Stringham 2008, Pierson et al. 2011), and, as people continue to build in the wildland-urban interface, can threaten property and lives (Cohen 2000, Radeloff et al. 2005).

The problems associated with changes in the fire regime have made fuel monitoring a critical process in maintaining the sagebrush steppe. Fuels are defined as live or dead biomass that contributes to the spread, intensity, and severity of a fire (Burgan and Rothermel 1984, Keene et al. 2001). The collection of fuel properties on a landscape is known as a fuel model (Anderson 1982) and can be used as inputs in fire behavior software to predict a wide range of fire behaviors (Burgan and Rothermel 1984, Reinhardt et al. 1997, Sandberg et al. 2001, Reinhardt 2003). Fuel characteristics are most accurately collected *in situ*, then applied to the ecosystem through the use of remote sensing data (Keene et al. 2001, Lentile et al. 2006).

Of available satellite sensor remote sensing technology, satellite images from the Landsat sensor has been commonly used for ecosystem level fuel mappings. These maps are 30x30-m pixels and are at a spatial scale suitable for representing variation in vegetation types. LANDFIRE products derived from Landsat have been used to predict fire behavior, monitor fire risk, and direct fuel mitigation projects (Cohen and Goward 2004, Lasaponara et al. 2006, Cochrane et al. 2012, Gralewicz et al. 2012). Landsat is capable of recording the heterogeneity of vegetation on the landscape (Lentile et al. 2006). This is important since fire models require detailed fuel load data for each vegetation type to provide accurate fire behavior predictions. The fuel model can be used to generate a landscape scale fuel map. This involves classifying the vegetation in a remote sensed image, then using *in situ* data to create a fuel model for each vegetation type classified. This will describe the fuel load across the landscape (Riano et al. 2002, Kerr and Ostrovsky 2003, Cingolani et al. 2004). However, as spatial scale increases there is a loss of heterogeneity recorded. At the 30-m scale some heterogeneity may be lost that could be recorded by instruments with finer spatial scales such as ASTER, IKONOS, QuickBird, and even aerial photography (Giakoumakis et al. 2002, Lasaponara and Lanorte 2007a, b, Arroyo et al. 2008). Though it is known that heterogeneity is lost at coarse spatial scales, the exact amount is difficult to quantify. This is problematic since one of the fundamental assumptions of fire predictive software is that fuels are homogenous within each fuel model across the ecosystem (Rothermel 1983). If fuels are heterogeneous, then the assumption of homogeneity is invalid, and the accuracy of the fire models would be questionable. Considering that land management decisions are often based on the output of fire models, it is important to understand the accuracy of the models' assumptions and how much variability exists in the ecosystem.

Fuel and Fire Models

Fuel models generated by the combined fine spatial scale and *in situ* data are used as inputs in fire models to calculate fire behaviors such as flame length, scorch height, rate of spread, and smoke output (Burgan and Rothermel 1984, Reinhardt et al. 1997, Sandberg et al. 2001, Reinhardt 2003). These fire behavior models are based on Rothermel's mathematical spread formula (Rothermel 1972, 1983) and require quantitative fuel bed data to be used as inputs. Initially, the fuel bed inputs revolved around four categories; Grass, Brush, Timber, and Slash. These categories were subdivided by Albini and Rothermel into the National Forest Fire Lab's (NFFL) 13 fuel models (Rothermel 1972, Albini 1976), which represented commonly encountered landscape fuel loads. The USDA Forest Service's National Fire-Danger Rating System (NFDRS) later created 20 more fuel models to represent additional fuel loads (Deeming et al. 1977, Burgan 1988). Land managers could collect data and manually calculate a fuel model for their site, or they could choose one or more of the pre-existing fuel models, often assisted by a photoseries of representative fuel types, as an appropriate rough approximation (Burgan et al. 1977, Anderson et al. 1982). The fuel model is then entered in a software package, such as BEHAVE, that predicted characteristics such as fire rate of spread, flame length, and fire intensity (Anderson et al. 1982, Burgan and Rothermel 1984).

Fire models have since grown in sophistication and complexity. The First Order Fire Effects Model (FOFEM), for instance, is capable of modeling emission production, canopy consumption, soil heating, and tree mortality (Reinhardt 2003, Reinhardt and

Dickinson 2010). However, to model these fire effects, FOFEM requires a more detailed fuel model, including additional fuel layers such as duff and crown fuels (Reinhardt et al. 1997). Several other fire models currently exist, each with their own specific requirements for fuel load inputs. Land managers tailoring their data collection to specific software created incompatible data sets (Keene et al. 2001). For instance, a fuel model created for BEHAVE would be incompatible and incomplete for FOFEM. The solution was the Fuel Characteristic Classification System (FCCS) which is a nationally consistent procedure in collecting all aspects of fuel data (Sandberg et al. 2001, Ottmar et al. 2007). The fuel bed is categorized into six strata: canopy, shrubs, non-woody vegetation, woody fuels, litter-lichen-moss, and ground fuels. These strata are subdivided, when necessary, into categories and subcategories (Ottmar et al. 2007). Shrubs can be subdivided into total shrub biomass, 1-hr (twigs 0-0.63cm in diameter), and 10-hr (branches 0.63-2.54cm) size classes (Frandsen 1983). Dead Woody fuels represent down woody debris (DWD) and is subdivided into particle size categories based on the time lag between ambient relative humidity and internal relative humidity (Fosberg 1970); 10-hr DWD are small branches (0.6-2.5cm diameter), 100-hr are medium branches (2.5-7.6cm diameter), and 1000-hr DWD are large branches and trunks (>7.6cm diameter). Non-woody fuels can be subdivided into live herbaceous and standing dead, which reflects the difference in moisture content. The resulting dataset is comprehensive and allows for greater interagency and international cooperation, and more flexibility in modeling fire behavior (Sandberg et al. 2001, Ottmar et al. 2007, Riccardi et al. 2007a, b).

