MODELING LONG-TERM CHANGES TO THE PONDEROSA PINE FORESTS OF GRAND CANYON NATIONAL PARK

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ABSTRACT

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Grand Canyon National Park's fire management program has utilized prescribed fire and wildland fire for over 20 years to reduce fuel loading and the potential for extreme fire behavior in order to obtain desired conditions of the forest. However, a better understanding of the long-term changes in stand structure and fuel characteristics following management activities is needed for the continued support of fire as a land management tool in Grand Canyon National Park. This study compares the long-term effects of several possible management scenarios across Grand Canyon National Park's ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) forests on the South Rim. The Fire and Fuels Extension (FFE) of the Forest Vegetation Simulator (FVS), a forest growth and yield model, was employed using data from Grand Canyon's fire effects monitoring program. Simulations were run in FFE-FVS using various management and regeneration scenarios to understand the possible trends in fuels characteristics, stand structure, and potential fire behavior for 50 years into the future. The scenarios included a no-treatment (control) scenario, an 8-year burn interval (the current fire management plan), a 16-year burn interval, and a scenario which alternated the burn interval between 8 and 16 years for the length of the simulation. We further simulated the natural variability of ponderosa pine regeneration in the Southwest by manually entering in low and high regeneration rates of the dominant tree species obtained from Grand Canyon's fire effects monitoring program data. These results have key management implications for Grand

Canyon's fire management program. The modeling results suggest that under a low regeneration rate burning every 8 years will produce stand structures, total surface fuel loadings, and potential fire behavior or fire effects that are within desired conditions for almost all of the variables we analyzed. The FFE-FVS modeling results also suggest that the long-term effects of the differing burn scenarios may be heavily dependent on the potential for high or low regeneration rates post-fire. Therefore, we recommend that a good management strategy be guided partly by the observed post-fire regeneration rates and we advocate for the use of all available information to guide the adaptive management cycle. Thus the alternating prescribed burn schedule may be a feasible option on the low post-fire regeneration sites. However, on the high post-fire regeneration sites we recommend that the 8-year burn interval will best achieve most management objectives or the pre-settlement reference conditions for these forests.

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Preface

Chapter two has been formatted for journal submission. This format may result in some redundancy throughout the chapters of this thesis. References have been placed at the end of each individual chapter.

Chapter One

Literature Review

Ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) forests in the Southwest have complex issues rooted in history and there are inherent challenges to sustainably manage and restore these forests. In order to address these challenges we need to understand the history of land management across the region. Past land management policies and perspectives have had long-lasting effects and shaped these forests into what they are today. Recognizing mistakes made in the past is the first step to developing better management strategies for the present and into the future. Therefore, this chapter reviews conditions of the ponderosa pine forests prior to Euro-American settlement and subsequent changes to these forests post-settlement. This chapter also reviews different management strategies currently practiced in ponderosa pine forests. Next, some background information is provided on the Forest Vegetation Simulator (FVS) and the Fire and Fuels Extension (FFE), a growth and yield model used for this project. Examples of current applications for FFE-FVS are given as well as limitations and weaknesses of the model. The work for this project was based in the ponderosa pine forests on the South Rim of Grand Canyon National Park. Thus, this literature review encompasses literature for both the South Rim specifically and the broader Southwest region to fully understand the depth of these topics.

Characteristics of southwestern ponderosa pine forests prior to Euro-American settlement

The historical range of variability (HRV) characterizes the normal fluctuations in ecosystem conditions or processes over time (Morgan et al. 1994). The HRV takes into account both the type of change and the rate of change (Morgan et al. 1994; Fulé et al.

1997). The HRV is particularly helpful in developing a useful point of reference for determining a range of desired future conditions, as well as to identify the limits of acceptable change in current ecosystem components and processes (Morgan et al. 1994). In the Southwest, many managers and ecologists agree that the pre-settlement era is a good benchmark in time to use as a guiding reference condition for management and restoration of ponderosa pine forests (Covington and Moore 1994a, 1994b; Moore et al. 1999; Fulé et al. 1997). Specifically, this refers to the pre-Euro-American settlement of the region which occurred in the late 1800's. Prior to Euro-American settlement of the area, the natural processes operated more or less uninhibited and ecological structures reflected both the recent evolutionary history and the dynamic relationship between these ecosystem components and disturbance patterns (Fulé et al. 1997). In short, the HRV as assessed from the pre-settlement era helps managers develop scientifically-based desired conditions for ponderosa pine forests.

Historical descriptions of southwestern ponderosa pine forests depict them as open, park-like stands with a frequent, low-severity surface fire regime maintaining and shaping the forests across the landscape (Covington and Moore 1994a; Moore et al. 1999). A thorough comparison of southwestern fire history data from Swetnam and Baisan (1996) highlights two main points regarding the intensity and severity of historical fires: 1) for ≥ 300-500 years prior to Euro-American settlement, southwestern ponderosa pine ecosystems historically experienced high-frequency, low-intensity surface fires; 2) high-intensity stand replacement fires were very rare, if they occurred at all. Surface fires were carried across these forests by a continuous grassy understory and by the long needle litter from the ponderosa pines. The frequent fires in these forests played a critical role in regulating tree density, species composition, amount and structure of dead biomass, and nutrient cycling in the dry, arid climate of the Southwest (Fulé et al. 1997). These frequent surface fires served to prepare localized seed-beds that were receptive to pine regeneration and regulated the amount of seedling survival which helped maintain the open, park-like stand structure (Bailey and Covington 2002).

The historical fire regime on the South Rim of Grand Canyon National Park has been quantified and described by analyzing written fire-records or by using dendroecological techniques such as fire-scar analysis. A study by Fulé et al. (2003b) quantified the pre-settlement mean fire return interval to be 6.9 years on their South Rim, Grandview study site. They were also able to show that the longest fire-free period during the pre-settlement era was 17 years. Other studies performed in ponderosa pine forests throughout the Southwest have reported similar fire return intervals in the range of ~2-25 years (Swetnam and Baisan 1996; Cooper 1960; Weaver 1951).

Many of these fires historically resulted from lightning strikes. There is also evidence that Native Americans across the Southwest set fire to some of their surrounding lands in order to drive game animals, promote areas for farming and/or gathering, and as a consequence of warfare in some cases (Pyne 1982). These fires are thought to contribute only slightly to the average fire return intervals, with the majority of fires being caused by lightning activity (Swetnam and Baisan 1996; Moore et al. 1999; Fulé et al. 2003a; National Park Service 2012). The timing, or seasonality, of a large percentage of these fires coincided with the summer monsoon weather pattern that passes through the southwest region of the United States primarily during the months of June through September. The convectional thunderstorms and the lightning strikes occurring across the region after spring winds had dried the fuels sufficiently were a key ignition source for many southwestern fires (Swetnam and Baisan 1996; National Park Service 2012). Fire seasonality on the South Rim was historically nearly evenly divided between spring fires (54% of recorded fires) and summer fires (46% of recorded fires) (Fulé et al. 2003b).

The historical stand structure of ponderosa pine forests has been reconstructed or described according to historical photographs, surveys, and through dendroecological reconstruction techniques. In a study performed on the South Rim by Fulé et al. (2002b) using dendroecological reconstruction techniques, they reported historical tree density ranging from 140 to 145 trees/ha and basal area ranging from 9.1 to 12.6 m^2/ha. They described these South Rim sites as historically dominated by few, large ponderosa pine trees with a distinct Gambel oak (*Quercus gambelii* Nutt.) component in the understory (Fulé et al. 2002b). In addition, according to an early survey of the forests around the time of settlement, Woolsey (1911) reported an average of 26 pine trees/ha (over 15.2 cm DBH) on the nearby Tusayan National Forest. And yet another reconstruction study done on the South Rim of the Grand Canyon reported an average tree density ranging from 93.8 to 176.3 trees/ha with an average basal area ranging from 10.6 to 20.3 m^2/ha (Fulé et al. 2002a). Similar numbers were found in nearby ponderosa pine forests as well. For example, an average of 148 trees/ha with an average basal area of 11.7 m^2/ha was reported in a ponderosa pine forest at Camp Navajo, Arizona, located south of Grand Canyon National Park (Fulé et al. 1997).

Post-settlement conditions in southwestern ponderosa pine forests

Over a hundred years of a strong fire suppression management policy, livestock grazing, and timber harvesting practices have increased the mean fire return interval (Covington and Moore 1994a, 1994b; Swetnam and Baisan 1996) and led to more dense, even-aged stands (White 1985; Bailey and Covington 2002) in southwestern ponderosa pine forests. The increase in tree densities was partly due to a decrease in competition as a result of reduced grass cover from grazing; this in conjunction with the absence of fire lead to high rates of seedling survival (Bailey and Covington 2002; Covington and Moore 1994a, 1994b). This shift to dense, even-aged stand structures was also partly caused by a "perfect storm" type of situation with extremely productive seed years which resulted from climate factors, further adding to a widespread regeneration event across the region (Schubert 1974). Previous research has shown that historical tree establishment ranged from about one to four trees/ha per decade over the 300 year time span before Euro-American settlement circa 1876 (Mast et al. 1999). This rate is several orders of magnitude below the hundreds to thousands of trees/ha which established post-settlement due to grazing and fire exclusion (Covington and Moore 1994b; Cooper 1960; Lang and Stewart 1910). After only three decades from the onset of the Euro-American settlement of northern Arizona, Lang and Stewart (1910) documented 268 new trees/ha, which was estimated to be a 337% increase in stand density from pre-settlement conditions. In a more recent study by Fulé et al. (1997), they showed that the stand density post-settlement on their sites had changed from 148 trees/ha to a more dense forest of about 1,265 trees/ha.

Also due to the absence of natural fires across the landscape, Fulé et al. (1997) reported that a species composition shift occurred on their site, as evidenced by a greater dominance post-settlement of Gambel oak and conifers less adapted to frequent fires, such as white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco). Similar findings have been supported by Savage and Mast (2005) on their study sites throughout Arizona and New Mexico.

In addition to the high rate of tree establishment and the resultant overly dense stand structures; ponderosa pine forests have also undergone changes in key ecosystem processes. The effects of removing the frequent, low-severity fire disturbance regime from these forests include: higher fuel loading, stagnated nutrient cycling, decreased decomposition rates, reduced tree growth and increased mortality (especially in the oldest trees), and overstocked patches of pole-sized trees and saplings (Covington and Moore 1994a, 1994b). In short, both the structure and the processes that historically occurred in these forests were disrupted during Euro-American settlement of the region.

Overall, the many changes that have occurred post-settlement in the ponderosa pine forests of the Southwest have lead to the increased susceptibility of these forests to both crown fires and insect or disease outbreaks (Allen et al. 2002; Fulé et al. 1997). The degraded stand structure of today's southwestern forests makes them much more prone to insect outbreaks and pathogens that spread easily in closed-stand conditions (Cooper 1960; Clark and Sampson 1995; Covington and Moore 1994b; Long and Shaw 2005). For example, root diseases such as *Armillaria* spp. (Stevenson) spread locally and result in an expansion of canopy gaps, therefore it follows that closed-stand conditions make it easier for these pathogens to spread and infect larger groups of trees creating even larger gaps across the forest (Castello et al. 1995). There have also been studies which have examined the relationship between increased relative stand density in ponderosa pine to an increase in susceptibility to attack by mountain pine beetle (*Dendroctonus ponderosae* Hopkins) (Long and Shaw 2005). While the exact mechanism underlying this density-susceptibility relationship is not yet fully understood, there is evidence for the effect to exist (Long and Shaw 2005; Negron and Popp 2004).

There also have been some studies performed which show that ponderosa pine is poorly adapted to stand replacing events or high severity crown fires (Savage and Mast 2005; Swetnam and Baisan 1996). By looking at the long-term effects and regeneration of the forest following high severity crown fires that occurred throughout the Southwest (fires occurring between 1948-1977 in Arizona and New Mexico), Savage and Mast (2005) were able to show that a species composition shift can occur following such fires. The two main outcomes defined by their study were: 1) abnormally dense ponderosa stands regenerated post-fire, resulting in further vulnerability to crown fires in the future, and 2) an eventual shift in forest type to an oak/manzanita (*Quercus* Nutt./*Arctostaphylos* Adans.) shrub field or grassland (Savage and Mast 2005). Furthermore, Savage and Mast (2005) were able to show that alternative stable states of this oak/manzanita shrub field could develop and persist for years after the crown fire. There are other examples in the literature which support this understanding that crown fires or stand-replacing events are thought to be rare in the pre-settlement ponderosa pine forests, further supporting that this species is poorly adapted to handling extreme events on a large scale and that these events could severely alter the successional trajectory of the forest (Covington and Moore 1994a, 1994b; Swetnam and Baisan 1996).

These examples of overly dense forests, altered fire regimes, shifts in species composition, and susceptibility to high severity wildfire or pathogen outbreaks outside of the HRV are worrisome issues for many managers. Fire and fuels managers today are now faced with these issues that are complicated and compounded because of past land management practices and their origins in the early post-settlement era. Managers today also acknowledge that they have the potential to further complicate these problems if the correct management decisions are not made now. The fire and fuels managers at Grand Canyon National Park are tasked with managing and/or restoring the ponderosa pine forests within the Park. All national parks are mandated by policy to preserve their natural ecosystems. The role of fire in Grand Canyon forests has long been recognized as one of the most important factors influencing the natural ecosystems here. And, one of their overall goals in the fire management plan is to restore and maintain park ecosystems in a natural, resilient condition (National Park Service 2012).

Different management strategies for the ponderosa pine forests of the Southwest

In order to restore historical forest structure and function to ponderosa pine forests, managers must remove the excess of small diameter trees, reduce the high surface fuel loadings, and reintroduce frequent low-severity surface fires back into the system. There are essentially three broad strategies currently being employed and experimented with across the Southwest to meet these objectives: 1) the use of fire alone, 2) mechanical treatments such as thinning or mastication, or 3) a combination of mechanical treatments and fire. All three strategies offer different ecological benefits, while also posing different costs or risks given the current conditions of the area to be restored.

Currently, the fire management program at Grand Canyon National Park is restricted to using fire alone across the majority of their forests. There is no mechanical thinning allowed within the Park except for the primary Wildland Urban Interface (WUI) management unit around Grand Canyon Village and the developed areas (National Park Service 2012). The WUI areas total 14,611 acres, which is only 1.22% of Park land (National Park Service 2012). Therefore, fires in the form of prescribed burns and wildfires managed for resource benefit are the primary land management tools used to work towards the desired future conditions across most of the South Rim ponderosa pine forest.

However, continued scientific support for the use of fire as the primary land management tool is essential in order to garner and maintain support for these methods employed by the fire management program. The public and many tourists visiting the Park often express a common misconception about fire in the forest, that all forest fires are bad. Educating the public and government officials about the proper use of fire as a land management tool in these ponderosa pine forests that historically evolved with frequent, low severity fires is critical to the long term success of the fire management program. A model like FFE-FVS can help foster public education and inform management decisions by visualizing and assessing the long-term effects that different management strategies can have on the forests. This can be accomplished by using the animations created with the Stand Visualization System in FVS, and by comparing the trends in variables of interest over time.

Some scientific studies have looked at the effectiveness of the management strategy of using fire alone. One such study was done on the North Rim of Grand Canyon by Fulé et al. (2004b). They assessed fire effects 6 years after a relatively intense prescribed fire had burned in the fall of 1993. They found that the average density (331 trees/ha) and basal area (28.5 m^2/ha) were reduced to levels similar to the reconstructed pre-settlement conditions (~ 246 trees/ha and 28.5 m^2/ha). Most of the mortality was found in the fire-susceptible species, especially white fir. Thus, the intense prescribed fire had also restored pre-settlement species composition by returning dominance to the ponderosa pine. Overall, Fulé et al. (2004b) concluded that more severe prescribed fires such as this example could restore the forest structure closer to the HRV. These structural changes could be achieved without the negative impacts attributed to thinning operations such as soil compaction, new roads, logging damage to surrounding trees, or the introduction of exotic species (Fulé et al. 2004b).

There is more support from different perspectives in the literature for the use of fire alone, especially fire of relatively high severity, in restoring ponderosa pine forests. The overly dense stand structure of southwestern forests are currently full of stagnated thickets of many small diameter trees, which likely established from the productive seed years of 1914 and 1919 (Schubert 1974). In the pre-settlement forest, the small diameter trees would have been young trees with thin bark and would have been easily girdled by the frequent, low-severity fires. In contrast, many of these small diameter trees in present-day forests are relatively old and can have thick bark because of the dense growing conditions (Sackett and Haase 1998; Sackett et al. 1996). Therefore, in such dense stands these small diameter trees can be difficult to kill with fire alone. Sackett and Haase (1998) observed resistance to mortality in these old, yet small diameter trees during prescribed burning. Instead, excessive heat to the crown (up to 75% crown scorch and/or consumption) was needed to effectively kill these small diameter trees (Sackett and Haase 1998; Sackett et al. 1996). However, it has also been reported that it is difficult to get prescribed burns to reach the lethal temperatures required to thin out these trees in dense stands due to the amount of shade and the related higher fuel moistures (Sackett et al. 1996). In addition, Sackett and Haase (1998) also observed that the lower layers of the forest floor in these dense stands did not burn well, rather only the top layer of litter was consumed entirely. All of this research supports the understanding that fire of higher severity may be needed to obtain the level of consumption required to decrease surface fuel loadings, increase mortality in the thickets of small diameter trees, and to achieve a stand structure that is closer to the HRV (Sackett and Haase 1998; Fulé et al. 2004b).

Others have argued that forest structure must first be restored through thinning before utilizing fire in order to most effectively restore these southwestern forests to the pre-settlement conditions (Covington et al. 2001; Sackett et al. 1996). A long-term study done near Flagstaff, Arizona comparing two sites undergoing a schedule of repeated low-severity burning found that pre-settlement conditions would be difficult to achieve using prescribed fire alone (Sackett et al. 1996). Instead, Sackett et al. (1996) recommended that a prescribed burn program should be supplemented by low-impact mechanical thinning in order to more closely achieve the pre-settlement structure. Other researchers in the Southwest have arrived at similar management recommendations (Covington et al. 2001; Harrington and Sackett 1990).

One of the biggest challenges facing land managers today is acknowledging the fact that the long term effects of our actions are often still not well understood or can be hard to predict. It is no different with Grand Canyon's fire management program. Grand Canyon managers have expressed that they would like scientific input about how to carry out the best management strategy for these ponderosa pine forests, ideally allowing the forests to function within their natural range of variability both now and into the future.

They have identified sections of the park as areas to either maintain or restore (National Park Service 2010). They want to visualize the long term effects of their management strategies on the land and to understand the potential changes to stand structure, fuels, and fire behavior characteristics. Models like FFE-FVS, while not perfect, can help with understanding the possible long-term effects of different management practices.

The Forest Vegetation Simulator and the Fire and Fuels Extension

The Forest Vegetation Simulator is a distance-independent, individual-tree based forest growth and yield model developed by the USDA Forest Service (Dixon 2002; Crookston and Dixon 2005). It is currently used by a variety of land management agencies across the United States including the USDA Forest Service, USDI Bureau of Land Management, and the USDI Bureau of Indian Affairs (Dixon 2002). It was originally developed in the northern Rocky Mountains, and then later expanded to now contain a national system of forest growth models maintained by the USDA Forest Service (Shaw 2009). Some of the uses of FVS modeling include silvicultural manipulations, wildlife habitat analyses, and fuel treatment evaluations. There have been several model extensions and post-processors built subsequently that have allowed for an even wider range of uses across many disciplines. These model extensions allow the user to model stand resistance and resilience in the face of anticipated disturbances due to insects, disease, and fire to name a few. The design of the FVS model interface allows the user to run a series of “what if” management scenarios to compare the long term effects to the growth and yield of the stands (Shaw 2009). These modeling results are intended to be treated as more of an educational tool rather than actual predictions of the future; the intent is to help users understand potential long-term effects that different management strategies can have to the land and how managers can achieve specific management objectives (Shaw 2009; Dixon 2002).

FVS is a descendant of the Prognosis Model which was first conceived by Al Stage in 1973 for the forests in the northern Rocky Mountains. Since then, the model has undergone dramatic development and included more complex processes and simulations to accommodate the national user base that is familiar with FVS today. The growth models have been continually refined for improved accuracy throughout the years. It also now includes more options for simulating management actions on a given stand, and the extensions to the model have allowed for more flexibility for today's land managers (Dixon 2002).

The minimum data required to run the FVS model include tree species, diameter, and the number of trees per acre. Other data such as tree height, canopy base height, and live crown ratio are good to collect in the field as well, although they can be estimated in FVS if these data are not available (Shaw 2009). Examples of output from FVS include trees per acre, the stand density index, basal area, size class distribution, and accretion or mortality within the stand (Dixon 2002). FVS was originally intended to work with the Forest Inventory and Analysis (FIA) data. These data requirements and much more are collected on all FIA plots across the country. This also happens to be the case with Grand Canyon's fire effects monitoring plots, i.e. all the essential data required to run plots as stands in FVS is collected on these plots.

The Fire and Fuels Extension (FFE) to the FVS model was built specifically to give fire and fuels managers a practical planning and/or evaluation tool (Reinhardt and Crookston 2003). Essentially FFE works with those predictions of stand dynamics created in FVS to further model fuels, potential fire behavior and potential fire effects within that stand. This is done internally within the program through linkages with fire effects and fire behavior models (Reinhardt and Crookston 2003). Examples of output from FFE include potential flame lengths, torching index, crowning index, potential smoke production of PM2.5 or PM10, surface fuel loading, soil heating estimates, and many other variables (Reinhardt and Crookston 2003).

Limitations and weaknesses of the FFE-FVS model

It is important to note that there are many inherent limitations and shortcomings to using a model like FFE-FVS. The FVS model is a statistical model so it lacks the process relationships to explore, for example, the potential effects of changing climate or a disturbance regime operating within an envelope of the natural range of variability (Fulé et al. 2004a). Although there has been a climate change extension recently built for use with FVS, simply using FFE-FVS alone (as we have done with this project) cannot address the potential effects of changing climate and related effects to growth of the stand. Therefore, without representing the stochastic nature of multiple changes to an environment into the future, an FFE-FVS user should acknowledge upfront that these models are simple representations of the real world. To reiterate an important fact, modeling results are not to be treated as accurate future predictions of what will happen, but rather should be viewed as an aid in the decision-making process through visualizing/assessing trends through time in response to various management scenarios.

Another weakness of the FFE-FVS model is the fact that the base model FVS simulates the growth and mortality of a stand on cycles of typically every 10 years. This is in contrast to the FFE, which simulates annually, or on a yearly cycle. This often leads to model behavior which is merely an artifact of working with both time steps within the model, and is not intended to represent a real event (Reinhardt and Crookston 2003). Discontinuous behavior related to pulses in the model such as regeneration, how much and when it is added to the stand, can also appear to tip the stand over a critical point/threshold at cycle boundaries with regards to things like torching index or crowning index (Reinhardt and Crookston 2003). Also, live fuels and non-tree vegetation such as herbaceous plants and shrubs, are poorly represented in the model and are only nominally represented as a fixed amount, depending on the percent cover and the dominant tree species present (Reinhardt and Crookston 2003; Crookston and Dixon 2005). In many fuel types, live fuels can contribute significantly to fire behavior.

