THE EFFECTS OF BURN ENTRY AND BURN SEVERITY ON STAND STRUCTURE AND COMPOSITION IN GRAND CANYON NATIONAL PARK

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A Thesis

Submitted in Partial Fulfillment of the

Requirements for the Degree of

Master of Science

In Forestry

Northern Arizona University

August 2011

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ABSTRACT

THE EFFECTS OF BURN ENTRY AND BURN SEVERITY ON STAND STRUCTURE AND COMPOSITION IN GRAND CANYON NATIONAL PARK

ANNA MARIE HIGGINS

Over one hundred years of fire exclusion in frequent-fire ponderosa pine and mixed conifer forests has created conditions outside of the historic range of variation. Increased tree densities, heavy fuel accumulations and an increase in late successional, fire-intolerant trees have resulted in more moderate- to high-severity fire occurring when fire is reintroduced. The reduction in seral species regeneration and the negative ecological impacts associated with uncharacteristic stand-replacing fires is of concern to managers, especially in the face of a changing climate.

Grand Canyon National Park began using prescribed fire in the ponderosa pine forests over 30 years ago and more recently, wildland fire in the mixed conifer forests. Manager's are increasingly using burn severity mapping to quantify above-ground vegetation change following fire, yet research is needed to determine post-fire vegetation response thus enabling future forest succession predictions. Our study focused on the effects of burn entry and burn severity on a subset of two forest types: ponderosa pine with white fir encroachment and dry mixed conifer. These forests have experienced a species shift along with densification and fire is the primary treatment being used to meet management objectives.

We stratified plots by forest type (ponderosa pine with white fir encroachment or dry mixed conifer), burn entry (unburned, first or second), burn severity (low or high) and years since last burn. We collected basic tree measurements and analyzed data using PERMANOVA with significance set at $\alpha \leq 0.05$. We found no difference in white fir densities in a single, low-severity burn compared to unburned areas in the ponderosa pine with white fir encroachment forest type. There was a significant difference in white fir overstory densities in a second-entry burn compared to the unburned areas. We found no white fir seedlings or saplings in a second-entry, low-severity burn. In the mixed conifer forest there was no difference in overstory composition and structure or white fir understory densities in unburned areas and low-severity burns. Ponderosa pine regeneration was not associated with every burn and thus the importance of repeated fire and a good seed crop is noted. Aspen regeneration significantly increased following a high-severity burn compared to both low-severity burns and unburned areas. Aspen regeneration was found on 90% of the high severity plots.

We did not find first-entry, low-severity fire to be effective in reducing white fir densities in either the ponderosa pine with white fir encroachment or dry mixed conifer forest types. Repeated entries are likely needed to reduce white fir densities and reduce new ponderosa pine regeneration in the ponderosa pine with white fir encroachment. The dry mixed conifer forests are experiencing a forest type shift to aspen following high-severity fire.

ACKNOWLEDGEMENTS

I thank my co-advisor Dr. Kristen Waring for her knowledge, input and assistance throughout the project. I thank my co-advisor Dr. Andrea Thode for her advice, support and assistance throughout the project. I am very appreciative of the many hours of work my advisors put into this project and their patience and encouragement over the past couple of years. I also thank my committee member Dr. Peter Fulé for his expertise, advice and input.

I thank Linda Kerr, Intermountain Region Fire Ecologist for working so hard to secure funding for this project. I thank Eric Gdula, Fire GIS Specialist, for his efforts to get this project going. I thank Windy Bunn, Fire Ecologist at Grand Canyon National Park, for her advice and input throughout the project, especially in the early phases. I thank Chris Marks, Deputy Assistant Fire Management Officer of Grand Canyon National Park, for his support of this project. I am very appreciative of all of the fire crew on the North Rim for keeping an eye on me and making sure I stayed safe out there. I thank the Ecological Restoration Institute for supplying me with wonderful field technicians. I thank my amazing field technicians: Chris Erickson, Ian Hyp, Jeremy McWhorter, and Diana Kennedy.

I am most thankful to my partner Trason Hirsch for his endless energy to get in one more plot and encourage me to follow my passion. I thank my family and friends for supporting me through it all and believing in me.

This project was made possible through funding from the National Park Service and the Colorado Plateau Cooperative Ecosystems Unit.

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PREFACE

This thesis contains one chapter intended for publication and is written in manuscript form. The manuscript is Chapter 3: Effects of burn severity and burn entry on stand structure and composition in ponderosa pine and mixed conifer forests.

CHAPTER ONE

Introduction

Recent increases in fire size (Stephens 2005), severity (Miller et al. 2009) and frequency (Westerling et al. 2006) have brought wildland fires to the forefront of public attention. Past management practices such as grazing, logging and fire suppression have changed forest conditions considerably thus affecting potential fire behavior. Prior to Euro-American settlement, frequent-fire forests in the southwestern US were shaped through climatic variables (Allen and Breshears 1998) and a wide array of disturbances (Swetnam and Betancourt 1998). A disruption of the natural fire regime has caused tree densities to increase tremendously in some forests (Covington and Moore 1994; Fulé et al. 1997) while other forests have experienced a shift in species composition (Mast and Wolf 2004; Heinlein et al. 2005). Current conditions increase the risk of high-intensity fires and natural resource managers use restoration and fuel treatments to reduce this risk. With climatic conditions predicted to change in the near future (Houghton et al. 1990), managers must understand how restoration and fuel reduction treatments will influence forest composition and structure to ensure perpetuation of healthy forests into the future.

Promoting resiliency within forests is one strategy managers utilize to protect forests in the face of a changing climate. Many treatments natural resource managers apply are designed to reduce fuel loads and tree densities and alter species composition, shifting stands towards more historic conditions.

Whether such treatments are successful in promoting resiliency in the face of climate change, reducing tree densities in dry, frequent-fire adapted forests will reduce the chance of future stand-replacing fires.

Uncharacteristically large, stand-replacing fires in forests adapted to frequent, low-intensity surface fires is worrisome as some forests lack the ability to naturally regenerate following high-intensity fire (Savage and Mast 2005; Dore et al. 2008; Goforth and Minnich 2008). Mechanical and prescribed or wildland fire treatments are used to reduce the risk of stand-replacing fire. Mechanical thinning treatments often focus on reducing the amount of small diameter trees to reduce ladder fuels and competition near large, more fire-tolerant trees. Prescribed burning often focuses on reducing both fuel loads on the forest floor and densities of small diameter trees. Recent research has concluded that single-entry, low-severity burns lack the ability to restore tree densities and species composition to more historical levels (Fulé et al. 2006; Youngblood et al. 2006; North et al. 2007). Whether treatments include mechanical or non-mechanical thinning, burning or a combination of both, consideration should be given to the regeneration which establishes post-treatment.