Variability is lost at coarser scales and may impact the accuracy of fire model predictions. If the spatial scale utilized for fire models were consistent, then the variability inherent in heterogeneous fuels would remain constant. However, Rothermel based fire models are independent of spatial scale, which allows for fuel bed inputs at fine spatial scales with high degrees of variability as well as coarse spatial scales with lower recorded variability. The accuracy of the fire model prediction between the different scales depends on the amount of heterogeneity lost as spatial scales increase. The purpose of this study is to compare *in situ* data collected at a 30-m spatial scale with data collected at the 5- m scale. The hypothesis is that the means of each fuel variable will be equal to each other, but the variance in each case will be larger at the finer scale. The variance can then be used to evaluate the assumption of homogeneity.

Methods

This study was conducted at three juniper woodland sites in conjunction with the Sagebrush Steppe Treatment Evaluation Project (SageSTEP; McIver et al. 2010). The SageSTEP project is located at 19 sites and is designed to monitor long-term changes to the landscape from different vegetation treatment methods in juniper woodland and cheatgrass invaded sagebrush steppe communities of the Intermountain West (Fig 3). Each of the 19 sites used a randomized design to create a permanent core plot per treatment. Core plot size varied at each site, but ranged from 20–80.94 ha. Core plots had permanent subplots established within them. Exact number of subplots per core plot varied between 14 and 24 and was dependent on the core plot’s size. Each subplot was established along a systematic grid with a minimum distance of 50-m between the centers

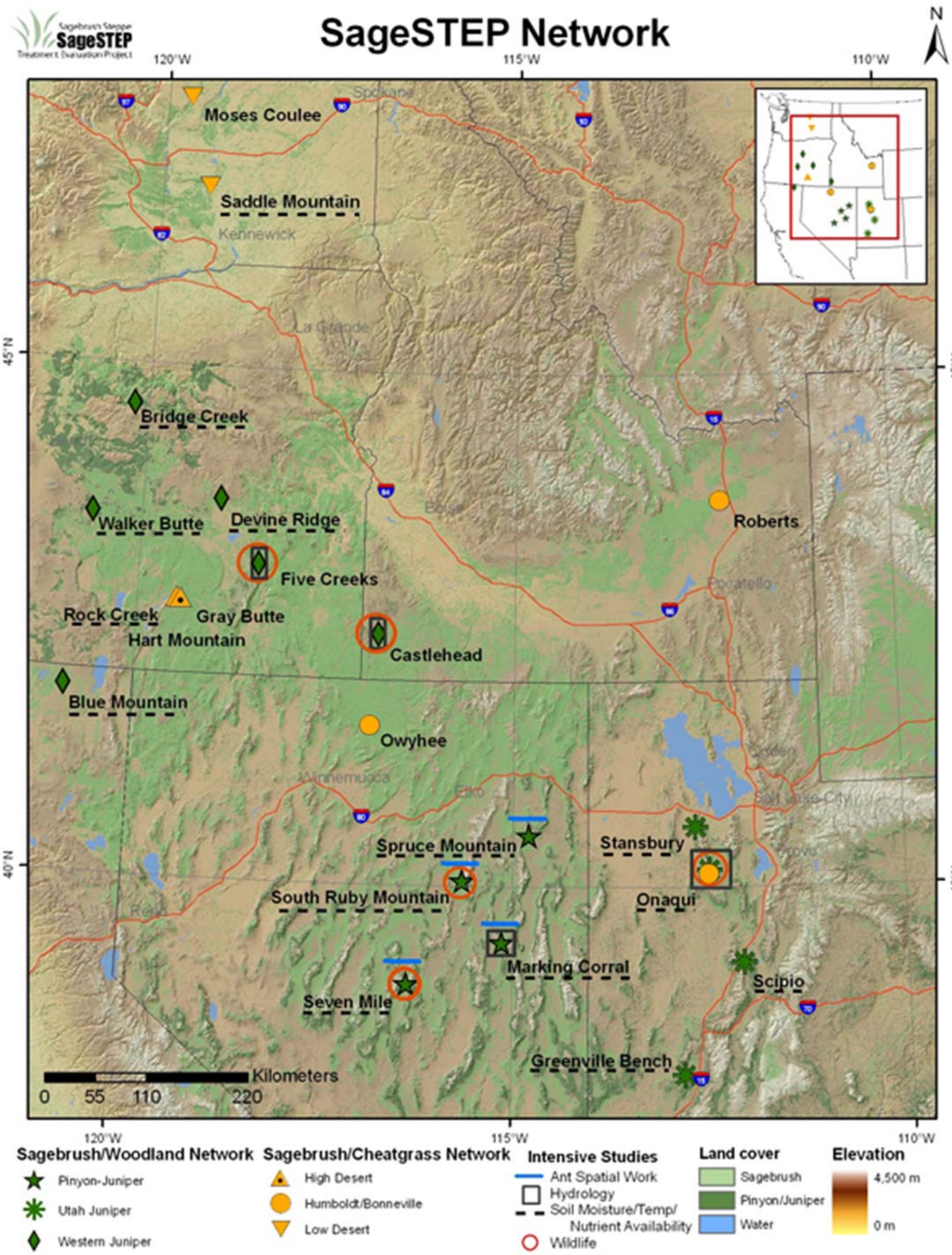

Figure 3: Map of SageSTEP Network sagebrush steppe sites (McIver et al. 2010).

of each subplot. The subplots were 30 x 33-m with 5 vegetation transects and 2 fuel loadings transect running parallel to the 33-m length (Fig 4).

