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Vegetation Disturbance Regimes and Conifer Succession in Gunnison Sage Grouse (Centrocerus Minimus) Habitat

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MS ENVIRONMENTAL BIOLOGY
CAPSTONE PROJECT

by

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CHAPTER 1. LITERATURE REVIEW: A REVIEW OF CURRENT FOREST FIRE OCCURRENCE POTENTIAL IN WESTERN NORTH AMERICA

Abstract

Many forests in the southwestern United States are undergoing a shift from low severity to high severity fire regimes, and therefore a process is desired to understand which forests are outside their historic range of fire frequency and may be a high priority for restoration. Here, I review the evidence regarding anthropogenic alternation of fire regimes, specifically the extent to which current forests may be outside their historic range of variability. A combination of published literature data and analysis showed that forest fires shape the landscape and disturb the ecosystems. A long period of forest fire exclusion policy has allowed fire intolerant tree species to increase, which promotes high severity fires. Many reports also stated that wind speed, drought, slope status, and unreasonable land use have accelerated forest fire prevalence. However, some researchers contend that high severity fire regimes and high tree density have been extensive in historical forests. Thus, predicted models based on multiplicity site index were urged to be established, however, there were no models that could be practiced in every forest due to the wide diversity of terrestrial communities. In conclusion, understanding variability in current forest fire areas is key to predict coming fire frequency and severity, benefits researchers

to decided which forests are worth studying, and helps foresters to set up better forest management practices.

Introduction

Wildfire is an important disturbance process that maintains or changes species composition, stand structure, and ecological functions in many terrestrial ecosystems (Pausas & Keeley, 2009). Over the past few decades, there has been an increasing global trend of fire disturbance, mainly caused by climate change and human activity (Bowman et al., 2011; Westerling et al. 2011; Oliveira et al., 2014; Marlon et al., 2009). Wildfires have impacted multiple ecosystems in North America (Ryan et al., 2013). In the southwestern United States, wildfires with multiple, high-severity burn patches (total or near-total canopy mortality) with large burned perimeters include the Cerro Grande fire (2000, Jemez Mountains, New Mexico, 19,425 ha total area burned), the Rodeo-Chedeski fire (2002, White Mountains, Arizona, 187,220 ha), and the Bullock and Aspen fires (2002 and 2003, Santa Catalina Mountains, 46,667 ha) (Margolis et al., 2007). Historical forest fire records quantified from tree rings (Buechling & Baker, 2004) and tree fire scars (Buechling & Baker, 2004; Margolis et al., 2007; Tepley & Veblen, 2015) have demonstrated that forest fire occurs less frequently now but with greater fire severities in the southwestern United States (hereafter southwest). After the 1940s, when fire suppression became a national policy, researchers found that fire exclusion caused vegetation to be much more susceptible to uncharacteristically severe fire (Odion et al., 2014).

This review combines data analysis and conclusions from published literature. The null hypothesis in this paper is that current forests are inside their historical range of fire frequency and fire effects. In the southwest, climate has not changed much in recent centuries (Sheppard et al., 2002), so vegetation, topographic, land use management and connectivity variables are assumed to be main drivers that cause forest fires. This paper focuses specifically on the relationships between fire occurrence and vegetation type, topography, land management and connectivity. I expect that high severity fire leads to changes in plant species composition and stand structure. Then forest fire regimes are changed after long fire intervals. I also expect forest topography and land management to affect forest structure, resulting in patterns of fire-induced mortality and frequency change. In order to study the current potential of fire regimes, I expect there might be a model to evaluate forest fire behavior based on stand composition and structure. Understanding fire regimes will enable prediction of future fire potential, assist scientists to choose study sites, and guide foresters to thin forests to reduce forest fire frequency and severity.

Factors causing forest fire

There is a widespread concern that forests in western North America have become more susceptible to severe fires compared to historical fire severity. One major cause of increasing fire severity in the western United States is climate change over the last several decades (Westerling et al. 2006). Forest fuels are no longer the main reason for fire occurrence in most of the southwestern US according to some recent climate-fire research (Miller et al., 2009). Therefore, while climate change may have played a limited role in area burned, I will focus on vegetation, topography, and land management as potential causes of forest fire.

Vegetation structure and composition influences fire susceptibility

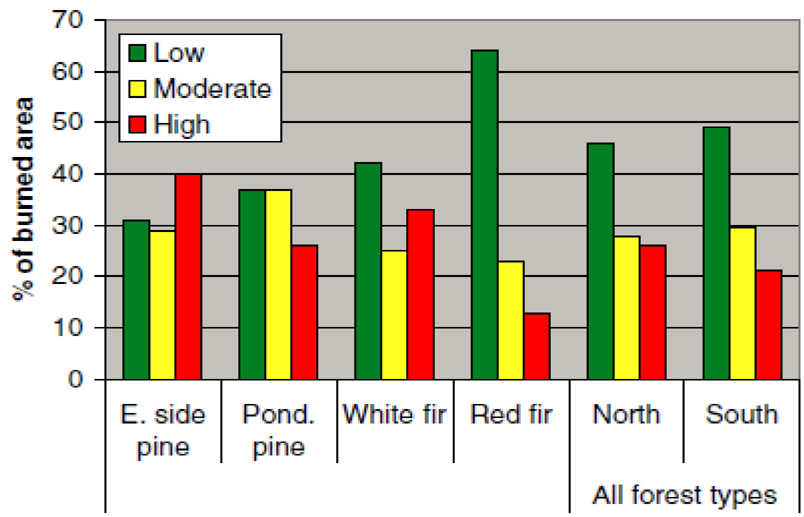

Figure 1. Low, moderate, and high severity fire as % of total burned area for four major forest types in the Sierra Nevada, 1984–2004, plus composites (excluding pinyonjuniper) for the north and south study regions. Only fires larger than 400 ha included (due to uneven spatial coverage of smaller fires); data from USFS lands only. E. side pine = P. jeffreyi and P. ponderosa stands, primarily on the east side of the Pacific Crest; P. pine = P. ponderosa, primarily on the west side of the Crest (Miller et al.,2009).

Different forest types have varied fire susceptibility levels. Areas mapped by vegetation type in 177 fires that occurred during 1984-2004 on forest service lands (Sierra Nevada and Southern Cascade Mountains, California and Nevada) indicated that eastside pine had 40% high severity fires, ponderosa pine had 26% high severity fires, mixed conifer had 29% high severity fires, white fir had 33% high severity fires, red fir had 13% high severity fires, and black oak had 23% high severity fire (Miller et al., 2009). Miller's study also showed that east side pine in the Sierra Nevada was the most susceptible forest type to high severity fires; red fir has the largest susceptibility to low severity fire occurrence and the lowest for high severity fire (Figure 1). After the fire suppression policy, fire exclusion has been found to cause vegetation to be much more susceptible to severe fire, especially drier forests dominated by ponderosa pine (*Pinus ponderosa Dougl. ex. Laws*) and Jeffrey pine (*P. jeffreyi Grev. & Balf.*), or those mixed with ponderosa/Jeffrey-pine and other conifer species (Odion et al., 2014), due to increased tree density and structure diversity. Historical fires may change forest structure and composition, as a result, forest fire susceptibility was altered. A fire exclusion experiment showed that fire was a dominant influence on plant age class diversity (Figure 2; Odion et al., 2014).

However, some researchers disagree that forest structure has changed relative to pre-settlement conditions. Laughlin (2011) stated that old age stands with large DBH have no potential for a severe fire. They also found that a second-entry prescribed fire in Grand Canyon National Park demonstrated that the fire treatments resulted in greater impacts on conifer seedlings <30 cm tall than the seedlings >30 cm tall (2011).

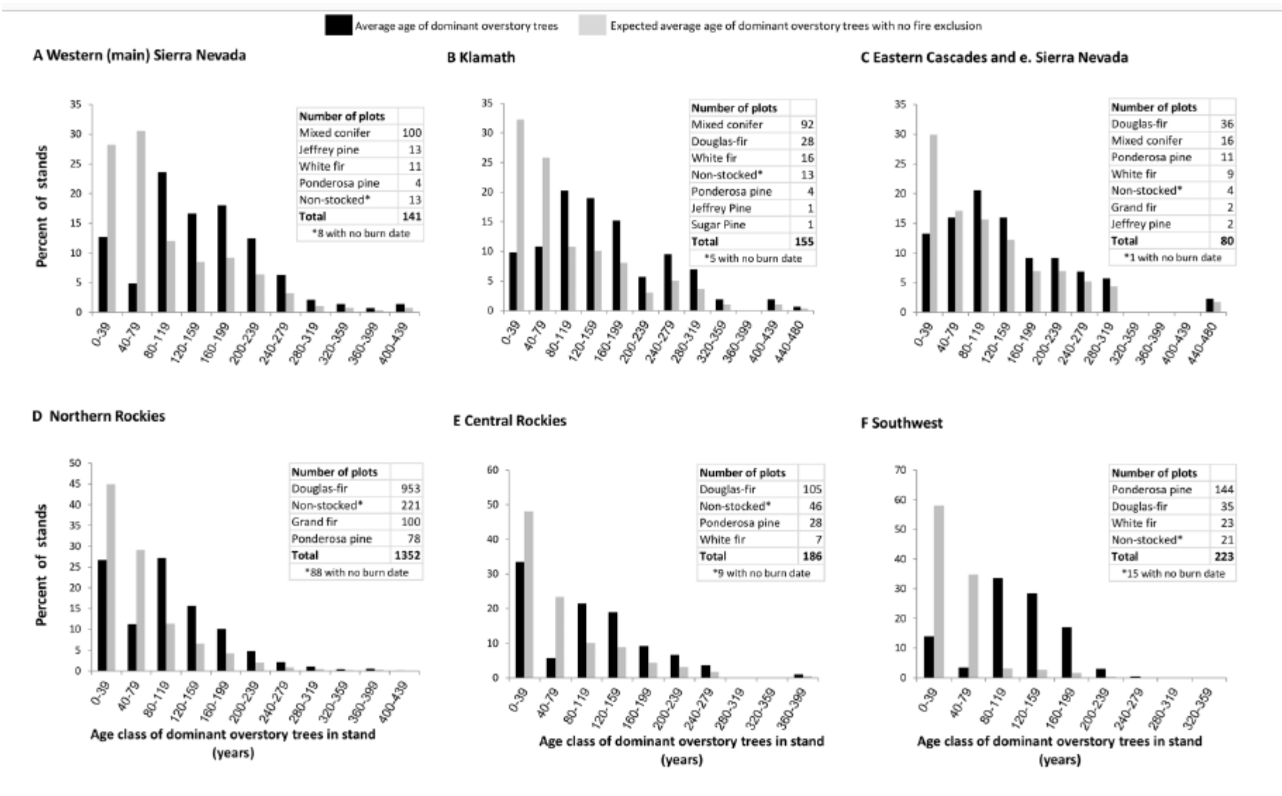

Figure 2. Age class distributions of dominant overstory trees. Data are from the US Forest Service Forest. Black bars are the current distributions of stand ages. Grey bars show an expected distribution (average age of dominant overstory trees with no fire exclusion) based on projecting the occurrence of the same age distributions that occurred from 1810–1889 into the most recent 80 year time period and rescaling these data. (Odion et al., 2014)

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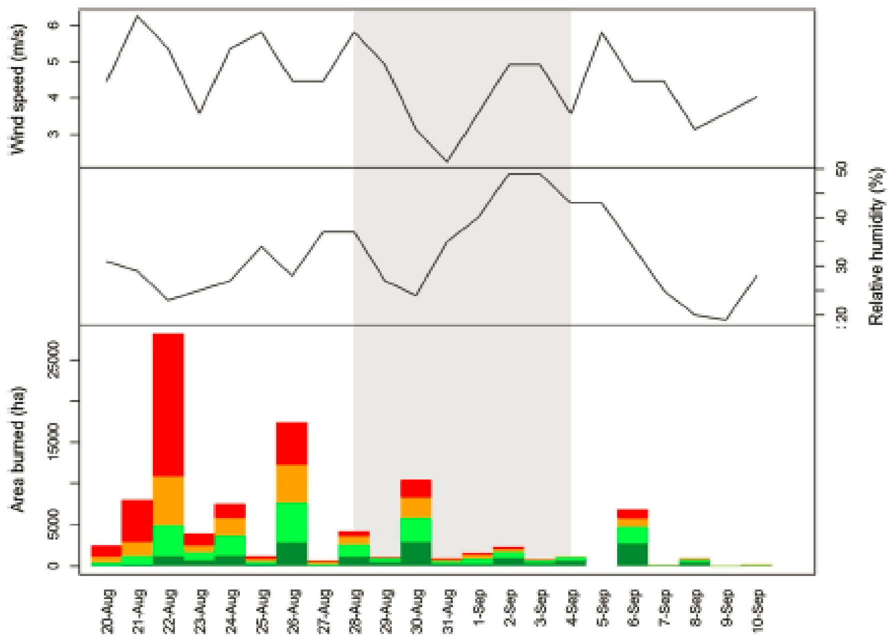

Figure 3. Daily area burned by the Rim Fire between August 20 and September 10, 2013 and weather conditions on those days. Each bar represents a 24-h period, roughly 12:00 AM–12:00 AM. The color of the bars represents severity: dark green is unchanged, light green is low, orange is moderate, and red is high (Harris, & Taylor, 2015).