Ponderosa pine, the most drought-tolerant tree found in the mixed conifer forests of the Southwest, has experienced reduced regeneration rates due to increasing tree densities and a lack of fire (Fulé et al. 2003; Mast and Wolf 2004; Heinlein et al. 2005). Aspen, an important tree for biodiversity, has also suffered reduced regeneration rates due to herbivore pressure and a lack of fire (Bailey and Whitman 2002; Fairweather et al. 2007). Treatments aimed at reducing tree densities and fuel loads could increase aspen and ponderosa regeneration as both species may respond positively to post-treatment conditions. With our research we aim to describe how multiple burn entries and burn severity affect forest structure, composition and regeneration in ponderosa pine and mixed conifer forests. Specifically, we analyze how the number of burn entries (0,1,2) affects overstory composition and structure and conifer and aspen regeneration success in the ponderosa pine with white fir forest type. We also analyze how burn severity (low vs. high) affects overstory composition and structure and conifer and aspen regeneration success in the dry mixed conifer forest type. In addition to the research for the thesis, we summarize burn severity by treatment (prescribed, wildland, suppression), forest type (ponderosa pine, mixed conifer, spruce-fir) and high severity patch size across forest types.

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CHAPTER TWO

Changes in species composition and stand structure in ponderosa pine and mixed conifer forests

Introduction

The reintroduction of fire into fire-dependent ecosystems has garnered much public attention in recent years as more public and private land is burned in wildfires. Many studies have described the changes associated with fire exclusion and the reintroduction of fire in pure ponderosa pine forests, yet less research has focused on mixed conifer forests in the Southwest. Mixed conifer forests occupy more hectares than ponderosa pine forests in the western United States (US) and vary considerably dependent upon longitude, topography, elevation, precipitation and past disturbance regimes. In Arizona (AZ), however, ponderosa pine forests inhabit more acres than mixed conifer forests (O'Brien 2002). Frequent fire forests have changed considerably due to logging, grazing and fire exclusion, resulting in problems arising with the reintroduction of fire into these forest types. More research is needed to study fire effects, such as fire severity interactions with post-fire regeneration, in mixed conifer and ponderosa pine forests, to determine future forest composition and structure and thus guide management decisions.

Grand Canyon National Park

The North Rim of Grand Canyon National Park is located in Coconino County in northern AZ on the Kaibab Plateau in the Colorado Plateau Providence. The elevation of the Plateau ranges from 1,830 m to 2,800 m and is bordered to the south by the Colorado River, dividing the North and South Rims. The soil is made up of a thick porous layer of Kaibab limestone contributing to the absence of streams (Rasmussen 1941). Weather data is recorded at the Bright Angel Point Ranger Station (www.wrcc.dri.edu) located at an elevation of 2560 m. Annual precipitation averages 64.2 cm. The majority falls as snow during the winter months, around 348 cm total, with the rest occurring as monsoonal moisture during the months of July and August, often accompanied by severe lightning storms. The months of May and June tend to be the driest. The winter temperature averages -1° C with summer temperatures averaging 14° C.

Prior to Euro-American settlement, the North Rim was used by several tribes during the summer and fall seasons. The Kaibab tribe from the north inhabited the area in the summer while the Paiute and Navajo tribes from the south used the area as a fall meeting and hunting place (Rasmussen 1941). Mormons were the first Euro-American settlers in the region in the late 1860's. Livestock were introduced in 1885 with numbers exceeding 200,000 sheep and 20,000 cattle by 1887 (Mann and Locke 1931; in Rasmussen 1941). The Grand Canyon Forest Reserve was created in 1893, followed by the Grand Canyon National Game Preserve in 1906. Congress created the Grand Canyon National

Park (GCNP) in 1919 with an additional 46,000 acres added in 1927. The creation of GCNP ensured the discontinuation of livestock grazing and a future without timber harvesting. Due to this early national park designation, GCNP contains the most acreage of unharvested forests in Arizona (Warren et al. 1982). However, grazing continued until the completion of a livestock fence in 1938 (NPS 1999). Currently there are several herds of unknown size, of an introduced cross between bison (*Bison bison*) and domestic cattle, grazing within the park.

The North Rim consists of three broad forest types over an elevation gradient (Rasmussen 1941; Moir and Ludwig 1979; Warren et al. 1982). The ponderosa pine (*Pinus ponderosa* var. *scopulorum* P. and C. Lawson) forest begins around 2250 m, closest to the rim's edge. Less dominant tree species include piñon pine (*Pinus edulis* Engelm.), Utah juniper (*Juniperus osteosperma* (Torr.) Little), and Gambel oak (*Querqus gambelii* Nutt.) at the lower elevations, white fir (*Abies concolor* (Gordon and Glendinning) Hoopes.) and quaking aspen (*Populus tremuloides* Michx.) at the higher elevations. New Mexico locust (*Robinia neomexicana* Gray) occurs throughout, mainly in a shrub state (Warren et al. 1982). The mixed conifer forest begins around 2450 m, and is comprised of ponderosa pine, white fir, Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco var. *glauca* (Beissn.) Franco), quaking aspen, Engelmann spruce (*Picea engelmanni* Parry ex *Engelmanni*) and blue spruce (*Picea pungens*) (Warren et al. 1982). The spruce-fir forest begins around 2650 m and consists of subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.), Engelmann spruce, blue spruce and the occasional ponderosa pine, Douglas-fir, white fir and aspen (Warren et al. 1982). Prior to Euro-American settlement each of these forest types was shaped through climatic factors (Allen and Breshears 1998) and disturbance processes such as insects, disease and wildfire (Swetnam and Betancourt 1998); creating heterogeneous forests with varying species composition and stand structure (Kaufmann et al. 2000).