A common measurement protocol was used across all sites. Fuel variables analyzed were total shrub biomass, shrub 1-hr, shrub 10-hr, 10-hr DWD, 100-hr DWD 1000-hr DWD, live herbaceous biomass, and dead herbaceous biomass. All shrub data was specific to *Artemisia tridentata* at all sites. Refer to Table 7 for specific methods and transects used for each reported variable (Bourne and Bunting 2011). Each variable corresponds to the FCCS' fuel stratum (Ottmar et al. 2007). Descriptive variables for all subplots included: region name, site name, subplot number, sampling year, UTM coordinates at zero corner, percent slope, aspect, macro-topography (ridgetop, sideslope, terrace, or bottom), micro-topography (flat, convex, or concave), and vegetation phase.

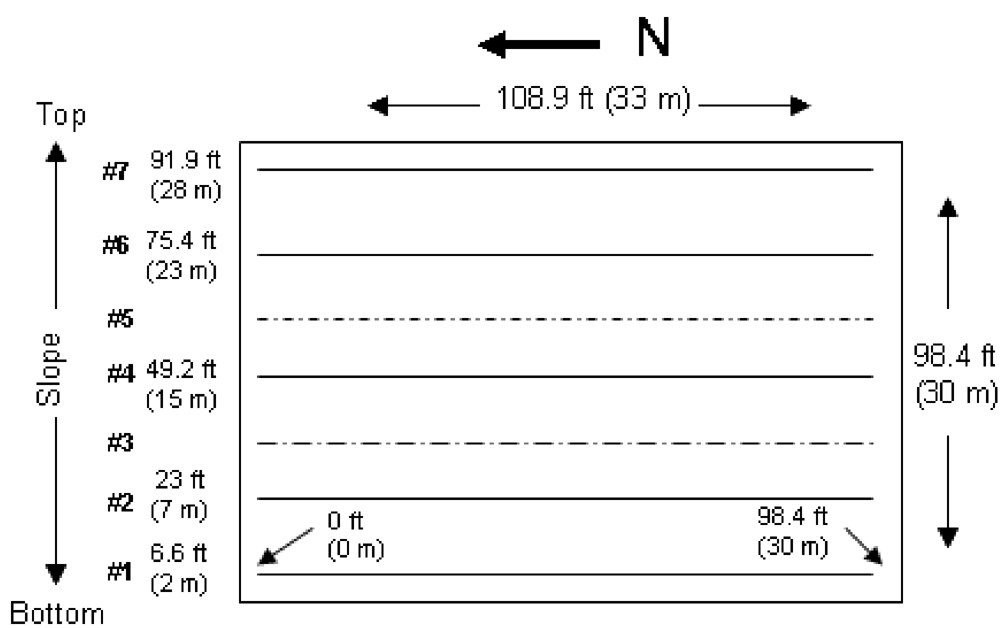

Figure 4. Subplot and transect layout. Solid lines signify vegetation transects; dotted lines denote fuels transects (Bourne and Bunting 2011).

Table 7: Sampling methods used for each of the reported variables by fuel stratum (Ottmar et al. 2007). Transect number refers to the corresponding number in figure 2. Table compiled by Bourne and Bunting 2011.

Stratum	Variable(s)	Method	Transect(s) #
Shrubs	Cover	Line point intercept (Bonham 1989)	1, 2, 4, 6, 7
	Height	Nested circular frame (Bonham 1989)	4
	Density	Belt transect (Krebs 1999, Saltzer 1994)	2, 3, 6
		Nested circular frame (Bonham 1989)	4
	Loading and Bulk Density	Harvest (Pechanec and Pickford 1937)	NA
		Nested circular frame (Bonham 1989)	4
Nonwoody fuels	Cover	Line point intercept (Bonham 1989)	1, 2, 4, 6, 7
	Height	50x50 cm quadrat (Bonham 1989)	3 in 2006; 5 in 2007
	Loading and Bulk Density	Harvest (Pechanec and Pickford 1937, Riser 1984)	3 in 2006; 5 in 2007
		50x50 cm quadrat (Bonham 1989)	3 in 2006; 5 in 2007
Woody Fuels	10-hour loading	Planar intercept (Brown et al. 1982)	2, 4, 6
	100-hour loading	Planar intercept (Brown et al. 1982)	2, 4, 6
	1000-hour Sound and Rotten loading	Planar intercept (Brown et al. 1982)	1, 2, 4, 6, 7

Quantifying Fuel Load Variability

This study took place at three of the SageSTEP sites: Marking Corral (White Pine County, NV, 2134-2256 m), Devine Ridge (Harney County, OR, 1463-1585 m), and Onaqui (Tooele County, UT, 1676-1890 m). All three sites are representative of juniper woodlands and their encroachment upon sagebrush steppe in the semi-arid regions of the Intermountain West. Each core plot ranged in size from 0.12 to 0.24 km^2 and contained 14 to 24 subplots. Nine core plots were used with a total of 182 subplots.

Marking Corral, Blue Mountain, and Devine Ridge contained 3 core plots while Onaqui had 4 core plots. Landsate 5 imagry was downloaded from the USGS's Global Visualization Viewer (glovis.usgs.gov). The landscapes downloaded represented pretreatment core plots, which allows for all 13 core plots to be used for vegetation classification in this study.