Wind speed, drought and slope conditions promote fire severity

In complex Yoemite national park terrain, flammability is impacted by changes in topography (Harris & Taylor, 2015). Climatic variables such as high speed of wind and drought conditions led to high fire severity (Figure 3). Fire spreads faster upslope than downslope due to convective pre-heating of upslope fuels from a fire burning below. In addition, fire spreads rapidly because of daytime upslope winds that develop from local-scale thermal pressure gradients that develop in mountainous terrain (Werth et al., 2011). Therefore, stands on middle and upper slopes receive higher fire line intensities (Harris & Taylor, 2015).

Over the past 20,000 years, climate changes have affected the high-elevation treeline in western North America (Malanson et al., 2013). A forest's composition and structure may change if its treeline has upslope or downslope movement. Malanson et.al. (2013) concluded that the change of treeline will not be a simple upward movement of a line. Thus, forest fire susceptibility will not be a simple switch from historical low mortality and short fire intervals to high mortality and long fire intervals.

Land management alters forest susceptibility to fire

In some land management processes, natural forest might be destroyed, leading to reductions in forest density and coverage. In contrast, a reasonable management plan might be refined forest structure and composition and develop a wildfire tolerant forest. In the late 19th and early 20th centuries, widespread livestock grazing and more advanced fire suppression caused the exclusion of surface fires from many fire-prone forests (Westerling et al., 2006). Previous research has shown that the vegetation type affects success of forest management strategies (Gustafson et al., 2004; Westerling et al., 2006). For example, fire exclusion in fire-prone forests that typically experienced frequent low-severity surface fires, stands structure and composition has become highly altered. In contrast, high elevation forest types that experienced infrequent, high-severity fires may have been minimally affected by systematic fire exclusion. (Westerling et al., 2006).

Predicted models to present relationships between multiple factors and fire range

Many studies focused on establishing a model to predict a forest fire. The LANDIS model (Gustafson et al., 2004) is a common model for identifying specific locations where interacting factors of land index, topography, management strategy and connectivity increase fire risk. This model has been useful to those not able to undertake the cost and effort of developing their own model, and it has provided a growing diverse set of test landscapes for the model (Mladenoff, 2004). Another method of fire prediction is to set up a regression or ANOVA model based on historical site index data. In this case, researchers can add variables specific to their study sites, such as historical severity of fire time in model, road density and distance from forest frontiers (Nepstad et al., 2001), fire and wind intervals, special silviculture history, soil depth and texture (Sherriff et al., 2014; Harris & Taylor, 2015), coarse woody debris (Harris & Taylor, 2015), fuels (Harris & Taylor, 2015; Odion et al., 2013; Miquelajauregui et al., 2016), rock (Harris & Taylor, 2015), biomass (Harris & Taylor, 2015; Miquelajauregui et al., 2016), day of burn (Harris & Taylor, 2015), wind speed (Harris & Taylor, 2015), fire behavior (Odion et al., 2013), carbon storage (Odion et al., 2013), and bark thickness (Miquelajauregui et al., 2016). However, there is not a model that can be used to predicate all forest fires at any time period. In addition, forest density, humidity, aspect, elevation, slope steepness, cover type, temperature, and precipitation are the basic factors to be considered when predicting the possibility of a forest fire occurrence. A model combining the basic factors and special factors of a studied forest is still able to present current forest ranges to some degree.

The effect of fire treatment on potential fire severity regimes

Managers of public and industrial forestlands are keenly interested in reducing the susceptibility of their forested landscapes to high-severity wildfires. The primary tools used to mitigate fire susceptibility are the reduction of fuel loads by thinning, manual fuel removal, and prescribed burning (Mutch, 1994). Outputs of fire model assisted managers to decide which areas to be treated first due to management strategies may affect less on large forest fire during extreme fire weather (Finney, 2005). Silvicultural practices also set up many forest ecosystem management guidelines based on natural disturbance. Present stand-level silvicultural strategies are realistically similar to natural disturbances under main goals of the large-scale use of cautious logging to protect progressive regeneration in ecosystems (Bergeron et al., 1999). Experiments were conducted in which fire was applied to forests in six western United States fire and fire surrogate sites, and the results of mean post-treatment combined fuel loads, mean percentage canopy cover and post fire live tree density directly tested the hypothesis that an anthropogenic forest fire reduces forest fire severity.

The most effective treatment for reducing crown fire potential and predicted tree mortality was mechanical treatments with prescribed burning or pile burning (Figure 4) because of low surface fuel loads and improved vertical and horizontal canopy separation. At the Southern Sierra Nevada site, both fall and spring fire-only treatments were still operative at removing surface fuels. In the forest with cutting down the density of trees up to 25 cm DBH, the fall fire treatment was more effective, while spring burning resulted in greater retention of large woody debris (Knapp et al., 2005). Previous studies in the Central Sierra Nevada site found “a significantly higher total standing volume of snags up to 15 cm DBH in the fire-only treatment when compared with the mechanical plus fire treatment which achieved a desired condition regarding potential fire behavior and effects in these forests" (Stephens et al., 2009). Fire hazards can be decreasing and more sustainable forest can be established through improvement/selection cutting, including removing trees through a low thinning and removing some low-vigor and more rich shade-tolerant trees from the main canopy (Fiedler et al., 2001). Mechanical-only treatment areas burned with higher severe fires than untreated areas (Raymond &Peterson, 2005). When the expectation was to reduce potential fire actions and severity, the most effective treatment is thinning forest from below, resulting in subsequent surface fuel reduction by fire. However, due to dense mid- and upper canopies and weighty amounts of shade tolerant species, this treatment may not be sufficient in some Rocky Mountain areas (Fiedler & Keegan, 2003; Fiedler et al., 2003).

Summary

Many scientists and foresters have tried to work out effective management plans to maintain forest disturbance regimes that approximate historical conditions. Historical forest structure and composition is a direct reference to decide current fire management. The relationships between factors causing forest structure and composition change and forest fires provide evidence to reject or accept the null hypothesis. Although some researchers argue that current forest fire areas are not outside the historical range of variability, most scientists believe that western North American forests are at risk for high severity fire due to increased fire intervals resulting in tree densities higher than those of the historical period. Therefore, forest management activities such as thinning need to be done to avoid potential high mortality wildfires. Future work that tests alternative hypotheses within the multiple interacting drivers framework could profitably expand our understanding of western North American forest fire areas and elucidate the complex difference between historical forest fires and current fire potential.

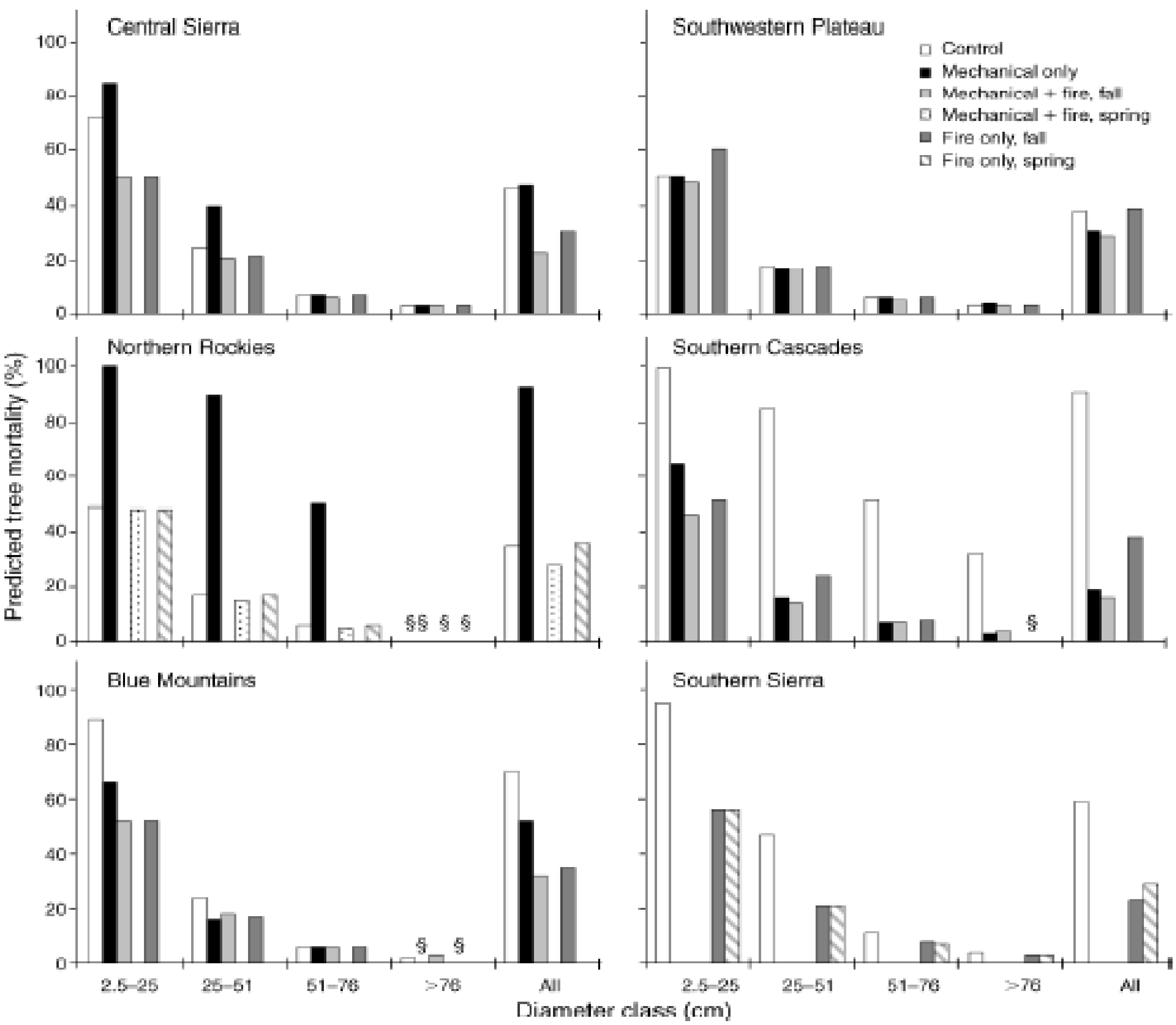

Figure 4. Modeled postfire tree mortality by dbh class under 80th percentile weather conditions for trees remaining at six western United States Fire and Fire Surrogate sites after treatments. When no trees were present in a given treatment, this absence of a given size class is denoted by §. If there is no bar, a treatment was not implemented at that site. (Stephens et al. 2009)

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CHAPTER 2. GRANT PROPOSAL: CONIFER INVASION OF GUNNISON SAGE GROUSE (*CENTROCERUS MINIMUS*) HABITAT

Abstract

The major reason for Gunnison sage grouse decline in southwestern Colorado is thought to be loss and fragmentation of its sagebrush habitat. Wildfire is considered as a major factor to cause sagebrush loss in southwestern USA due to the invasion of juniper and pine after fire. This project works to determine whether Rocky Mountain juniper and pinyon pine forest is expanding its range into sagebrush area at low elevations in Gunnison County, Colorado. Fire, grazing, or other anthropogenic activities will be assessed to explain juniper and pine forest invasion patterns on sagebrush steppe. I hypothesize that recent land use, climate variability, over grazing, and fire will influence tree invasion of sagebrush. Thus, the lower treeline will advance and encroach upon sagebrush habitat. Field work will be done in late summer and early fall 2017 to collect data on the establishment dates of trees invading sagebrush. Juniper and pine forest structure, density and lower treeline elevation, fire history will be analyzed using tree cores and forest inventory data. Climate, fire history and livestock will be extracted from published database and literature. The outcome of this project will help land managers to understand current lower treeline conifer condition, and make up better management strategies to protect Gunnison sage grouse.