Forest Fire

The ponderosa pine forests experienced the most frequent and least severe fire regimes; lighting started fires occurring on the interval of every 4-7 years (Fulé et al. 2003a). The shortest fire return interval was 1 year with the longest fire free interval being 12 years (Fulé et al. 2003a). Fires tended to occur in dry years following wet years, probably as a result of increased herbaceous growth during wet years (Swetnam and Baisan 1996; Fulé et al. 2000; Fulé et al. 2003b). Most lightning started fires are associated with the monsoon season thus fire scars tend to occur in the middle earlywood (Fulé et al. 2003b). These small, frequent fires shaped the forest into a park-like structure with widely spaced, large trees. Fulé and others (2003a) estimated the ponderosa pine forests supported 159.1 trees per hectare in 1880. A survey by Lang and Stewart (1910) on the Kaibab Plateau noted an average of 124.8 ponderosa pines per hectare larger than 15.2 cm. The open canopy promoted herbaceous growth in the understory allowing fires to spread through the fine herbaceous fuels. Due to the thinning effect of fire, Mast and Wolf (2006) found that the overstory ponderosa pine on the North Rim tended to be in even-aged patches with an average interpatch distance of 5.0 m. Cooper (1961) found a similar mosaic of even-aged groups sized at 1/5 acre or more in other Arizona ponderosa pine forests.

The mixed conifer forests experienced a frequent fire, mixed-severity fire regime (White and Vankat 1993; Fulé et al. 2003a). South and west facing slopes experienced the shortest fire return interval (Fulé et al. 2003a). The frequent, low-severity fires maintained stands dominated by seral, fire-tolerant species, such as ponderosa pine, white fir and Douglas-fir, and a forest structure of large, open grown trees. North and east facing slopes were able to support more dense forests made up of less drought-tolerant species. Due to the differences in fuel moisture levels and fuel characteristics, the north and east facing slopes experienced less frequent fires but more moderate- and high-severity fires when fire did occur (Fulé et al. 2003a). The large amount of aspen found on the North Rim is thought to be an indicator of past high-severity fires (Fulé and Laughlin 2007). The historical level of fire severity depended on topography, elevation and forest type (White and Vankat 1993; Wolf and Mast 1998; Fulé et al. 2003a)

Fire Exclusion

Wildfires have shaped species composition and forest structure in ponderosa pine and mixed conifer forests across the western United States for thousands of years (Arno 1980; Pyne 1982; Agee 1993; Swetnam 1993; Swetnam and Baisan 1996). Forests adapted to a frequent fire regime have undergone considerable change since the onset of grazing and fire exclusion, due to the number of missed fire cycles. The importance of fire in southwestern ponderosa pine forests has long been documented (Pearson 1933; Weaver 1951; Cooper 1960; White 1985; Covington and Moore 1994a,b; Covington et al. 1997). More recently, studies have focused on the historic fire regime of the frequent fire, mixed conifer forests (Mast and Wolf 1998; Fulé et al. 2003a; Beaty and Taylor 2001; Beaty and Taylor 2008) and the changes associated with fire exclusion (Parsons and DeBenedetti 1979; White and Vankat 1993; Minnich et al. 1995; Goforth and Minnich 2008). Fulé and others (2000, 2003a,b) found an abrupt cessation of the natural fire regime in 1879 in GCNP most likely due to the removal of fine herbaceous fuels from over-grazing by livestock. Active fire suppression began following designation of GCNP in 1919. Complete fire suppression continued until the 1978 when GCNP's fire management program began (GCNP Fire Management Plan 2005).

In the ponderosa pine forests, grazing and over 100 years of fire exclusion has created a dense overstory, creating a homogenous canopy, susceptible to uncharacteristic crown fires (White and Vankat 1993; Covington et al. 1994; Moore et al. 2004; Fulé et al. 2004b). On the North Rim, as in many other ponderosa pine forests, an increase in tree densities is the most notable change occurring since fire exclusion. Along with the densification of the overstory comes a reduction in the amount and diversity of understory communities (Covington and Moore 1994; Huisinga et al. 2005; Laughlin et al. 2005) in turn affecting fire spread. Without frequent fire, litter and duff accumulate on the forest floor inhibiting ponderosa pine regeneration (Pearson 1950) and contributing to increased fire behavior. Fire-intolerant and shade-tolerant trees, most notably white fir on the North Rim, have migrated down from the mixed conifer forests into the historically pure ponderosa pine forests (Mast and Wolf 2004). Similar migrations are noted in other frequent-fire ponderosa pine/mixed conifer forests (Fulé et al. 1997; Heinlein et al. 2005; Keeling et al. 2006). Frequent fires reduced white fir densities in the past. The invasion of white fir has reduced ponderosa pine health and regeneration success through resource competition (Mast and Wolf 2004). The increase in white fir has also altered the historic fire regime in the ponderosa pine forests. The long ponderosa pine needles burn quickly and more intensely, contributing to rapid fire spread, compared to the shorter fir needles (vanWagtendonk et al. 1998; Stephens et al. 2004). However, white fir saplings are more susceptible to surface fire due to their thin bark (Thomas and Agee 1986; Laacke 1990). Once established, older white fir trees generate thick bark allowing the trees to withstand multiple surface fires. White fir and other fire-intolerant species create ladder fuels to the canopy increasing the probability of torching and passive crown fire.

In the mixed conifer forests of the North Rim, fire exclusion has also contributed to a change in species composition and forest structure (Fulé et al. 2003a; Fulé et al. 2006; Mast and Wolf 2006). As in other frequent-fire, mixed conifer forests throughout the West, an increase in tree densities and percentage of shade-tolerant trees has occurred (Keeling et al. 2000; Fulé et al. 2003a; Crocke et al. 2005; Fulé et al. 2009). The increase in tree densities reduces overstory vigor (Mast and Wolf 2004), increasing susceptibility to insects, such as mountain pine beetle (*Dendroctonus ponderosae* (Hopkins)) and fir-engraver (*Scolytus ventralis* (LeConte)) and diseases, such as mistletoe (*Arceuthobium vaginatum subsp. cryptopodum* (Engelmann) Hawksworth and Weins) and root disease (*Armillaria* spp.). The fire-intolerant and later successional species, such as white fir, Engelmann spruce and blue spruce, have expanded outside of their historical range (Mast and Wolf 2006), creating water and nutrient stress in older trees (Covington and Moore 1994b). Fulé and others (2002) found ponderosa pine to be the oldest living species in the ponderosa pine and mixed conifer forests on the North Rim. They noted the following establishment dates for the oldest living trees by species: ponderosa pine (1537), Gambel oak (1650), Utah juniper (1770), white fir (1793), Douglas-fir (1796), aspen (1813), New Mexico locust (1904), and Engelmann spruce (1932). Several studies have recreated historic ponderosa pine forest stand structure by dating the establishment of older trees (White 1985; Mast et al. 1999; Abella 2008). Recreating historic stand structure in mixed conifer forests is more difficult due to the relatively young age of Douglas-fir, white fir and aspen. These species have higher instances of decay that may be introduced through fire-scars (Arno et al. 1995) and are more susceptible to insect damage from drought stress than ponderosa pine. The invasion of late successional species has reduced seral species regeneration rates and homogenized stand structures across the landscape (Mast and Wolf 2004; Cocke et al. 2005; Fulé et al. 2009). When fire is reintroduced, stands that historically experienced low-severity fire may now experience an increased amount of moderate- to high-severity fire (Fulé et al. 2004a; Fulé et al. 2004b; Mast and Wolf 2004).