FFE-FVS applications in research or management, and the accuracy of the model

FFE-FVS is widely used to evaluate different fuel treatment options for a given stand, to assess the potential long-term effects management can have on wildlife habitat, fuels, and potential fire behavior (Reinhardt and Crookston 2003). One example of a research application was carried out in a study performed by Fulé et al. (2004a) in Grand Canyon National Park. They used FFE-FVS to predict changes to canopy fuels and potential fire behavior across the park and through time (Fulé et al. 2004a). They applied detailed forest reconstruction data to initialize the FFE-FVS modeling starting in the year 1880, and spanning 160 years, they modeled changes in canopy biomass, canopy bulk density, species composition, and the crowning index to the year 2040. They were able to determine that the model was running relatively accurate, as they compared the model output in the year 2000 with the field data they collected in 1997 to 2001. The simulated densities and basal areas in 2000 were within 20% of the values they collected in the field (Fulé et al. 2004a).

The changes in stand dynamics and potential fire behavior varied across the landscape depending on the initial fuel type and forest conditions. The authors of this study concluded that that different management goals or strategies should be devised according to the forest type and elevation, as there are inherent differences in fire ecology and HRV for each vegetation group (Fulé et al. 2004a). Through modeling with FFE-FVS in a very similar location to our own study area in Grand Canyon National Park, they were able to quantify potential changes across the landscape through time. They identified areas that were susceptible to active crown fires in the Park, and compared these values to their historical conditions and the natural range of variability in these forest types when explaining the management implications of this study. FFE-FVS when used in such applications is a very powerful tool but like all models, it should be applied to management situations cautiously.

FFE-FVS can also be used for other purposes as well. In a study performed by Sorensen et al. (2011), they used FFE-FVS to simulate the effects of an intense wildfire compared to thinning and repeated prescribed burning on stand carbon storage in ponderosa pine forests across northern Arizona. They were able to identify different management options for carbon storage in these forests, while taking into account other activities such as logging or a severe wildfire that occurred once during the 100 years of the simulation.

The accuracy of the model has been tested in other regions as well. One study that looked closely at the accuracy of modeling results from the base FVS model was done by

Leites et al. (2009). They used the North Idaho (NI) variant and the South Central Oregon and Northeast California (SO) variants. The main goal of this study was to assess the accuracy of these two variants in predicting the crown ratio values and how the accuracy of those predictions would affect the diameter growth and basal area increment through time (Leites et al. 2009). When the crown ratio is not measured in the field, FVS estimates the crown ratio by using species-specific allometric relationships (Leites et al. 2009; Dixon 2002). Leites et al. (2009) found that the NI variant over-predicted the crown ratio when the measured values were below 40%, but it under-predicted crown ratio when the measured values were above 60%. While the SO variant tended to over-predict the crown ratio when measured values were less than 60% (Leites et al. 2009). These kinds of differences are why the FVS user's guide strongly recommends collecting crown ratio in the field, even though it is considered a highly subjective field measurement which could introduce error into the data (Dixon 2002; Leites et al. 2009).

In another study by Parresol and Stedman (2005) the height-diameter prediction equations were tested using the Southern variant of FVS on several eastern hardwood species. Using a large dataset (9,236 trees from 301 plots) they found the average bias for all species combined to be 4.2% and the prediction accuracy was 82.4% (Parresol and Stedman 2005). They identified a few species that appeared to be estimated especially poorly, such as American sycamore (*Plantanus occidentalis* L.), chinkapin oak (*Quercus muehlenbergii* Engelm.), and black locust (*Robinia pseudoacacia* L.) (Parresol and Stedman 2005). However, given the bias and prediction accuracy listed here, they are both within the accepted standards. This allows users to have confidence that FVS is predicting these height-diameter relationships and basic stand dynamics fairly accurately, at least for the Southern variant (Parresol and Stedman 2005).

Conclusion

This chapter described the pre-settlement conditions of southwestern ponderosa pine forests and subsequent changes to these forests post-settlement. As a consequence of these post-settlement changes to both the historical structure and processes, many southwestern ponderosa pine forests now exhibit overly dense, even-aged stand structures that are highly susceptible to insect and disease outbreaks and high severity wildfires (Allen et al. 2002; Clark and Sampson 1995; Cooper 1960). Different management strategies are currently practiced in ponderosa pine forests to restore historical forest structure, composition and disturbance processes. There are essentially three broad management strategies employed across the Southwest: 1) the use of fire alone, 2) mechanical treatments such as thinning or mastication, or 3) a combination of mechanical treatments and fire. All three strategies offer different ecological benefits, while also posing different costs or risks. Models like FFE-FVS can be used to evaluate different strategies by looking at a series of “what if” scenarios and their long-term effects. Applications of FFE-FVS include but are not limited to: aiding in the development of silvicultural prescriptions, fuel treatment evaluations, potential fire behavior and fire effects assessments, and carbon stand storage analysis.

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Chapter Two

Modeling long-term changes in the ponderosa pine forests of Grand Canyon National Park using the Forest Vegetation Simulator with the Fire and Fuels Extension

Introduction

Wildfires of increasing severity and extent are becoming more common in ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) forests in the southwestern United States compared to historical conditions, especially pre-Euro-American settlement in the late 1800's (Westerling et al. 2006; Savage and Mast 2005; Roccaforte et al. 2008). This is largely a result of warmer and earlier springs in recent years (Westerling et al. 2006) in conjunction with past management practices such as fire exclusion, livestock grazing, and selective timber harvesting (Covington and Moore 1994a, 1994b; Mast et al. 1999). Many southwestern ponderosa pine forests consequently now exhibit overly dense, even-aged stand structures that are highly susceptible to insect and disease outbreaks and high severity wildfires (Allen et al. 2002; Clark and Sampson 1995; Cooper 1960). Such fires can result in forest type conversions because ponderosa pine has not evolved with high severity fire historically (Swetnam and Baisan 1996; Savage and Mast 2005). Due to the degraded state of these southwestern ponderosa pine forests, managers today are charged with restoring forest structure and function through practices like thinning and prescribed fire.

Some managers, such as the fire and fuels managers at Grand Canyon National Park (GCNP) in northern Arizona, U.S.A., have experienced limitations in what they can do on public lands. Currently, management plans do not permit mechanical fuel reduction treatments anywhere besides the primary Wildland Urban Interface (WUI) management unit around Grand Canyon Village and the most developed areas (National Park Service 2012). The primary WUI areas total 14,611 acres, which is only 1.22% of Park land (National Park Service 2012). Therefore, fires in the form of prescribed burns and wildfires managed for resource benefit are the primary land management tools used to work towards the desired future conditions across most of the forested landscape within the Park. There are many reasons for not approving mechanical thinning fuel treatments, a few of which include the high cost of thinning operations (Lynch et al. 2000) and the potential for soil damage and/or compaction associated with the use of heavy equipment (Tiedemann et al. 2000). Despite the costs and potential for damage associated with thinning operations, it has been suggested that returning these forests to their historical range of variability (HRV) through ecological restoration treatments – often including thinning treatments for structural manipulation before returning fire to the system, is the most effective way to manage these forests (Allen et al. 2002; Covington et al. 2001; Sackett et al. 1996). Nonetheless, GCNP is restricted to using fire alone on the majority of the landscape. To date, approximately 88% (52,757 acres) of these ponderosa pine forests have been burned, and most of these acres have burned multiple times (up to three) due to the active prescribed burn program (National Park Service 2012).

To meet resource management goals, GCNP implemented a prescribed burn program in 1980 and has also managed wildfires for resource objectives since 1987. GCNP has maintained an active fire effects monitoring program since 1990. The fire effects monitoring program is intended to provide managers with a means to evaluate the effectiveness of the prescribed fire program and thus inform the need to adapt management practices (National Park Service 2003). A central goal of the fire management program at GCNP is to promote a science-based program that relies on current, best-available information and thus to implement adaptive management practices (National Park Service 2012).

The fire management plan (FMP) at GCNP is currently outlined to burn all their South Rim ponderosa pine burn units every 8 years (National Park Service 2012), which is consistent with the historical fire regime in these forests (Swetnam and Baisan 1996; Covington and Moore 1994a, 1994b; Fulé et al. 2003). As part of the adaptive management process, the Park is interested in exploring the long-term effects of the current management strategy on these forests. The Park is also interested in assessing potential alternatives or in making adjustments to the current FMP as new data becomes available and provides support for such changes. To achieve this, there needs to be a solid understanding regarding the effectiveness of prescribed fire in meeting management objectives for stand structure, fuels and potential fire behavior. Research addressing this need will help managers evaluate the effectiveness of established management practices and to also recognize new management strategies for meeting long term objectives. One way to answer this need is through modeling with a program like the Forest Vegetation Simulator (FVS) (Dixon 2002) using the Fire and Fuels Extension (FFE) (Reinhardt and Crookston 2003). By using this model with data from GCNP's fire effects monitoring program, we can answer important questions regarding GCNP's forests and make management recommendations based off the effectiveness of both current and proposed management strategies in the Park.

The Forest Vegetation Simulator is a distance-independent, individual-tree based forest growth and yield model developed by the USDA Forest Service (Dixon 2002;

Crookston and Dixon 2005). The design of the FVS model interface allows the user to run a series of "what if" management scenarios to compare the long term effects to the growth and yield of the stands (Shaw 2009). The Fire and Fuels Extension to the FVS model was built in light of many fuel treatment options used today to give fire and fuels managers a practical planning and evaluation tool based on current stand conditions (Reinhardt and Crookston 2003). Essentially FFE works with those predictions of stand dynamics created in FVS to further model fuels dynamics, potential fire behavior and fire effects within that stand. This is done internally within the program through linkages with fire effects and fire behavior models (Reinhardt and Crookston 2003).

The objectives of this study were to evaluate in FFE-FVS over 50 years how: 1) prescribed fire interval could affect predicted stand structure, fuel loading, potential fire behavior and fire effects; and 2) different regeneration rates could affect predicted stand structure, fuel loading, potential fire behavior and fire effects. Modeled results were compared to desired targets or thresholds for stand structure, fuel loading, potential fire behavior and fire effects variables to determine how well a particular fire interval might meet management objectives in the long-term. The different prescribed fire intervals were modeled under two contrasting regeneration scenarios to determine if certain fire management strategies might be appropriate for sites experiencing more or less regeneration. We hypothesized that the regeneration rate simulated would drive the results for all management scenarios, except for the most frequent burning interval. In other words, we hypothesized that the most frequent burning schedule would overcome any regeneration effects and still produce desirable conditions or close to pre-settlement reference conditions, even under high regeneration rates. Whereas with the longer prescribed fire intervals, we hypothesized that the ability to achieve desired conditions would depend on the regeneration rate.

Ponderosa pine regeneration in the Southwest is characterized as episodic and highly variable between years. This is partly due to the fact that good seed crops are produced only every 4 to 6 years, with little to no viable seed produced in intervening years. Other factors such as seed predation, climatic variability between years, or lack of receptive seed bed further limit the viability of ponderosa pine seedlings from year to year (Shepperd et al. 2006; Schubert 1974). Previous research has reported pre-settlement regeneration rates for ponderosa pine in northern Arizona to range from about one to four trees/ha per decade (Mast et al. 1999). However, after only three decades from the onset of settlement of northern Arizona, Lang and Stewart (1910) in a survey estimated a 337% increase in stand density from pre-settlement conditions. Such evidence points to the fact that regeneration rates can be strongly affected by different land management practices and concurrently we acknowledge that regeneration rates are likely to have significant impacts on the longevity of fuel treatments. Thus, we included the second objective of assessing different regeneration rates.

Several variables from the modeling output were compared through time to assess the potential long-term changes to stand structure, fuels characteristics, and potential fire behavior and fire effects (Table 2.1). The three stand structure variables that were explored include tree density, basal area, and the relative density index (RDI). Used in conjunction these variables lend an understanding of competition within the stand, availability of growing space, stand stocking, and overall health of the stand. The RDI is particularly helpful for understanding the trajectory of a stand through time because it is the stand density relative to the maximum possible stand density index (SDI). The SDI developed by Reineke (1933) is based on the inverse straight-line relationship between average DBH and the number of trees per acre. The index value from the SDI is the number of trees for stands that have an average DBH of 25 cm (Smith et al. 1997). Therefore, relative density can be summarized as the number of trees actually in a stand divided by the maximum number of trees of that average size that could exist (Smith et al. 1997).

We also evaluated fuel loading of litter and duff (data not shown), fine woody debris (fuels 0 to 3 inches in diameter), coarse woody debris (fuels > 3 inches in diameter), and the total surface fuel loading. We assessed potential fire behavior using the torching index and potential surface flame lengths under severe weather conditions, which provides fire managers with an easy way to identify potential scenarios that may pose resistance to control of fires. Lastly, to evaluate fire effects we looked at potential smoke production of particulate matter less than 2.5 microns in diameter (PM2.5) under severe weather conditions. Production of PM2.5 is particularly important for managers to evaluate, because these fine particles have many negative implications for human health (Nel 2005) and this is also regulated under the Clean Air Act. In addition, soil heating estimates during simulated prescribed fires were evaluated. Soil heating estimates can provide a picture of potential mortality and long-term affects to the soil itself resulting from prescribed fires.

Methods

Study area

Grand Canyon National Park is located on the southwestern edge of the Colorado Plateau in northern Arizona. The forests within GCNP are spread across both the north and south rims, with ponderosa pine forests occurring between elevations of 1,950 m to 2,600 m. There are important differences between the more mesic ponderosa pine forests found on the North Rim and the drier ponderosa pine forests on the South Rim. South Rim ponderosa pine forests have been described historically as being both structurally and functionally different than the North Rim ponderosa pine forests, due to lower tree densities and more frequent fire return intervals (Fulé et al. 2003). This study focuses on the lower elevation, drier forests which characterize the ponderosa pine forests found on the South Rim.

The ponderosa pine forests on the South Rim of the GCNP make up roughly 6.6% of land area in the park, which is approximately 60,000 acres (National Park Service 2010). Temperatures on the South Rim range from an average July maximum of 29.6° C to an average January minimum of -7.8° C (WRCC 2011, Grand Canyon NP2 weather station). Average annual precipitation is 41.0 cm and average annual snowfall is 111.3 cm (WRCC 2011, Grand Canyon NP2 weather station). Soils on the South Rim are classified as Lithic Arguistolls, Vertic Paleustalfs or Lithic Haplustalfs; which range in texture from loamy-skeletal to fine or clayey soils weathered from a calcareous sandstone parent material (Lindsay et al. 2003).

Study design

The GCNP fire effects monitoring program collects pre- and post-fire stand composition and structure data as well as the response of the stand and understory components to fire. These data are collected according to the standard protocols outlined in the Western Region Fire Monitoring Handbook (FMH) (National Park Service 2003). These data were used with the Central Rockies variant (Keyser and Dixon 2008) of FFE-FVS to assess different management scenarios and their effects on stand structure, fuel loading, and potential fire behavior 50 years into the future. The GCNP fire effects monitoring program has 113 FMH plots installed in different forest types across both the north and south rims.

In this study we focused on the pure ponderosa pine forest type on the South Rim. Pure ponderosa pine forest was defined by a 100% ponderosa pine species composition specifically in the pre-settlement trees. Pre-settlement trees were identified as having a DBH of approximately 40 cm or greater (White 1985, Mast et al. 1999). Within the ponderosa pine forest type, we focused on the plots that have burned three times at low severity, leaving 11 FMH plots to use for modeling. The average time between the first and second prescribed fires was 6.3 years and the average time between the second and third prescribed fires on these plots was 7.9 years. This shows the ability of GCNP to closely adhere to the 8-year burn interval outlined in the FMP (National Park Service 2012). These plots represent the condition that the Park would like to continue managing from their present state into the future, and they are interested in obtaining additional information about the effectiveness of the current management strategy of burning every 8 years.

The FMH plot visitation schedule includes a pre-treatment read right after plot installment is completed. After the plot burns in any type of fire an immediate post-fire read is done, and then the plot is read 1-year post-fire, 2-years post-fire, 5-years post-fire, and 10-years post-fire (National Park Service 2003). If the plot burns again at any point along this timeline, it re-sets the whole plot visitation schedule and it continues from the immediate post-fire read through time again. We focused on the point in time that most closely represented the current status of the 11 ponderosa pine FMH plots, the first-year read after the third fire.

Development of the modeling scenarios

Data was exported from the Park's FEAT/FIREMON Integrated (FFI) database to run simulations in FFE-FVS. See Appendix A for more details on this process. We defined the FVS management scenarios with the help of Grand Canyon National Park fire and resource management staff. A total of four management scenarios were used: 1) a control/no-treatment scenario; 2) an 8-year prescribed burning interval; 3) a 16-year prescribed burning interval and; 4) an alternating prescribed burn interval. The control scenario received no prescribed fire or other treatments, and simply grew the stands for 50 years. The 8-year management scenario is based on the current fire management plan for the park, which is to burn every South Rim ponderosa pine burn unit every 8 years (National Park Service 2012). The 16-year burning interval doubles that management timeline from the current fire management plan. In addition, a mix of the two scenarios was simulated by alternating the prescribed burn schedule between 8 and 16 year intervals. Lastly, in order to better represent the episodic nature of ponderosa pine regeneration in the Southwest (Shepperd et al. 2006; Schubert 1974) two regeneration scenarios were compared: a low and high regeneration scenario. Thus we had a total of eight simulation scenarios: the four management strategies, each at a low and high regeneration rate.

Field methods – FMH data collection

Data collection followed protocols outlined in the Western Region Fire Monitoring Handbook (FMH) (National Park Service 2003). The plot design was a 20 by 50 meter rectangular plot, as described in detail in the FMH handbook (Figure 2.1). Data specific to this project included overstory and pole-sized tree data, seedling data, and fuels data. Overstory tree data was collected on the whole plot and included: species, DBH, tree height, crown base height, live crown ratio, live or dead designation, and any damage. Pole-size tree data was collected on half of the plot and included: species, DBH, live or dead designation, and the height class. Seedling data was collected on one quarter of the plot (10 by 25 m) and included the species, live or dead designation, and height class. Fuels transects radiate at random azimuths every ten m along the centerline of the plot and provided the down woody fuel loading data on a per plot basis (Brown 1974).

Additional measurements of total tree height and live crown ratio were taken. These measurements are not included in the standard FMH protocols, but were collected during the field seasons of 2009 and 2010. These data were collected because a similar modeling study performed in Grand Canyon found FVS to overestimate tree heights (Fulé et al. 2004a). Since the live crown ratios would also be affected by any inaccuracies in the tree height projections, this ultimately could affect predictions of stand structure, canopy fuel, and fire behavior variables into the future. With 847 tree heights collected on the four tree species encountered in the field, we were able to test FVS predictions for tree height by species. See Appendix B for additional information regarding this accuracy check.

Model parameters

The specific parameters used throughout the modeling exercises for this project varied somewhat depending on the management scenario and the regeneration scenario under each hypothetical situation. The simulations were run for 50 years into the future. Park managers felt that 50 years into the future was a practical management/planning horizon. Severe weather conditions were defined as 97th percentile based off historical weather data from the Tusayan weather station from years 1966 to 2010 using a date range of May 1st to October 31st. These values are shown in Table 2.2. Season of prescribed fires were set to fall. Burn conditions, or conditions at the time of the prescribed fires, are also given in Table 2.2. Seventy percent of the stand area burned during each simulated prescribed fire. Fuel model TL8 (Scott and Burgan 2005) was used and weighted at 100%.

A maximum stand density index (SDI) of 450 was specified for ponderosa pine and SDI was calculated using the summations method (Long and Shaw 2005). Regeneration rates were manually entered by species for each of the low and high regeneration scenarios (Table 2.3) and sprouting was turned off for all the scenarios. All species for both regeneration scenarios were set to 100% survival. A uniform distribution was used throughout each stand. To determine the low and high regeneration rates to input into FVS, the seedling data from the 11 true ponderosa pine plots were analyzed. The low and high regeneration rates were determined by using the seedling data from the first-year read after the third fire, and picking out the lowest and highest recorded post-fire regeneration of each species on these plots.

Only live seedlings and only those seedlings taller than 0.15 cm were included from the original dataset to determine realistic regeneration rates. Seedlings 0.15 cm in height or smaller were not included due to a recent study conducted in the ponderosa pine forests of southern Utah (Johnson 2011). This study reported that there is a very high rate of mortality in the seedlings of this smallest size class within the first five-years post-fire (Johnson 2011). Furthermore, Gambel oak regeneration was not added at all, as FVS appeared to produce unrealistic results in the early stages of this project with the addition of any Gambel oak.

Identifying desired conditions or thresholds

Model outputs were evaluated against desired conditions for each variable of interest (Table 2.1). Desired conditions were based on targets outlined by the GCNP fire management plan, historical conditions, and other sources from the literature. The GCNP objective for tree density is to achieve ponderosa pine densities of 106 to 162 trees/ha in these forests (National Park Service 2012). Previous research has reported reconstructed pre-settlement total (including all species) densities within similar ranges of GCNP's desired conditions: 148 trees/ha (Fulé et al. 1997), 94 – 176 trees/ha (Fulé et al. 2002a), and 140 – 145 trees/ha (Fulé et al. 2002b). These previous studies add support for the use of GCNP's desired conditions range as a general density guideline. Our desired conditions for basal area ranged from 9.2 – 18 m^2/ha (Sánchez Meador et al. 2010). Desired targets for RDI were defined as 35-60%, based on thresholds for full site occupancy and the onset of self-thinning or density-related mortality (Reineke 1933; Drew and Flewelling 1979; Long and Shaw 2005).

Desired conditions for total surface fuel loading in ponderosa pine forests were defined by GCNP to be 0.2 – 20.9 Mg/ha, as measured immediate post-fire (National Park Service 2012). Although litter and duff depths were also evaluated in this study, a desired conditions range or threshold for litter and duff depths was not identified for this particular fuels component. A desired target for fine woody debris fuel loading was identified to be 6.50 – 6.72 Mg/ha (Rothermel 1972; Stevens-Rumann et al. 2012), which allows adequate amounts to maintain surface fire spread. Desired conditions for coarse woody debris (CWD) fuel loading were identified as 11.2 – 44.9 Mg/ha using Brown et al.'s (2003) optimum range of CWD for warm, dry forest types; which include ponderosa pine and Douglas-fir forests. This range of CWD fuels was determined to meet the widest range of resource benefits by addressing soil productivity, wildlife habitat, and historical conditions without creating unacceptable levels of fire hazard or undesirable levels of soil heating (Brown et al. 2003).

A threshold of 16 km/hr was determined for the torching index using FireFamily Plus (Bradshaw and McCormick 2000) software and the 20-foot wind speeds under 97th percentile weather conditions, or severe fire weather conditions. We used the Tusayan weather station (number 020207) to determine the torching index threshold. Potential flame lengths of 1 meter are considered a standard threshold for firefighter safety (NWCG Fireline Handbook, Appendix B, 2006). Potential smoke production of PM2.5 was used to review fire effects; however a practical threshold for the potential production of PM2.5 was not identified in this study. Lastly, estimates of soil heating were assessed and a threshold of 60° C was identified, as this is often considered the lethal temperature for most living organisms (Reinhardt 2003).

Results

Changes in stand structure through time

The effects to the stand structure variables varied depending equally on both the regeneration rate and the management actions simulated in FFE-FVS (Figures 2.2 to 2.4). For stand density, the control scenario ended with higher values than the other three management strategies, then in order of decreasing magnitude it would follow: 16-year burn interval, alternating prescribed burn interval, and the 8-year burn interval. In some cases, there was little difference among the 8-year, 16-year, and alternating burn intervals. This was true for both regeneration rates. The regeneration rate appeared to have a strong effect on the tree density results in comparison to burn interval as there were large differences among the ending results between high and low regeneration rates for all management scenarios (Figure 2.2). The high regeneration rate simulations ended in tree densities ranging from approximately four times higher to ten times higher than the low regeneration rate simulations (Figure 2.2).