On June 18 and 19, 2009, an aircraft equipped with a Vexcel Ultracam Xdigital Camera with forward motion compensation, airborne GPS capabilities, and an ApplAnix inertial measurement unit was used to take aerial imagry in color/NIR, multispectral, and hyperspectral bands at the 6-cm, 0.5-m, and 1-m scales, respectively. Two ground targests were placed at opposite corners of each core plot and a Trimble Geo XT GPS unit was used to average 100 points of data per target to accurately align the aerial images to the Landsat 5 satelite images. ITT Visual Information Solutions' graphical software ENVI 4.5 was used to create a supervised vegetation classification of the core plots and surrounding area of the landsat image into three phases based on visual interpretation of the control core plot 6-cm color/NIR aerial image. The three phases are based on juniper dominance. Phase 1 is sagebrush steppe dominated with a presence of juniper, Phase 2 is a codominant mix of juniper and sagebrush steppe, and Phase 3 is juniper dominated (Miller et al. 2005).

Once classification was completed, a digital elevation model layer was added to provide topographical information at each site. Then a new core plot, called the adjacent core plot (ACP), was generated adjacent to the control core plot (CCP) with a minimum buffer of 800 m. The ACP had similar dimensions and topography as the CCP and was used for destructive sampling of live herbaceous, dead herbaceous, and DWD. Fine-scale data were collected in microplots for each fuel strata at the 5-m scale during the summer of 2010 (Fig. 5). Microplots were 5x5 m and had transects running at the 0, 2.5, and 5-m marks, parallel to any slope, for a total transect length of 15-m. Microplots were placed upon the landscape using ArcGIS 9.2 random point generator with a 20-m buffer betwee each point, roads, and fences. Twenty microplots were created within each CCP, and fifty microplots were created within each ACP. All microplots were located in the field using a Garman GPS unit.

Data collected at each microplot within the ACP were as follows: tree and shrub cover was estimated over the entire microplot; shrub height, length, and width was measured for each shrub greater than above 10-cm tall for each shrub in the microplot; live herbaceous, dead standing herbaceous, litter, duff, line-point cover, and woody fuels data were collected on all three transects following the same methods of the subplot

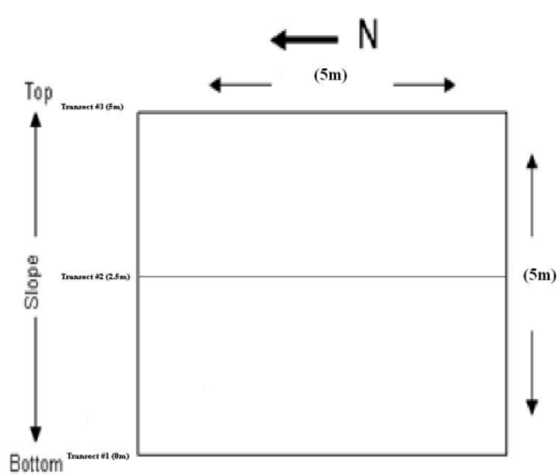

Figure 5: Five-meter microplot with transects at the 0-m, 2.5-m, and 5-m points along baseline transects.

described in Table 7. Data collected in microplots within the CCP included tree and shrub cover and shrub height, length, and width, and line-point cover. Multiple linear regression was used to calculate microplot fuel variables within the CCP using destructively sampled fuel data from the ACP. The calculation was done in SAS 9.2 for shrub total biomass, shrub 1-hr, shrub 10-hr, DWD 10-hr, DWD 100-hr, live herbaceous, and dead herbaceous with the following equation:

$$y = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \beta_3 X_3 + \varepsilon$$

where the response variable (y) is the fuel variable, such as shrub biomass, and the explanatory variables X_1, X_2, and X_3 were a combination of up to three of the following variables; shrub cover, shrub length, shrub width, shrub height, shrub volume as calculated by multiplying the shrub dimensions, or tree cover. This resulted in 20 CCP microplots with fuel data for each fuel variable. A factorial ANOVA was used to compare the means of the 5-m fuel variables with their corresponding 30-m variables for each site. Variance was analyzed with a two-sample test of variance against a null hypothesis that the 30-m:5-m variance ratio is greater than or equal to 1.

Results

Multi-linear regressions resulted in equations with R^2 above 0.75 for all shrub variables ($p<0.0001$; Table 8). DWD 10-hr had R^2 of 0.08 for Onaqui ($p=0.0521$), 0.42 for Marking Corral ($p=0.0030$), and 0.26 for Divine Ridge ($p=0.0156$). DWD 100-hr had R^2 of 0.86 for Onaqui ($p=<0.0001$), 0.30 for Marking Corral ($p=0.3300$), and 0.44 for Divine Ridge ($p=0.3381$). Live herbaceous biomass R^2 ranged from 0.30 to 0.40 ($p<0.0040$) while dead herbaceous biomass had R^2 of 0.28 for Onaqui ($p=0.0006$), 0.31 for

Marking Corral (p=0.0090), and 0.11 for Divine Ridge (p=0.0956). Mean and Variance analysis was not conducted on Live herbaceous biomass, dead herbaceous, 10-hr DWD, and 100-hr DWD, due to their low R^2 values.

The factorial ANOVA comparing 30-m with 5-m plots determined that means were not equal for total shrub biomass in Onaqui (p=0.0017), Marking Corral (p=0.0265), and Divine Ridge (p=0.0195; Table 9; Fig 6). Shrub 1-hr means were not equal for Marking Corral (p=0.0212) and Divine Ridge (p=0076; Fig 7). Shrub 10-hr means were not equal for Marking Corral (p=0.0059) and Divine Ridge (p<0.0001; Fig 8).

Table 9: Fuel variables, locations, sample size, mean, and standard deviation for the 30-m and 5-m shrub data and the factorial ANOVA comparing the means at the two spatial scale.
**= Means significantly different at the 95% confidence interval.