The North Rim mixed conifer forest has historically varied in stand structure and species composition, and the recent changes affect not only future fire behavior but ecosystem health as well. The downward migration of less fire-tolerant species is a main concern to GCNP managers (Chris Marks 2011, personal communication). White fir is less drought-tolerant and more fire-tolerant than ponderosa pine. Climate prediction models are predicting temperatures to increase with precipitation decreasing in the Southwest (Seager et al. 2007). Hotter, drier conditions in turn lead to more fire across the landscape, leading to larger and more severe fires (Westerling et al. 2006). Drought related tree mortality is predicted to increase in forested ecosystems (van Mantgem 2009; Allen et al. 2010) and there is concern that climate change may outpace species migration rates (Malcolm et al. 2002). Fossil records have shown *Pinus* species have historically migrated with changing climatic conditions, most notably as the climate dried (Axelrod 1986). Natural ponderosa pine regeneration should be a top priority in GCNP in the mixed conifer forests as climate conditions continue to change, as ponderosa pine is the most adapted to survive in warmer and drier environments.

Reintroduction of Fire

Many agencies began reintroducing fire in the ponderosa pine forests as the low-severity fires which historically occurred are easier to control and suppress than moderate- to high-severity fires. In addition, the ecological effects of frequent, low-severity fire were realized to be critical to the ecosystem. GCNP began reintroducing fire into the ponderosa pine forests in the late 1970’s. GCNP’s objective to restore fire and reduce tree densities and fuel loads is achieved through prescribed burns and naturally ignited wildland fires.

Fire has successfully been reintroduced into many ponderosa pine forests throughout the west. Ponderosa pine is naturally self-pruning so when fire is reintroduced the potential for a surface fire to transition into a crown fire is minimal. In contrast, mixed conifer forests may have heavy ladder fuels, created by trees with full length crowns, which can contribute to more surface to crown fire transitions. Objectives of prescribed burns are typically to restore ecosystem function, reduce fuel loads created by an excess of dead and down woody debris and to reduce ladder fuels by decreasing small tree densities and increasing overstory crown base heights (Agee and Skinner 2005). Tree mortality from prescribed fires is typically concentrated in the smaller size classes. Harrington (1981) noted mortality was greatest in the smallest size classes following several prescribed fires in the Santa Catalina Mountains in AZ and was concentrated in stands with the highest tree densities. Not all mortality following prescribed fires is concentrated in the smallest trees; Kaufmann and Covington (2001) found increased mortality in pre-settlement ponderosa pines in areas burned on the North Rim compared to unburned areas. Mortality varied by fire type and seasonality with the highest mortality seen in a prescribed fire turned wildfire, lower mortality rates in a prescribed fire only and the lowest mortality seen in a spring time burn when soil and fuel moisture levels were higher following snow melt. More recent studies suggest that while pre-settlement ponderosa pine mortality may increase following prescribed fires, mortality levels decrease after 10 years post-treatment (Erickson 2011).

Fewer studies have evaluated the effects of prescribed fires in mixed conifer forests than in ponderosa pine forests in the Southwest, largely due to the amount of prescribed burning occurring in ponderosa pine forests compared to mixed conifer forests. Objectives of prescribed fires in the mixed conifer forests are similar to those in ponderosa pine forests and include restoring ecosystem function, reducing fuel loads, tree densities and basal area. In the mid-altitude forests on the North Rim, Fulé and Laughlin (2007) found a 54% reduction in total tree densities and a 23% decline in total basal area following a wildfire in 2003. A significant reduction in tree densities was found in ponderosa pine, white fir and aspen while a significant reduction in basal area was only seen in white fir and aspen. Tree mortality was greatest in smaller diameter trees. They found no significant difference in regeneration pre- or post-fire at the mid-altitude sites. In northeastern Oregon, Youngblood and others (2006) found that one prescribed burn in ponderosa pine/Douglas-fir forests reduced smaller size Douglas-fir, but had little effect on the overstory. The study showed that while species composition was affected by the reduction of smaller Douglas-fir trees, basal area was not reduced and therefore crown fire potential remained high.

Prescribed fires in southwestern aspen stands are carried out less frequently than prescribed fires in ponderosa pine forests. Pure aspen stands have a low probability of burning yet aspen stands with heavy fuel loading and a mixed conifer component burn more readily (Brown and Simmerman 1986). Aspen are highly susceptible to fire due to their thin bark (Baker 1925) and even light surface fire can kill trees or create a fire scar. Such trees may be more susceptible to pathogens (Davidson et al. 1959), ultimately leading to tree mortality.

Regeneration

Ponderosa pine regeneration in the Southwest is mostly episodic (Schubert 1974; White 1985) and requires a good seed crop, favorable climatic conditions (Cooper 1961; Savage et al. 1996) and bare mineral soil (Pearson1950), typically achieved through fire. Ponderosa pine germinates following summer rains associated with the monsoon season. Following germination, ponderosa pine seedling mortality may occur from grass competition (Pearson 1943; Larson and Schubert 1969), frost heaving, drought (Pearson 1910; Larson and Schubert 1969; Heidmann 1976), or browsing by ungulates or rodents (Pearson 1950). Ponderosa pine can germinate in the duff layer yet rarely becomes established because the roots do not have access to sufficient moisture (Sackett 1984). Most ponderosa pine seedling mortality occurs within 3 years of germination (Shepperd et al. 2006; Johnson 2011). Ponderosa pine seeds also provide the main food source to the endemic Kaibab squirrel (*Sciurus aberti kaibabensis*), resulting in direct seed predation. Seed caching may actually promote regeneration as Hall (1981) observed Kaibab squirrels burying ponderosa pine seeds in mid-October, following a good seed crop, and not all caches may be revisited. Hall (1981) also noted that the Kaibab squirrel forages on the phloem, twigs and apical buds of ponderosa pines, adversely impacting future seed production.