The effects on the RDI (Figure 2.3) help to illustrate some of these density changes through time in a different way by evaluating site occupancy and the competition for growing space within the stand. The general pattern under both regeneration rates was the control producing the highest RDI values, then the other scenarios following in order of decreasing magnitude: the 16-year burn interval, alternating burn interval, and the 8-year burn interval. Under a high regeneration rate all the scenarios increased in average RDI, with the control increasing the most. The low regeneration rate permitted the RDI to actually decrease under the 8-year and alternating burn intervals, with the 16-year burn interval decreasing only slightly. It is important to note that these decreases may be an even smaller change in truth, given the standard error (Figure 2.3).

The last stand structure variable analyzed was basal area (Figure 2.4). Again, the same pattern in the basal area results is evident as with the other stand structure variables: the control ending much higher than the other scenarios, followed by the 16-year burn interval, the alternating interval, and the lowest is the 8-year burn interval. One distinction here is the fact that the basal area results are nearly identical under both regeneration rates.

Changes in fuel loadings through time

The effects to the fuel loadings also varied through time depending on both the regeneration rate and the prescribed burn interval (Figures 2.5 to 2.7). Under both regeneration rates, the control scenario ended up with the highest amounts of total surface fuel loading, whereas the 8-year burn interval ended up with the lowest fuel loading. There was little difference between the 16-year and the alternating burn intervals, as they ended with nearly the same average surface fuel loadings under both regeneration rates. Furthermore, the differences in the results between the low and high regeneration rate scenarios were relatively small. In this case, the prescribed burn schedule appeared to have a stronger influence on surface fuel loading than regeneration rate alone.

Litter and duff depths combined were reviewed but varied little by burning interval or regeneration rate (data not shown). Fine woody debris (FWD); defined as all fuels ranging from 0 to 3 inches in diameter, were also analyzed through time (Figure 2.6). The results are similar to the trends displayed in Figure 2.5. The ending results are very similar between the three burning scenarios under both regeneration rates; with only small differences between the low and high regeneration scenarios. In this case, regeneration did not appear to have a large effect on FWD fuel loading.

The last fuels component that was analyzed was coarse woody debris (CWD), defined as any fuel greater than three inches in diameter (Figure 2.7). There was no real difference in CWD fuel loadings between the three burning scenarios. The control scenarios ended with the highest CWD fuel loading under both regeneration rates, but only slightly higher than the three burning scenarios under each regeneration rate. Overall, there were only slight differences in CWD fuel loading results between the two regeneration rates, with the largest difference in the control scenario.

Changes in potential fire behavior and fire effects over time

Average surface flame lengths under severe fire weather conditions were projected to be nearly the same between the low and high regeneration scenarios for the four management strategies (Figure 2.8). In this case, under both regeneration rates the control scenarios ended with the lowest potential surface flame lengths, with the 16-year burn cycle having the second lowest flame lengths. The alternating burn cycle had intermediate flame lengths and the 8-year burn cycle ended with the highest potential flame lengths. Overall, there were only slight differences between the regeneration scenarios and among the three burn scenarios results.

Average torching index (TI), the 20-foot wind speed required to cause torching of some trees under severe fire weather conditions (Reinhardt and Crookston 2003), was analyzed over time (Figure 2.9). There were large differences between the regeneration rates in TI results, particularly evident within the control scenarios. However, there were only slight differences among the three burning scenarios' ending TI results when looking at each regeneration rate respectively. Under the high regeneration rate we see very large but brief swings in the TI as the TI dips down to below 15 km/hr and then comes back up to over 150 km/hr for the burn scenarios. These swings in TI coincide with the pulses of new regeneration under the high rate, and the subsequent swings back up to high TI values are a result of most of those seedlings being burned in the simulated prescribed fires.

The average potential production of PM2.5 under severe weather conditions displayed the common pattern of the control scenario with the highest value, followed by the 16-year burn cycle, the alternating cycle, and the 8-year burn cycle being the lowest value (Figure 2.10). Differences in potential smoke production of PM2.5 were very small between low and high regeneration rate scenarios.

Estimated soil heating from prescribed fires at the surface were derived on the years that prescribed fires were simulated for each of the three burning scenarios (Table 2.4). For ease of comparison, data is only provided for those years in which prescribed fire was simulated for all three fire management scenarios (Table 2.4). There is no data provided for the control scenarios, because there were no prescribed fires scheduled. The soil heating estimates show that the 16-year burn cycle is likely to produce the highest potential soil heating, followed by the alternating burn cycle, and the 8-year burn cycle will likely produce the lowest average soil heating. There was no difference between the regeneration rates in soil heating estimates. The prescribed burn interval appeared to be driving the results in this case.

Discussion

Long-term effects to stand structure

Modeling results should always be interpreted with caution. The FVS model assumes homogenous stand conditions and is not able to account for spatial variability across a site in any way. Therefore, the results are a simplified version of what may be observed in reality. However, despite these assumptions the modeling results still provide valuable information and a means to understand potential changes in the stand structure in response to various management strategies over time.

Burning every 8 years under the low severity conditions that were parameterized in FFE-FVS illustrates that this scenario is both an effective burn schedule and fire severity to achieve desired conditions for tree density over time. In the year 2058, under the low regeneration scenario, the average tree density dips down to 131 trees/ha. These results fall within the range of pre-settlement reference conditions for the South Rim ponderosa pine forests and similar forests throughout the Southwest (Fulé et al. 2002a, 2002b; Fulé et al. 1997). Two studies on the South Rim have shown through dendroecological reconstruction an average total density ranging from 94 – 176 trees/ha (Fulé et al. 2002a) and a total density range of 140 – 145 trees/ha (Fulé et al. 2002b). In yet another study in northern Arizona located at Camp Navajo, an average 148 trees/ha for all tree species was reported (Fulé et al. 1997). In addition, these results also fall within desired density targets for GCNP which are to achieve 106 – 162 trees/ha for ponderosa pine specifically (National Park Service 2012). It is important to note that the density results presented here include all tree species present, not just ponderosa pine.

Other stand structure variables support the same conclusion that the 8-year burn interval is moving the stand structure towards desired conditions. The basal area stays relatively consistent throughout the simulations by remaining only slightly above 18 m^2/ha, the high end of average pre-settlement basal area for ponderosa pine forests in the Southwest (Sánchez Meador et al. 2010). For most of the simulation the RDI for this scenario stays within the target range of 35 – 55% (Reineke 1933; Long and Shaw 2005). The upper end of this desired conditions range can actually extend up to 60% (see Figure 2.3) based on more recent research in ponderosa pine forests (Long and Shaw 2005). Previous studies have suggested that relative densities in the range of 35-40% are an appropriate target for capturing "near maximum" stand growth, which characterizes full site occupancy (Long 1985; Long and Shaw 2005). Therefore a management strategy that is focused on maximizing volume production would likely include this lower limit of 35% RDI (Long and Shaw 2005). Otherwise, managers not concerned specifically with volume production may not recognize this lower RDI limit. Nonetheless, it does follow that under such low levels of competition when the RDI drops below 35%, the stand structure may be more conducive to successful regeneration in ponderosa pine stands. Of course, there are many other factors which were previously discussed regarding ponderosa pine regeneration (Shepperd et al. 2006) that must also line up with the timing of such stand structure conditions. Overall, our results suggest that reducing tree density with low-severity fire alone can take a long time to reach desired conditions. In the case of the 8-year, low regeneration scenario it takes about 32 years and four prescribed burn cycles, to reach desired conditions.

This study supports other work in ponderosa pine forests throughout the Southwest which show reducing tree density with low severity prescribed fire alone can be difficult (Covington et al. 2001; Sackett et al. 1996; Sackett and Haase 1998). A long-term study done near Flagstaff, Arizona comparing two sites undergoing a schedule of low severity repeated burning found that pre-settlement conditions would be difficult to achieve by using fire alone (Sackett et al. 1996). Instead, Sackett et al. (1996) recommended that a prescribed burn program should be supplemented by low-impact mechanical thinning in order to more closely achieve the pre-settlement structure. Other research in the Southwest has arrived at similar management recommendations (Covington et al. 2001; Harrington and Sackett 1990). Fulé et al. (2002a) compared the effects of different ecological restoration alternatives including using fire alone compared to several thinning techniques. They found that the burn-only technique thinned out about 2,300 trees/ha of small-diameter trees but the site was still far above historic levels for density and basal area (Fulé et al. 2002a). This is very similar to what we found with our FFE-FVS modeling results for both the 16-year burn interval and the alternating burn interval with regards to density.

Additionally, this study also supports the idea of utilizing prescribed fire of higher severity to more closely achieve desired conditions. There is support from different perspectives in the literature for the use of fire of higher severity in restoring ponderosa pine forests to desired or pre-settlement reference conditions (Fulé et al. 2004b; Sackett and Haase 1998; Sackett et al. 1996). One such study was done on the North Rim of Grand Canyon by Fulé et al. (2004b). They assessed fire effects 6 years after a relatively severe prescribed fire had burned. They found that the average density (331 trees/ha) and basal area (28.5 m^2/ha) were reduced to levels similar to the reconstructed pre-settlement conditions (~ 246 trees/ha and 28.5 m^2/ha) (Fulé et al. 2004b). Most of the mortality was found in the fire-susceptible species, especially white fir, thus also restoring pre-settlement species composition to being ponderosa pine dominant (Fulé et al. 2004b). They concluded that more severe prescribed fires could help to restore the forest structure closer to the HRV. These structural changes could also be achieved without the negative impacts attributed to thinning operations such as soil compaction, construction of new roads, logging damage to surrounding trees, or the potential introduction of exotic species. Similarly, in another study excessive heat to the crown (up to 75% crown scorch and/or consumption) was needed to effectively kill the thickets of small diameter trees in order to attain desired levels of density (Sackett and Haase 1998; Sackett et al. 1996). Such methods could be employed in dense stand conditions or on sites experiencing high rates of regeneration. Perhaps managers may want to consider the benefits of utilizing prescribed fire of higher severities in the more dense stands in order to alter the stand structure in a shorter timeframe. This could lend both ecological and safety benefits in a shorter timeframe as well. Lastly, it could take less burn entries to achieve desired conditions for some variables and less maintenance over time if higher severity prescribed fire were employed in such areas.

The alternating prescribed burn interval under the low regeneration rate produces results that are similar to those from the 8-year burn interval under low rates. Stands are less dense with less basal area than inventory conditions; however they remain above desired conditions for the length of the simulation. The RDI remains low throughout the simulation, indicating there is growing space available. While the scenario doesn't quite meet desired targets, it does come fairly close.

The 16-year burn interval under a low regeneration rate also shows the stand structure becoming less dense with less basal area over time. The RDI stabilizes and remains below thresholds for density-related mortality. However, compared to the other two burn intervals the 16-year burn interval is not as effective at achieving specific objectives or targets. Although tree density and basal area were an improvement from the inventory conditions they were still a high compared to the range of desired conditions.

The stand structure results are heavily dependent on the regeneration rate simulated. In all high regeneration scenarios, none of the burn intervals were sufficient to reduce tree density to desired levels. Even in the case of the 8-year burn interval, the high rate of new seedlings coming in after the simulated prescribed burns overrides the effects from the frequent schedule of burning. As a point of reference, the ponderosa pine regeneration rates presented in Table 2.3 fall within the range of data Fulé et al. (2002a) reported on their burn sites post-treatment. However, both the pinyon pine and juniper species regeneration rates appear to be rather high compared to what Fulé et al. (2002a) reported for those species. The RDI values are also higher overall with the high regeneration rate scenarios, and exceed the upper 55 – 60% threshold with the 16-year burn interval. The alternating prescribed burn interval also nearly surpasses this threshold under a high regeneration rate. These stands are likely to be stressed due to density-related mortality and competition for resources on that site.

Overall, these results indicate that regeneration rates play an important role in determining the long-term effectiveness of a particular prescribed burn interval. When simulated under a low regeneration rate, the most frequent burn interval of 8 years most closely achieved desired conditions in all stand structure variables. However, when simulated under the high regeneration rate, the most frequent burn interval was unable to attain desired density targets. With a relatively consistent basal area over time between the low and high regeneration scenarios, but with higher RDI values and higher tree densities for the high regeneration rate scenarios – this all indicates that there would be an abundance of smaller diameter trees present in these stands. Such stand structures would not be desirable from both an ecological and a safety standpoint. These results indicate that using low-severity repeated prescribed fire alone may be very difficult to attain desired density conditions on sites experiencing high rates of regeneration. Therefore, fire of higher severity or mechanical fuel treatments may be warranted in order to more closely achieve desired conditions on such sites.

Long-term effects to fuels characteristics

The average total surface fuel loading results displayed similar trends under both regeneration rates. Grand Canyon's desired conditions for the average total surface fuel loading in these ponderosa pine forests on the South Rim is defined as being between 0.4 and 20.9 Mg/ha, as measured immediate post-fire (National Park Service 2012). Clearly, both control scenarios surpass this high end of the desired fuel loading conditions beginning in the year 2034 (24 years from the start). In the high regeneration scenario the new seedlings coming in produce more mortality from competition and dense stand conditions, further adding to the total surface fuels.

Since the total fuel loading desired conditions range is intended to be used immediate post-fire (National Park Service 2012), all three of the burn scenarios were effective at keeping total surface fuels within desired post-fire targets after each prescribed burn. Regeneration rate had little effect on the total surface fuel loading with the most frequent burn interval, since both 8-year regeneration scenarios stayed well below the upper threshold of 20.9 Mg/ha for the length of the simulations. The trends from this burn interval varied very little throughout time. In fact, the peaks and dips in the trends are nearly identical under both regeneration rates. Thus, the 8-year prescribed burn interval is very effective at keeping average total surface fuel loading consistently within the desired conditions range at all times; even on sites experiencing high rates of regeneration. The 16-year burn interval and alternating prescribed burn interval trends varied more between high and low regeneration rates and did not maintain the fuels within as consistent a range over time as the 8-year burn interval. Under the high regeneration scenario, the 16-year burn interval gained higher average surface fuels than the other scenarios (23.0 Mg/ha before the last prescribed burn). The overall trend shows total surface fuels gradually increasing over time, with each prescribed burn becoming slightly less effective at decreasing the total fuel loading. The same is true for the 16-year, low scenario and both alternating prescribed burn scenarios.

Since the range of Grand Canyon's desired conditions for fuel loadings are intended to be measured and used immediate post-fire (National Park Service 2012), some may worry that burning every 8 years is actually too often. Especially when we note that the predicted fuel loadings for the 8-year burn interval are well below the upper end of this desired range for the duration of the simulations. It is a valuable question to ask whether or not keeping the fuels within such a consistent range throughout time could be detrimental to these forests. Especially when we consider that all three burn scenarios rarely, if ever, accumulated more fuels than the high end of this *immediate post-fire* desired conditions range during the years in between prescribed burns. Although none of the scenarios ever drop below the lower end of this desired conditions range (0.4 Mg/ha) evidence suggests that this lower threshold is really too low and not representative of a healthy forest ecosystem which would promote surface fire spread (Rothermel 1972).

Litter and duff depths varied little by burning interval or regeneration rate (data not shown). As for the FWD fuel loading component, both control scenarios had gained adequate amounts of FWD to maintain surface fire spread (Rothermel 1972; Stevens-Rumann et al. 2012). In contrast, the three prescribed burn scenarios did not gain adequate amounts of FWD to maintain surface fire spread. This suggests that GCNP could have (both currently and into the future) difficulties in getting prescribed burns to effectively spread throughout these forests. It is important to note that the FWD target identified for this study may underestimate the potential for fire spread, as it does not include litter, which is also an important carrier of fire in these systems. However, some managers feel that they have already encountered such problems during prescribed burns on the South Rim, where little to no fire spread has been recorded in some cases during prescribed burning in this section of the Park (W. Bunn, GCNP fire ecologist, personal communication, 2012). If such problems persist, then longer intervals between fires may be necessary to allow for more surface fuel accumulation.

The coarse woody debris results varied by management strategy and regeneration rate simulated. Only the control, high regeneration scenario gained enough coarse woody debris by the end of the simulation to fall into the lower end of the optimum range of CWD for warm, dry forest types which include ponderosa pine and Douglas-fir forests (Brown et al. 2003). This optimum range of CWD was determined to meet the widest range of resource benefits by addressing soil productivity, wildlife habitat, and historical conditions of CWD without creating unacceptable levels of fire hazard or undesirable levels of soil heating (Brown et al. 2003). Our results suggest that these stands on the South Rim of the Grand Canyon may be deficient (both currently and into the future) in desirable levels of CWD. Given the total post-fire fuel loading desired targets defined by GCNP, we do not advocate that GCNP adopt the high end of the optimum range for CWD (44.9 Mg/ha). However, we propose that the lower end of the CWD optimum range (11.2 Mg/ha) is an appropriate target for GCNP. This would mean that about half of the fuels complex should be composed of CWD. Since our results show that only the control/no-treatment, high regeneration scenario gained desirable levels of CWD; these sites may be low in CWD. Thus, managers at GCNP may want to look at methods which would protect more of the CWD present during prescribed fires, or perhaps take measures to create more CWD by producing snags or downed logs with higher severity fires. Such measures may be warranted in order to provide wildlife habitat and ensure higher levels of soil productivity in these forests. Although we realize that the requirements for CWD in providing adequate wildlife habitat can vary widely by species, and few studies have quantified the specific amounts needed (Brown et al. 2003).

Long-term effects to potential fire behavior and fire effects

The potential surface flame length under severe weather conditions were very similar across all the management scenarios and under both regeneration rates and none of the scenarios surpass the safety threshold for flame lengths. Flame lengths actually decreased through time in both control scenarios. This counter-intuitive result was probably caused by the stand becoming denser in both scenarios and thus the mid-flame wind speed decreasing over time. Both control scenarios produced much higher tree densities compared to the other management scenarios, which in turn created more closed stand conditions and a higher wind reduction factor.

The torching index, or the 20-foot windspeed under which one would expect torching to occur, was compared to the 97th percentile 20-foot windspeed for the area (16 km/hr). None of the low regeneration scenarios come close to this low wind threshold. In fact, the TI increased over time, in all scenarios. These results suggest that the crown base height may be increasing slowly over time as the stand ages and the trees grow; while at the same time there isn't a lot of regeneration to sufficiently develop into many ladder fuels. Both factors would reduce the potential for torching. Under the high regeneration rate the control, the 16-year burn interval, and the alternating burn interval all fall below this wind threshold at some point and the control stays consistently below this threshold. Since these brief but dramatic dips in the torching index do not occur in the 8-year, high regeneration scenario it appears that these large swings in TI are resulting from both the high regeneration rate in combination with the longer intervals between prescribed burns. These results point to the idea that burning every 8 years on sites with high post-fire regeneration could safeguard against the probability of torching occurring under high severity fire weather conditions.

There are a number of important caveats with these fire behavior results worthy of pointing out. The first is the fact that potential fire behavior and fire effects modeling results should always be viewed conservatively. A real fire will often burn with a degree of variation across a site or a landscape; being affected by slight variations in fuel moistures, wind speed (or wind reduction factor), weather, topographical influences and even time of day. A second caveat is the fact that sprouting was turned off in FVS for every simulation. Therefore, FFE-FVS may be overestimating the TI. These TI projections can appear rather unrealistic, especially considering that Gambel oak is present in these forests and does provide ladder fuels.

The last caution in interpretation of these TI results is regarding the wind threshold we used to evaluate our results against. Previous studies have suggested that the 97^{th} percentile wind speed is not representative of extreme wind conditions often associated with severe fire events (Fulé et al. 2001; Dicus and Zimmerman 2007). Therefore, we should not assume that the 16 km/hr wind threshold used here is an accurate representation of the 20-foot wind speeds that may occur during severe fires.

The long term potential fire effects were analyzed by assessing potential production of PM2.5 under severe weather conditions and soil heating estimates from prescribed fires. Under the high regeneration rate, all scenarios produced more PM2.5 than the low regeneration scenarios because of the higher fuel loadings associated with the high regeneration scenarios. Particularly important to potential PM2.5 production is not only the total surface fuel loading available, but also the amount of litter and duff and

FWD fuel loadings available. However, the one scenario which did not produce more PM2.5 under high regeneration rates was the 8-year burn interval.

GCNP does have desired conditions for potential smoke production of PM2.5 but this range is not directly comparable to the way FFE-FVS produces potential smoke values. GCNP follows the Fine Particulate Air Quality Index which was created by the Environmental Protection Agency. The Fine Particulate Air Quality Index states that PM2.5 values averaged over a 24-hour period cannot exceed 40.5 – 65.4 $\mu g/m^3$ for sensitive groups of individuals. For the general public and healthy individuals, PM2.5 smoke production averaged over a 24-hour period cannot exceed 65.5 – 150.4 $\mu g/m^3$ (http://www.epa.gov/oar/particlepollution/health.html). These values are not directly comparable to the values presented here as FFE-FVS reports smoke production of PM2.5 in tons/acre (converted to Mg/ha for results presented here). This tons/acre value is not necessarily intended to be used as a 24-hour average value; rather it should be interpreted as the total potential smoke production from a given fire.

Soil heating estimates can provide a picture of potential mortality resulting from prescribed fires since 60°C is often considered the lethal temperature for most living organisms (Reinhardt 2003). Soil heating estimates are also helpful for managers to assess because if the 60° C threshold is surpassed, this may point to potential problems with soil productivity and stability into the future (Reinhardt 2003). By looking at soil heating estimates managers can evaluate potential issues regarding mortality and soil productivity and/or stability as a result of management actions through prescribed burning.

The average soil heating from the 16-year burn interval came close to exceeding the 60° C threshold for mortality, and some of the individual plots did exceed 60°C (data not shown). Regeneration rate had little effect on the soil heating estimates. The differences in the soil heating estimates were due to the variations in fuel loadings corresponding to the varying intervals between prescribed burns, with the 16-year burn interval producing the highest soil heating estimates partly because of the largest amounts of surface fuels to burn. The more surface fuels available, the hotter a fire will potentially burn and more mortality to the stand may result. FOFEM soil heating estimates are also highly sensitive to the duff depths available (Reinhardt 2003), and since these ponderosa stands on the South Rim don't have much duff due to repeated burning, these soil heating estimates are consequently not very high.

Conclusions and management implications

In conclusion, the modeling results depended equally on both the regeneration rate simulated and the management scenario for most variables that we analyzed. A few variables that were analyzed, such as the soil heating estimates and basal area appeared to be heavily driven by the management strategy and unaffected by the regeneration rate. However, most of the results support the understanding that both model inputs are equally important in determining the long-term effects to the variables of interest. Therefore we conclude that the best fire management strategy be guided, in part, by the observed post-fire regeneration rates across a site or burn unit. In other words, we strongly advocate for the use of the adaptive management feedback loop by using new data (the seedling/regeneration data from the FMH plots specifically) as it becomes available to inform plans and future decisions. For example, if a given site is experiencing low post-fire regeneration rates then perhaps the alternating prescribed burn interval could be a feasible burning option. On the other hand, if high rates of post-fire regeneration are observed then the 8-year prescribed burn interval is going to best achieve most objectives.

The 8-year burn interval achieves desired conditions or pre-settlement reference conditions in most of the variables that were analyzed. These desired targets were reached within the 50 years of the FFE-FVS simulations. Under the low post-fire regeneration rates, the 8-year burn interval achieved density objectives within four burn cycles into the future. Based off these modeling results, we suggest that a good time for a review of the effectiveness of the current fire management strategy across these sites that have already burned three times at GCNP could be carried out ~32 years in the future. In other words, after four more burn entries (if adhering to the 8-year burn interval) average density objectives should be attained, or nearly so. Fuels and potential fire behavior should be well within desired conditions at this point as well. Specific elements of the stand structure may require further evaluation, such as an assessment of the diameter distribution by species (Appendix C).