Background/Rationale /Significance

Gunnison sage grouse (*C. minimus*) inhabited large portions of woody sagebrush (*Artemisia*) steppe and sagebrush semi-desert plant assemblages in western North America (West 1983; Crawford et al 2004). In winter, sage grouse are dependent solely on sagebrush leaves for food, and they require areas dominated by sagebrush for cover (Oyler-McCance et al 2001). In the spring, nesting occurs in thickly vegetated areas usually dominated by sagebrush and during brood rearing, hens with chicks are typically found in diverse habitats including areas of sagebrush (Oyler-McCance et al 2001). The population of sage grouse are declining throughout most of their range in recent years due to plant community change in the 19th and 20th centuries (Crawford et al 2004). Factors responsible for plant community change have included alterations in fires regime; excessive livestock grazing; proliferation of non-native plant species; conversion of rangeland seeded pastures, cropland and roads; and other land alterations (Braun 1995). Crawford (2004) also found that critical habitat components include adequate canopy cover of tall grasses ($\geq$ 18 cm) and medium height shrubs (40-80 cm) for nesting, abundant forbs and insects for brood rearing, and availability herbaceous riparian species for late-growing season foraging. Oyler-McCance (2001) documented a loss of 20% or 155,673 ha of sagebrush-dominated areas in southwestern Colorado between 1985 and 1993. Loss rates of the amount of sagebrush-dominated area were much lower in the Gunnison Basin (Oyler-McCance et al 2001). Oyler-McCance (2001) also found that 37% of their sampled plots were underwent substantial fragmentation of sagebrush vegetation. If current trends of habitat loss and fragmentation continue, Gunnison sage grouse may become extinct (Oyler-McCance et al 2001). Thus, preventing the sagebrush habitat from further loss is an important topic in ecosystem management.

The reasons for sage grouse abundance decline are difficult to understand because it needs to integrate sage grouse population ecology and habitat requirement, as well as the ecology and management of plant communities. For example, livestock grazing can have negative or positive impacts on sage grouse habitat depending on the timing and intensity of grazing and which habitat element is being considered (Crawford et al 2004). Wildfire is an important disturbance process that maintains or changes species composition, stand structure, and ecological functions in many terrestrial ecosystems (Pausas & Keeley, 2009). Fire ecology of sage grouse habitat changed dramatically after European settlement (Crawford et al 2004), therefore wildfire is commonly considered as a disturbance to destroy sagebrush habitat. Like other sage grouse abundance decline reasons, the impact of wildfire patterns on sage grouse would be complex in different ecosystems. In high elevation sagebrush habitat, fire return intervals have increased (from 12-24 to > 50 years) resulting in invasion of conifers and a consequent loss of understory herbaceous and shrub canopy cover (Crawford et al 2004). In contrast, fire return intervals have decreased (from 50-100 to <10 years) in lower elevation sagebrush habitat, still leading to loss sagebrush habitat. Crawford (2004) explained that annual greases have opportunities to invade open land after wildfires and occupy native bunchgrasses and shrubs habitat. Hence, understanding patterns of reduction in fire frequency is essential to study relationship between fire occurrence and sage grouse distribution.

The Rocky Mountain juniper (*Juniperus Scopulorum*) and pinyon pine (*Pinus Edulis*) woodlands under study in this project are those woodlands that occupy the lower slopes of the mountains and foothills and the ecotones bordering the sagebrush in the Rocky Mountains of Gunnison County, Colorado. Managers of public and industrial forestlands are keenly interested in conservation of sage grouse, thus reducing the susceptibility of their forested landscapes to high-severity wildfires is a key point to protect sagebrush habitats. The primary tools used to mitigate fire susceptibility are the reduction of fuel loads by thinning, manual fuel removal, and prescribed burning (Mutch, 1994). This project will focus on the forest structure and composition in relation to land management and historical forest disturbances based on historical records and present day databases. Current forest fire regimes will be predicted to answer questions of whether current fire regimes are outside historical regimes or not. These results will provide managers with a fuller understanding of the ecological roles of the conifer woodlands, resulting in more informed management decisions on the ground.

This project aligns with Regis University's mission in regards to environmental responsibility. I believe that testing data on land use activities, climate change and grazing activities from Rocky Mountain Juniper, Pinyon Pine and mixture forests and sagebrush area would benefit future forest fire management strategies and protect sage grouse habitat effectivenely. Conducted in a university setting, this project's results can be reference material of southwestern American forest fire study.

Purpose and Specific Aims

The objective of this project is to quantify stand composition, structure and vegetation dynamics of Rocky Mountain juniper and pinyon pine at lower slopes of mountains and foothills in the Gunnison Basin of western Colorado. I seek to determine whether these conifer forest is expanding its range into sagebrush area at low elevations, and if these dynamics are related to fire regimes, grazing, or other anthropogenic activities.

Hypotheses:

I. Human land use management alters fire return intervals, resulting in changing forest structure and composition.

I expect that land-use history has reduced the frequency of fires by putting out a fire quickly. This lack of fire may have enabled increased recruitment of Rocky Mountain juniper and pinyon pine at low elevations.

II. Rocky Mountain juniper and pinyon pine forest is expanding into non-forest vegetation areas at low elevation because of climate change.

I expect that more and more Rocky Mountain juniper and pinyon pine grow at lower elevation non-forest areas due to less available water and harsh climate at upper elevation. Thus, current forest fire regimes are changed since more conifer forest has established in lower elevation non-forest area.

III. Grazing activities accelerate Rocky Mountain juniper and pinyon pine expand into non-forest vegetation area.

I expect that grazing activities reduce the density of grass land and create more open land for these conifer trees. Livestock become vehicles to transport conifer seeds from forest to non-forest area, resulting these tree invading sagebrush area. This helps forest fire regimes expand into historical non-forest area.

Method

Study sites will be selected across the sagebrush communities and conifer woodland transitional stages in Gunnison County, Colorado. Currently, there are seven known sage grouse breeding areas in Colorado. The largest one is in the Gunnison Basin, and smaller satellite populations are located throughout southwestern Colorado. I will locate suitable study sites in a geographic information system (GIS) using satellite imagery and vegetation maps. I will then locate five study sites in the Gunnison basin and five additional sites in suitable satellite populations. Juniper and pine density, height and canopy cover will be measured in each sites. Belt transects will be established from forest to sagebrush in each study site. The number, length and width of transects will vary according to tree density and the pattern of tree distribution at each site. I will set up two to five belt transects with length from 15 to 50 m and width from 3 to 10m (Taylor, 1990). All juniper and pine will be counted and separated into 4 classes; height: juvenile <30 cm, sapling 30 cm to 3 m, mature > 3 m (Miller & Rose 1999). Both tree and shrub canopy cover will be estimated and recorded into 6 cover classes; percent cover: <1, 1-5, 6-10, 10-20, 21-35, >35 (Miller & Rose 1999). Increment cores will be extracted from all trees with DBH (diameter at breast height) greater than 2.0 cm within the transects (Taylor, 1990). Ages of all trees will be determined by counting their annual growth rings, and each year's juniper and pine density will be calculated based on growth ring results. Finally, the number of standing dead trees, stumps, and downed logs will be also recorded for each plot transects.

I will use fire reports from fire history databases to estimate the date of effective fire suppression at my study area. According to Vale (1981), date of effective fire suppression can be estimated by a diagram of the ratio of total acreage burned in a year to maximum annual acreage burned for the fire record. I will also use tree scars to identify fire history in study areas. Miller (1999) pointed that examining fire history using juniper is difficult because the thin barked tree is susceptible to fires, particularly within the first 50 years of growth. Thus, fire history will be inferred from fire scares in fire-tolerant pine (Miller & Rose 1999). This will provide evidence to test hypothesis I.

Annual, spring and summer precipitation and temperature data will be obtained from weather stations to study the relationships of climate variability and conifer tree establishment. A regression model will be established between precipitation and number of trees, temperature and number of trees year by year and different seasons. Differences of juniper and pine abundance between wet and dry years will also be analyzed to see how precipitation affects conifer forest growth. I will use dummy model (spring=1, summer=2) to test how precipitation under different temperature influence conifer forest growth. If there is juniper and pine abundance difference between wet and dry years, then the hypothesis II will be accepted.

There may be no direct and detailed data on livestock use at study sites. I therefore will use regional grazing trends data which have the number of sheep and cattle permitted to graze within a regional allotment. However, these numbers may be substantial in my study time period. So, governmental reports and correspondence, and local accounts of settlement will be examined to determine relative numbers of stock. Relationships of animal stock and juniper and pine abundance will be studied to see if livestock number affects conifer forest distribution regimes. This will help to test hypothesis III.

Work plan

Fire historical records, sage grouse habitat areas records, grazing activities data, and temperature and perception data will be collected in summer and fall 2017 from public databases. Field work will happen during the late summer and early fall in 2017. The days that my advisor and I go to Gunnison County will be based on the weather situation. After field work, I will analyze tree cores and scars at Regis University. I will start to do data statistical analysis and draw a GIS map at the end of August or at the beginning of September. Once finish data analysis, I will write a paper to discuss results and contribute my paper to a journal.

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URSC project budget justification table

Name(s): Panmei Jiang

Project Title: Conifer invasion of Gunnison sage grouse (*Centrocerus minimus*) habitat

Items ***(Please itemize amounts below)***	**Description**	**Funds requested from URSC**	**Funds requested from other sources**	**Source of other funds**
Supplies				
Fuel	750 miles = approximately 50 gal. x $3/gal	150		
Food	Meals for five people for one week	500		
Research assistants				
Undergraduate student stipends	3 students @ $100	0	300	Advisor's
Others				
Lodging	Camping at BLM campgrounds ($20/night)	140		
Total URSC Request		790		

Faculty Advisor name: John Sakulich

Faculty Advisor signature and date: John Sakulich 17 April 2017

URSC project budget justification narrative

Please describe **each** item you listed in the budget table. The description should enable reviewers to understand a) how the cost of each item was computed, and b) how the budget items relate to your project objectives.

Supplies:

The Biology Department already owns the necessary field equipment for this project. The main expenses will be travel costs for the fieldwork. Roundtrip travel to Montrose, Colorado (in the center of the study area) is approximately 600 miles. Additional driving between various study sites will also be necessary. Using a fuel calculator 750 miles of diving equates to approximately 50 gallons of gasoline. At $3 per gallon, this will cost $150. Food for a field crew of five people will cost approximately $500.

Research assistant(s):

Three undergraduate students will be hired to work in the field. We will pay their transportation, food and lodging costs and pay them a small stipend of $100. Collecting vegetation data in plots is labor intensive, so field assistants are necessary

Other

We will camp in campgrounds near the study areas. This is the most economical and efficient way to stay in the field. There is a fee charged at campgrounds managed by the BLM ($20 per night).

TOTAL AMOUNT REQUESTED FROM URSC: $ 790

Application to current coursework

This project not only improves my former training in field forestry and lab techniques but is also in line with my current graduate coursework at Regis University. Field sampling techniques I have learnt in Forest and Vegetation Ecology and Management class and Field Ecology Lab will assist me to core trees in limber pine forests. GIS, which I will learn on the coming spring semester, will prepare me to study limber pine forests structure and composition, and fire regime shifting. Analysis methods that I have learned in Biostatistics will be used to establish regression models to test hypotheses. Grant writing class enhances my writing skills which were valuable for this grant proposal and future paper writing.