After ponderosa pine seedlings become established they remain susceptible to fire injury until they reach a height of 2 m (Bailey and Covington 2002). In Colorado, it takes approximately 22 years for a ponderosa pine tree to reach a height of 1.4 m (Shepperd et al. 2006). As most wildland fires burn in a mosaic pattern, ponderosa pine seedlings may survive low-intensity, surface fires yet could face competition from neighboring sprouting species, such as aspen and New Mexico locust, which respond to fire via asexual reproduction. New Mexico locust responds by prolific sprouting following overstory release (Gottfried 1980) and fire (Carmichael et al. 1978). Aspen suckers outgrow conifer seedlings as well, with some achieving a height of 1.4 m in 2 to 5 years (Jones 1974).

White fir is a prolific seeder, producing from 200,000 (Laacke and Fiske 1983) to1.5 million seeds per hectare (Laacke 1990) depending on overstory composition. Since most seed dispersion occurs through gravity with fewer seeds carried by the wind, the majority of seeds fall within a distance of 1 to 1 ½ times the height of the seed tree (Laacke 1990). White fir germinates following snowmelt and prefers cool and moist soils (Franklin 1974). Although shade-tolerant, white fir grows best in the open once established (Laacke 1990). White fir can survive in low light levels in stunted form, releasing with an increase in sunlight either from a canopy opening or when the leader emerges above surrounding vegetation. White fir with its thin, resin blistered bark is highly susceptible to fire up to pole size (Laacke 1990). As the tree ages, self-pruning and development of thicker bark make it more resistant to fire (Youngblood 1985). White fir is more shade-tolerant than Douglas-fir.

Douglas-fir bears seeds at a younger age than white fir and with winged seeds the seeds are transported by wind and gravity (Owston 1974). Douglas-fir germinates following summer moisture. Once established, Douglas-fir remains fire susceptible up to an age of 40 years (Fischer and Bradley 1987) due to its thin resinous bark. Douglas-fir is more fire-tolerant than white fir and less fire tolerant than ponderosa pine. In the frequent fire forests of the Northern Rockies, ponderosa pine frequently dominated over Douglas-fir as the two trees reached fire resistance at different times (Arno 1980). The relationship between ponderosa pine and Douglas-fir in the Northern Rockies is similar to the ponderosa pine/white fir relationship in Arizona.

Aspen reproduces both sexually and asexually. Aspen seeds mature in the early summer although most aspen regenerates through suckers. Aspen roots produce suckers, which in turn grow into the overstory, thus increasing the size of the clone. Aspen suckers have an advantage over aspen seedlings due to their pre-formed root system (Graham et al. 1963). Aspen suckers also require less water than seedlings for establishment (DeByle and Winokur 1985). Aspen is shade-intolerant and shorter lived than conifers, surviving up to 200 years in optimal conditions.

Regeneration following fires

Fire is both beneficial and detrimental to tree regeneration. Some species require exposed mineral soil for germination, yet many tree seedlings are highly susceptible to fire due to their thin bark. Fire reduces competition from other seedlings and overstory trees yet if all of the overstory is killed by fire then the seed source is lost. Most conifer seeds remain viable for only 1-2 years after release.

Post-fire, conifer regeneration is dependent on a viable seed source and many studies have shown a decline in regeneration with an increase in distance from seed source. However, Shatford and others (2007) observed tree densities of 84 - 1,100 trees per hectare greater than 300 m from seed source following high-severity fires in northern California and southern Oregon. In the mixed conifer forests of southern California, Minnich (1977) found that seeds originating outside the burned area contributed to white fir regeneration 10 years post-fire but found little regeneration 5 years post-fire.

Ehle and Baker (2003) observed that historically more ponderosa pine regeneration occurs within 10 years following low-severity fire than within 10 years of high-severity fire in the Rocky Mountains, however they found that overall tree densities were greatest in stands initiated following high-severity fire, possibly due to lack of overstory competition. Bailey and Covington (2002) found that while prescribed fires prepare a suitable seedbed, mortality did occur in previously established seedlings in AZ. Dore and others (2008) found no ponderosa pine regeneration 20 years after high-severity fire in a pure ponderosa pine forest outside of Flagstaff, AZ.

A viable seed source does not always equate to regeneration as Bock and Bock (1977) found no white fir regeneration, even with a seed source, 17 years after a stand replacing fire had occurred in California. Although many conifers germinate in bare mineral soil, conditions may be less than ideal for germination and establishment. Jones (1995) found unshaded soil surfaces in Idaho reached lethal temperatures in the summer, detrimental to seedling establishment and survival. He also noted the effect of shrubs on regeneration was dependent on the shrub species, with only certain shrubs showing a positive relationship to seedling establishment and growth. In Oregon, Minore (1986) found greater Douglas-fir seedling survival in clearcuts where stumps provided shading compared to seedlings without any shade protection.

Not all trees reproduce by seed following fire as seen by the asexual reproduction of aspen and New Mexico locust. While aspen is known for its clonal properties, Quinn and Wu (2005) found aspen regenerating by seed following stand replacing fires in southeastern AZ. Aspen can regenerate without fire although rarely succeeds in entering the overstory due to competition from other aspen and/or conifers. Keyser and others (2005) found that light burning stimulates aspen suckering although the regeneration densities were similar to unburned areas in the Black Hills of South Dakota. They also observed the greatest amount of aspen suckering following complete overstory mortality due to stand-replacing fire. Brown and DeByle (1989) found similar trends with stands burned at moderate to high severity producing the most aspen suckers.

In Wyoming, Bartos and others (1994) found aspen sucker densities of 10,000 to 20,000 per hectare 3 years post fire. In Idaho, Brown and DeByle (1989) found aspen sucker densities of 17,000 to 39,000 per hectare following moderate- to high-severity prescribed fires. These high densities quickly self thin as Jones (1975) found over a third of 3 to 4 year old suckers dead following a clearcut in AZ. High aspen densities may contribute to a lack of shade-intolerant ponderosa pine regeneration. Eventually shade-tolerant conifers establish in the understory of mature aspen stands.

As the amount of fire increases across the landscape managers must be aware of how fire affects natural tree regeneration. Tree regeneration can both be positively and negatively affected by fire. On a finer scale, regeneration is correlated with burn severity. As burn severity mapping becomes more accessible to resource managers it is being used to help predict future vegetation trajectories.