Fuel Variable	Location	N		Mean		Standard Deviation		ANOVA	
		30-m	5-m	30-m	5-m	30-m	5-m	N	p-value
Total Shrub Biomass	Onaqui	18	20	4640	1754	3369	1707	38	0.0017**
	Marking Corral	15	23	1299	2584	1117	1945	38	0.0265**
	Divine Ridge	17	20	516	849	322	476	37	0.0195**
Shrub 1-hr	Onaqui	18	20	760	776	627	743	38	0.9413
	Marking Corral	15	23	470	916	396	640	38	0.0212**
	Divine Ridge	17	20	216	418	135	266	37	0.0076**
Shrub 10-hr	Onaqui	18	20	1009	801	827	778	38	0.4292
	Marking Corral	15	23	411	938	348	635	38	0.0059**
	Divine Ridge	17	20	221	998	138	612	37	<0.0001**

The two sample variance statistical test determined that variances were significantly greater at the 5-m spatial scale for Marking Corral Total shrub biomass (p=0182), Shrub 1-hr (p=0.0340), and Shrub 10-hr (p=0.0119); and Divine Ridge shrub 1-hr (p=0.0042) and shrub 10-hr (p<0.0001; Table 10). A follow up two sample variance test was conducted on the remaining fuel variables and determined that Divine Ridge total shrub biomass, Onaqui shrub 1-hr, and Onaqui shrub 10-hr 30-m and 5-m variances were equivalent while Onaqui total shrub biomass had a higher variance at the 30-m spatial scale (p=0.0027).

Table 10: Two sample test comparing the 30-m and 5-m variance for each fuel variable.
**= Variance significantly greater for the 5-m spatial scale at the 95% confidence interval.

Two Sample Test for Variance						
Fuel Variable	Location	N		variance		H_0: 30m/5m >= 1
		30-m	5-m	30-m	5-m	p-value
Total Shrub Biomass	Onaqui	18	20	11350486	2914943	0.9973
	Marking Corral	15	23	1248633	3784885	0.0182**
	Divine Ridge	17	20	103541	226813	0.5930
Shrub 1-hr	Onaqui	18	20	393133	552126	0.2430
	Marking Corral	15	23	156762	409402	0.0340**
	Divine Ridge	17	20	18155	70507	0.0042**
Shrub 10-hr	Onaqui	18	20	683306	604850	0.6040
	Marking Corral	15	23	121081	403405	0.0119**
	Divine Ridge	17	20	19043	373998	<0.0001**

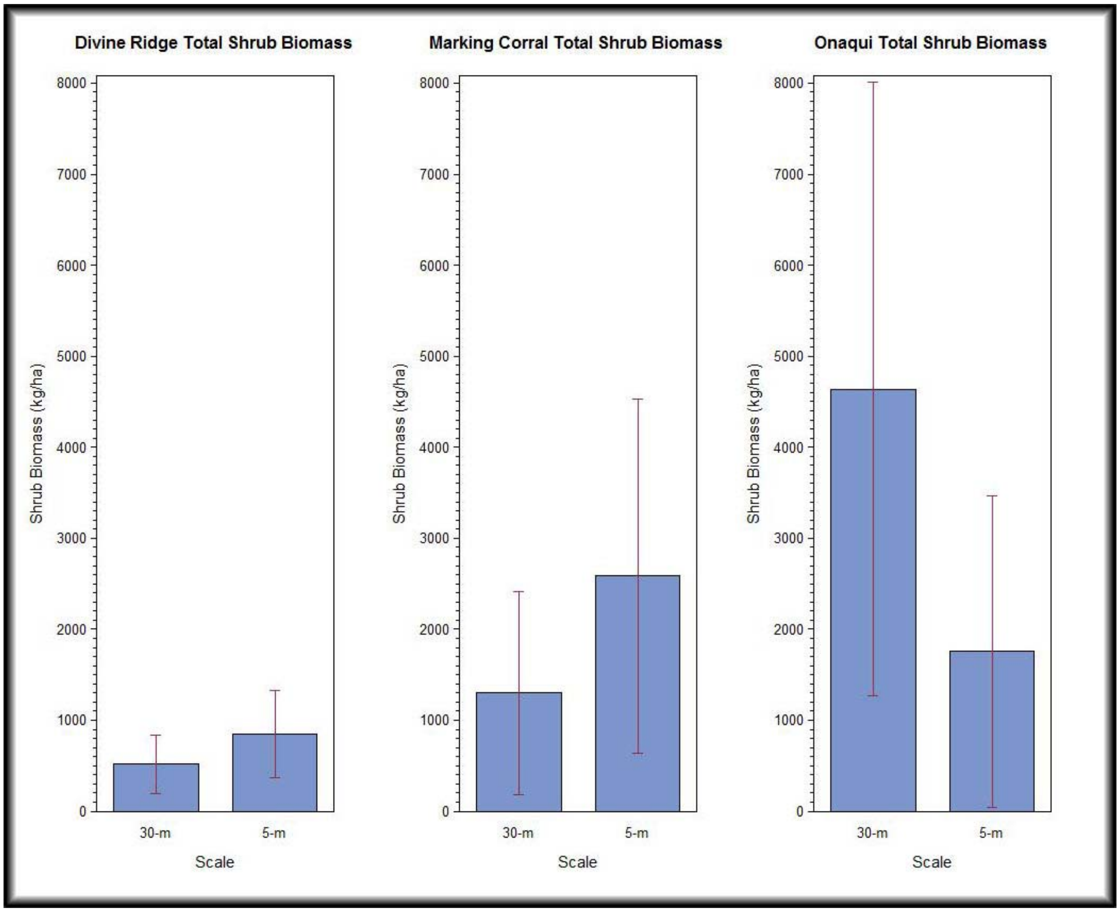

Figure 6: Total shrub biomass means and variances for Divine Ridge, Marking Corral, and Onaqui.