CHAPTER 3. JOURNAL MANUSCRIPT: VEGETATION DISTURBANCE REGIMES AND CONIFER SUCCESSION IN GUNNISON SAGE GROUSE (*CENTROCERUS MINIMUS*) HABITAT

Abstract

The major reason for Gunnison sage-grouse decline in southwestern Colorado is thought to be loss and fragmentation of its sagebrush habitat by pinyon and juniper encroachment. This encroachment could be driven by both anthropogenic disturbance and natural climate variation. This project determines these potential drivers of Rocky Mountain juniper and pinyon pine forest expansion into sagebrush area at low elevations in Gunnison County, Colorado. Fire, grazing, and other anthropogenic activities was assessed to explain juniper and pine forest invasion patterns on sagebrush steppe. I hypothesize that recent land use, climate variability, over-grazing, and fire will drive pinyon or juniper tree invasion of sagebrush. Field work done in late summer and early fall 2017 collected data on the establishment dates of trees invading sagebrush. Juniper and pine forest structure, density and lower treeline elevation, and fire history were analyzed using tree cores and forest inventory data. Climate, fire history and livestock data were extracted from published database and literature. We found that the tree establishment in the mid-and later- twentieth century (post 1970s) was caused by wet and cool growth season after the great drought period in the 1950s. The outcome of this project will help land managers to understand current lower treeline conifer condition, and decide whether current tree removal treatment is necessary to improve sage-grouse habitat.

Introduction

Anthropogenic activities have altered disturbance regimes in the arid forest and rangeland ecosystem of the southwestern U.S. Extensive expansion of pinyon-juniper woodland forest coincided with the onset of livestock grazing and fire exclusion during the late 1800s and early 1900s, when Euro-American settlers arrived in this region (Romme et al., 2009). These coincidences support the idea that Euro-American settlers' activities, namely reduction in the frequency of extensive surface fires and onset of heavy grazing, caused an increase in tree density. However, in many locales, anthropogenic activities are less well-documented so conclusively linking these changes in stand regimes to vegetation changes was not possible. In addition, there were two very wet periods in the southwest, which coincided with the beginning of grazing and fire exclusion (Romme et al., 2009). Since these factors interact with each other, the precise mechanisms driving recent changes in vegetation structure and composition are not well understood.

Extensive infill and expansion of trees has happened in sagebrush of the Great Basin since the mid-19th century (Romme et al., 2009). Sagebrush (*Artemisia tridentata*) is a major food source and provides breeding habitat for sage-grouse (*Centrocercus minimus*). The loss of sagebrush to forest encroachment disrupts sage-grouse populations. For example, before 1860, only 16-67% of current woodland stands were present across northwest Utah, central Nevada, southwest Idaho and southeast Oregon; after 1860, the area occupied by trees has increased 150-625% (Miller et al., 2008). Because sage-grouse is an endangered species, conservation and restoration of sagebrush are urgently by needed in order to protect sage-grouse. Some local forest management strategies involve cutting down conifer trees to create open areas for sagebrush to recover (U.S. Department of the Interior Bureau of Land Management, 2017). A challenge facing the management of sagebrush communities by removing conifers is that disturbance regimes and succession in pinyon-juniper woodlands are not well understood. A comprehensive understanding of how natural and anthropogenic disturbance factors impact sage-grouse habitat decline is needed to provide management strategies to protect sage-grouse habitats.

During the past 150 years, tree density and canopy coverage has increased in pinyon-juniper woodland (Romme et al., 2009). However, the factors that drive conifer forest expansion remain unclear. Factors hypothesized to be responsible for plant community change have included alterations in fire regimes, excessive livestock grazing, proliferation of non-native plant species and conversion of rangeland to seeded pastures, cropland and roads (Braun, 1995). Thus, the reasons for conifer forest infill and expansion are difficult to understand because of many co-occurring factors. In addition, these factors impact vegetation dynamics differently in locations, like pinyon-juniper woodlands, pinyon-juniper savannas, and wooded shrublands. Anthropogenic activities such as grazing and fire suppression may drive the expansion of the Rocky Mountain juniper (*Juniperus scopulorum*) and pinyon pine (*Pinus edulis*) into sagebrush communities in Gunnison County, Colorado. Nevertheless, other locations have not changed or declined after such anthropogenic changes. For example, tree age structures of woodland stands in the Great Basin indicated a gradual shift from substantial increases in pinyon and juniper through the middle of the twentieth century to relatively limited establishment at the end of the century (Miller et al., 2008). For example, lower elevation conifer species expanded their range throughout the Holocene (the past ~12000 years), when climate became warm and moist (Romme et al., 2009). Therefore, understanding of the disturbance history and patterns of vegetation dynamics is crucial for informing management strategies aimed at preserving sage-grouse habitat.

One of the most common disturbances in the sagebrush habitat is fire. Historical fire type or a occurrence intervals may influence stand structure in pinyon-juniper woodland. Pinyons and junipers are fire-intolerant species because of their thin bark and typically low crowns. In the past, fire intervals in most pinyon-juniper woodlands exceeded 400 years (Romme et al., 2009). However, after the mid-1980s, large and severe fires occurred in these woodlands with shorter fire intervals (Westerling et al., 2006). For example, the burned proportion of pinyon-juniper woodlands during the decades between 1995 and 2005 was greater than the previous 200 years at Mesa Verde, Colorado (Floyd et al., 2004). These increasing trends in large fire frequency and total area burned have been implicated in pinyon-juniper woodland extension in some regimes (Romme et al., 2009). Nevertheless, some fire behavior may not change woodland structure. For example, a low-intensity surface fire has a limited role in changing pinyon and juniper landscape because this kind of fire can spread via fine fuels without killing dominant trees or shrubs (Romme et al., 2009). Additionally, in some events, some fire-promoting species have expanded due to recent broad-scale environmental changes, such as warming temperature. This may lead to shorter fire intervals. Hence, fire exclusion may not be the principal mechanism responsible for infill of woodland. But, understanding the current fire regime and fire severity is essential to predict pinyon-juniper infill in sagebrush habitat.

In addition to wildfire, livestock grazing also disturbs vegetation structure and dynamics, and causes forests to expand their ranges into non-woodland vegetation. In the southwestern United States, two mechanisms may contribute to the effect of grazing on tree infill: (1) tree seedlings have higher survival because heavy grazing reduces herbaceous competition with seedlings; and (2) shrub density and coverage increase after heavy grazing, and act as nurse plants for tree seedlings (Soulé et al., 2004). Once forest density and tree biomass are increased, forest fire frequency is changed because grazing reduces the abundance of grass fuels and fuel connectivity across the landscape (Sakulich & Taylor, 2007). However, the precise reasons for forest invasion into sage-grouse habitat are difficult to link to grazing because of multiple post-disturbance trajectories. For example, livestock grazing can have negative or positive impacts on sage-grouse habitat depending on the timing and intensity of grazing, and which habitat element is being considered (Crawford et al., 2004). Livestock grazing could increase sagebrush cover by excluding other understory herbaceous species. In later summer, excessive grazing may damage shrubs and cause deterioration of recovery. Thus, the impact of grazing has not been fully studied in the southwest United States because the focus has been on the more noticeable effects of logging and fire suppression (Belsky & Blumenthal, 1997). Additionally, the grazing effects could vary by soil type within different climate zones (Romme et al., 2009). Since the empirical evidence for grazing effects is sparse, a study examining the influence of its effect on tree line variation is necessary. In the Rocky Mountain conifer forest of the southwestern United States, such studies are especially urgent where vegetation and structure change may fragment and disturb wildlife habitats, such as those of sage-grouse.

Except for grazing, woodland expansion to lower elevations can also be driven by the effects of climate change because tree growth and regeneration are controlled by a region's temperature and precipitation (Grace et al., 2002). The onset of extensive infill and expansion of pinyon and juniper coincided with the end of Little Ice Age in the late 19th century, therefore, current tree expansion and contraction may have been a normal part of climatically driven fluctuations in woodland density (Romme et al., 2009). In the late 1500s, there was widespread tree mortality during a megadrought (Shinneman & Baker, 2009). After that, savannas and wooded shrublands were dominated by trees again when a warmer and moister climate returned. Data sets from desert and semidesert areas of southern New Mexico and Arizona showed that the twentieth century winter precipitation was high in areas where woody plants occupied areas previously dominated by herbaceous vegetation (Brown et al., 1997). Additional evidence that changes in density are climatically driven is that tree density and coverage have continued to increase both on grazed and ungrazed sites (Soulé et al., 2004). These findings support the idea that recent pinyon and juniper infill patterns may be caused by natural variation in climate, rather than anthropogenic disturbances. Since these factors are not exclusive of one another and different drivers may be more important in different regions, a study of how woodland structure and composition change as climate varies is worthy of our attention.

In this study, stand composition, structure, and disturbance history of Rocky Mountain juniper and pinyon pine woodlands are quantified to determine whether the conifer forest is expanding its range into sagebrush shrubland, and to evaluate possible drivers of these vegetation dynamics. Dendrochronology is used to analyze tree growth and forest structure in old forest. Based on the foregoing, we hypothesize that: (1) Rocky Mountain juniper and pinyon pine forest is expanding into non-forest areas. (2) Grazing activities accelerate Rocky Mountain juniper and pinyon pine expansion into non-forest areas, and (3) Human land use management lengthens fire returns intervals. We expect that grazing activities reduce competition from grass and create more opening for conifers to establish in shrubland. We also expect that land-use history has reduced the frequency of fires due to fire suppression. The lack of fire may have enabled increased recruitment of Rocky Mountain juniper and pinyon pine at the study area. We evaluate patterns of tree establishment to determine whether conifer establishment coincides with changes in climate and disturbance (fire or grazing activity).

Methods

Study area

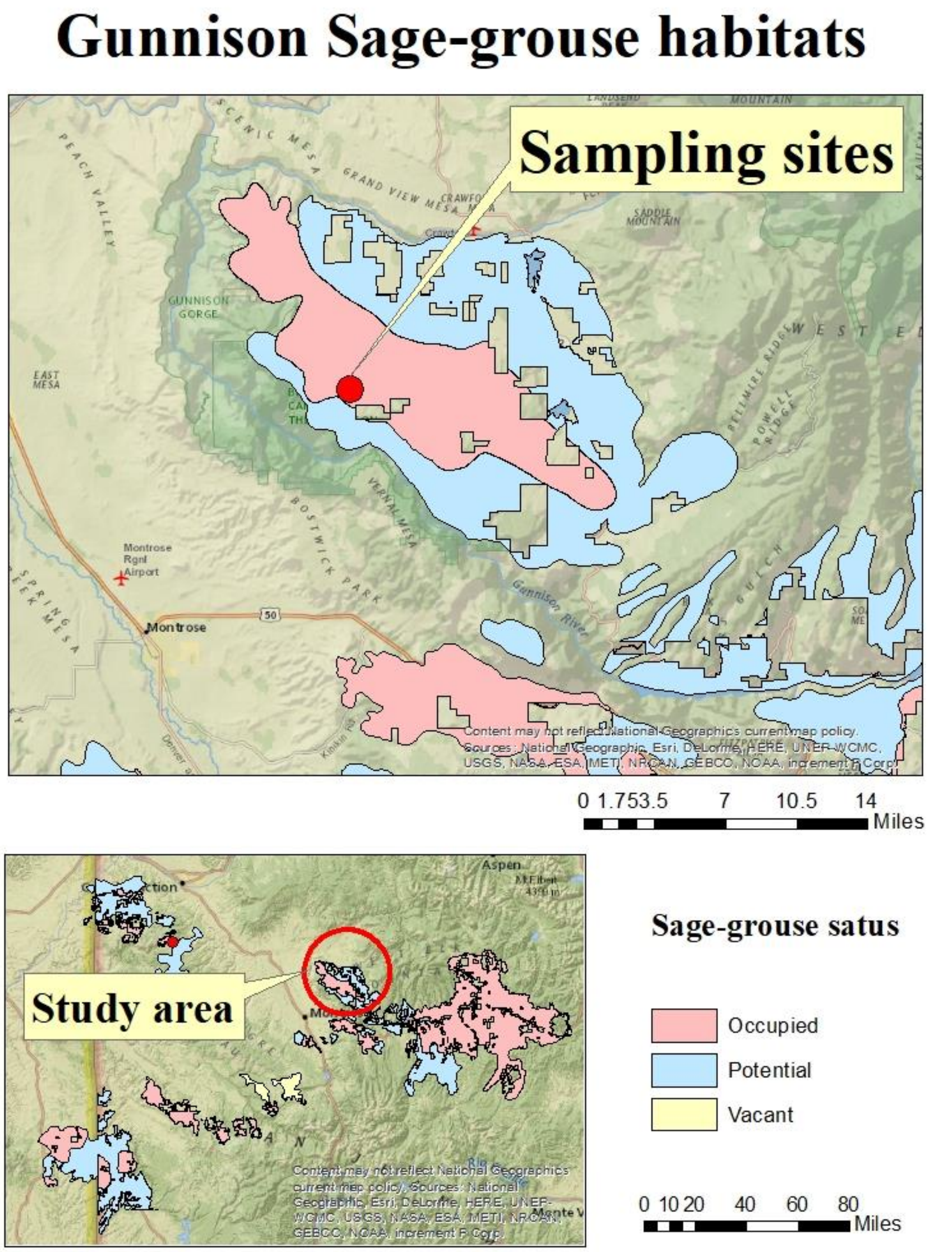

Figure 1. Sage-grouse habitat at Gunnison County. The red circled area is the region where study sites were established.