Burn severity mapping

As wildfire severity (Miller et al. 2009) and frequency (Westerling et al. 2006), has increased in recent years, natural resource managers are continually looking for more reliable ways to quantify burn severity across the landscape. Burn severity, as quantified by GCNP, refers to the degree in which aboveground vegetation is altered due to fire. Burn severity affects both above and below ground biota and affects subsequent aboveground vegetation regeneration dependent on species reproduction strategies. Managers are increasingly using burn severity mapping to make informed land management decisions (Eidenshink et al. 2007). Some uses of burn severity mapping include prescribed fire planning analysis, tactical wildfire planning, fuels management, post-fire assessments, and landscape and ecosystem monitoring.

Another use of burn severity mapping is to monitor changes to wildlife habitat, most notably Mexican spotted owl (MSO) (*Strix occidentalis lucida*) habitat. The MSO is listed as threatened by the US Fish and Wildlife Service (USFWS) and management within its habitat is governed by the Endangered Species Act (1973). In the northern part of the MSO range, which includes the North Rim of GCNP, owls primarily nest in rocky canyons (Ganey and Balda 1989). Even though the MSO is believed to roost mainly in the caves below the rim (Willey and Ward 2001a,b), most of the mixed conifer forest on the North Rim is considered prime habitat for the threatened bird by the USFWS. High-severity fire is believed to be detrimental to MSO habitat as it destroys old growth forests and reduces roost and forage structure in the overstory. Although the mixed conifer forest on the North Rim historically burned at a mixture of severities, recent large, high-severity fires have caused concern to wildlife biologists over the amount of high-severity fire across the landscape and how this will affect the forests, and thus MSO habitat, into the future (Shaula Hedwall 2011, personal communication).

To date, the Grand Canyon Fire and Aviation program has mapped burn severity on all fires, occurring since 2000, greater than 100 acres, for a total of 35 fires covering 90,652 acres (NPS 2009). One method for mapping burn severity is the delta Normalized Burn Ratio (dNBR) (Key and Benson 2005b). Severity values using the dNBR are determined through Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper (ETM +) imagery, recorded at a resolution of 30 m since 1984. The Normalized Burn Ratio (NBR) is the difference in TM bands 4 and 7: NBR = (R4 – R7)/ (R4 + R7). Band 4 detects near infrared while band 7 detects short wave infrared. The dNBR is then calculated using pre- and post-fire NBR values: dNBR = NBRprefire – NBRpostfire (Key and Benson 2005b). A project is currently underway to remap burn severity on fires in GCNP from 1984 to 1999. All burn severity is field verified using Composite Burn Index (CBI) plots (Key and Benson 2005a). To date GCNP has installed over 800 CBI plots.

National Park Service policy and management

The National Park Service has long understood the beneficial role fire has played in many Western forests (van Wagtendonk 2007). Current and past policies have led the park to re-establish natural processes on the landscape using the best available science. All prescribed fire in National Parks is required to have a prescription, found in each Park's management plan.

GCNP, as with most National Parks, differs from other land management agencies in its limited ability to mechanically treat fuels to reduce the risk of high-severity fires. Current non-fire treatments include pruning, thinning, lop/scatter, piling and burning, and chipping/mulching (GCNP Fire Management Plan 2005) with all of the treatments except chipping/mulching accomplished through the use of chainsaws. GCNP uses prescribed fire and wildland fire in lieu of mechanical treatments used by other agencies, yet these treatments are not always successful in meeting management objectives (Fulé et al. 2006), particularly with one first-entry burn. One objective of the GCNP Resource Management Plan (1997) is to restore fuel loads and ecosystem structure within the natural range of variability.

A large portion of the North Rim is currently being considered for wilderness designation. Under a wilderness designation the typical treatments are naturally started wildland fires or fire suppression to meet resource objectives. Fire suppression in wilderness areas is usually carried out using the minimum tool concept, whereas current fire suppression in GCNP employs the use of helicopters, bulldozers, fire engines and chainsaws.

To significantly reduce fuel loads and reduce the potential of stand-replacing fires, several factors must be addressed including: reducing surface fuels, increasing live crown base height, decreasing crown bulk density, and retaining large trees of fire-resistant species (Agee and Skinner 2005). In addition to reducing the potential for stand-replacing fires, fire-resistant tree regeneration is required to replace the aging cohorts. One time, low-severity burns have been shown to not substantially reduce crown fire potential (Youngblood et al. 2006; Goforth and Minnich 2008). Multiple burns at a mixture of severities are predicted to significantly reduce both fuel loading and future crown fire potential (Fulé et al. 2004a,b)

Conclusion

Currently, limited research is available comparing post-fire regeneration density and species composition in southwestern forests across burn severity. Additionally, there is no research comparing stand structure and regeneration after multiple entry burns with single entry burns and unburned areas. As managers continue to use fire to reduce fuel loading and restore functioning ecosystem processes, research is needed to determine the various effects mixed-severity fires have on vegetation responses in the mixed conifer forest. In GCNP, where lightning started fires are frequent, managers must choose which fires to let burn and which fires to suppress. Therefore managers must understand what effects low and high-severity fire has on regeneration rates across the landscape. To continue to manage forests using natural disturbance processes managers must understand how fire is changing current forest conditions to ensure desired regeneration is able to eventually reach the overstory. In accordance with National Park Service management policies managers must rely on scientists and their findings to inform management decisions (The Departmental Manual Part 620,1998)

As climate change increases temperatures, drought, insect and disease outbreaks, and the frequency of fires, there will be increased tree mortality across the landscape, thus current regeneration must be addressed to ensure the perpetuation of ponderosa pine and mixed conifer forests in GCNP.

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CHAPTER THREE

The effects of burn entry and burn severity on forest structure, composition and regeneration in Grand Canyon National Park

Introduction

Ponderosa pine and mixed conifer forests adapted to frequent fire regimes have experienced great change due to over 100 years of fire exclusion in many areas including Arizona (Cooper 1960; Covington and Moore 1994a; Fulé et al. 1997), Montana (Keeling et al. 2006), California (Goforth and Minnich 2008) and in Colorado (Fulé et al. 2009). Most forests adapted to a frequent fire regime have changed accordingly with the number of fire return intervals missed, while some forest types experience more change then others (Stephens 2004). Species shifts and the homogenization of horizontal and vertical canopy structure due to increasing tree densities have led to changes in forest structure and composition (Parsons and DeBenedetti 1979; White and Vankat 1993; Cocke et al. 2005; Heinlein et al. 2005; Abella et al. 2007; Sakulich and Taylor 2007). These changes, combined with an increase in dead and down woody debris, have led to a change in fire behavior when fire is reintroduced (Keane et al. 2002; Goforth and Minnich 2008). Forests adapted to frequent, low-severity forest fires in the Southwest are at risk of experiencing more moderate- to high-severity fires when fire is reintroduced (White and Vankat 1993; Covington and Moore 1994a,b; Fulé et al. 2004b).