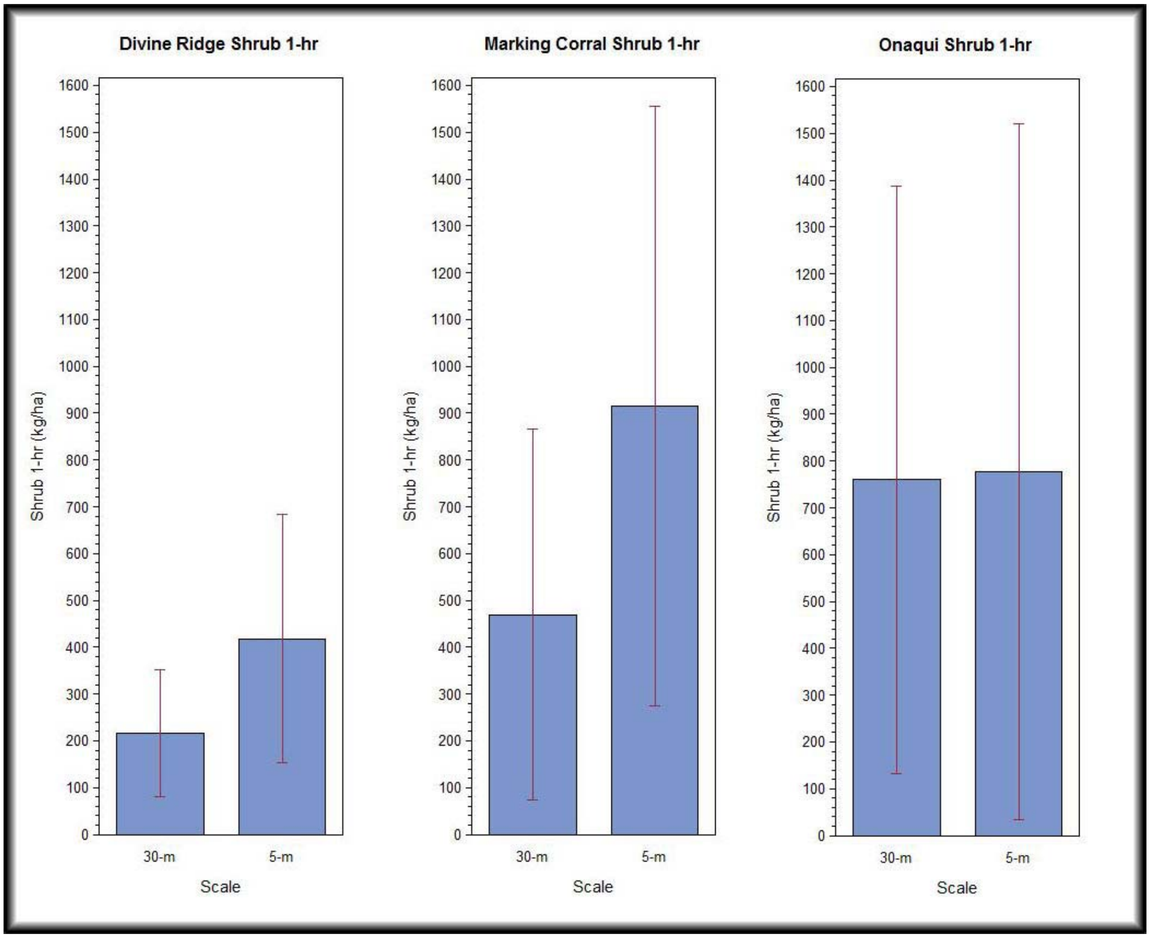

Figure 7: Shrub 1-hr biomass means and variances for Divine Ridge, Marking Corral, and Onaqui.