The study site is located on Fruitland Mesa in Montrose County, Colorado (from 38.647691N, 107.778548W to 38.648289N, 107.754020W). This region is characterized by the Gunnison uplift, which formed during the Laramide orogeny and is a transition zone between the Southern Rocky Mountains on the east and the Colorado Plateaus on the west. (Hansen & Peterman 1968; Hansen 1965). This study site represents one of the seven known Gunnison sage-grouse breeding areas in Colorado (Figure 1). Soils in the study area are shallow, well-drained loams derived from sandstone and shale alluvium, which support pinyon-juniper conifer woodlands and shrublands. In the study area, Rocky Mountain juniper (*Juniperus scopulorum*) and pinyon pine (*Pinus edulis*) are the dominant canopy tree species and sagebrush (*Artemisia tridentata*) is the major shrub species in the wooded shrubland. The property is managed by the federal Bureau of Land Management, and this agency has instituted tree removal treatments to preserve sagebrush cover and protect sage-grouse habitat by restoring sagebrush. Study sites with young trees that appear to be invading shrubland were selected in regions across the sagebrush communities and conifer woodland transitional stages. An additional study site with mature trees and old-growth forest structure was established on a nearby ridge around 4 km away.

The climate in the study area is defined as a continental montane climate. Monthly temperature ranges from an average minimum of -7.2° F in January to an average maximum of 80.8° F in July (Western Regional Climate Center, period of record: 1/1/1900 to 12/31/2005). Mean monthly precipitation ranges from 0.58 inches in November to 1.49 inches in August (Western Regional climate center, period of record: 1/1/1900 to 12/31/2005). Annual snowfall approximates 50.5 inches according to the Western Regional Climate Center give year of source record from January, 1900 to December, 2005.

Wildfires and livestock grazing are two major anthropogenic disturbances that affect lower elevation vegetation structure in the study area. Although fire scars were found in pinyon and juniper woodland, no major fires were recorded after the Gunnison National Forest in 1905, and no fire has been recorded since the time of permanent settlement in the area in the 1870s (Ken Holzinger, personal communication).

Field methods for vegetation dynamics

To reconstruct disturbance history and quantify patterns of conifer establishment, data on vegetation structure and composition of pinyon-juniper wooded shrublands were collected in two 50m by 50m sampling plots in October of 2017. The plots were established throughout areas of sagebrush shrublands believed to have been recently invaded by conifers based on the BLM's record. The plots were established in areas identified by local BLM ecologists. Most of the trees are small with stem diameters less than 20 cm at base. Ten tree increment cores were collected from each species in each plot. Trees were all cored at a height of 30 cm above the ground. Then, inventory was done to record each tree's basal diameter, canopy height, decay degree and life status. All juniper and pine were counted and separated into 3 classes based on their height: seedling <30 cm, sapling 30 cm to 3 m, mature > 3 m (Miller & Rose 1999). Both tree species' canopies were estimated and recorded into 5 classes: Dominant, Co-dominant, intermediate, sub-canopy, suppressed. The degree of decay was divided into 6 categories: 1= needless present, 2= branchless, 3= all bark, 4= bark is more than 50%, 5= bark is less than 50%, 6= no bark. For life status, 4 levels were designated: live, dead, cut stump, dead down. To estimate the time it takes for trees to reach coring height (30 cm), saplings and seedlings were cut down randomly to get upper (30 cm above ground) and lower (at ground surface) basal cross-sections. Trees of young plots were cored to establish tree age structure as well.

Field methods for disturbance history

To analyze disturbance history of the study area, I developed a tree-ring chronology of conifer tree growth by coring old-growth trees in a nearby remnant stand. This chronology was used to quantify tree annual growth rate and disturbance. Sixteen pinyon pines and six junipers were targeted to age in their old growth morphology. Trees with old growth morphology were selected. Since these old-growth trees have large diameters, two roughly orthogonal tree cores were collected from each tree without crossing the center of tree. For each core, tree bark was saved to make sure that there were no missing rings in recent growth years.

Lab methods

In the laboratory, cores were dried, placed onto mounts, sanded to a high polish so that ring structure of wood was clearly visible under an optical microscope. Cores were analyzed using standard techniques (Stokes & Smiley, 1968). Rings were counted to assign tree ages and their width were measured by using a sliding stage micrometer. The program COFECHA was used to control tree-ring measurement, synchronization, dating and quality (Grissino-Mayer, 2001). COFECHA calculates correlation coefficients between overlapping segments of each core and the remaining cores in the data set. The segment length to calculate a correlation coefficient was 50 years of series with 25-year overlapping periods against the master chronology. Segments are flagged by COFECHA if their correlation coefficients fall below a threshold of statistical significance (A = correlation under 0.3281 but highest as dated; B = correlation higher at other than dated position). Segments flagged by COFECHA were re-examined under the microscope to check for crossdating errors.

Tree age structure was established based on cores of trees, stems of saplings and seedlings and basal diameter of tree inventory data from young plots. Tree’s age was represented as tree ring number. If the tree cores did not contain the pith, estimated missed rings number would be added to the present ring number in the increment cores. The number of missing rings in each increment core was estimated by a transparent concentric ring template (Petruccelli et al., 2014). Tree age was classified using 20-year classes to analyze the temporal pattern of tree establishment. Tree basal diameter data were grouped into 5-cm classes to develop current stand structures. The Kolmogorov-Smirnov test (KS test) was used to compare whether overall shape of pinyon and juniper distributions were similar or not (Lopes, 2011). The number of rings were counted from basal cross sections and at 30 cm height from stems of saplings and seedlings. These stems' ring numbers were used to adjust tree ages as an age-to-coring height correction factor.

A tree ring index was computed using cores collected from old growth trees. The techniques for processing and crossdating were the same as for the cores from the stand structure plots. After crossdating, cores were detrended using the detrend function in the dplR package in R version (Bunn, 2008; Bunn, 2010 ; R core team, 2016-03-16). A cubic smoothing spline with a flexibility of 50 years was fit to each raw measurement time series, and index values were calculated by dividing ring-width values by predicted values (from the spline). This results in a unitless index with a mean of 1.0, where years with wide rings have an index greater than one and narrow years have an index less than one. The detrended series were then combined into a single standardized tree-ring chronology using the chron function in dplR.

Analyses

The influence of climate variables on tree growth was analyzed using treeclim package in R (Zang & Biondi, 2015). Monthly maximum and mean temperature, precipitation, drought for Colorado climate division 2 were downloaded from the National Climatic Data Center (NOAA, 2017). I then tested significance of correlations between climate variables and the tree-ring chronology, from June to December of the year prior to growth and from January to September of the current growing year. The significant months were picked to set up a multiple regression model of climate response analysis. Residuals of tree growth and the predicted value from the model in each year were used to analyze years with unusually larger or smaller growth rates than predicted from climate variations alone.

The chronology was further investigated to detect stand disturbances. The TRADER package in R was used to determine the date of release events (Altman et al., 2014). When disturbances such as fire kill understory trees, surviving overstory trees experience a temporary increase in growth rate resulting from the loss of nearby competitors. This technology reconstructed disturbance events based on raw measurement data of old tree cores. Implementation of radial-growth averaging criteria was computed by function growthAveragingAll (Altman et al., 2014). In this method, average radial growth over the preceding 10-years period were calculated. The minimum thresholds used for release were set up in 25% growth change for moderate and more than 50% for major release. Years after 1700 with major release frequency more than 25% were examined to see whether fire or grazing activities occurred during the same time period.

Results

Stand structure

Stand structure plots in the pinyon-juniper shrubland are dominated by small trees with low canopy heights (Figure 2). Most trees were suppressed or subcanopy, and a few individuals emerged from this low canopy into dominant positions.

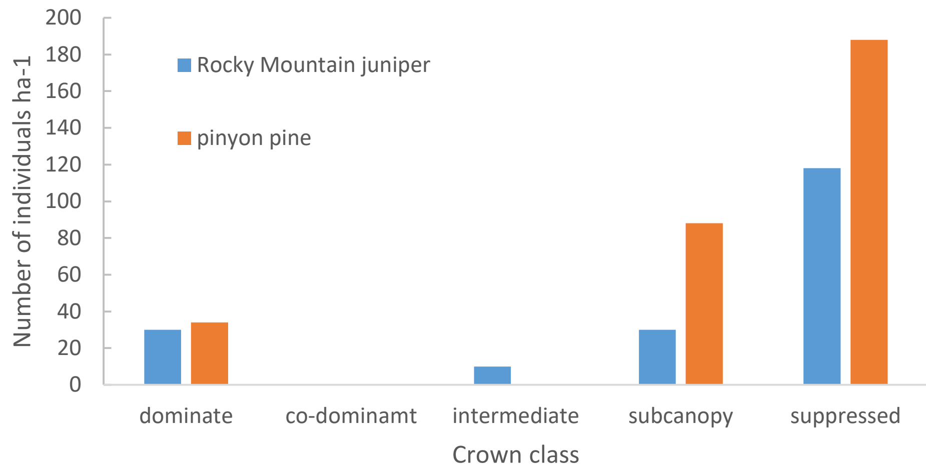

Figure 2. Crown class distributions per hectare. The suppressed group have the largest tree individual number. The second big group is subcanopy. This distribution indicated that research areas are dominated by small trees with low canopy heights.

76.6% of Rocky Mountain juniper and 88.4% of pinyon pine were small with basal diameters less than 20 cm (Figure 3). 10.6 % of Rocky Mountain junipers and 0.6 % of pinyon pine had basal diameters exceeding 50 cm. Both species had similar size class distributions (KS test: D = 0.10309, p-value = 0.5628) with many small trees and a few individuals reaching larger diameters.

Most trees in the plots established in the mid- to late- twentieth century, and the age structure pattern of pine and juniper are marginally similar (KS test: D = 0.17345, p-value = 0.0489) (Figure 4). From 1917, a succession of juniper tree establishment crossed in the recent century, followed by pinyon pine. Pine establishment reaches a peak in the 1957-1977 period. For juniper, the highest establishment time period is from 1957 to 1997. The oldest tree sampled in stand structure plots is juniper, which is between 280 and 300 years old. The next oldest tree species is juniper as well with an inner date between 180 and 200 years old. Pine establishment was continuous from 1837 through 1897. After this period, there was no new pine establishment until 1937. From 1737 to 1817, no new trees established in the study area.