Without fires' naturally thinning properties, fire-intolerant species have migrated down from higher elevation forests that historically experienced less frequent fire (Fulé et al. 1997; Fulé et al. 2003a; Mast and Wolf 2004; Cocke et al. 2005; Heinlein et al. 2005). This downward movement is opposite to future migration predictions. As climatic conditions continue to change, with the Southwest predicted to be warmer and drier (Seager et al. 2007), species are predicted to migrate up in elevation to move with the shifting conditions (Davis and Shaw 2001).

In the Southwest, the increase in white fir densities in the historically pure ponderosa pine forests is of concern to resource managers as the young fir trees compete with the older, more fire-resistant trees for water and nutrients (Mast and Wolf 2004). The increased competition coupled with increasing drought and temperatures can create stress in older ponderosa pines making them more susceptible to mortality from insects and diseases (Covington and Moore 1994b; van Mantgem et al. 2009; Allen et al. 2010). The densification of the ponderosa pine and mixed conifer forests is also of concern due to reduced ponderosa pine regeneration (Fulé et al. 2002; Mast and Wolf 2004), reduced herbaceous growth (Moore and Deiter 1992; Laughlin et al. 2005), and increased fuel loading and fire behavior (Fulé et al. 2003a; Fulé et al. 2004b). The abundant white fir under- and mid-story create ladder fuels where fire may enter the canopy and subsequently increase tree mortality across the landscape (Fulé et al. 2004a,b; Mast and Wolf 2004).

To ensure the perpetuation of frequent fire-adapted ponderosa pine and mixed conifer forests in the Southwest, natural regeneration of ponderosa pine is imperative. Natural ponderosa pine regeneration in the Southwest is mostly episodic (Schubert 1974; White 1985), requiring a good seed crop, favorable climatic conditions (Cooper 1961; Savage et al. 1996) and bare mineral soil (Pearson 1950; Sackett 1984), typically achieved through fire. Following germination, ponderosa pine seedling mortality may occur from grass competition (Pearson 1943; Larson and Schubert 1969), frost heaving, drought (Pearson 1910; Larson and Schubert 1969; Heidmann 1976), or browsing by ungulates or rodents (Pearson 1950). Ponderosa pine can germinate in the duff layer yet rarely becomes established because the roots do not have access to sufficient moisture (Sackett 1984). As ponderosa pine requires bare soil for establishment, prescribed fire and wildland fire are treatments that are successful in propagating ponderosa pine regeneration. Due to social, safety, and funding constraints, far less fire is applied to the landscape than occurred historically. Following many years of successful fire suppression the amount of fire across the landscape is increasing, and some areas have now experienced multiple-entry fires since fire exclusion. As more wildfire and prescribed fire occurs a better understanding of the interactions of multiple burn entries, burn severity, and forest structure and composition is needed.

Lack of fire in southwestern mixed conifer forests has resulted in a decrease in regeneration of seral species, such as ponderosa pine and aspen, as tree densities have increased through time (Mast and Wolf 2004, 2006;

Heinlein et al. 2005). A lack of high-severity fire allows for conifer encroachment and aspen mortality in mixed aspen forests (Smith and Smith 2005). While some mixed conifer forests historically experienced a mixed-severity fire regime (Fulé et al 2003a; Beaty and Taylor 2008), little is known of tree regeneration following mixed-severity fires. While several studies have focused on the effects of either low-intensity or stand-replacing fires on tree regeneration (Bock and Bock 1977; Minnich 1977; Quinn and Wu 2005; Shatford et al. 2007), many studies describe regeneration densities for only one species and few compare regeneration by fire severity (Ehle and Baker 2003; Keyser et al. 2005).

The acceptance of fire treatments has grown in recent years yet most prescribed fires are relatively small and typically burn at low intensities. Several studies show that first-entry, low-intensity prescribed burns are not sufficient to restore forest structure and fuel loading to more historical conditions (Stephens and Moghaddas 2005; Fulé et al. 2006; Youngblood et al. 2006; North et al. 2007; van Mantgem et al. 2011). As managers are preparing for repeated entry prescribed fires in some areas, and more wildfires are burning at higher severities (Miller et al. 2009), more research is needed to determine regeneration success in the ponderosa pine and mixed conifer forests following fire.

Grand Canyon National Park

On the North Rim of Grand Canyon National Park (GCNP) the natural fire regime was disrupted in 1879 (Wolf and Mast 1998; Fulé et al. 2000, 2003a,b), probably due to livestock overgrazing. The lower elevation ponderosa pine forests experienced the most frequent and least severe fire regimes; lightning started fires occurring every 4-7 years (from 10% of all samples scarred in the same year) (Fulé et al. 2003a). Fires tended to occur in dry years following wet years, probably as a result of increased herbaceous growth during wet years (Swetnam and Baisan 1996; Fulé et al. 2000; Fulé et al. 2003b). Situated higher in elevation, the mixed conifer forests experienced a mixed-severity fire regime (Fulé et al. 2003a). South and west facing slopes experienced the shortest fire return interval (Fulé et al. 2003a). North and east facing slopes were able to support denser forests composed of less drought-tolerant species and experienced a less frequent, moderate- to high-severity fire regime (Fulé et al. 2003a).

In GCNP, managers primarily use fire as a prescriptive treatment to meet natural resource management objectives (USDI 2005). Prescribed fire and wildland fire are both used to reduce tree densities, basal area and fuel loading in the fire-dependent forests (Abella et al. 2007). Other public agencies, including federal and state agencies, use mechanical and fire treatments to reduce stand densities and fuel loading. National park status, access, topography and an impending wilderness designation contribute to the dependence on fire treatments in GCNP. Grand Canyon's first prescribed burns occurred in the ponderosa pine forests. After successfully reintroducing fire into the ponderosa pine forests GCNP began prescribed burns in the mixed conifer forests with several fires on the North Rim burning at a mixture of severities.