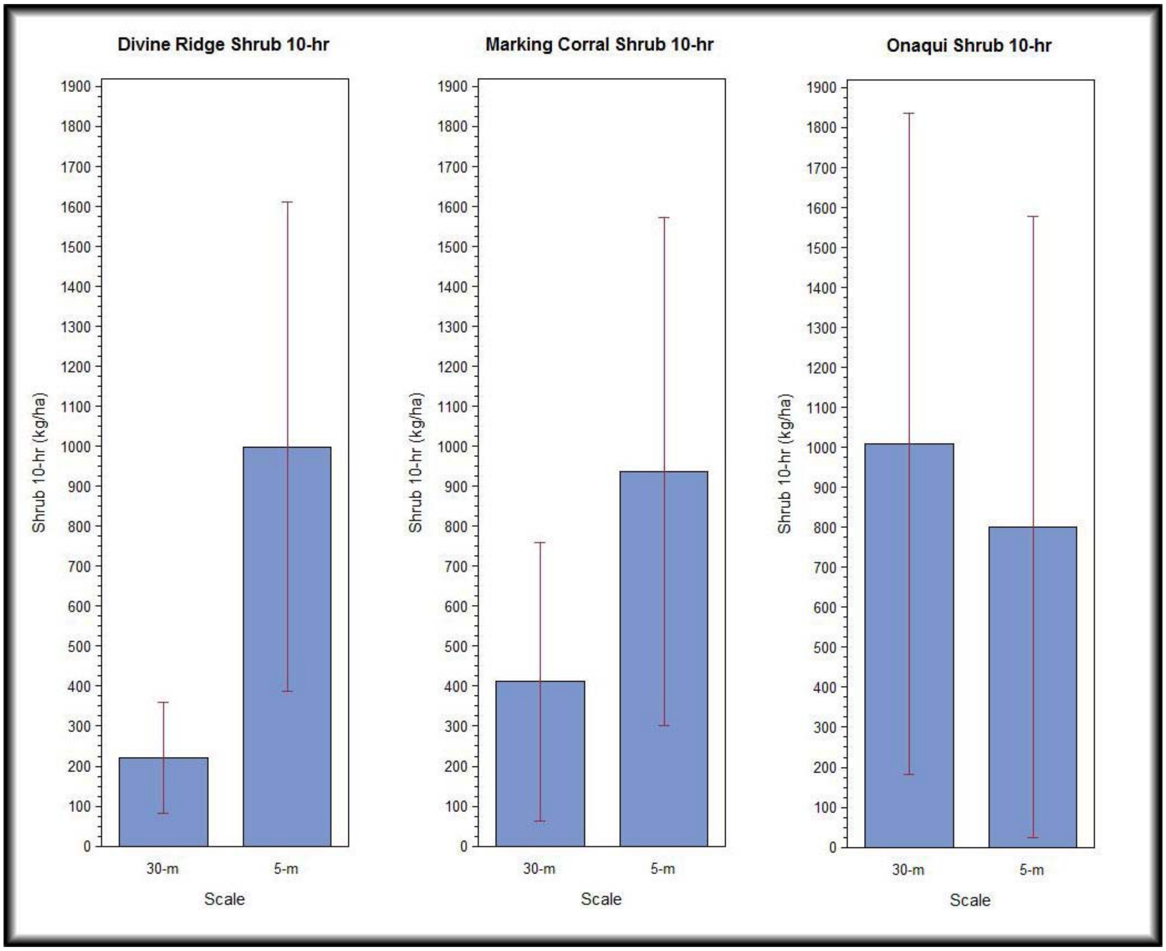

Figure 8: Shrub 10-hr biomass means and variances for Divine Ridge, Marking Corral, and Onaqui.

Discussion

Multi-linear regression for total shrub, shrub 1-hr, and shrub 10-hr biomass resulted in R^2 values higher than 0.75 for each site (Table 8). This high correlation indicates a strong relationship between the physical dimensions of a shrub (length, width, height, and volume) and its total biomass. These regression calculations were site specific and could not be used across sites. Accurate comparisons between the 5-m and

30-m scales could not be made with DWD and herbaceous biomass due to low regression R^2 values.

DWD 10-hr and 100-hr fuels appeared poorly correlated with tree cover, shrub cover, and shrub physical dimensions (Table 8). Fuel deposition studies have found that stand structure and vegetation type does contribute to the fuel load of a landscape (Hirabuki 1991, Keane 2008a, b, Van Wagtendonk and Moore 2010). Keane (2008a) used Leaf Area Index and Tree Basal Area to indirectly measure fuel deposition in the Rocky Mountains. Van Wagtendonk and Moore (2010) used crown height, crown ratio, stem height, and stem diameter to indirectly measure fuel deposition in the Sierra Nevada mountain range. Both studies found higher correlation for fine fuels (foliage, 1-hr DWD) and lower correlation for coarse fuels (10-hr and 100-hr DWD). Though deposition from the vegetation adds to the fuel load, stand history may be more important in determining fuel bed characteristics. Disturbance events such as fire, bark beetle outbreaks, and blowdowns are heterogeneous and can alter fuel deposition, creating higher variability of fuels within a stand than variability of fuels between vegetation types (Brown and Bevins 1986, Velblen et al. 1994, Kulakowski and Veblen 2002, Jenkins et al. 2008, Klutsch et al. 2009). Without a detailed account of a stand's history, it may not be feasible to indirectly measure fuel loads accurately across the landscape.

Live and dead herbaceous biomass appeared poorly correlated with tree cover, shrub cover, and shrub physical dimensions (Table 8). It is well documented that as juniper trees increase in density, there is a rapid decrease in sagebrush steppe vegetation which includes the herbaceous fuel strata (Blackburn and Tueller 1970, Bunting et al.

1999, Robert and Jones 2000, Rowland et al. 2011). However, the distinction specific to the phases of juniper encroachment is lost at the 5-m scale. A 5-m plot could be placed in a barren interspace within a dense stand of juniper and record no juniper cover. While a 5-m plot in a Phase 1 site, yet centered on a juniper, would record a high percentage of the juniper cover. This would confound the idea that juniper cover and shrub variables correlate with herbaceous biomass. Stand history may also be important in determining herbaceous fuel bed characteristics. Heterogeneity in coarser fuels could influence fire intensity and increase herbaceous mortality (Thaxton and Platt 2006) while a history of grazing may alter vegetation structure and fire frequency (Kerby et al. 2007, Mitchel et al. 2009).

Divine Ridge had several problems that confounded comparisons between spatial scales. The ACP had evidence of grazing which rendered the herbaceous regression calculation inaccurate, was located within a mix of western juniper (*Juniperus occidentalis*) and ponderosa pine (*Pinus ponderosa*), and the shrub fuel strata had a large percentage of stiff sagebrush (*Artemisia rigida*). The CCP had grazing excluded, only one ponderosa pine, and had a shrub fuel strata primarily composed of big sagebrush. The difference between the CCP and ACP, despite their proximity, demonstrates the need for field reconnaissance and information beyond remote sensing data sets prior to vegetation classification.