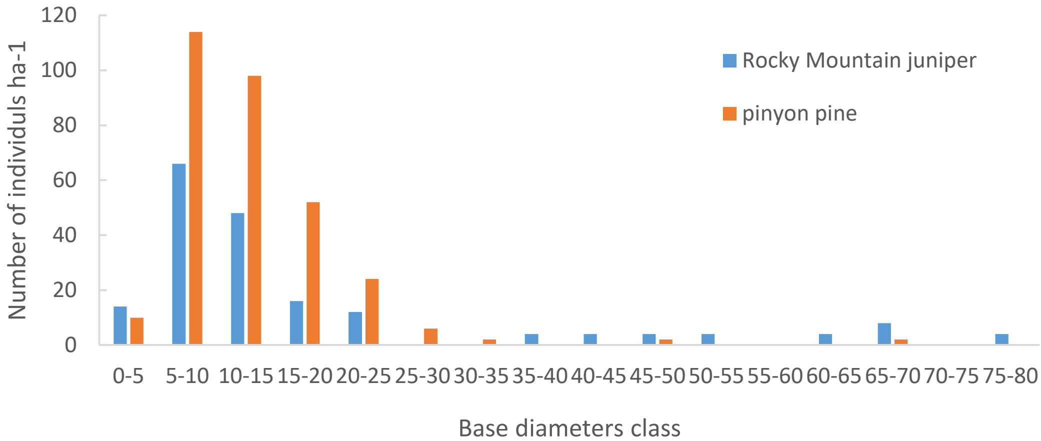

Figure 3. Basal diameter size class distribution. Both species have many young trees and fewer old trees. Most of trees have a basal diameter less than 20 cm.

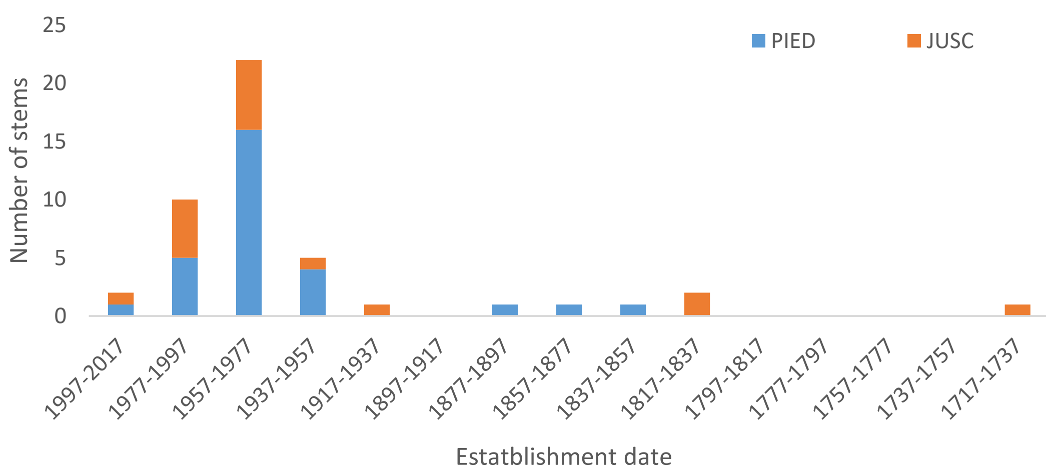

Figure 4. Age structure. Every twenty years combines as one group. Most of trees were established in the mid- and late- 20th century. From 1957 to 1977 period, tree establishment rate reached its peak.

Dead trees in study areas were only 3.5% (nine individuals) of total. The large majority of dead trees were junipers (eight individuals, 88.9% of all dead) rather than pines (one individual, 11.1%). The only dead pine was small with 11.4 cm basal diameter (Table 1). The basal diameter distribution of dead juniper was bimodal. 62.5% of dead junipers were small (basal diameter, Mean $\pm$SE= 16.4 $\pm$7.3 cm). The rest of the dead junipers had a large basal diameter with average 65.6 $\pm$ 9.9 cm. Most dead trees (89% total) were in the late stages of decay and are classified in bark less than 50% or no bark categories. All dead trees with large basal diameters (> 50 cm) are in advanced decay condition with little or no bark (Table 1). One pine with all bark was found with a small basal diameter (11.4 cm).

Table 1. Dead tree distribution. Juniper has more dead trees with late stages of decay. Most dead trees had in small basal diameters. Only three dead junipers had basal diameter larger than 50 cm.

plot	tree#	species	base D(cm)	Status	Decay
C1	99	Rocky Mountain juniper	58	dead	no bark
C2	1	Rocky Mountain juniper	55.8	dead	< 50% bark
C2	2	Rocky Mountain juniper	73.9	dead	< 50% bark
C2	3	Rocky Mountain juniper	19	dead	no bark
C2	4	Rocky Mountain juniper	12	dead	no bark
C2	5	pinyon pine	11.4	dead	all bark
C2	23	Rocky Mountain juniper	12	dead	no bark
C2	51	Rocky Mountain juniper	10.8	dead	< 50% bark
C2	66	Rocky Mountain juniper	28.2	dead	no bark

Tree-ring chronologies

A well-replicated tree-ring chronology for pinyon pine was developed in a remnant old-growth stand near the stand structure plots (Figure 5). I successfully produced a 436-year tree-ring chronology extending from 1581 to 2017 from 15 tree core series selected from 44 tree core samples. Only series 8 during the period from 1725 and 1774 was marked flag B, which means that the correlation was not high. The remaining core serious had a higher correlation (Series intercorrelation 0.729). After re−examination of the flagged segment, we determined that these specimens were free of crossdating errors. The average interseries correlation for the samples (n=15) was 0.692 and the mean sensitivity was 0.295 of the 161 segments tested using COFECHA.

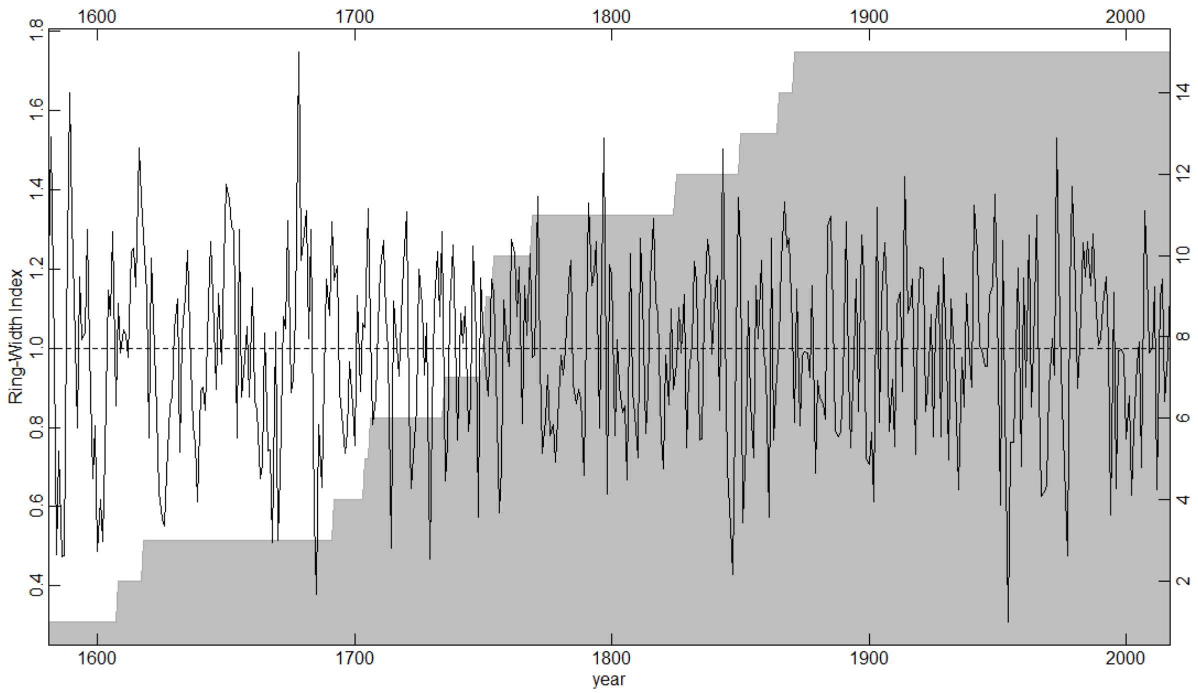

Figure 5. Residual tree-ring chronology and number of samples for pinyon pine from an old-growth forest. The index was calculated from 15 series of annual ring-width measurements. Sample depth is indicated by gray shading. Ring-width indices have a mean of 1.0; values larger than 1.0 are wide rings, and values less than 1.0 are narrow rings.

The largest negative index value post-1700 (where sample depth is sufficient to assess extreme values) occurred in 1954. This coincides with one of the most severe modern-era droughts in the western United States (Woodhouse et al., 2010).

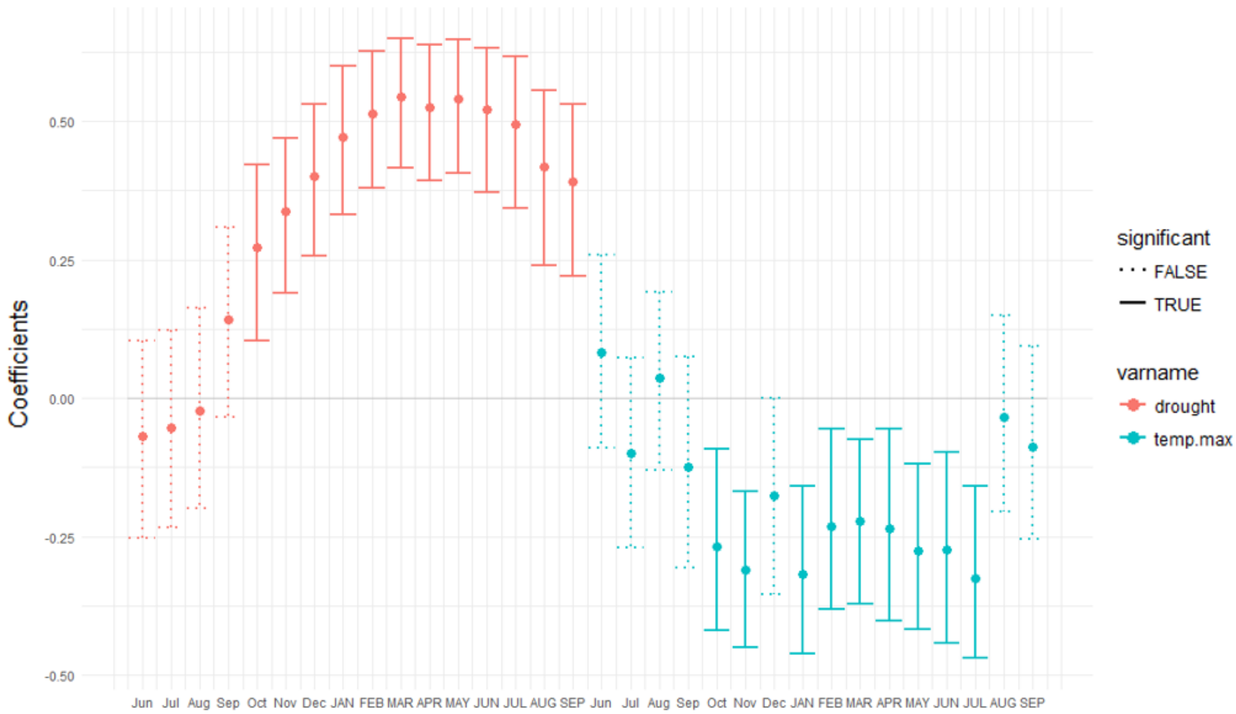

Figure 6. Climate response analysis. The current year's months are capitalized in x-axis. The figure indicated that growing season drought and temperature had significant negative influences on annual tree growth. PDSI was used as a drought index. Low PDSI means drought while wet year shave a high PDSI value. This figure also presented that seasonal drought had a positive impact on tree growth.

Climate-Growth Relationships

Current year Palmer Drought Severity Index (PDSI) is positively correlated with tree growth, while monthly temperatures during the growing season are negatively correlated with annual growth. These correlations indicate that tree growth was enhanced when moisture stress is lower during the growing season. From previous year October to current year September, lack of drought positively correlated with annual tree growth (Figure 6). The monthly maximum temperature of previous year October and November, and the first six months of the current year have a significant negative correlation with annual tree growth (Figure 6). The absolute value of residuals in 1954 are bigger (residual is -0.500688 $\pm$ 0.1624443, p-value=1.914*10^$^{-07}$), which means that growth in that year was significantly lower than that predicted from drought severity and temperature alone (Figure 7).