However, using the 8-year burn interval may pose problems in the future through the lack of CWD and the various services CWD can provide the ecosystem (Brown et al. 2003). The 8-year burn interval may also create a situation with inadequate amounts of FWD to maintain surface fire spread (Rothermel 1972). Thus, fire managers might have difficulty in carrying out prescribed burns that frequently at all. Also, the potential smoke production from more frequent prescribed fires may be undesirable to some managers at the park whose focus is to protect GCNP's Class I airshed. In the case of smoke production, it becomes a question of whether smaller amounts of smoke created more frequently over time (with the 8-year burn interval) are more appropriate than larger amounts of smoke which are created less often (e.g. 16-year burn interval).

Results from the FFE-FVS simulations point to the 8-year scenario as being the best fire management strategy for producing low density stands, for maintaining the lowest surface fuels, and for posing the smallest risk of extreme fire behavior or fire effects under both post-fire regeneration rate scenarios. However, the alternating prescribed burn interval can also produce results that are desirable on the low post-fire regeneration sites but this management strategy is probably not feasible on the sites with high post-fire regeneration. Advantages to using an alternating burn interval compared to the 8-year burn interval include: fewer resources required, decreased cost to the FMP at Grand Canyon, comparable levels of smoke production compared to the 8-year burn cycle, and increased production and/or retention of high quantities of CWD.

In summary, we advocate for the implementation of adaptive management practices and recommend that the best fire management strategy be influenced by the observed rates of post-fire regeneration. Our results reaffirm the importance of the FMH program since these post-fire regeneration rates can be captured by adhering to the normal FMH plot visitation schedule. However, if there are no FMH plots installed in an area that burns and it is an area to be managed in the future, a seedling sampling protocol can be employed in rapid assessment type plots throughout the area. If using the 8-year burn interval, managers may want to take measures to create or preserve higher quantities of CWD during burning, which in turn will promote several other aspects of health and biodiversity in the forest (Brown et al. 2003). If utilizing an alternating burn interval, managers should be aware that specific density targets and stand structure elements may take longer than 50 years to achieve. In addition to the management strategies being guided by post-fire regeneration rates, we also would to like to point out that this study provides some evidence for the use of higher severity prescribed fire. Since our modeling results suggest that it can take up to 32 years to achieve desired density conditions on low post-fire regeneration sites, even by adhering to the current FMP of burning every 8 years, we propose that employing higher severities than typical prescribed burns may allow for much quicker changes in the stand structure. Thus managers would likely see desired conditions achieved in other variables of interest in a shorter timeframe as well, since fuels and fire behavior are partly driven by the stand structure.

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Table 2.1. FFE-FVS output variables analyzed. Desired conditions are presented in bold where applicable. Thresholds identified as undesirable to cross are presented in bold and italics. The sources of each threshold or desired conditions range are presented below.

Stand structure variables	**Fuels characteristics**	**Potential fire behavior and fire effects**
Tree density (trees/ha) [A] **94 -176 trees/ha**	Fine woody debris (fuels < 3 inches in diameter) in Mg/ha [D] **6.50 – 6.72 Mg/ha**	Torching Index (km/hr) [G] ***16 km/hr***
Relative Density Index [B] **35-60% RDI**	Coarse woody debris (fuels > 3 inches in diameter) in Mg/ha [E] **11.2 – 44.9 Mg/ha**	Potential flame lengths (m) under severe fire weather conditions [H] ***1.0 m***
Basal area (m^2/ha) [C] **9.2 – 18 m^2/ha**	All dead surface fuels combined (Mg/ha) [F] **0.4 – 20.9 Mg/ha**	Potential smoke production of PM2.5 (Mg/ha) under severe conditions
		Soil heating (°C) [I] ***60° C***

[A] **Fulé et al. (2002a)**
[B] **Reineke (1933); Long and Shaw (2005)**
[C] **Sánchez Meador et al. (2010)**
[D] **Rothermel (1972); Stevens-Rumann et al. (2012)**
[E] **Brown et al. (2003)**
[F] **GCNP Fire Management Plan (National Park Service 2012)**
[G] **FireFamily Plus (Bradshaw and McCormick 2000)**
[H] **NWCG Fireline Handbook, Appendix B (2006)**
[I] **Reinhardt (2003)**

Table 2.2. Weather and fuel moisture parameters for severe burning conditions set in FFE-FVS. Weather and fuel moisture conditions at the time of the simulated prescribed fires are also given.

	Severe fires (97^{th} percentile weather)	Burn conditions (Conditions at the time of the prescribed fires)
1- hr timelag fuel moisture (%)	4	6
10- hr timelag fuel moisture (%)	4	5
100- hr timelag fuel moisture (%)	8	6
1000- hr timelag fuel moisture (%)	9	25
Live woody fuel moisture (%)	71	150
Live herbaceous fuel moisture (%)	58	150
20-ft. wind speed (mph)	10	7

Table 2.3. Regeneration rates set in FFE-FVS.

	Low Regeneration (every 16 years)	High Regeneration (every 8 years)
Ponderosa pine (trees/acre)	16.2	162
Common pinyon (trees/acre)	16.2	226
Utah juniper (trees/acre)	16.2	113

Table 2.4. Estimates of soil heating from prescribed fires simulated in FFE-FVS. Data is presented for years in which fires were simulated for all scenarios. No prescribed fires were simulated in the no-treatment (control) scenarios. Standard error for each burn year is reported in parentheses.

Management Scenario	Beginning result in 2010 (°C)	Result after treatment in 2042 (°C)	Result after last treatment in 2058 (°C)
8 year – Low	40 (2.2)	25 (1.5)	22 (0.5)
High	40 (2.2)	26 (1.6)	22 (0.6)
16 year – Low	40 (2.2)	44 (5.1)	39 (2.9)
High	40 (2.2)	45 (5.1)	41 (2.9)
Alternating - Low	40 (2.2)	32 (2.9)	32 (1.7)
High	40 (2.2)	33 (3.0)	33 (1.7)

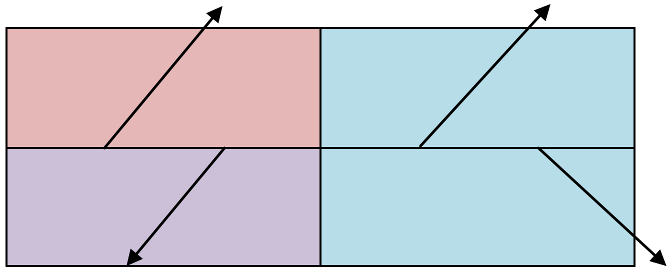
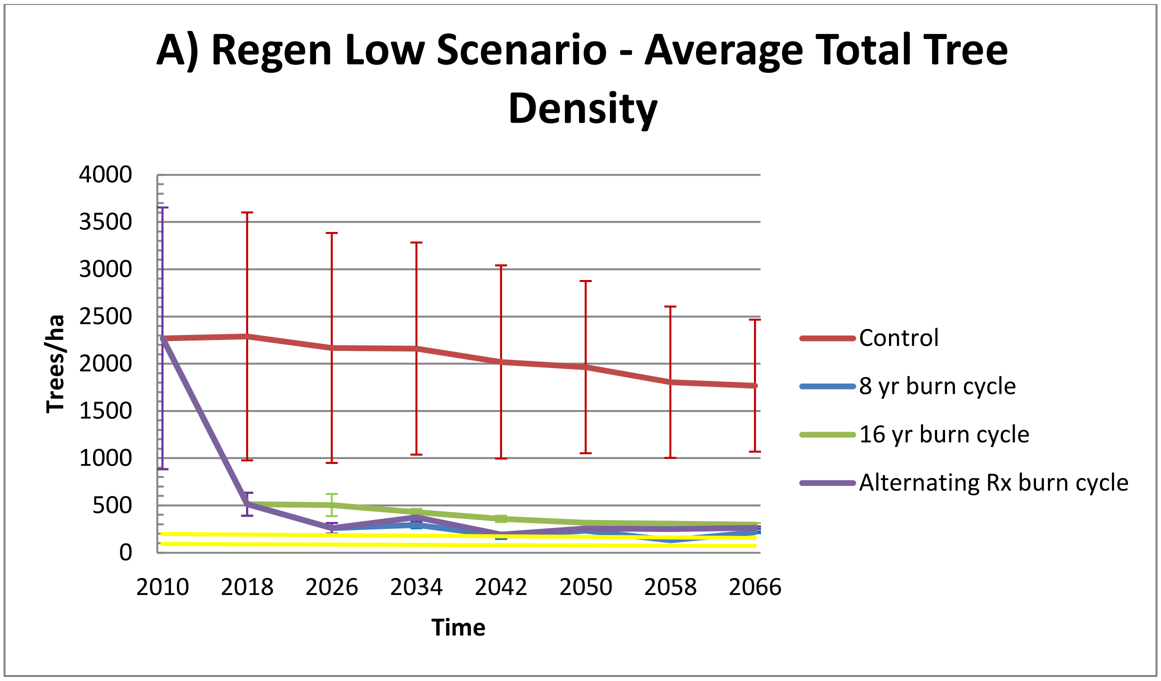


Figure 2.1. FMH plot design. Fire Monitoring Handbook (FMH) plots are 20 by 50 m rectangles (National Park Service 2003). The arrows indicate the fuels transects (Brown 1974) stemming from the center of the plot in random directions. Overstory data are collected in all 4 quarters of the plot, pole tree data in the pink and purple quarters, and seedling data in only the pink quarter.



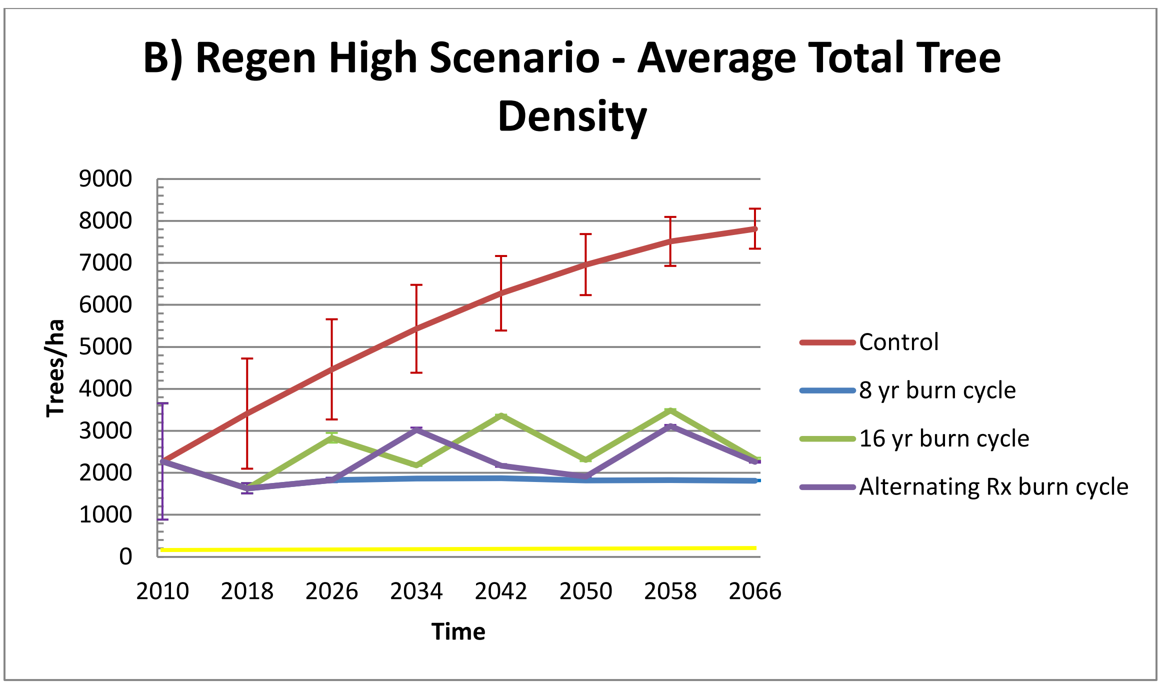
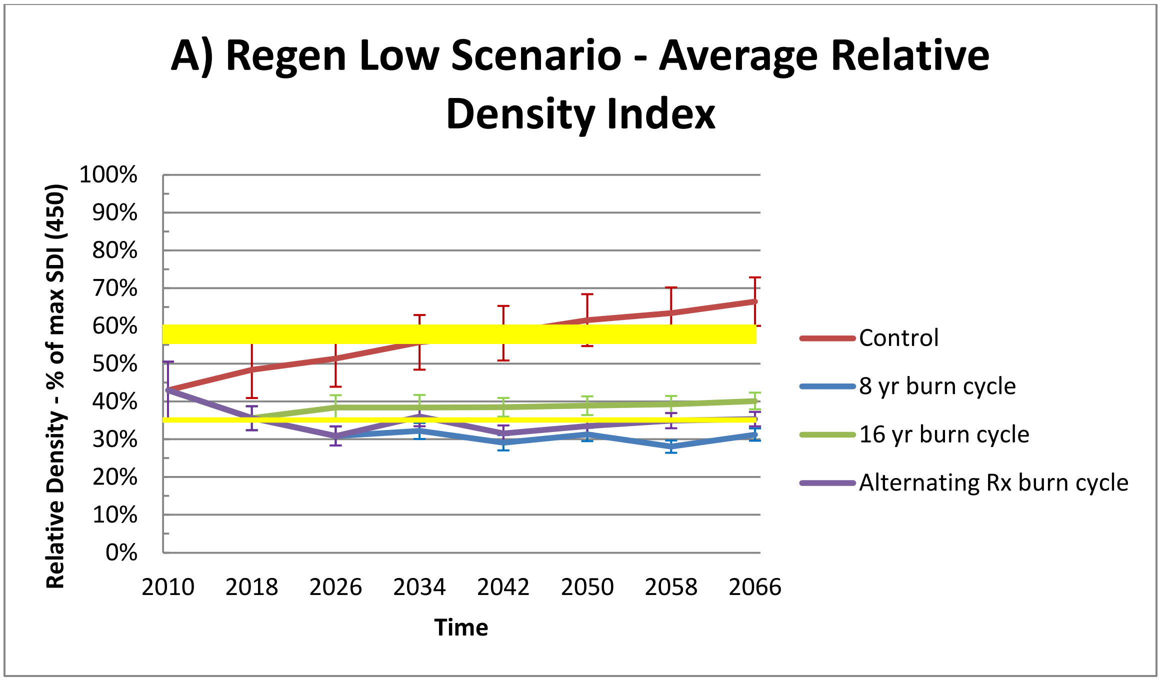


Figure 2.2. The average total (including all species) tree density in trees/ha displayed through time under: (A) low regeneration rate and (B) high regeneration rate. Standard error bars are shown every 8 years for each of the management scenarios. The yellow lines represent a range of pre-settlement reference conditions: 94-176 trees/ha (Fulé et al. 2002a), 140-145 trees/ha (Fulé et al. 2002b), and 148 trees/ha (Fulé et al. 1997).



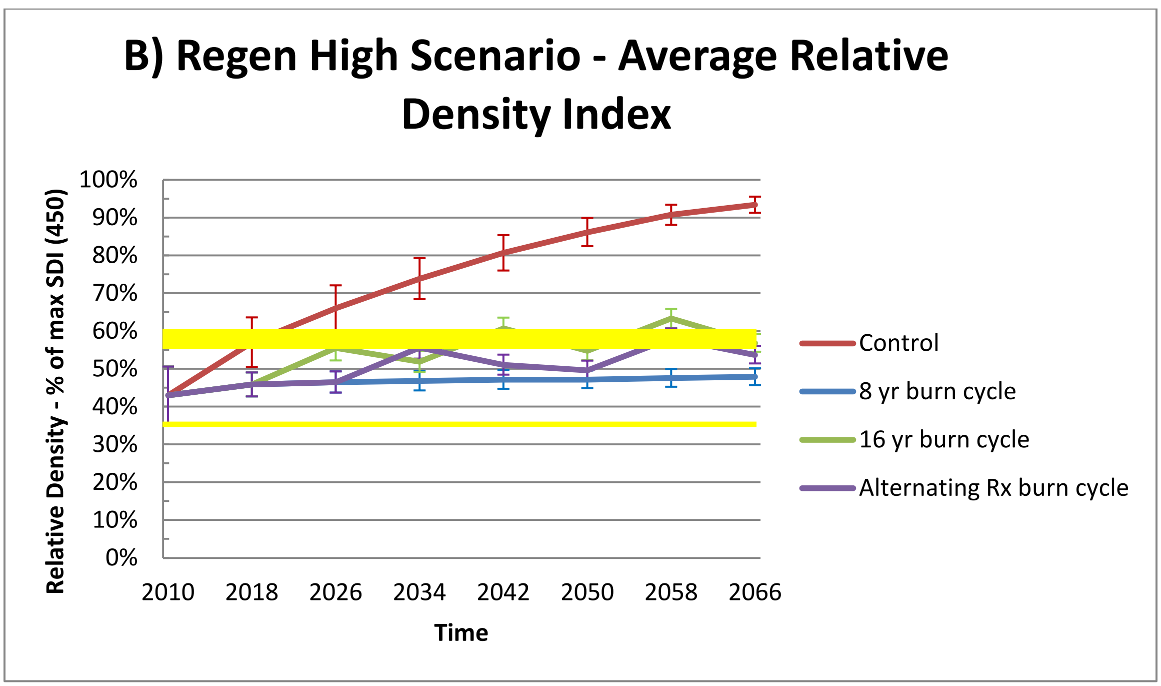
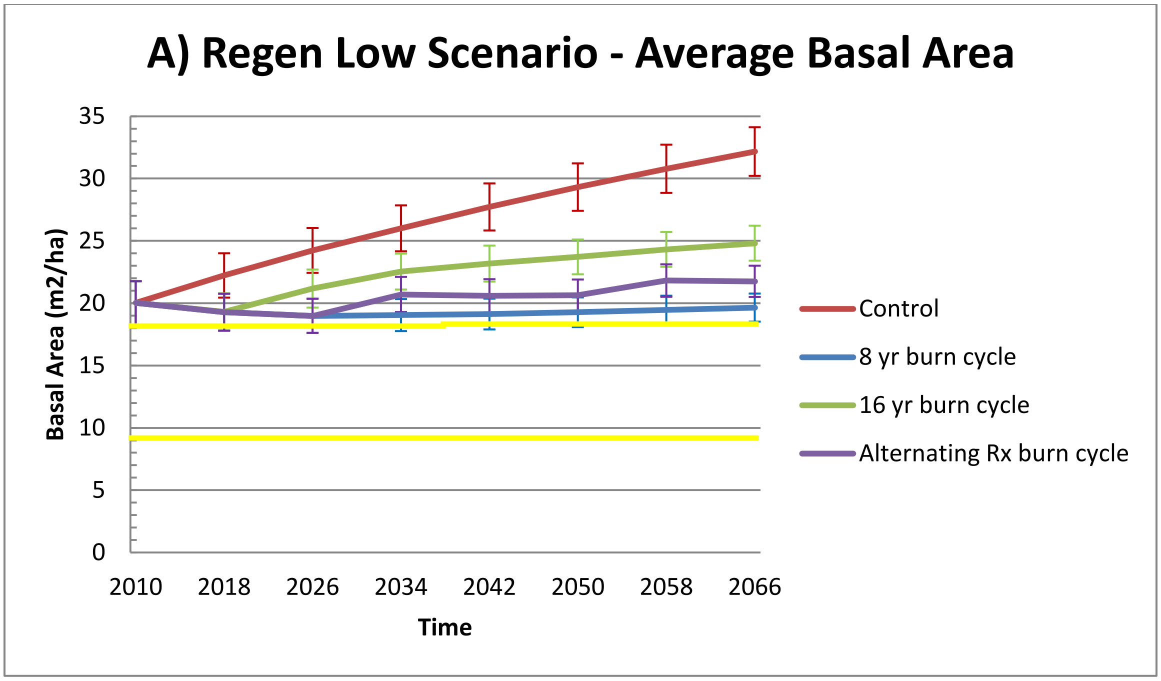


Figure 2.3. Average relative density index displayed through time under: (A) low regeneration rate and (B) high regeneration rate. Standard error bars are shown every 8 years. The lower yellow line highlights 35% RDI, which is the lower limit of "full site occupancy" (Long and Shaw 2005). The upper yellow box highlights 55-60% RDI which is the lower limit of the self-thinning zone, or where density-related mortality begins to occur (Reineke 1933; Long and Shaw 2005).



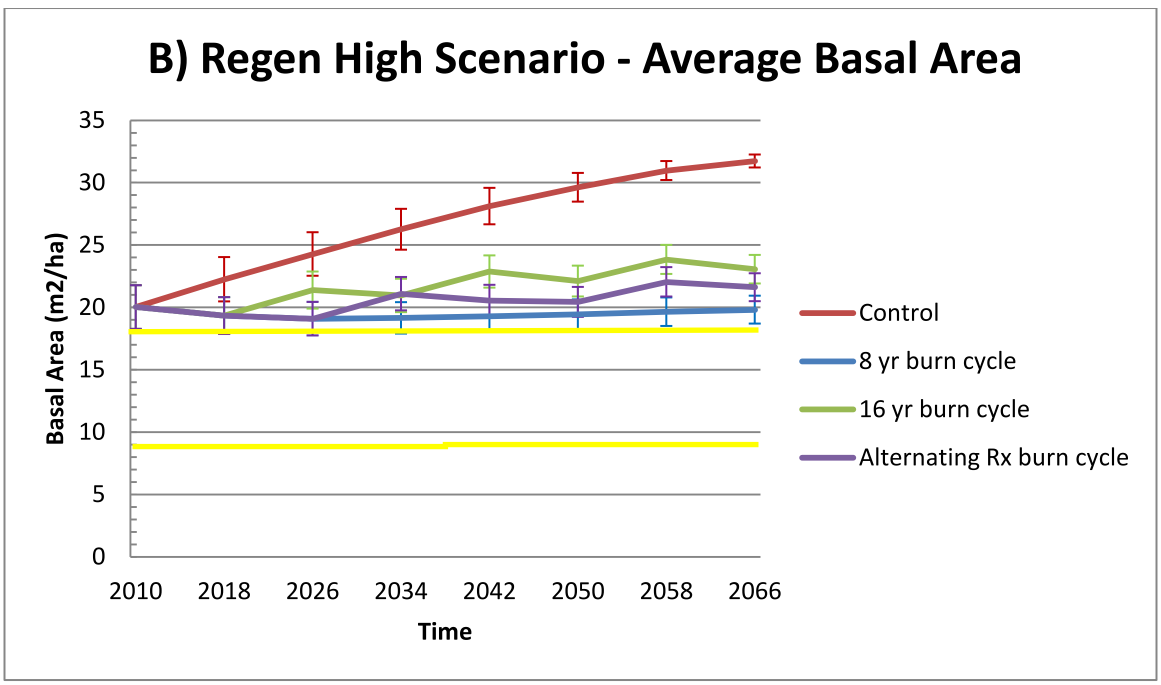
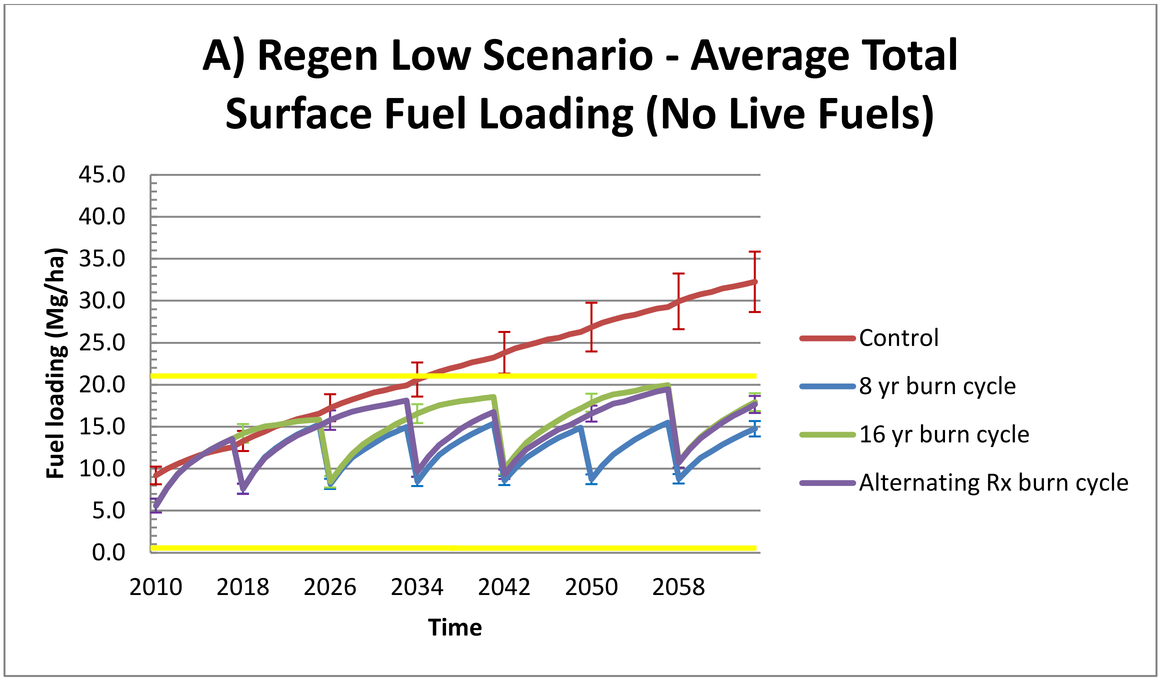


Figure 2.4. The average basal area of all species in m^2/ha displayed through time under: (A) low regeneration rate and (B) high regeneration rate. Standard error bars are displayed every 8 years for each of the management scenarios. Yellow lines represent the average pre-settlement reference conditions of 9.2 – 18.0 m^2/ha reported by Sánchez Meador et al. (2010).