Comparing Scales

Divine Ridge and Marking Corral variances for total shrub biomass, shrub 1-hr, and shrub 10-hr were greater in the 5-m than 30-m scale (Fig. 6 and 7; Table 10). A larger variance in the 5-m scale was predicted. At finer scales, differences in vegetation heterogeneity become more pronounced. A clump of shrubs or a cache of DWD would heavily influence and even dominate a 5-m plot, while that same patchiness would be less apparent in a 30-m plot. Variability in the landscape also confounded vegetation classification. The landscape was delineated into three phases based on juniper tree dominance at the 30-m scale. At this scale the electromagnetic reflectance value of several individual trees and shrubs are averaged to represent a single pixel. This pixel can be characterized into a vegetative community and can be classified into vegetation types. In contrast, a 5-m pixel may completely represent a barren interspace between juniper trees in a Phase 3 woodland, or be dominated by a single tree in a Phase 1 woodland. Demonstrating that a classification system based in one spatial scale may not be applicable at other spatial scales.

The three sites lacked consistency for most fuel characteristics when compared to each other (Fig 6-8; Table 9, 10). Total shrub biomass variance was greater at the 30-m spatial scale in Onaqui, greater at the 5-m spatial scale in Marking Corral, and there was no difference between the two spatial scales in Divine Ridge. Shrub 1-hr and shrub 10-hr was slightly more consistent in that Marking Corral and Divine Ridge recorded greater variances in the 5-m spatial scale, while Onaqui recorded equivalent variances for both fuel variables. The difference and inconsistency between sites indicates that comparisons of spatial scale may be site specific, or that the data collection methods used are density dependent. Onaqui 30-m scale measured biomass nearly four times that of Marking Corral and almost ten times that of Divine Ridge, while the 5-m scale were similar at the three sites, with Marking Corral having greater biomass.

The means at Divine Ridge and Marking Corral for total shrub biomass, shrub 1-hr, and shrub 10-hr fuels were greater when sampled at the 5-m scale than at the 30-m scale (Table 9). Onaqui had greater means at the 30-m scale for total shrub biomass, while shrub 1-hr and 10-hr means were equivalent across scales. This was unexpected since both the 5-m and 30-m data were collected in the same location. This might be a result of differences in method between spatial scales. At the 30-m scale, a nested circular frame method was used. Five circles were located along a 30-m transect, each with a radius of 1, 2, or 3-m depending on the density of shrubs in the plot. At the 5-m scale every shrub rooted in the plot was measured, which may allow for some over estimation of shrub biomass. Both methods resulted in a calculation for shrub biomass, however it may be that the difference in method caused the variability of biomass means between scales.

In addition, the data collection process may have been influenced by the modifiable areal unit problem (MAUP). MAUP exists where areal units are arbitrarily created and may be aggregated to form units that represent different spatial scales and/or a variety of shapes. It is characterized as a scaling problem and a zonation problem. The scaling problem is apparent variability in statistical results as finer spatial scale data is aggregated into coarser spatial scales. This causes a blending effect resulting in less variability reported as data are aggregated into coarser scales. The zonation problem is apparent variability in statistical results within the same scale and unit size, but different unit shape. Its effect on means and variance is not as predictable as the scaling problem (Fotheringham and Wong 1991, Jelinski and Wu 1996, Svancara et al. 2002, Dark and Bram 2007). Both spatial scales in this study collected biomass then adjusted by its collection area to represent kg/ha within that plot. At the 5-m scale, shrub biomass was collected within a 5x5-m plot then divided by 25-m^2 to represent kg/m^2. The 30-m scale had some variability in collection size that may be vulnerable to MAUP. Five circles on one transect were measured, with radii dependent on the density of the plot, resulting in the biomass being divided by 16-m^2, 63-m^2, or 141-m^2 for 1-m, 2-m, and 3-m radii, respectively. The difference in zonal size could affect the means and variance calculated between 30-m plots which could influence the comparison between the 30-m and 5-m spatial scale.

The difficulty described above demonstrates that caution should be taken when making comparisons across spatial scales. Vegetation classifications and definitions created at one spatial scale may be diminished or may not exist at coarser or finer spatial scales. Common data collection methods designed at one spatial scale may become cumbersome and problematic at different spatial scales, and any scale related adjustments made to those methods may influence scale comparisons.

Conclusion

The exact amount of heterogeneity lost based on variance as scales increased was difficult to ascertain. Indirect measurements for DWD and herbaceous biomass yielded low correlation to surrounding vegetation, which in turn made comparison between the 30-m and 5-m scale unreliable. Fuel deposition studies found lower correlation with coarser fuels (Keane 2008a, Van Wagtendonk and Moore 2010), and it may be that stand history is more influential as a correlating variable for DWD and herbaceous biomass.

The results indicate that indirect measurements utilizing tree cover, shrub cover, and shrub dimensions are not recommended.

Shrub biomass regression equations' R^2 values were generally greater than 0.75, however the comparisons between the 30-m and 5-m scale were inconsistent. Means were not equivalent as expected, suggesting possible incompatibilities between data collection method or errors attributable to the modifiable areal unit problem. Two sites, Divine Ridge and Marking Corral, did have larger variances at the smaller spatial scale. However, this was not always the case, and in Onaqui larger variance was found at the 30-m spatial scale for total shrub biomass. The inconsistent results indicate the problems inherent in comparing across scales. The methods of data collection between the two scales were nearly identical, however the methods were specifically designed at the 30-m spatial scale. As a result, some data collecting methods, such as shrub biomass collection, was adjusted to be practical at the finer scale. These small adjustments may have been enough to impact the comparison.

These results demonstrate the difficulty inherent in comparing data collected at different spatial scales. Compatibility between methods designed at different scales, MAUP, and the variability in the landscape all contribute to confounding comparisons between scales. Unless these can be overcome, comparisons between scales may be imprecise, impractical, and should be avoided.

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