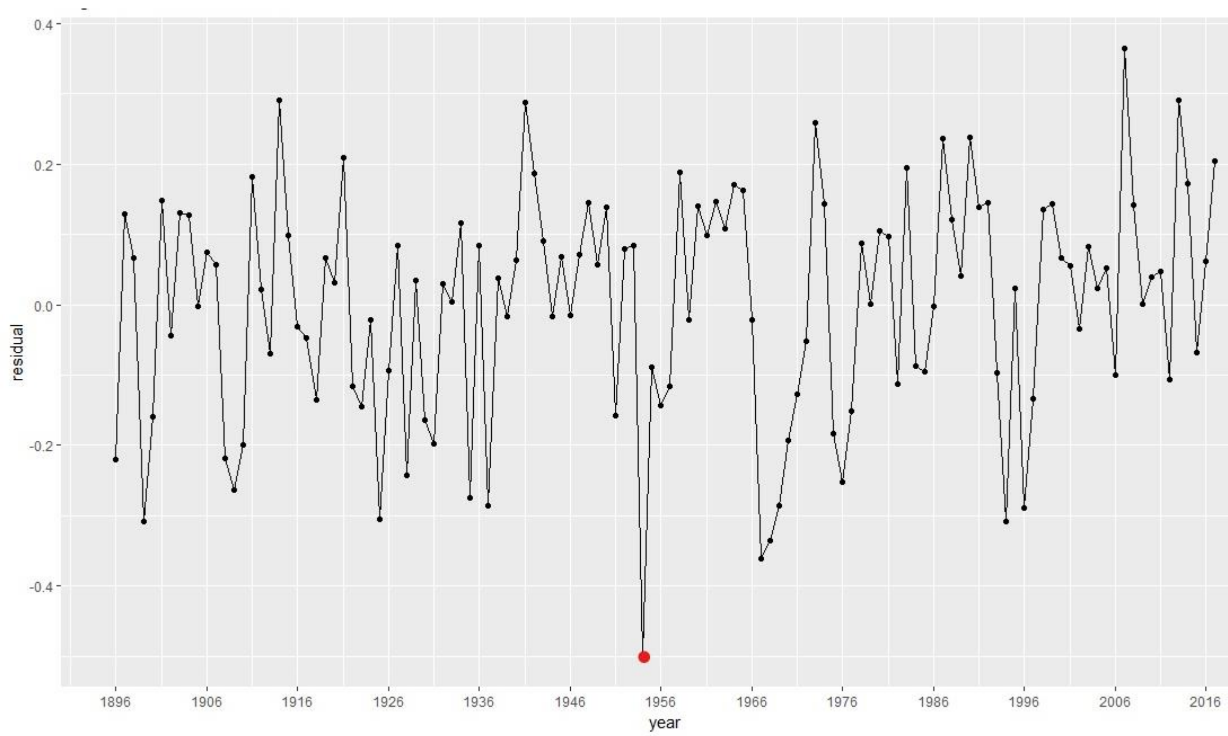

Figure 8. Residuals of tree ring width from the modeled climate response. The absolute value of residuals of some years are bigger, which means that non-climatic disturbance may have had a major influence on tree growth in these years.

Since only four years met the disturbance events analysis threshold (major releases >25%) in the late twentieth century (1902, 1937, 1957, 1978), drought condition in the study area was checked to detect whether late twentieth century's tree establishment was also influenced by climate. A pulse of tree establishment in the 1960s and 1970s coincides with the end of the 1950s drought and a return of wet conditions (Figure 8). Thus, drought may be an important driver of tree establishment patterns in this woodland.

Table 2. Tree ring analysis of disturbance events. Released years are selected after 1700. Number of major releases are bigger than 25%. This analysis indicated the years that disturbance events happened and impacted tree growth.

All Releases Year	All Releases Frequency	Number Of All Releases	Number Of All Trees
1700	25	1	4
1701	25	1	4
1717	33.33	2	6
1759	50	5	10
1780	27.27	3	11
1789	54.55	6	11
1810	27.27	3	11
1864	30.77	4	13
1883	33.33	5	15
1902	42.86	6	14
1937	40	6	15
1957	40	6	15
1978	40	6	15
2006	33.33	5	15

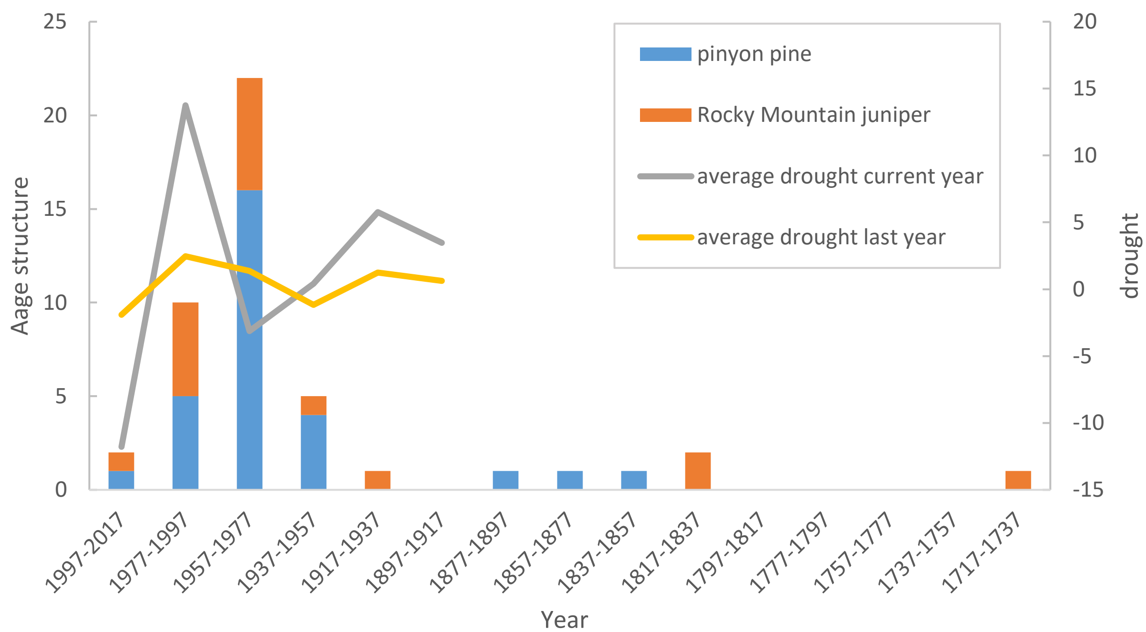

Figure 8. Age structure of sample plots in 20-year age classes. Most trees established in the second half of the 20th century. The largest pulse of tree regeneration coincided with an wet decade in 1970s, which was after a “megadrought” period in 1950s .

Discussion

The biggest rate of tree establishment in our southwestern Colorado studying plots occurred in the mid- and late- twentieth century. Only four years had disturbance events which were big enough to change vegetation composition. Tree established in the 1970s after a large drought period in the 1950s. Thus, drought variation is the major reason that conifer trees established at low elevation in southwestern Colorado. Rather than this effect of climate, anthropogenic changes in vegetation have been supported by other studies in southwestern US (Allen et al., 1998; Lenihan et al., 2003; Swetnam & Betancourt 1990). In conclusion, the climate variation played a major role to cause sage-grouse habitats loss, in comparison to by the disturbance events of fire and grazing.

Most conifer establishment in sagebrush shrubland occurred during the mid to late twentieth century beginning a shift in vegetation type from shrubland to woodland. A few of the oldest trees began establishing in the stand structure plots after 1917. Then, tree establishment reached a peak between 1957 and 1977. Most trees in the plots were small, with basal diameters ranging from 5 to 10cm, indicating that trees establishing in sagebrush shrubland are young. The stand structure plots were representative of most of the upland area of Fruitland Mesa, where sagebrush is mixed with small diameter pinyon pine and Rocky Mountain juniper. My age and size structure data suggest that the widespread succession of shrubland to forest is a recent occurrence, with the majority of trees establishing in the late twentieth century. The similar patterns of tree establishment pattern over twentieth century was also found in other places, such as Marshall Forest Preserve in the Southern Appalachian Mountains, Georgia (Jacobs, 2008; Petruccelli et al., 2014).

In order to study the tree establishment patterns in my study areas, I established a chronology from 1581 to 2017 which was used to assess relationships to climate and disturbance. A few areas of the study region consisted of ridges with steep slopes. These ridges were vegetated with mature, closed canopy, old-growth pinyon-juniper forest. The tree-ring chronology was used to develop a tree growth index for future work on climate response analysis. In addition, these raw measurement data of tree ring chronology were used to understand the effect of disturbance events on tree growth. For example, climate data is lacking for the year of 1789 AD, but tree ring width was larger than expected based upon climatic relationships which may be the result of a disturbance event.

Annual growth of pinyon pine in this study area is limited by growing season moisture. The climate response concluded that tree growth was limited by growing season moisture because monthly maximum temperature negatively correlates and the PDSI positively correlates with tree growth index. Years of greater than average tree growth are years with high precipitation and low temperatures during the spring and summer months. The extreme drought of the past 400 years has had pervasive effects on tree establishment (Swetnam & Betancourt, 1998). The 1950s was a megadrought period which was broken by wetter cool seasons in 1976 in the Southwest (Stahle et al., 2007; Swetnam & Betancourt, 1998). This wet period coincided with peak tree establishment. The same vegetation dynamic pattern has also been found in other species in the Southwestern USA in previous work in ponderosa pine forests in Arizona (Savage et al., 1996). This analyses explains how current drought seasons benefit conifer trees as conifers outcompete shrubs. The dormant seeds of trees have stronger resprouting ability than grass or shrub seeds (Higgins et al., 2000). Once the climate turns wetter after drought season, tree stem growth rate reaches a maximum where trees begin to infill shrubland (Higgins et al., 2000).

Consequently, these conifer trees occupy areas that are previously treeless. However, in our study several trees established from 1817 to 1897. Since climate data were recorded not until 1987, my disturbance analysis assessed examined whether tree extension and infill was caused by climate variation or disturbance.

Fire regimes in pinyon-juniper woodlands are typically characterized by long fire return intervals and large, stand replacing fires (Romme et al., 2009). In my analysis of disturbance, a widespread growth release (rapid increase in annual growth) event in 1789 may have resulted from a large fire that initiated the development of this large patch of sagebrush. This disturbance release analysis indicated that 1789 has the biggest major release frequency after 1700. A major fire may have killed small trees in the area now occupied by sagebrush and now reverting to woodland. However, the old-growth patch growing on the nearby isolated ridge may have been on the edge of that fire allowing canopy trees to survive and experience a growth release following the fire-induced mortality of competing understory trees. In addition, the El Niño of 1788–96 as the most dramatic in the world (Grove, 2006; Ward et al., 2014). Several ENSO events that occurred before 1880 had effects at least as intense and wide ranging as those associated with the current event (Grove, 1998). Other studies in the Southwest US found that fires were significantly related to phases of ENSO because the ENSO caused severe droughts (Sibold &Veblen 2006; Ward et al., 2014). Since settlement and livestock grazing did not begin until 1870, wildfire may be the driver behind the observed increase in growth rate.

No historical records of wildfire in this area are known since the time of Euro-American settlement in 1870. There are some basal fire scars visible on dead trees around the study area, but I did not collect cross-sections because of time constraints and lack of proper permits during field work. Future research should seek to reconstruct past fire occurrence using fire-scarred trees to determine whether growth releases in the old-growth stand correspond with fire dates. Although we found several fire-scarred trees in our study field, there is no official record of fire history. In order to provide more additional evidence in support of the hypothesis that fire causes earlier conifer forest expansion and infill of shrubland, a fire scar analysis should be used to develop a fire history.