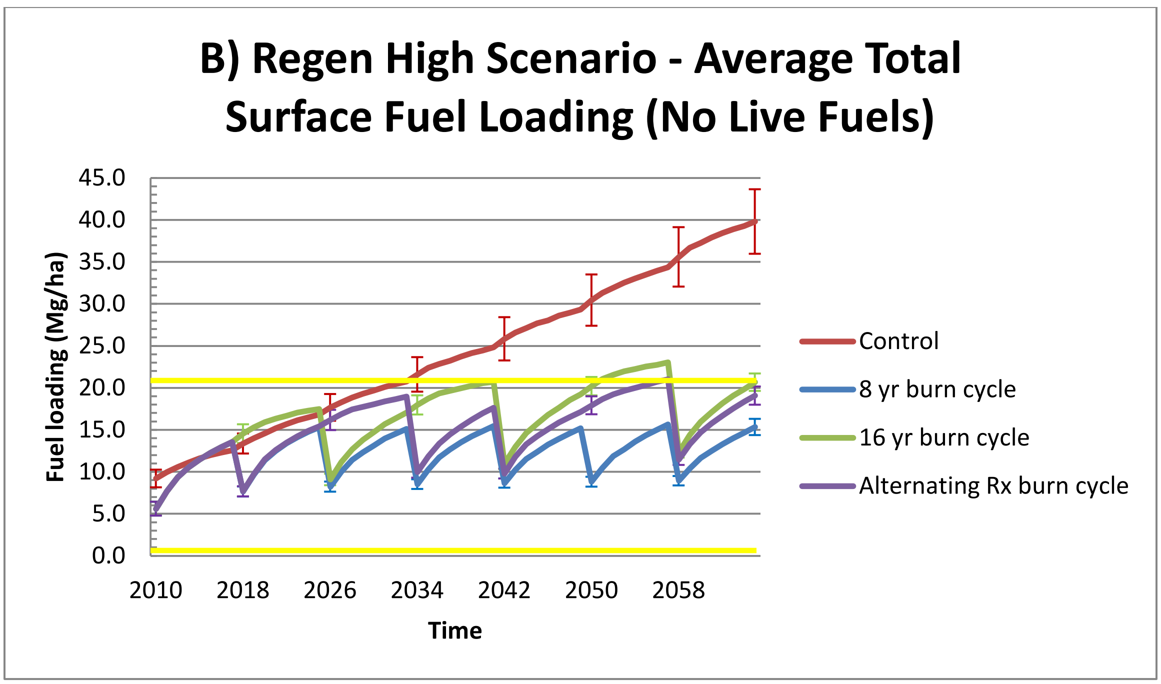
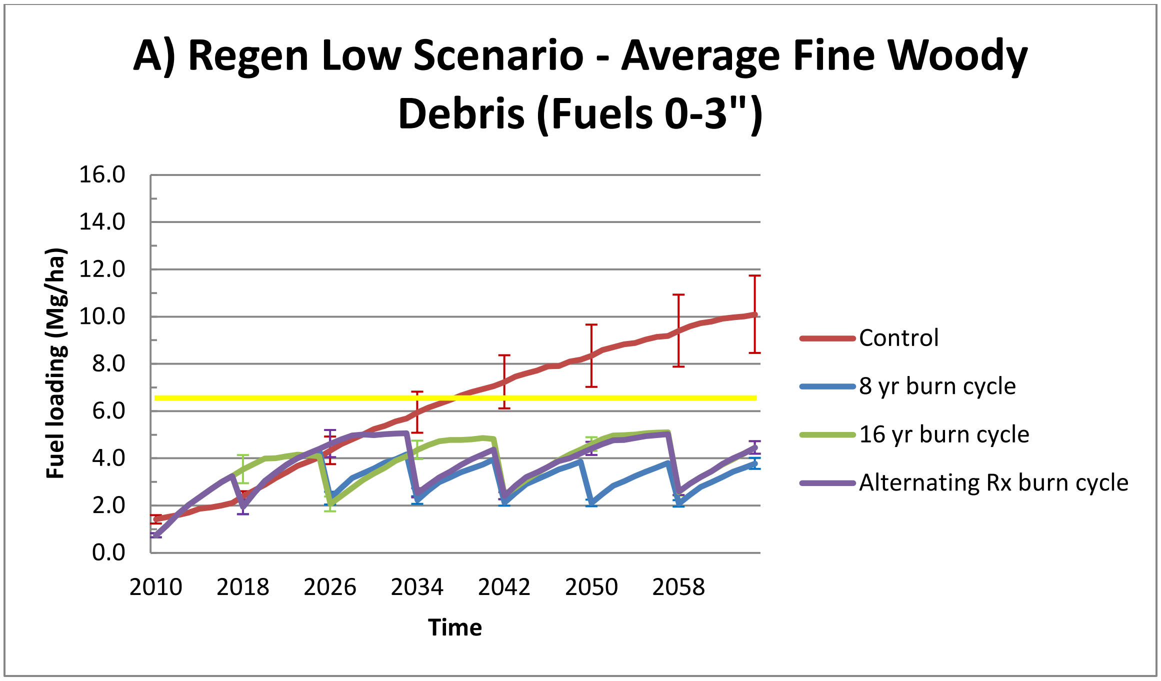


Figure 2.5. Average total surface fuel loading (not including any live fuels) displayed through time in Mg/ha under: (A) low regeneration rate and (B) high regeneration rate. Standard error bars are displayed every 8 years. Grand Canyon's desired conditions for these forests are defined as between 0.4 – 20.9 Mg/ha, measured immediately post-fire (National Park Service 2012). The desired conditions for post-fire fuel loadings are designated by the yellow lines.



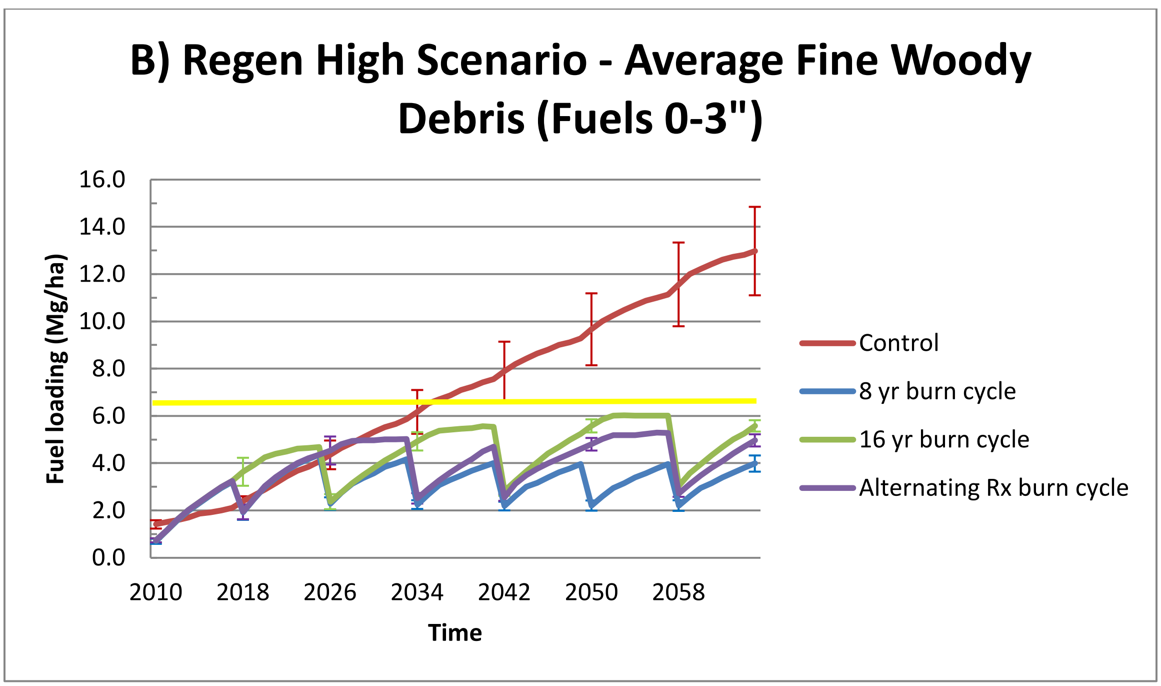
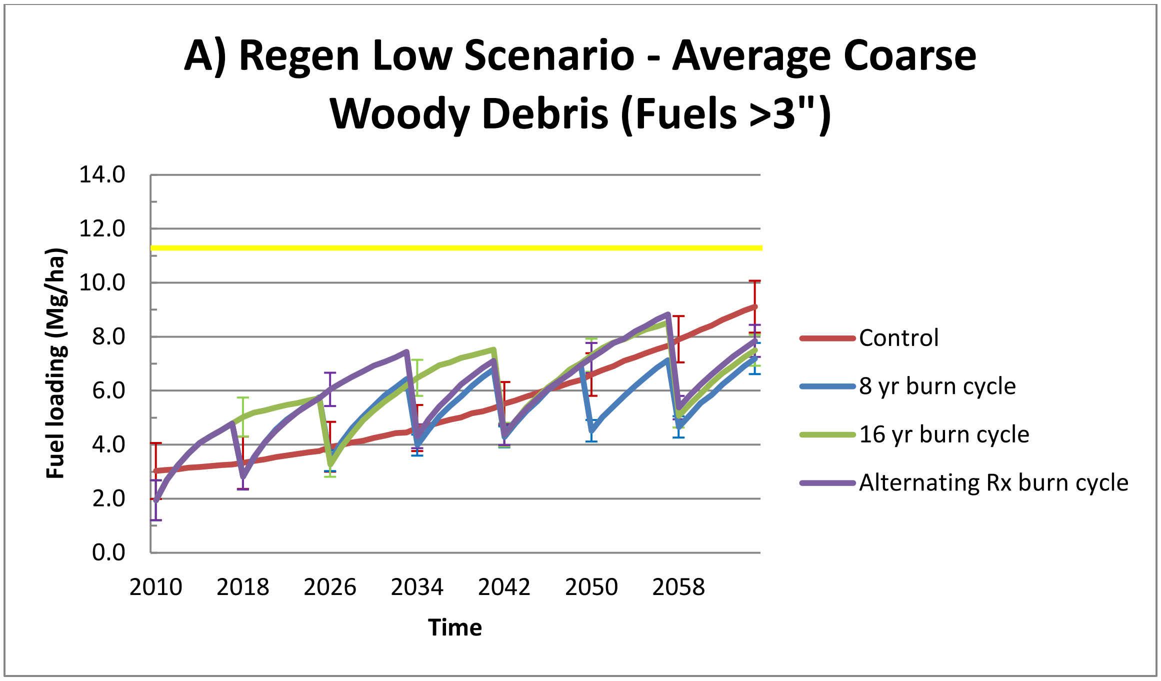


Figure 2.6. Average fuel loading of fine woody debris (fuels 0-3 inches in diameter) displayed through time in Mg/ha under: (A) low regeneration rate and (B) high regeneration rate. Standard error bars are displayed every 8 years. The yellow line represents the fine woody debris fuel loading which is adequate to maintain a surface fire for this fuel model, this ranges from 6.50 – 6.72 Mg/ha (Rothermel 1972; Stevens-Rumann et al. 2012).



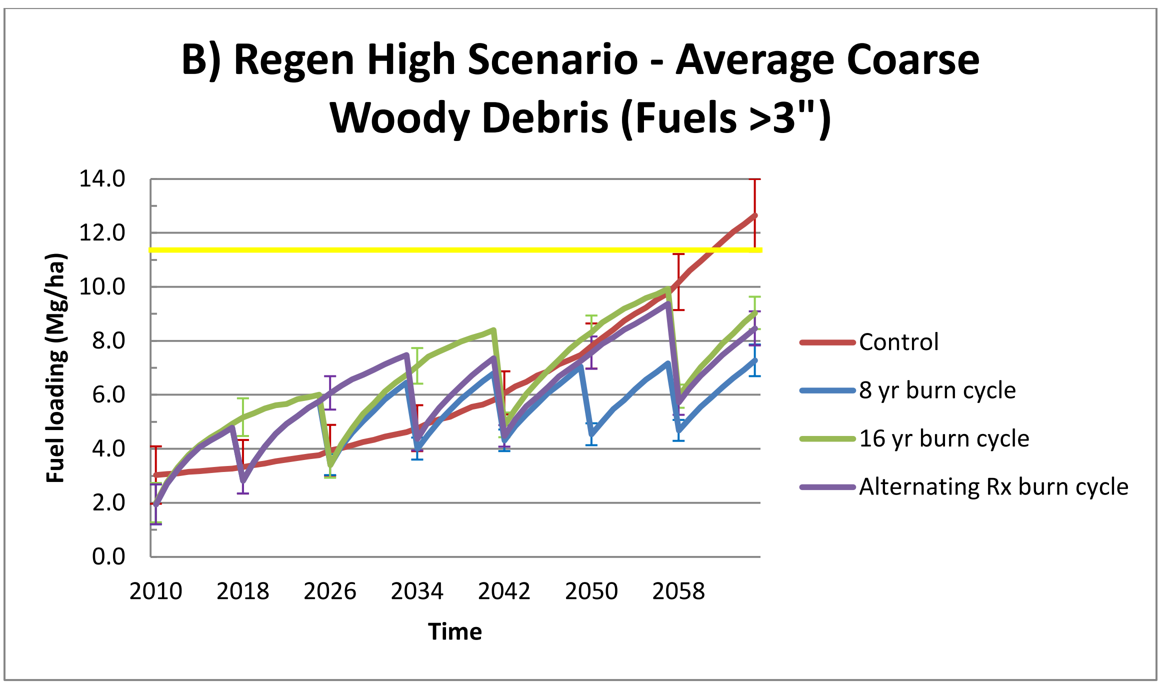
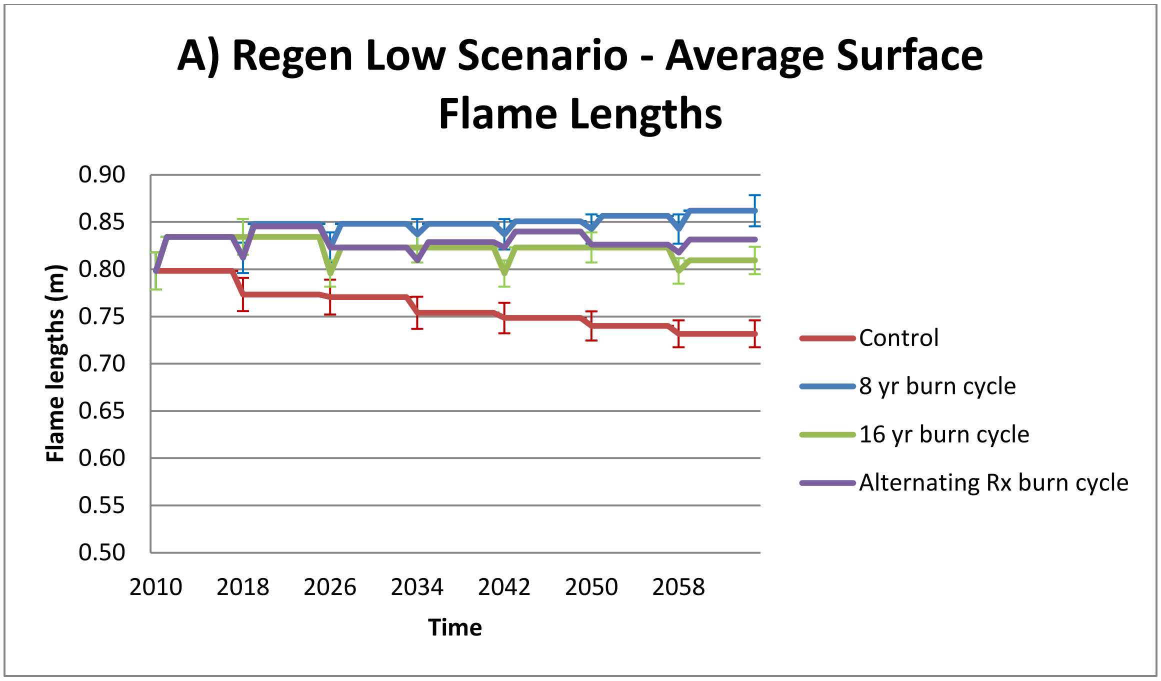


Figure 2.7. Average fuel loading of coarse woody debris (fuels greater than 3 inches in diameter) displayed through time in Mg/ha under: (A) the low regeneration rate and (B) high regeneration rate. Standard error bars are displayed every 8 years. The yellow line represents the lower end of the optimum range of coarse woody debris for dry, warm forest types as 11.2 Mg/ha (Brown et al. 2003). The high end of this range ends at 44.9 Mg/ha (Brown et al. 2003).



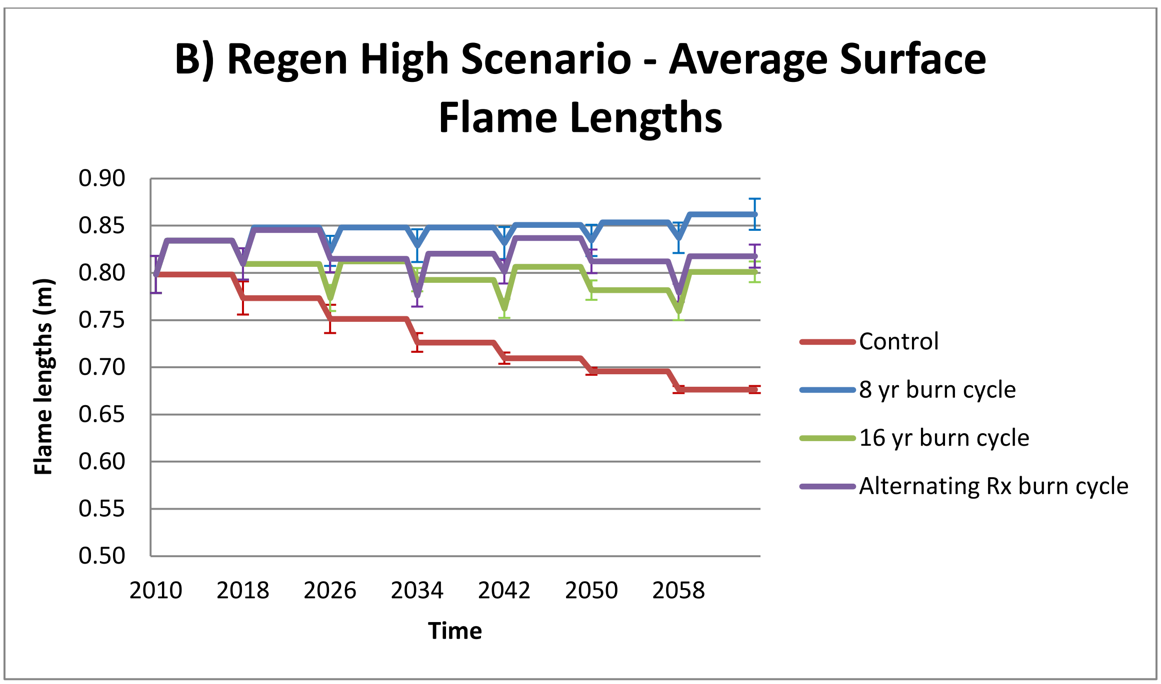
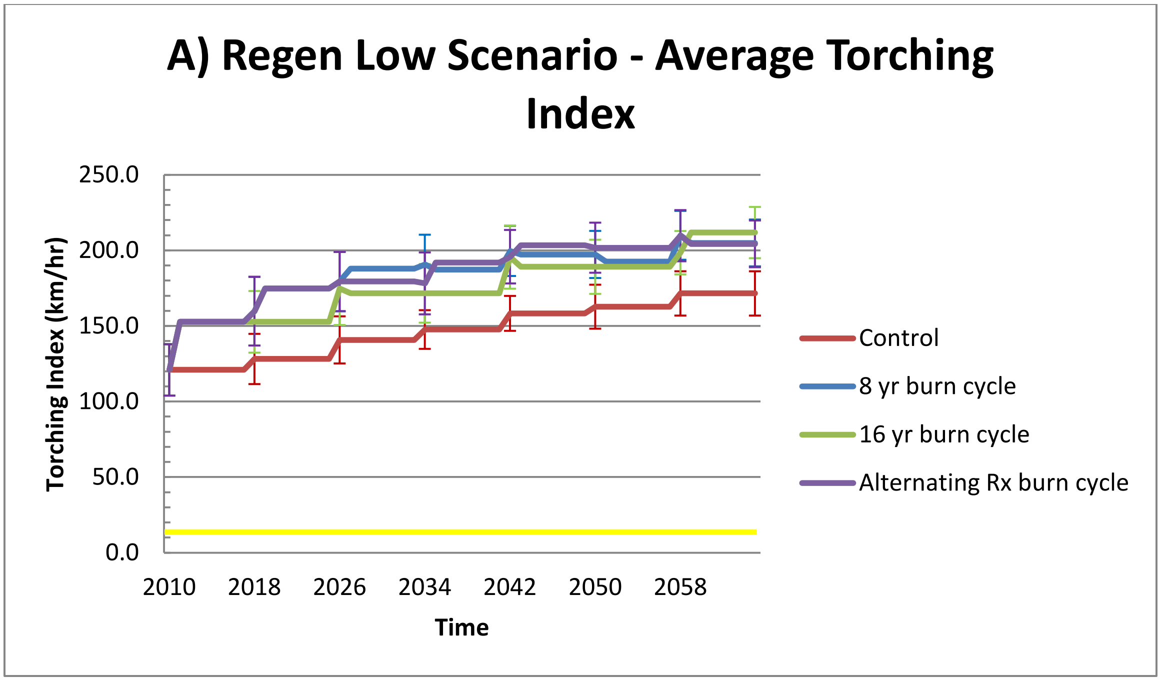


Figure 2.8. Average surface flame lengths (under severe fire weather conditions) displayed through time in meters under: (A) the low regeneration rate and (B) high regeneration rate. Standard error bars are displayed every 8 years. Flame lengths of 1 meter are considered a standard threshold for firefighter safety (NWCG Fireline Handbook, Appendix B, 2006).



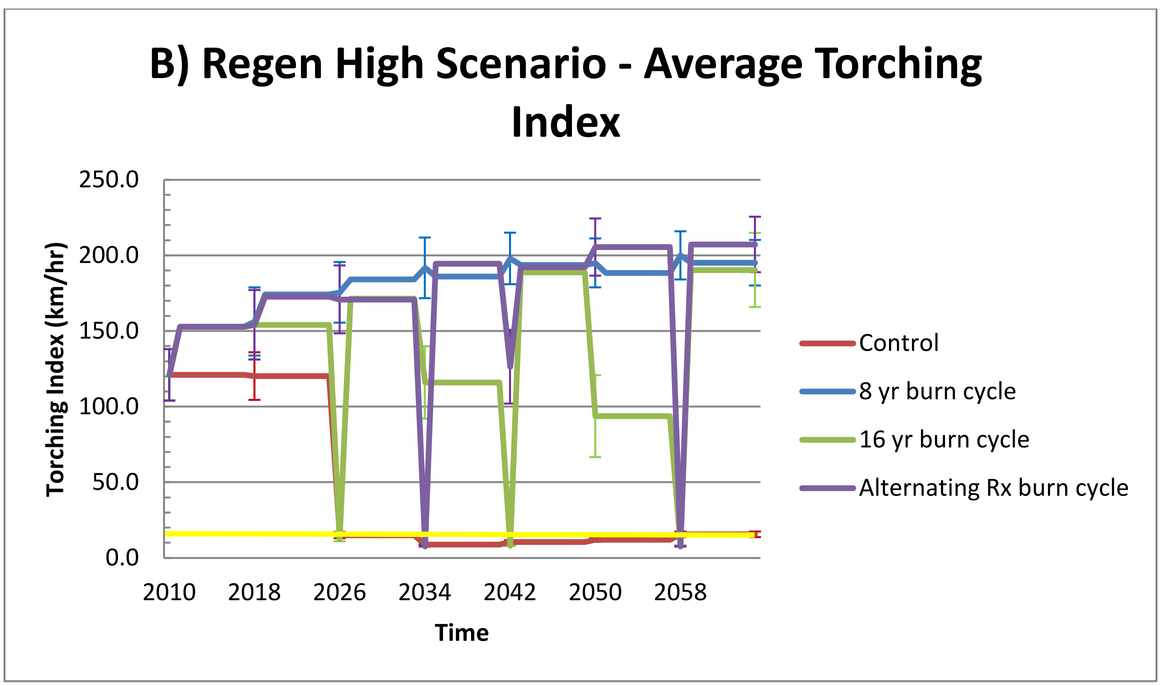
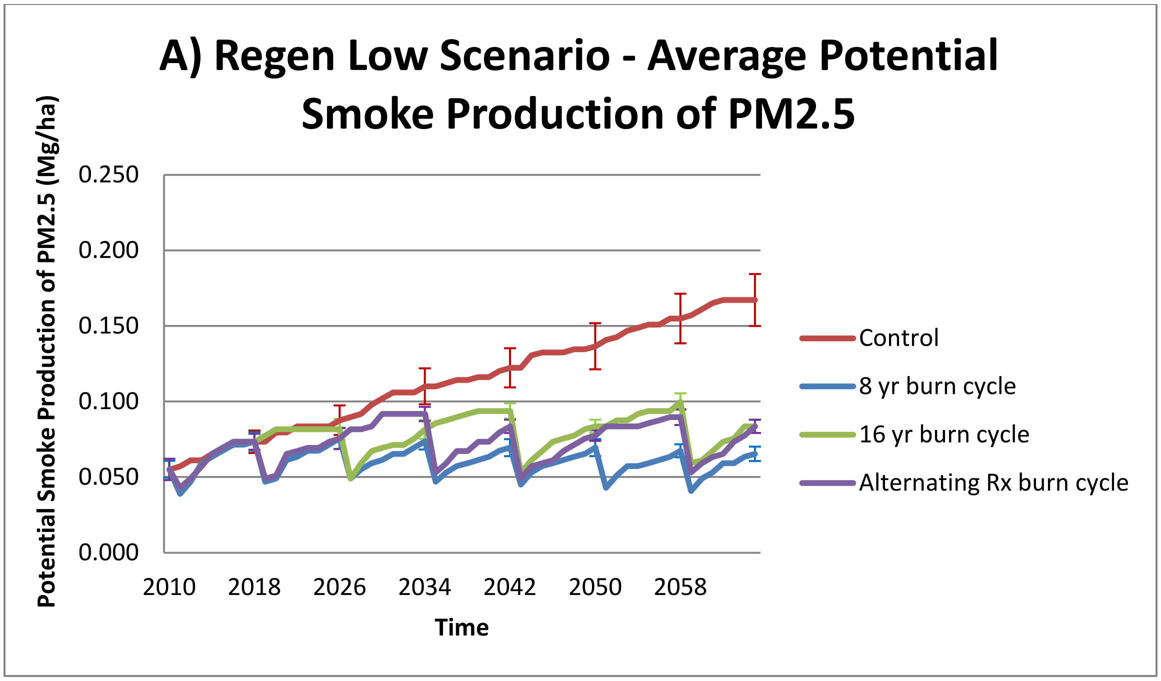


Figure 2.9. Average torching index (under severe fire weather conditions) displayed in km/hr under: (A) the low regeneration rate and (B) high regeneration rate. Standard error bars are displayed every 8 years. The yellow line represents severe fire weather conditions using FireFamily Plus (Bradshaw and McCormick 2000) and data from the Tusayan weather station, which shows 20-foot wind speeds of 16 km/hr.



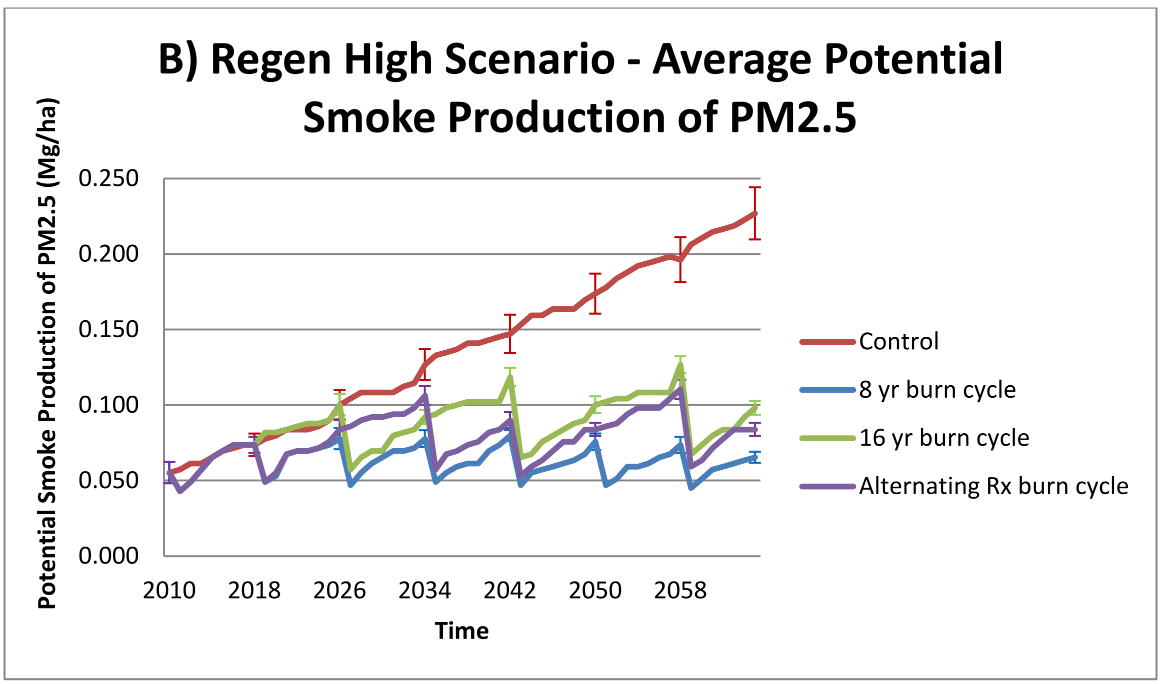


Figure 2.10. Average potential production of PM2.5 (under severe fire weather conditions) displayed in Mg/ha under: (A) the low regeneration rate and (B) high regeneration rate. Standard error bars are displayed every 8 years. No thresholds or desired conditions were identified for this variable.

Chapter Three

Management Implications

Wildfires of increasing severity and extent are becoming more common in recent years in ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) forests in the southwestern USA compared to historical conditions, especially pre-Euro-American settlement in the late 1800's (Westerling et al. 2006; Savage and Mast 2005; Roccaforte et al. 2008). This is largely a result of warmer and earlier springs (Westerling et al. 2006) in conjunction with past management actions such as fire exclusion, livestock grazing, and selective timber harvesting (Covington and Moore 1994a, 1994b; Mast et al. 1999). Many southwestern ponderosa pine forests consequently now exhibit overly dense, even-aged stand structures that are highly susceptible to insect and disease outbreaks and high severity wildfires (Allen et al. 2002; Clark and Sampson 1995; Cooper 1960). Such fires can result in forest type conversions because ponderosa pine has not evolved with high severity fire historically (Swetnam and Baisan 1996; Savage and Mast 2005). Due to the degraded state of these southwestern ponderosa pine forests, managers today are charged with restoring forest structure and function through practices like thinning and prescribed fire.

To restore forest structure and function in Grand Canyon National Park (GCNP) managers have implemented a prescribed burn program since 1980, and they have managed wildfires to meet these resource objectives since 1987 as well (National Park Service 2010). GCNP has also maintained an active fire effects monitoring program since 1990 (National Park Service 2010). The fire management plan (FMP) at GCNP is currently outlined to burn all their South Rim ponderosa pine burn units every 8 years (National Park Service 2012). As part of the adaptive management process, the park is interested in exploring the long-term effects of the current management strategy as well as identifying potential alternative management strategies. In addition to these goals, the current issues and stand structures in southwestern ponderosa pine forests today emphasize the critical need for a solid understanding regarding the effectiveness of prescribed fire in meeting management objectives for stand structure, fuels and fire behavior. Hence this modeling project was intended to provide management feedback on the effectiveness of the current FMP at GCNP by using the fire effects monitoring data as input to the Central Rockies variant (Keyser and Dixon 2008) of the Forest Vegetation Simulator (FVS) (Dixon 2002) model using the Fire and Fuels Extension (FFE) (Reinhardt and Crookston 2003). We modeled different prescribed burn intervals under two contrasting regeneration rates to assess the impacts of different management actions on the land.

The modeling results from this project should be interpreted with caution, as they represent a simplified version of reality. However, despite the caveats of using modeled data for our purposes, modeling with a program such as FFE-FVS certainly provides valuable data by looking at the long-term trends of differing prescribed burn intervals. In this study we evaluated the modeling results against desired conditions or thresholds, where available, for each variable of interest. Through the methods employed in this study we assessed the long-term effects of differing burn intervals on sites experiencing low and high rates of post-fire regeneration. The regeneration rates we input into the FFE-FVS model came from the seedling data collected on the fire effects monitoring plots using Fire Monitoring Handbook (FMH) protocols (National Park Service 2003). The data we used came from plots that have already been burned three times. Thus, we were able to accurately depict a range from low to high post-fire regeneration rates by species. The modeling results from this project suggest that a combination of the low regeneration rate and burning every 8 years (as the current FMP describes) can produce results within desired conditions or pre-settlement reference conditions for stand structure, total surface fuel loading, and potential fire behavior and fire effects.

We concluded from this study that the best fire management strategy may depend heavily on post-fire regeneration. We found that under high post-fire regeneration rates, many specific objectives would be very difficult to achieve with any prescribed burn interval by using low-severity repeated fire alone. Therefore, we concluded that the most frequent burn interval, the 8-year burn interval, would best achieve desired conditions or pre-settlement reference conditions for most variables under both regeneration rates. When simulated under a low regeneration rate, these desired conditions were reached within the 50 years of the FFE-FVS simulations for most variables. Under low post-fire regeneration rates, the 8-year burn interval began to see desired conditions met within 32 years, or four burn cycles/entries, into the future. Therefore, based off these modeling results we propose that a good time for a review of the effectiveness of the current fire management strategy across these sites could be carried out ~32 years into the future. In other words, when evaluating the areas that have already been burned three times, and if four more burn cycles or burn entries were performed with the 8-year burn interval in the future, our results suggest that desired conditions should be achieved for many variables. Specifically, average density objectives, fuels and potential fire behavior should be within desired conditions (or nearly so) at this point. Other elements of the stand structure outside of average density may require further evaluation, such as an assessment of the diameter distribution by species.

When we compared the different burn intervals to each other, the 8-year burn interval produced results that mangers would find encouraging in terms of probable resistance to control of wildfires, potential soil heating, average tree density, potential smoke production, and maintaining low average surface fuel loading consistently over time. The 8-year burn interval can be summarized as the safest scenario from a fire management standpoint. However, the modeling results from the 8-year burn interval also showed a number of potential ecological drawbacks to using this management strategy. The first drawback is that the CWD component with this burn interval would be well below Brown et al.'s (2003) optimum range of CWD. Specifically, Brown et al. (2003) recommended 11.2 to 22.5 Mg/ha of CWD to maintain soil productivity in warm, dry ponderosa pine and Douglas-fir forest types. In addition, the CWD component may not be sufficient to sustain wildlife habitat (Brown et al. 2003). Another potential ecological drawback to using the 8-year burn interval is that this scenario may create a condition with inadequate amounts of FWD to maintain surface fire spread (Rothermel 1972). Thus, fire managers might have difficulty in carrying out prescribed burns that frequently over the long-term.

Therefore, we recommend that if employing the 8-year burn interval, managers may want to take measures to create or preserve higher quantities of CWD during prescribed burning. Higher quantities of CWD could be produced by creating more snags or downed logs during prescribed burns, and existing CWD could be protected by scraping handline around the base of snags. Such activities, in turn, could promote several other aspects of forest health and can provide habitat for wildlife (Brown et al. 2003). Although we acknowledge that wildlife needs for desirable levels of CWD are not well understood and can vary greatly depending on species (Brown et al. 2003). To address the FWD ecological concerns, we recommend that slightly longer intervals in between prescribed burns may be needed to allow for more surface fuel accumulation and successful fire spread.

Despite the fact that the 8-year burn interval was identified as the best overall fire management strategy for achieving specific objectives and for being the safest strategy; the alternating burn interval can also produce results that are desirable on the low post-fire regeneration sites. Advantages to using the alternating burn interval compared to the 8-year burn interval include: fewer resources required, decreased cost to the FMP at GCNP, comparable levels of smoke production compared to the 8-year burn interval, and increased production and/or retention of higher quantities of CWD and FWD. Disadvantages to using the alternating burn interval include the likelihood that specific density targets and stand structure elements may take longer than 50 years to achieve. This is especially true if using low severity fire alone, as our results from this study suggested. Although we recommend that the alternating prescribed burn interval may be utilized on low post-fire regeneration sites, it is probably not a feasible management option on the high post-fire regeneration sites. Lastly, we found that the 16-year burn interval was the least effective at achieving stand structure objectives specifically within the 50 year horizon; and it would likely take even longer than the alternating prescribed burn interval to do so.

Given the wide variety of management objectives and goals for these ponderosa pine forests that we have attempted to address, we recommend that fire management strategies be guided by the observed post-fire regeneration rates. We also advocate for the use of all available information through the implementation of the adaptive management cycle and for continued monitoring on the fire effects plots, as these plots are an excellent way to obtain post-fire regeneration data. In a sense, this project has reaffirmed the importance and the value of the FMH program. By utilizing the data collected with the FMH program this would allow GCNP to achieve one of their central goals outlined in the FMP: to "promote a science-based program that relies on current and best-available information" (National Park Service 2012).

The 8-year burn interval is the most effective at achieving desired conditions for most variables of interest on both sites of low and high post-fire regeneration rates. However, the alternating prescribed burn interval may also be a feasible option on sites experiencing low post-fire regeneration. On these low post-fire regeneration sites, the alternating burn interval produced results that are within or very close to the desired conditions or pre-settlement reference conditions for stand structure, fuels, and potential fire behavior and fire effects. Utilizing the alternating burn schedule would likely take more than 50 years to achieve the desired density conditions, but there could be benefits gained in other areas. These benefits include the production or maintenance of more CWD, fewer resources required over time, and decreased cost to the FMP at Grand Canyon.

In addition to our recommendation that management strategies be guided by post-fire regeneration rates, we also would to like to point out that our study provides some evidence for the use of higher severity prescribed fire. Since our modeling results suggest that it can take up to 32 years to achieve desired density conditions on sites with low post-fire regeneration rates, even by adhering to the current FMP of burning every 8 years, we propose that employing slightly higher severities than typical prescribed burns may allow for much quicker changes in the stand structure. Average tree density desired conditions may be achieved in a shorter timeframe. Thus managers would likely see desired conditions achieved in other variables of interest in a shorter timeframe as well, since fuels and fire behavior are partly driven by stand structure. These results are similar to what Fulé et al. (2004) reported on the North Rim. In addition, other research has reported that dense thickets of old yet small diameter trees (due to dense growing conditions) are very resistant to mortality from low severity surface fires – further supporting the use of higher severity fires to achieve density objectives in some areas (Peterson et al. 1994; Sackett et al. 1996). Although the implementation of higher severity prescribed fires may present managers with a new set of challenges and risks, the ecological benefits to these forests may be warranted in some areas where overly-dense stand structures dominate.

It is important to properly manage and/or restore both the structure and function of a healthy ponderosa pine forest to these South Rim forests. Through this project, we looked at the combined effects of repeated, low severity burning at various intervals followed by high or low rates of post-fire regeneration. Explaining the patterns and results from modeling projects such as this one are not always simple and straightforward, they are oftentimes more nuanced. Nonetheless, these results provide managers with a means to evaluate the effectiveness of the current FMP and to identify possible alternatives to the currently employed management strategy on certain sites. The modeling data from this project can be used to answer different questions in the future, such as those focusing on specific requirements for wildlife habitat for a species of interest or to compare these results to the effects of a simulated thinning regime or to a thin-and-burn type scenario. The relationship between management actions and the influences or affects we can have on natural processes (e.g. post-fire or post-disturbance regeneration) across a landscape are dynamic and long lasting. We hope that the data provided in this study will help the reader to think about the long-term effects of land management and how such actions can affect regeneration, stand structure, and the resultant fuels and potential fire behavior or fire effects.

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Appendix A –Creating files that are ready for use with FVS from the Grand Canyon FMH database (Using FFI version 1.04.01 export utility)

All the years of FMH data are currently stored in GCNP's FEAT/FIREMON Integrated (FFI) database. There is an FVS export utility built into FFI (found under the "Reports and Analysis" tab in FFI) that creates Microsoft Office Access Database files containing most of the FMH plot data required to run simulations in FVS. The Access database files are created in conjunction with a .loc file, a .kcp file, and a .slf file when using the export function in FFI. All file types are required to be saved in the same folder in order to work properly in FVS.

There was an extended period of troubleshooting in order to get these files to all work as they should in FVS. Initially, the Access database files that were being exported/created by FFI were blank files or were missing data. In some cases, the data that was exported were simply columns of 0's and 1's indicating an absence or presence of data collected in the field. Another example of some of the issues I ran into was when I noticed that the sub-plot fraction wasn't indicated correctly in the TreeCount column in the database tables, essentially it was representing the sampling protocol for both the overstory trees and the pole trees as being the same across the whole plot. When in fact, pole trees are only recorded or sampled in two quarters of the FMH plots whereas the overstory trees are recorded across the whole plot in all four quarters. Such issues required fixing before any simulations could be run in FVS due to the effects that density and tree counts, or missing data would obviously have on the accuracy of the simulations. However, after working in close conjunction for several months with two individuals (Duncan Lutes from the U.S. Forest Service for FFI support, and John Caratti from the

Systems for Environmental Management development team) in charge of support for the National Park Service's FFI software program, I was able to obtain nearly ready-to-use Access database files using this export utility. Also a newer version of FFI was released in the early spring of 2011, incorporating most (but not all) of the changes that I had discussed would be a significant help to FVS users. The Access database files now exported out of this latest version of FFI no longer have major issues such as missing data or blank files. They now contain the data necessary for FVS to read them and treat each FMH plot as a stand in the FVS simulations. As I mentioned previously, they are *nearly* ready-to-use files, but they do still require some corrections for FVS to fully read all the FMH data available and to avoid error messages when running FVS.

When creating files from FFI version 1.04.01, here are the corrections that are necessary to complete for properly functioning files in FVS.

1. The FVS_GroupAddFilesAndKeywords table must be copied and pasted into the Access database. A default copy was obtained originally from Dr. Kristen Waring in the School of Forestry at Northern Arizona University. A copy of this table can be provided upon request.
2. The default name for the .mdb database should be changed and simplified under the FVSKeywords section of the FVS_GroupAddFilesAndKeywords table, i.e. take out all the directory structure and just leave the name of the database itself (for example, the default C:\My Documents\...etc.\PIPO_03YR01.mdb should be changed to PIPO_03YR01.mdb)

3. Under the FVS_TreeInIt table select the entire Species column. Using the Find and Replace functions on the toolbar, remove all the 1's after the species codes (e.g. PIPO1 should be changed to PIPO). Save your changes.
4. Next, manually enter in the seedling data for each plot included in that database file. (FFI does not currently export *any* of the seedling data into the FVS_TreeInIt table because the seedlings are collected under a separate FMH protocol.)
    a. For each seedling enter a diameter of 0.1 inches, assign the appropriate code for live (1) or dead (8), enter the height in feet, enter the species code, and under the Tree_Count column show the corresponding density of that line of data from FFI by using the correct trees/acre expansion factor. (For example, if 1 seedling was recorded, it would need 16.188 to be in the Tree_Count column. If 2 seedlings were recorded, it would be 32.376 and so on). Don't forget, all height measurements will need to be converted from metric (as they are exported from FFI) into English units for FVS.
    b. Take care to assign the seedling data to the correct plot if more than one plot is included in the database.
5. Under the FVS_StandInIt table, add a new column for the variant (name it Variant) and enter in CR for the Central Rockies variant.
6. Scroll to the PV_Code column under the FVS_StandInIt table, and change this default code for the habitat type to 011030 for all plots that fall within the South rim ponderosa pine forest type. This code stands for ponderosa pine with a blue grama understory.

7. Save *all* the changes made to *all* tables within the Access Database file.
8. Open up the .kcp file corresponding to the .mdb file (opens in Notepad). On the third line of data, take out all the directory structure to exactly match how the .mdb database is named in Microsoft Access from step #2 above. Save your changes.