In summary, conifer encroachment into sage-grouse habitat is related to a regeneration pulse, and as the major cause of sage-grouse habitats loss in southwest US in the late 20th century. A wet period after the 1950s severe drought season drove tree seeds to sprout and occupy non-woodland areas. In this condition, removal of small trees where sagebrush is present may improve grouse habitat. However, this strategy would not work in stands of closed-canopy pinyon-juniper forest that are more than 400 years old. Since their understory lacks sagebrush, tree removal treatment will unlikely produce suitable sage-grouse habitat. Another factor commonly considered for driving sagebrush habitat loss was disturbance. A major disturbance in 1789 may have initiated tree establishment in non-woodland areas. However, large-scale disturbances seem to be infrequent according to the tree-ring chronology. Thus, the pattern of infrequent disturbance will be difficult to implement with management actions because large, stand-replacing fires have damaged sage-grouse habitat for decades to centuries and this is incompatible with the goal of preserving key habitat over the short term. Although sagebrush protection restoration management and research efforts go back as far as the 1930s, breeding populations of sage-grouse have declined by at least 17-47% (Girard 1937; Connelly et al., 2000). This is because the tree removal, prescribed fires and grazing management results in complex and varied ecosystem conditions. Hence, a successful method in one site may not be suitable in other sites. Since sage-grouse populations are closely allied with sagebrush vegetation, understanding vegetation dynamic composition and factors driving conifer forests expansion into shrubland benefits land managers to develop more efficiency conservation and restoration strategies for sagebrush habitat. This research also benefits to conservation organizations, ranchers and gas drillers to developed management at conservation scale.

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CHAPTER 4. ENVIRONMENTAL STAKEHOLDER ANALYSIS: STAKEHOLDER INVOLVEMENT IN GUNNISON SAGE GROUSE (*CENTROCERUS MINIMUS*) HABITAT PROTECT: A CASE STUDY OF CONIFER INFILL AND EXPANSION INTO PREVIOUSLY NON-WOODLAND

Introduction

Sagebrush (*Artemisia tridentata*) is a major food source and breeding habitat for Gunnison sage grouse (*Centrocerus minimus*), which is federally listed endangered species. Sagebrush areas are lost to forest which encroachment disrupts sage grouse populations. Pinon-juniper woodland tree density and canopy coverage has increased in many areas during the past 150 years, so pinon-juniper forest infills and expands into shrublands (Romme et al 2009). In order to protect sage grouse, a tree removal treatment has become a common method of conservation and restoration in sagebrush habitat (Egan and Johnson Basin Restoration Project 2013). The challenge with this method of management is that disturbance regimes and succession in woodlands are not well understood. Factors driving conifer forest expansion remain unclear because of many co-occurring factors, such as alterations in fire regime, excessive livestock grazing, proliferation of non-native plant species, conversion of rangeland to seeded pastures, cropland and roads, and other land alterations (Braun 1995). Generally, a science study is an efficiency way to understand how vegetation dynamics mechanism works, and then executive management strategies should be developed to decide which kind of industrial activities should be stopped and whether a prescribed fire or tree manual removal treatment should be continued or not. Without enough research data, tree removal treatment is a common execution currently which works well in low canopy density forest with sagebrush in the under story (Scholes & Archer 1997). If the area is occupied by sage grouse, grazing or gas drilling should be forbidden in these places. However, some sage grouse habitats are on private land. Since many land owners prefer woodland over shrubland, educating land owners about how important it is to protect sage grouse combined with government compensation for maintaining sagebrush habitats would encourage landowners to conserve these sage brush habitats (Innes et al 1998).

One of the hypotheses for conifer forest infill and expansion is that anthropogenic activities altered the landscape. Since the late 1800s and early 1900s, when Euro-American settlers migrated to the southwest, their activities such as livestock grazing and fire exclusion, have altered disturbance regimes in the arid forest and rangeland ecosystem of the southwestern U.S (Romme et al 2009). However, an anthropogenic cause may not completely explain increased tree densities in many savannas because human activities were less well documented in many areas (Romme et al 2009). There were two very wet periods in the southwest in the 20th century, during which some have suggested that climate variability caused the observed forest infill and expansion (Grace et al 2002). Furthermore, anthropogenic and climatologic factors may interact with each other to cause changes in vegetation structure and composition.

Disturbance events are another potential driver of pinon-juniper forest expansion into areas which were previously non-woodlands. One major disturbance type is wild fire. Pinon and juniper are fire-intolerant species because of their thin bark and typically low crowns, hence historical fire type, intervals, and rotations are considered a reason to influence stand structure. After the mid-1980s, large and severe fires occurred in southwestern woodlands with shorter fire intervals, which burned a larger proportion of pinon-juniper woodlands during the past few decades (Westerling et al 2006). However, low-intensity surface fire has a limited impact on the pinion and juniper landscapes because this type of fire does not kill dominant trees or shrubs (Romme et al 2009). Along with wildfire, livestock grazing also disturbs vegetation structure and dynamics, and may cause forests to expand their ranges into non-woodland areas (Boyd et al 2014). In the southwestern United States, two mechanisms may be involved in the processes through which grazing causes extensive tree infill: tree seedlings have higher survival because heavy grazing reduces herbaceous competition with seedlings. Another one is that shrubs density and coverage are increased after heavy grazing, which creates nurse plants for tree seedlings (Soulé et al 2004). However, the precise reasons for forest invasion into sage grouse habitat are difficult to link to grazing because of multiple post-disturbance trajectories. The impact of grazing has not been fully studied in the southwest United States because of a focus more on the more noticeable effects of logging and fire suppression (Belsky & Blumenthal 1997). Besides, grazing effects could be variable across different soil types under different climate zone (Romme et al 2009). Hence, a study should be done to understand the mechanisms by which a reduced frequency of extensive surface fires and an increase in heavy grazing may cause increased tree density.

Another important factor to be considered is the effects of global warming and climate change. The onset of extensive infill and expansion of pinion and juniper coincided with the end of the Little Ice Age in the late 19th century. Therefore, a hypothesis has been developed that current tree expansion and contraction may have been a normal part of climatically driven fluctuations in woodland density (Romme et al 2009). Strong support for this hypothesis is the fact that tree density and coverage have continued to increase both on grazed and ungrazed sites (Soulé et al 2004). These pieces of evidence support the explanation that pinion and juniper infill patterns in the late 21th century may be caused by a natural biogeographical process rather than unnatural expansion. Thus, it is worth studying climate to understand how these factors control low elevation conifer forests abundance and structure in established in Rocky Mountain conifer forest in the southwestern United States.

Stakeholders

The main industrial stakeholder are oil and gas drillers and ranchers. Drillers build roads or well pads during their production activities causing vegetation fragment. Ranchers want to continued access to range land without additional restrictions, hence they are also interested how much of the region have conservation restrictions in the future. Their activities influence vegetation composition and structure, especially in dry areas (Reynolds et al 1992). For example, almost 20 years ago, many local ranchers in Gunnison Basin have changed grazing management practices, fenced riparian areas, and added conservation easements totaling > 40000 acres (Knapp et al 2013). This resulted in approximately 4000 sage grouse with numbers within the Gunnison Basin remaining stable-to– increasing (Knapp et al 2013). Thus, drillers and ranchers would be concerned about any areas that would be limited access or forbidden access. The ranchers and drillers would be interested in reading any research talking about wheatear their activities driving current landscape vegetation variation.

For sage grouse protect programs, conservation organizations are considered as directly-related stakeholders. Conservation organizations, such as the Audubon society, that are concerned with the conservation of birds, are interested in which region would be designed as sagebrush conservation or restoration habitats (Kress, 2006). The Western Association of Fish and Wildlife Agencies (WAFWA) has a goal to maintain and enhance population and distribution of sage grouse (Connelly et al 2004). Thus the WAFWA is interested in habitat conservation and land use, habitat restoration, science, data management and information, regulatory mechanisms, and integration and coordination across range and jurisdictions. They established North American Sagebrush Ecosystem Conservation Act (NASECA) (Connelly et al 2004). In this action, the WAFWA pointed out that all adaptive management models need significant commitment and rigorous application of technique, in this way the land management would use management at conservation scale. Therefore, the data analysis of how human activities and climate disturbance disadvantage or advantage sage brush restoration or conservation benefit conservation organization's working goals. They are interested in what management will do to prevent conifer tree infill and expansion into brushland.

Nearby private land owners of all professions are stakeholders in sage-grouse related project. They mostly want woodland rather than shrubland on their property. Thus they do not want grouse on their land because it restricts their land use. In addition, many private land owners prefer trees because they provide shade and privacy for their houses. Therefore, a sagebrush vegetation dynamic study could provide evidence whether it is necessary to set up a sage grouse habitat in a targeted areas or not. This also could find a balance point to fix the conflict issue between conservation organizations and the private land owners and ranchers.

Recommendation

Design tree removal treatment

Tree removal is commonly used to restore structure and function to these communities. In many areas of the southwestern United States, pinion and juniper are expanding and infilling in rangelands at an unprecedented rate, dominance of conifer alters fire regimes, decreases shrub and herbaceous cover, and diminishes wildlife habitat. Thus manual tree removal could reduce canopy fuel loads and increase understory cover, which will meet conservation organizations' profits. In addition, grazing and drilling activities would be forbidden because sage brush need time to recover. However, successional trajectories following disturbance are dependent on disturbance severity and residual species abundance, composition and resulting structure on site. A prerequisite to make this treatment come out successfully is that abundant sagebrush is present in the understory. After tree removal, if shrub and herbaceous cover are already at a lower percentage, the new open areas in the community may be replaced by invasive species. Therefore, tree removal treatment may not be an appropriate strategy for BLM faculties to fix conifer infill and expansion issues in every sites. In addition, this method might be difficult to be executed in private land if the lander owners prefer trees rather than shrubs. Thus in order to balance conflicts between private landowners and conservation organizations, tree removal treatment general executed in BLM managed areas.

Execute prescribed fire

Prescribed fire has an advantage of reducing woody fuel loads and eliminating most of the trees much better than mechanical methods. After prescribed fire, the fire initially increases annual and exotic forb cover and decreased tall grass cover, which would satisfy conservation organizations' goal: sage brush restoration. On wetter and cooler sites with high perennial grass cover and where sagebrush cover is not a priority, fire will enhance the perennial herbaceous community and reduce the risk of high-intensity fire. However, a disadvantage of this method is that prescribed fire has the liability of slow recovery of shrubs, especially sagebrush. Thus, both ecologists and land managers use former wildfire history as reference information to consider if prescribed fires are necessary. Although, new land could be opened for grasses or shrubs after prescribed fires, the ranchers potentially may not be allowed to graze because the grazing activities may cause non-wood vegetation population to decline and lead tree recovery. Since there was still conflicts between conservation organizations, private landowners, ranchers and gas drillers, this management were executed in regimes where are in charge of BLM. Or government would compensate private landowners when prescribed fire was processed in private lands.

Establish dendrochronology analysis

A dendrochronology study is a way to explore what disturbs tree growth rate in the past centuries. The tree growth analysis information would help ecologists understand factors that impact on tree establishment where tree growth and environmental factors are not well documented. Policy makers would develop environmental laws and BLM faculties would make field work guidelines based on the dendrochronology analysis result. Conservation organizations could use the data analysis to test whether past managements benefit sage grouse protection or not. For instance, if a new conifer forest establishes in a non-woodland area, their increment tree core would indicate what factors influence the trees' grow in rate. This would help researchers understand whether the current tree infill and expansion process is a normal natural occurrence evolution or not. Whether it is necessary for humans to interfere in the current landscape. And whether ranchers and driller should be reduced or forbidden their industrial activities. Thus, dendrochronology analysis aid ecologists, policy makers, and land managers to develop advanced land management strategies on conservation scale, which balance conservation organizations, industries and private landowners' profits.

Recommendation: use multiple methods together

Since every single method has its disadvantages, combining two or more above methods together may develop more efficient strategies to reduce conifer forest infill and expansion. It would be better to conduct a study to understand which factors drive tree establishment before taking other executive action. The reason is because if management executions were done directly without understanding working areas ecosystem patterns, more environmental issues would come later. The oil and gas drillers or livestock businessmen might be interested in the results of such a study, because this might give them a prediction whether their production activities would be limited or not in future. And then if most of the forest is composed of young stands with sagebrush under story, the tree removal treatment may be an appropriate strategy to restore sage grouse habitat. A prescribed fire execution may be effective in the forests that have many other grass or shrub species in the understory. Manual tree removal and prescribed fire also could be used in one program if these management sites already have declined sagebrush cover. In this condition, the burn management may kill other grasses or shrub species and give lower opportunities for invasive species growth after tree removal treatment. At the same time, cutting down seedlings would reduce their competition with sagebrush. In this way, the sage brush would have higher possibility of successful recovery. This would meet conservation organizations' profits of protect sage grouse habitat. Private landowners would benefit from understanding if it is necessary for them to do tree removal or prescribed fire treatment in their lands.

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