Appendix B – Accuracy check of the FVS modeled stand structure attributes (data from the 2009 and 2010 field seasons)

In an effort to provide further management assistance for the fire effects monitoring program at Grand Canyon National Park, I compared how well FVS was estimating the tree heights when tree height data is not included in the Access database files to run FVS simulations. I also compared FVS modeled crown ratio values to field collected data. My goal was to obtain an idea of how accurately FVS is handling basic growth predictions of the dominant tree species on the South Rim. Thus, I hoped to lend a level of confidence to future FVS users for the Park (and in the region) if additional parameters such as tree height are not collected in the field or are unavailable for any reason. Total tree height, crown base height, and live crown ratio are all recommended parameters to collect in the field according to the FVS user's manual (Dixon 2002). However, when such data is not included, FVS uses dubbing submodels to estimate these stand structure descriptors (Dixon 2002). FVS uses species-specific height to diameter relationships to estimate these variables (Dixon 2002).

During the 2009 and 2010 field seasons, the fire effects monitoring crew at Grand Canyon collected tree height, crown base height, and live crown ratio on all FMH plots that were read according to the regular FMH plot read schedule. Data was collected on 847 trees during these two years, and included data on all four South Rim tree species. To compare the modeled to the field collected values, I removed the tree height, crown base height, and live crown ratio data from the Access database files I was using to run FVS simulations. Without this data, I ran a simple growth simulation (no disturbance) in FVS for all the plots. Next, I pulled out the tree heights that FVS predicted in the initial year of the simulations to compare these values to the field collected tree height data. I compared the FVS modeled values to field collected values on a species by species basis. This relationship was displayed in graphical form, and then was fit with linear regression equations to obtain an R^2 value. Refer to Figures B.1 through B.4. Two outliers representing two ponderosa pine trees were discarded from the tree height analysis. This was due to probable error in the field resulting in unlikely (very tall) tree height values.

The relationship is very strong for the ponderosa pine species (Figure B.1). With an R^2 value of 0.8555 using linear regression and using a sample size of 623, this indicates that FVS is predicting the tree heights of ponderosa pine accurately. However, the relationship between FVS modeled tree heights and the field collected tree heights was not that strong for the pinyon pine, Utah juniper, or Gambel oak tree species (Figures B.2, B.3, and B.4). These tree species exhibit growth patterns that are more difficult to correlate with their diameters and thus the tree heights are harder to predict. Parresol and Stedman (2005) reported similar findings when checking on the accuracy of FVS height predictions, that some species were accurately represented in the FVS program while others were not (discussed in Chapter One, Literature Review).

The same comparison between FVS modeled and field collected data was done for the crown ratio values. This comparison was also done on a species by species basis. These data are displayed in Figures B.5, B.6, B.7 and B.8. The crown ratio analysis included fewer trees than the tree height analysis because crown ratios were not collected on the pole-sized trees or dead trees; although tree heights were collected on both poles and dead trees. The relationship between FVS modeled crown ratios and the field data is not very strong for any of the tree species. When crown ratio data is missing from the FVS tree data input records, FVS uses embedded equations based on tree and stand attributes to predict a crown ratio (Dixon 2002). However, an important caveat is that these crown ratio equations are not calibrated from the input data (Dixon 2002). This is unlike the tree height equations which, if collected on at least three trees per plot/stand, are calibrated using the diameters of the sample trees and the tree heights available (Dixon 2002). The fact that the crown ratio equations are not calibrated using input data is likely the reason for the patterns that we see in Figures B.5, B.6, B.7 and B.8. Many of the crown ratio values were predicted in a similar range for many of the sample trees of each species. Thus, the relationships are not very strong for any of the species analyzed since there is much more variability in the field which FVS is unable to represent due to the method of crown ratio calculation.

In summary, I would recommend that future FVS users should not be too concerned about collecting ponderosa pine tree heights in the field if time and money are running short. However, if tree height data could be sub-sampled (collecting the data on at least three trees per plot), that would enable to FVS to calibrate the tree heights according to the input data available. For the other species, given the weak prediction capability of tree heights for Utah juniper, pinyon pine, and Gambel oak; I recommend continuing data collection on these species' tree heights if at all possible. FVS tended to predict much taller tree heights for these three species than the field collected values often reflected. One important qualification regarding these recommendations is the fact that ponderosa pine represented the largest sample size of all the tree species present. Therefore, by performing this same accuracy check with a larger sample size of the other three tree species that are simply less prevalent across these forests, one might find a different result in the accuracy of those FVS predictions. Furthermore, I recommend that crown ratio values continue to be collected in the field for all tree species. The FVS embedded equations are not calibrated using the input tree data, so sub-sampling would not be of benefit. Since the relationships between FVS modeled values and the values collected in the field are very weak, this suggests that there is more variability in the field than FVS is capable of accurately predicting.

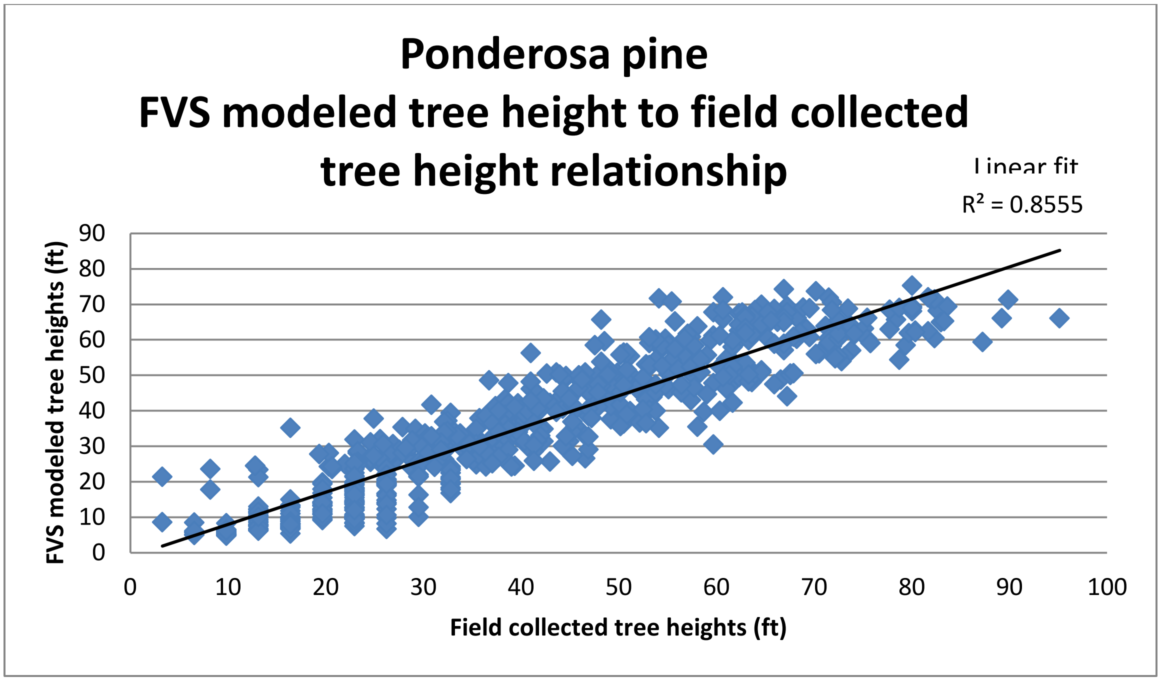

Figure B.1. Ponderosa pine species comparison of FVS modeled tree heights to field collected tree heights. Linear fit, $R^2 = 0.8555$, n = 623

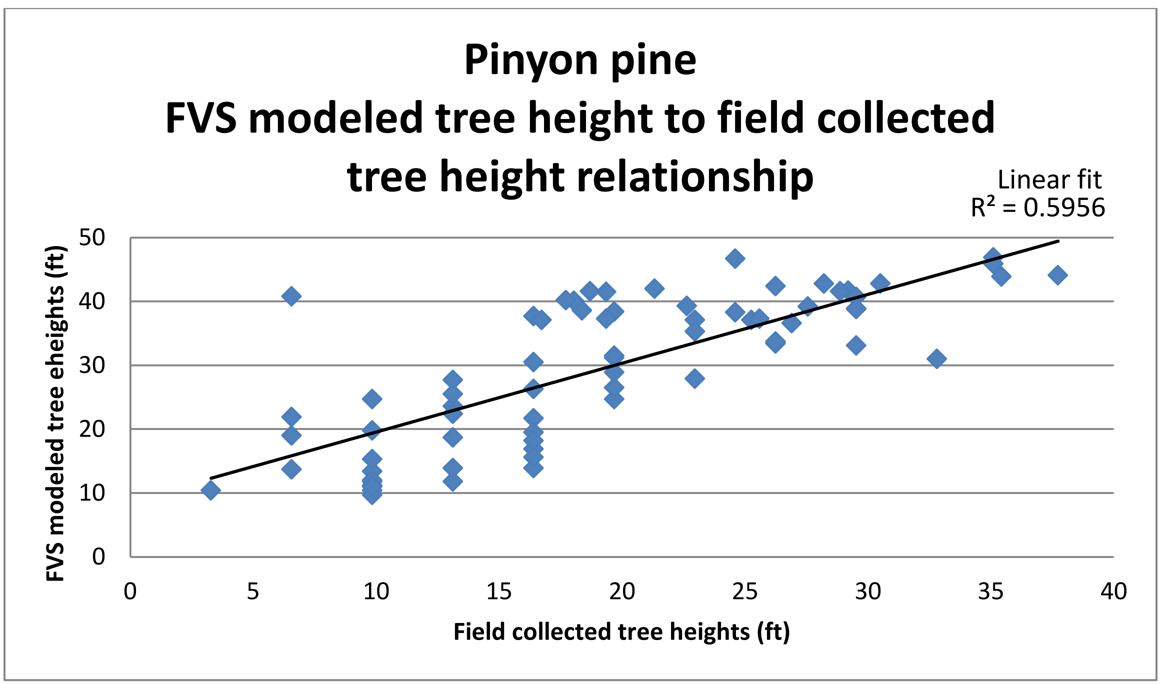

Figure B.2. Pinyon pine species comparison of FVS modeled tree heights to field collected tree heights. Using a linear fit, $R^2 = 0.5956$, n = 72

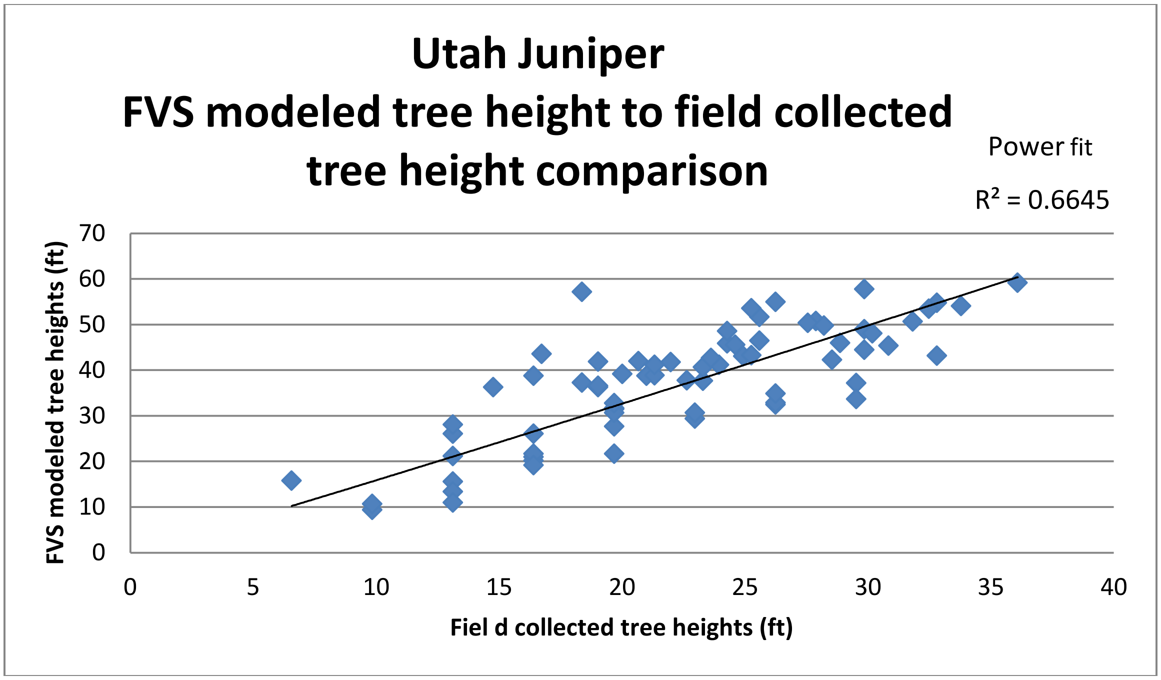

Figure B.3. Utah juniper species comparison of FVS modeled tree heights to field collected tree heights. Using a power fit, $R^2 = 0.6645$ (Using a linear fit, $R^2 = 0.6324$) n = 71

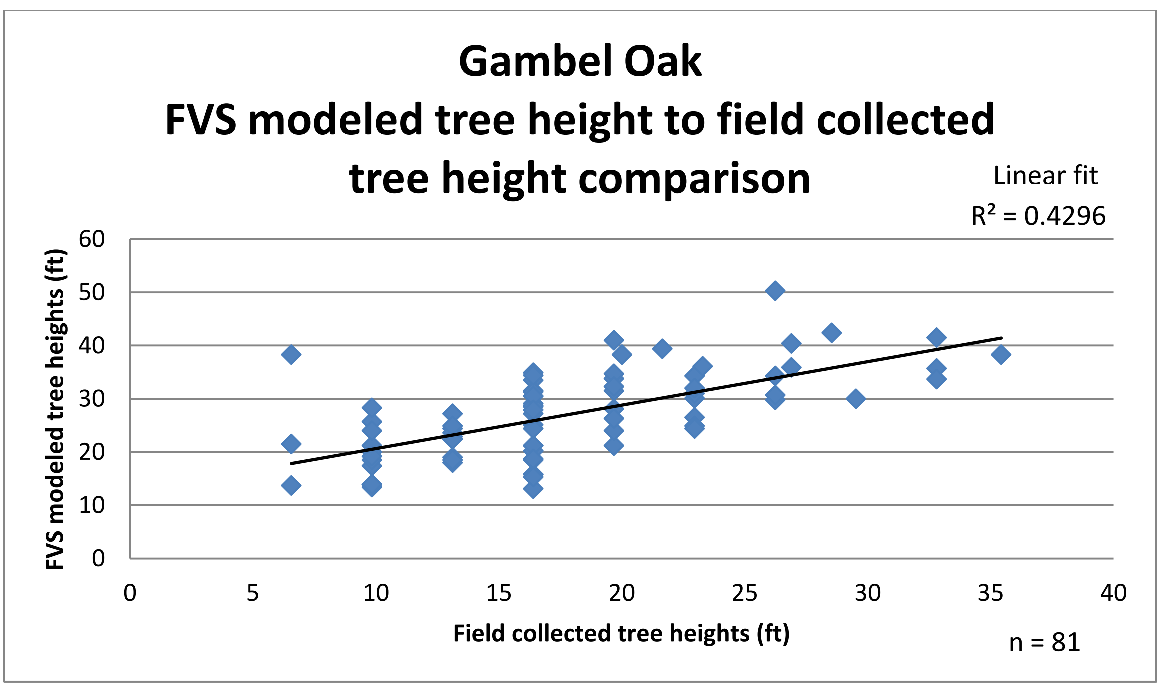

Figure B.4. Gambel oak species comparison of FVS modeled tree heights to field collected tree heights. Using a linear fit, $R^2 = 0.4296$, n =81

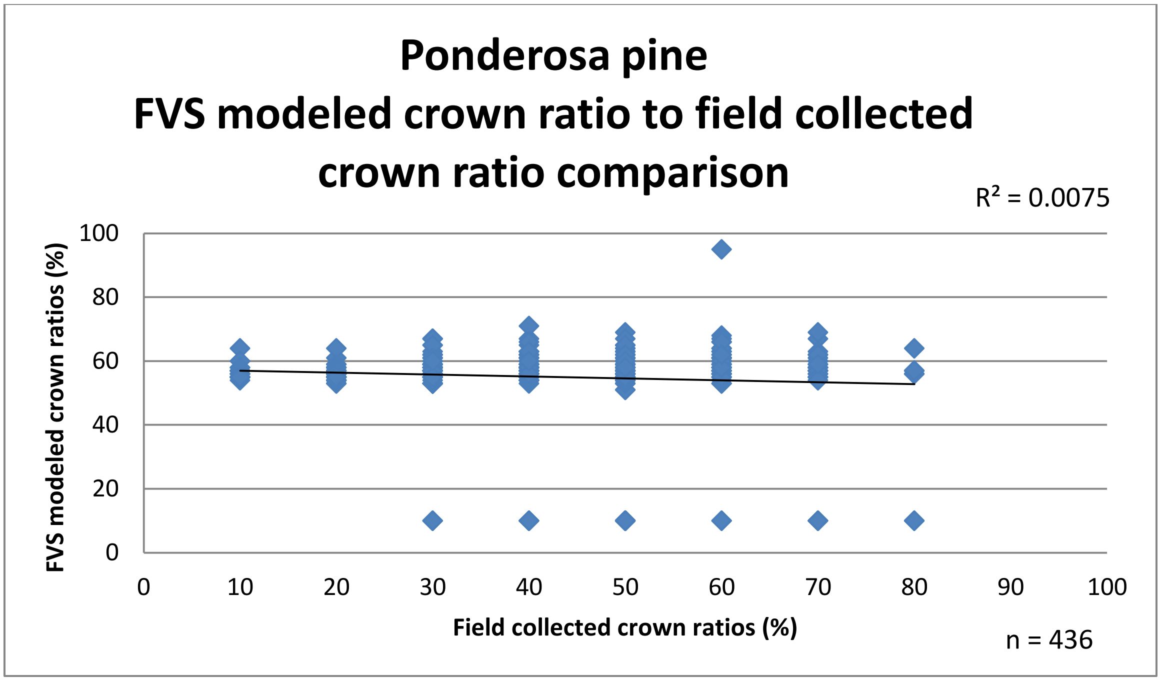

Figure B.5. Ponderosa pine species comparison of FVS modeled crown ratios to field collected crown ratios. Using a linear fit and a sample size of 436, the $R^2 = 0.0075$.

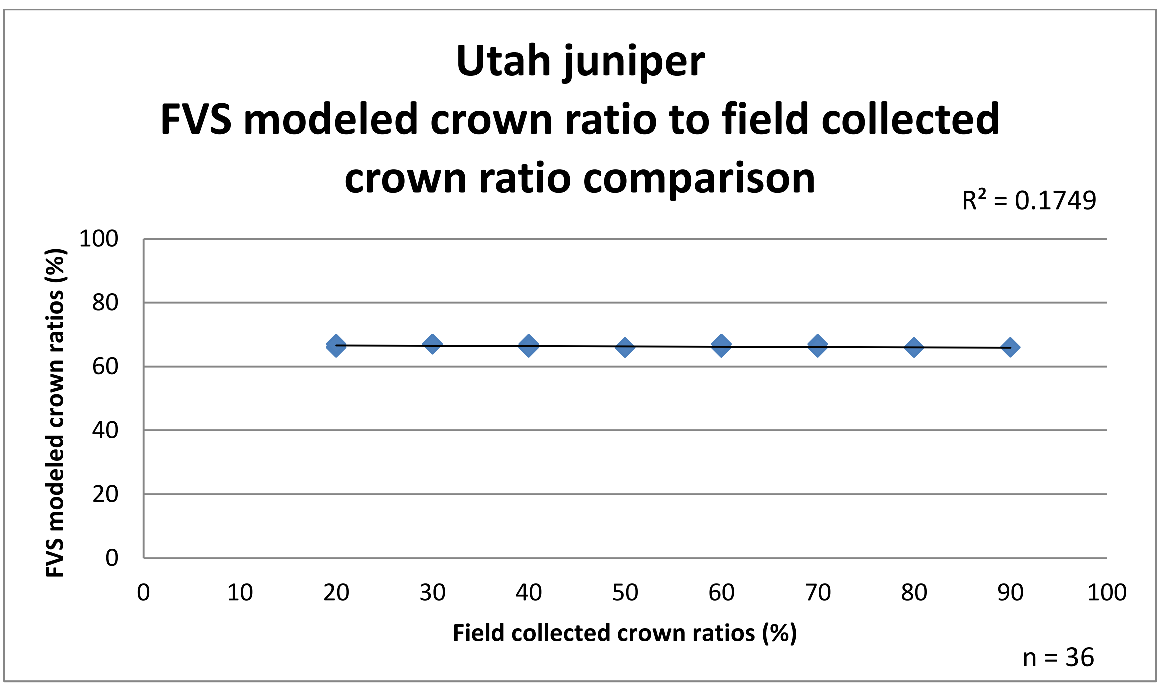

Figure B.6. Utah juniper species comparison of FVS modeled crown ratio values to field collected crown ratios. Using a linear fit and a sample size of 36, $R^2 = 0.1749$

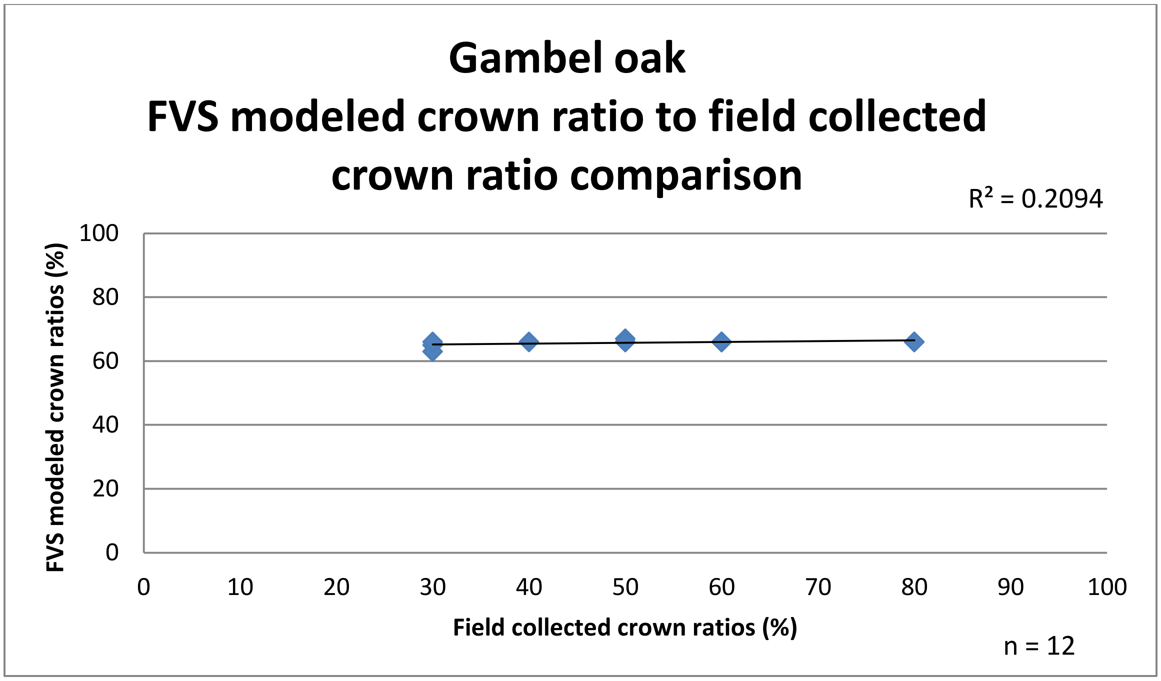

Figure B.7. Gambel oak species comparison of FVS modeled crown ratio values to field collected crown ratios. Using a linear fit and a sample size of 12, $R^2 = 0.2094$.

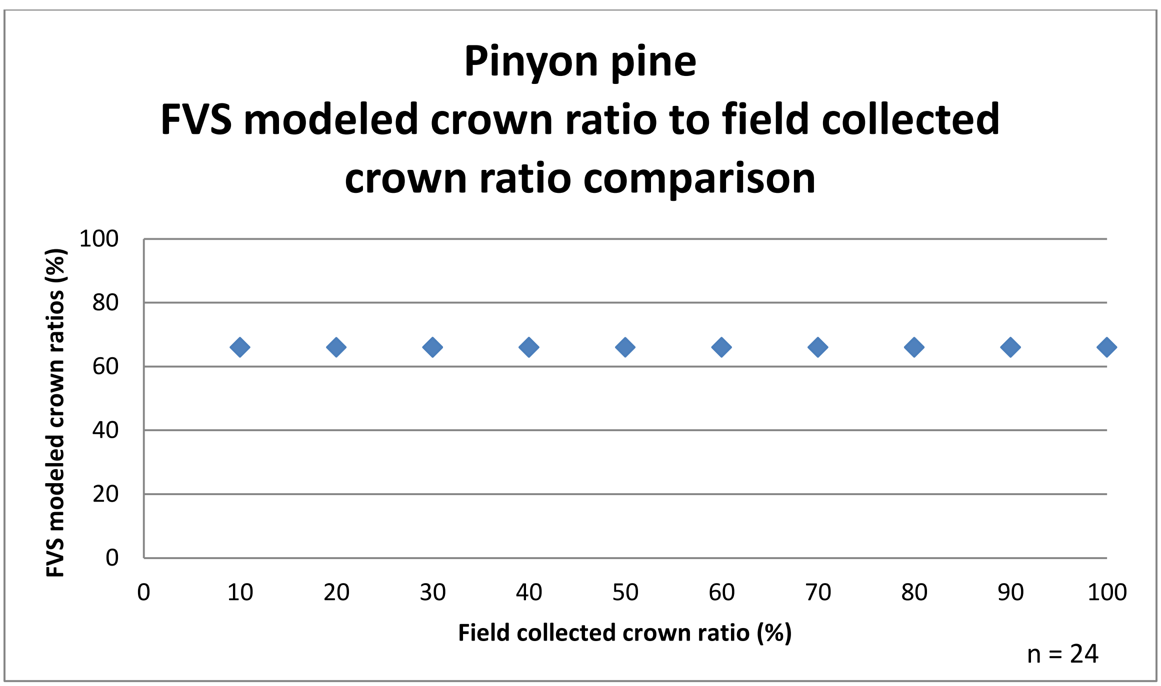

Figure B.8. Pinyon pine species comparison of FVS modeled crown ratio values to field collected crown ratios. Sample size was 24 and no suitable fit could be found. All the pinyon pine crown ratio values were modeled to be the same (66%). The field collected data displays a large degree of variability in crown ratios.

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Appendix C – Diameter distribution changes over time

The changes in diameter distributions over time were assessed by species (Figures C.1 to C.4). With the inventory conditions at the beginning of the simulation (Figure C.1) Gambel oak made up over 90% of the species composition in the smallest diameter class in these stands (data not shown). There was also a small component (5-10%) of pinyon pine (*Pinus edulis* Engelm.) and Utah juniper (*Juniperus osteosperma* (Torr.) Little) in the smaller diameter classes. Pinyon pine was present in low densities up into the 25 cm diameter class range, and Utah juniper is present in low densities throughout all the diameter classes (less than 20 trees/ha except for the smallest diameter class, where it is more abundant). Ponderosa pine made up nearly 100% of the species composition in the larger diameter classes (>41 cm), with just over 100 trees/ha total. The largest diameters of ponderosa pine present at this time are in the range of 81-86 cm in diameter; although the next size class down from this diameter range, 76-81 cm, had no ponderosa pine (data not shown).

The shifts in diameter distributions to the end of the simulations are given for each of three burn scenarios under the low and high regeneration rates in Figures C.2 to C.4. The 8-year low scenario filled in the missing gap in the 76-81 cm diameter range and continued to grow ponderosa pine trees up into the 91-97 cm diameter class (data not shown). Species composition strongly favored ponderosa pine throughout all the diameter classes, except for the smallest class which was composed of nearly equal parts pinyon pine, Utah juniper, and ponderosa pine. This is due to the regeneration methods employed in FVS. The 8-year high scenario produced a similar pattern of the ponderosa pine continuing to grow into the same larger diameter classes, but this scenario had higher densities in the smallest diameter class. There was ten times more ponderosa pine present in the smallest diameter class in the 8-year high than in the 8-year low scenario. There was also about nine times more pinyon pine and seventeen times more Utah juniper in the smallest diameter class in the 8-year high scenario compared to the 8-year low scenario. The 16-year low scenario produced similar diameter distribution results to the 8-year low scenario, except that the 16-year scenario had much higher densities of each species in the smaller diameter classes in comparison to the 8-year scenario. The same is true for the 16-year high scenario compared to the 8-year high scenario, with the 16-year high scenario producing even higher densities. The alternating prescribed burn scenario results are very similar to the 16-year burn scenario results when comparing the low and high regeneration rates to each other. Furthermore, there was little to no Gambel oak present by the end of any of the simulations for all three burn scenarios.

In summary, these changes in the diameter distributions show that the low regeneration rate scenarios are decreasing in the number of trees in the smallest diameter class. There is some growth in the density of large diameter trees (>41 cm), indicating that the pre-settlement trees are being retained for the most part and the mid-size trees are growing up into this diameter class. Also, the species composition strongly favors ponderosa pine throughout all the diameter classes except for the smallest diameter class. The biggest difference in the high regeneration rate end-of-simulation figures is the density of pinyon pine trees in the smallest diameter class. FVS appeared to be creating higher mortality rates in the Utah juniper and ponderosa pine species in this smallest diameter class; so that the density was not evenly distributed across all three species, as was the case with the low regeneration scenarios.

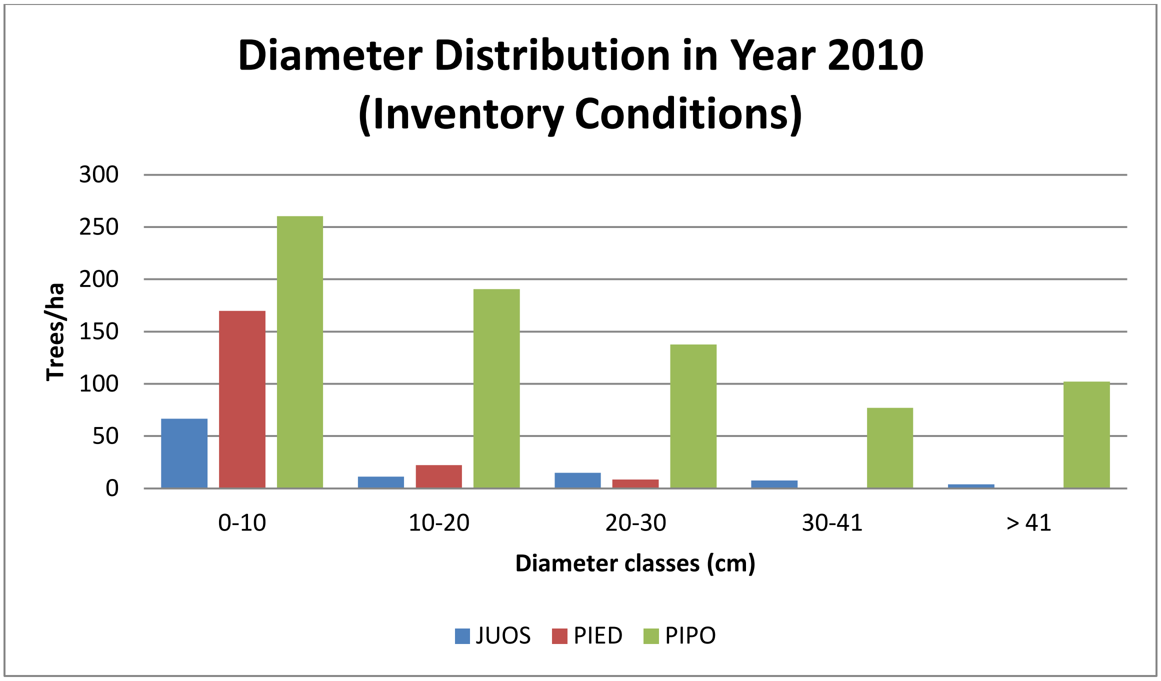


Figure C.1. The diameter distribution displayed for each species in the year 2010, inventory conditions. Diameter classes are broken up into ~10 cm classes, and anything larger than 41 cm is grouped together in the largest diameter class. The trees/ha displayed are the total trees/ha for each species within each diameter class. Species codes are as follows: JUOS = Utah juniper, PIED = pinyon pine, and PIPO = ponderosa pine. QUGA is present but the density is not shown.

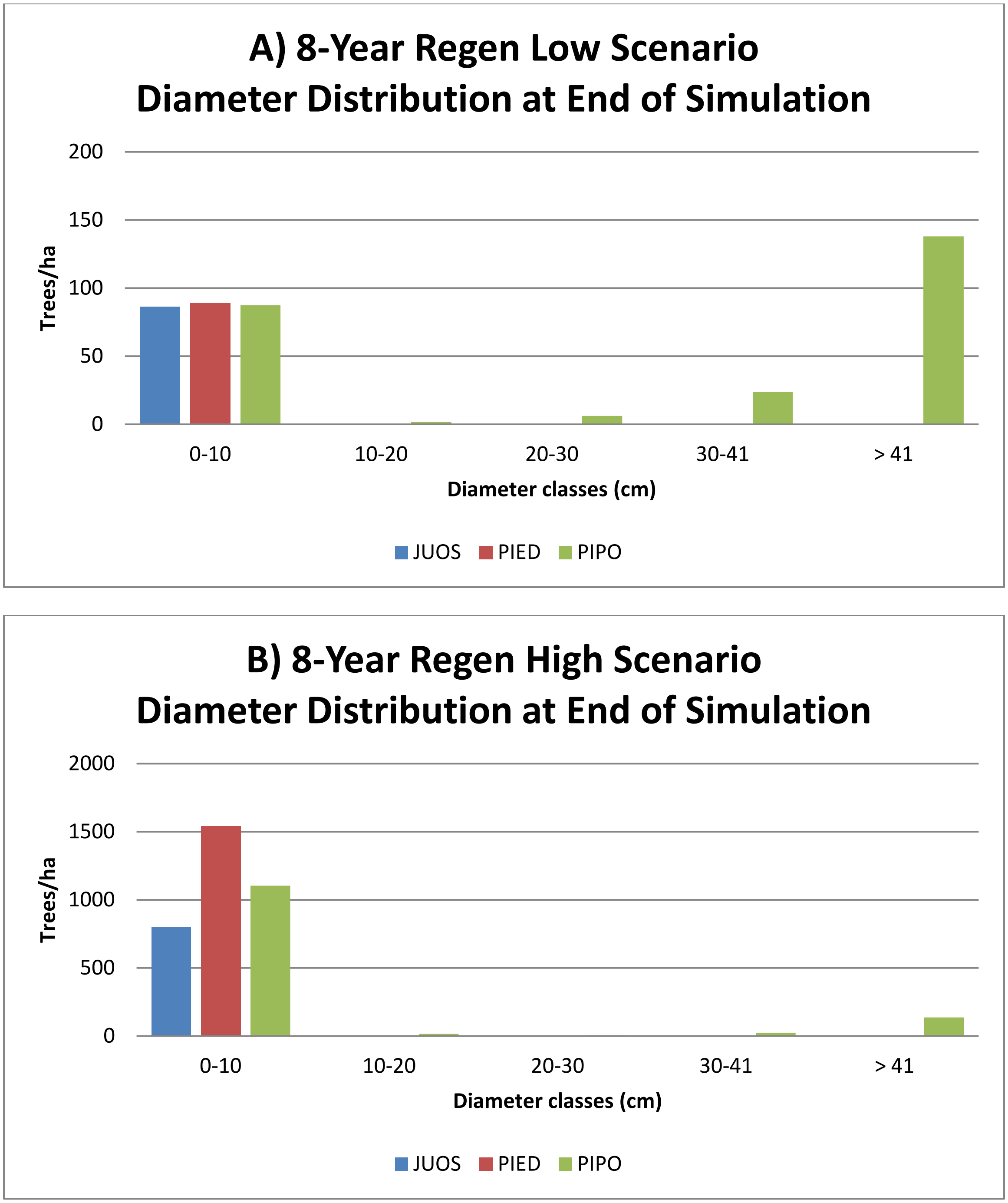
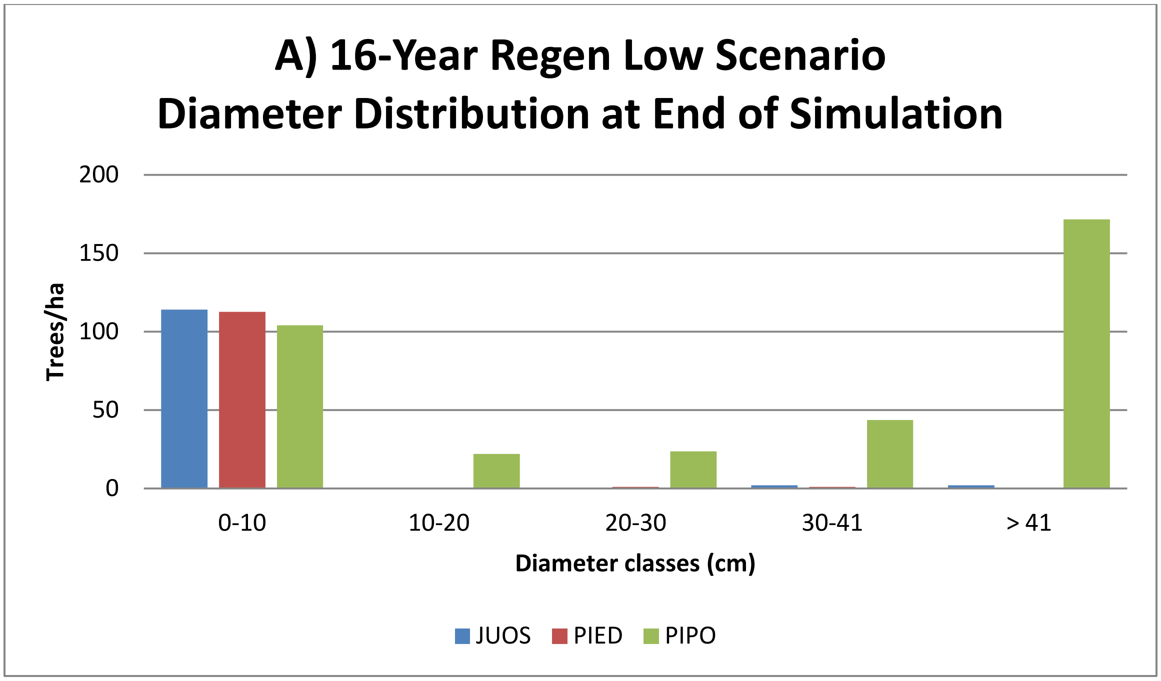


Figure C.2. Diameter distribution at the end of the simulation (year 2066) for the 8-year prescribed burn interval under: (A) low regeneration rate and (B) high regeneration rate. Diameter classes are broken up into ~10 cm classes, and anything larger than 41 cm is grouped together in the largest diameter class. The trees/ha displayed are the total trees/ha for each species within each diameter class. Note the different scales in the y-axes from (A) to (B).



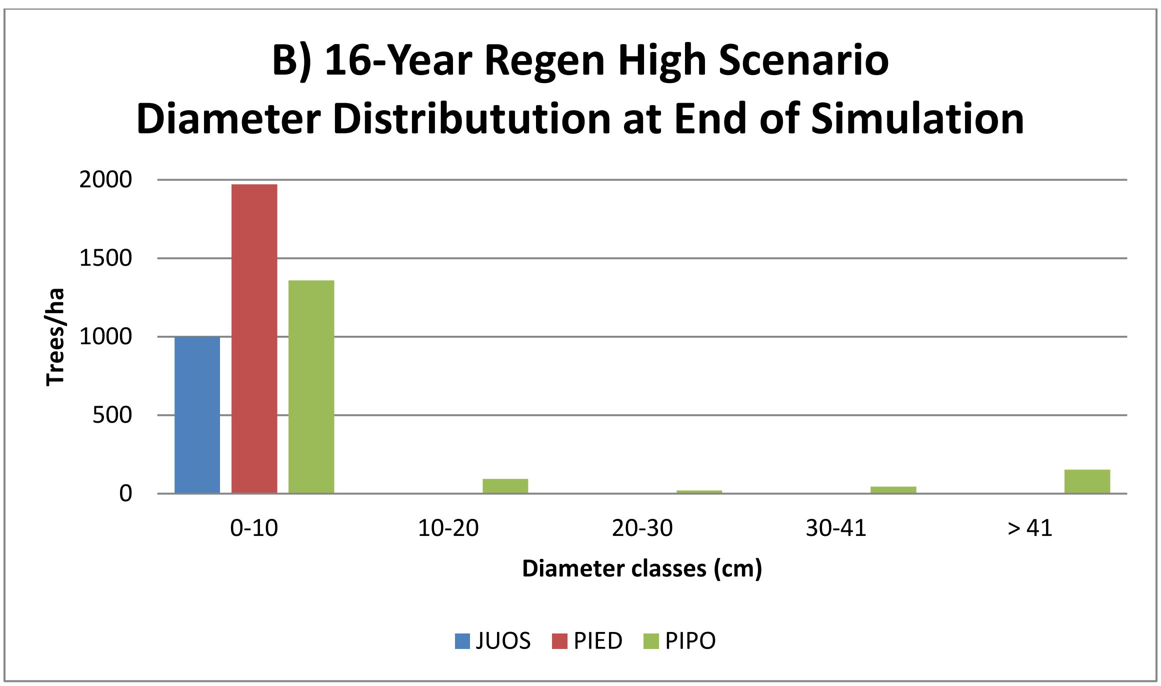
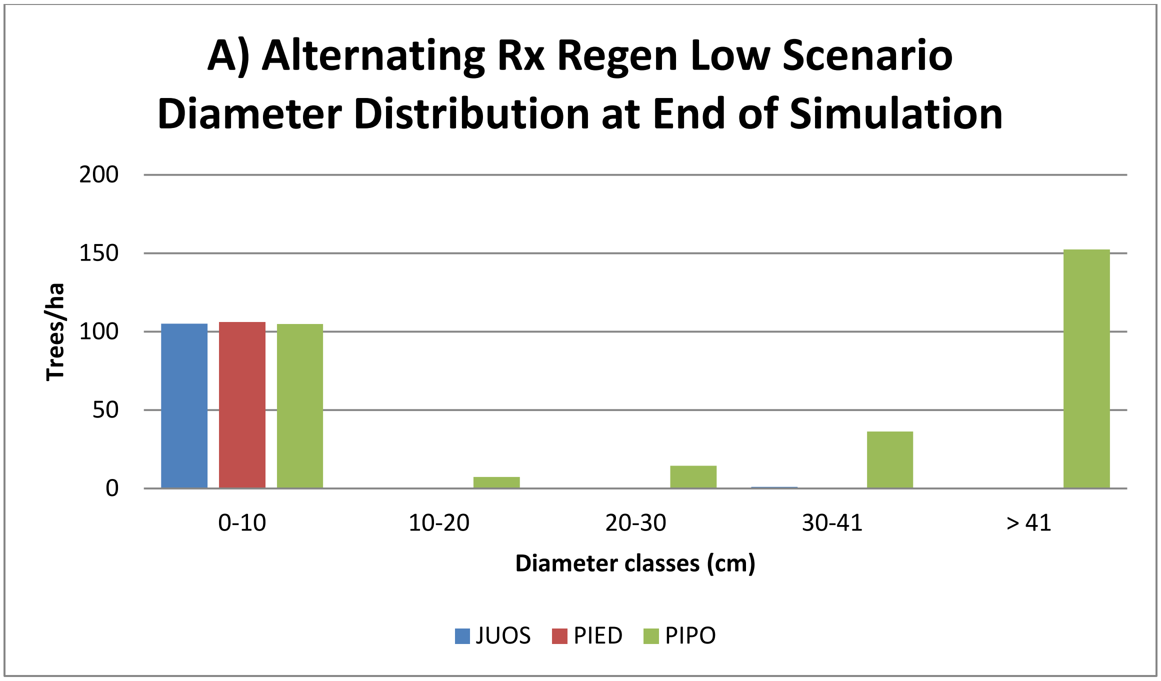


Figure C.3. Diameter distribution at the end of the simulation (year 2066) for the 16-year prescribed burn interval under: (A) low regeneration rate and (B) high regeneration rate. Diameter classes are broken up into ~10 cm classes and anything larger than 41 cm is grouped together in the largest diameter class. The trees/ha displayed are the total trees/ha for each species within each diameter class. Note the different scales in the y-axes from (A) to (B).



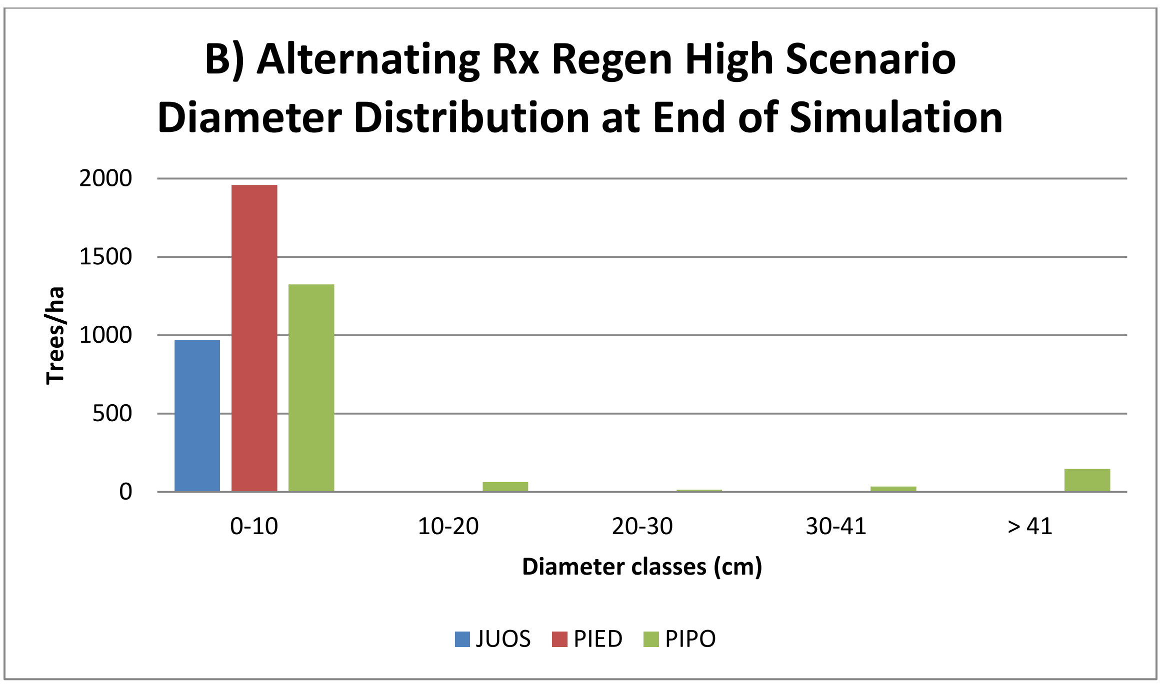


Figure C.4. Diameter distribution at the end of the simulation (year 2066) for the alternating prescribed burn interval under: (A) low regeneration rate and (B) high regeneration rate. Diameter classes are broken up into ~10 cm classes, and anything larger than 41 cm is grouped together in the largest diameter class. The trees/ha displayed are the total trees/ha for each species within each diameter class. Note the different scales in the y-axes from (A) to (B).