One objective of GCNP's fire management plan is to "restore the vegetative structure to the natural range of variability, which must provide diverse habitats for native species". To achieve this objective there are specific objectives for each forest type. In the ponderosa pine with white fir encroachment forest type the objectives are to: (1) reduce white fir pole sized tree densities (2.5 – 15 cm dbh) to less than 247 trees per hectare; (2) retain ponderosa pine trees greater than 40 cm dbh at densities of 47 – 62 trees per hectare; and (3) limit high-severity fire to patches less than 25 hectares in size on less than 15% of the forest type across the landscape. The only specific management objective in the mixed conifer forest type is to limit moderate-high and high-severity fire to less than 30% across the landscape. This number was based on consensus from park and wildlife managers due to the majority of the mixed conifer forests designated as restricted Mexican spotted owl habitat.

Grand Canyon's fire management plan requires monitoring of all burned areas to ensure management objectives are met. Monitoring plots are permanent and are typically visited prior to fire, immediately post-burn, one year post-fire, five years post-fire and ten years post-fire. Currently there are fire handbook monitoring (FMH) plots (USDI 2003) established in three subtypes of the ponderosa pine forests: South Rim ponderosa pine, North Rim ponderosa pine and the ponderosa pine with white fir encroachment. There are few FMH plots in the mixed conifer forests on the North Rim.

Our research focuses on the effects of multiple burn entries and burn severity on stand structure and composition in the ponderosa pine with white fir encroachment and the dry mixed conifer forest types in Grand Canyon National Park, Arizona. Our research focused on the mixed conifer forest typically found between the historically pure ponderosa pine forests and the mixed conifer forests where no ponderosa pine occurs. This forest type is sometimes referred to as dry or xeric mixed conifer as opposed to wet or mesic mixed conifer. Dry mixed conifer is defined as occupying drier sites, supporting ponderosa pine and adapted to a frequent-fire regime with wet mixed conifer occupying more mesic sites and little to no ponderosa pine and a more mixed-severity fire regime (Smith et al. 2008). In the ponderosa pine with white fir encroachment forest type, our objectives were to determine the effect of first- and second-entry fires on (1) overstory structure and composition and (2) conifer and aspen regeneration success. We specifically hypothesized that:

1. White fir and aspen overstory densities would be different in second-entry burned areas compared to densities in unburned and first-entry burned areas.

2. Ponderosa pine overstory densities would not be different in first- or second-entry burned areas compared to unburned areas.

3. White fir understory densities would be different in second-entry burned areas compared to unburned and first-entry burned areas.

4. Ponderosa pine and aspen regeneration would be different in first-entry and second-entry burned areas compared to in unburned areas.

In the dry mixed conifer forest type, our objectives were to determine the impact of fire severity (low vs. high) on (1) overstory structure and composition and (2) conifer and aspen regeneration success. We specifically hypothesized that:

1. Overstory species composition and densities would not be different in low-severity burned areas compared to unburned areas.

2. White fir understory densities would not be different in low-severity burned areas compared to unburned areas.

3. Ponderosa pine regeneration would be different in low-severity burned areas compared to unburned and high-severity burned areas.

4. Aspen regeneration would be greatest in high-severity burned areas compared to unburned and low-severity burned areas.

In addition to the above hypotheses, we had the opportunity to investigate a time since fire effect on regeneration in the dry mixed conifer forest type.

Methods

Study Area

The study sites were located on the North Rim of Grand Canyon National Park in northern Arizona on the Kaibab Plateau (Figure 3.1). The elevation of the Plateau ranges from 1,830 m to 2,800 m and is bordered to the south by the Grand Canyon and Colorado River, dividing the North and South Rims of GCNP. Plot locations ranged in elevation from 2410 m to 2684 m. The soil consists of a thick porous layer of Kaibab limestone, contributing to the absence of streams (Rasmussen 1941).

Weather data is recorded at the Bright Angel Point Ranger Station (www.wrcc.dri.edu) located at an elevation of 2560 m. Annual precipitation averages 643 mm. Most of the precipitation falls as snow during the winter months, around 3500 mm total, with the rest occurring as monsoonal moisture during the months of July and August, often accompanied by severe lightning storms. The months of May and June tend to be the driest, with occasional dry lightning occurrence. The winter temperature averages -1° C with summer temperatures averaging 14° C.

Above the rim of the Grand Canyon, the lowest elevations are dominated by the ponderosa pine forest type, followed by the dry and wet mixed conifer forest types at mid-elevations and the spruce-fir forest type at the highest elevations and along drainages. For this study we focused on a subset of 2 forest types: ponderosa pine with white fir encroachment (Encroachment) and the dry mixed conifer (Mixed conifer). We defined the encroachment forest type as a historical overstory comprised of 90 – 100% ponderosa pine with an influx of white fir since fire exclusion. We defined the mixed conifer forest type as having a historical overstory comprised of 40 – 90% ponderosa pine. Historical was defined as a tree established prior to fire exclusion, around 1879. Conifers greater than 40 cm at diameter at breast height (DBH) were considered to have established prior to fire exclusion (White 1985; Mast et al. 1999; Fulé et al. 2002; Fulé et al. 2003a). Forest type was visually confirmed on site following random plot placement using vegetation maps from Warren and others (1982). A pilot study found no ponderosa pine regeneration in fires that burned in 2007. As ponderosa pine only produces good seed crops every 4-6 years (Shepperd et al. 2006) we did not study any fires occurring after 2005.

Prescribed fires in the encroachment forest type began in the 1990's and several first- and second-entry burns have since occurred. The area we studied for first-entry, low-severity fire was burned by the Tower Fire, ignited in August 2001 (Figure 3.2). On a separate plateau southeast of the Tower fire we studied an area that had experienced a second-entry, low-severity burn (Figure 3.3). The first entry was the Outlet 1 (1999) prescribed fire, initiated in October 1999. In June of 2005, lightning ignited a fire within the old Outlet 1(1999) perimeter and park managers allowed the fire to burn as wildland fire use (WFU), now termed wildland fire. The Dragon WFU burned through much of the Outlet (1999) fire in addition to a good portion of the plateau previously unburned.

After the successful reintroduction of fire to the majority of the ponderosa pine forests in GCNP, managers began using prescribed and lightning-ignited fires within the mixed conifer forests. The Outlet 2 Fire (2000) (Figure 3.4) began as a prescribed fire but soon turned into a wildland fire. An unexpected wind event occurred several weeks following initial ignition and the low-intensity prescribed fire quickly turned into a running crown fire. On May 10, 2000, the Outlet 2 Fire (2000) overtook contingency lines with the majority burning at moderate to high severity.

Three years later in September of 2003 a lightning-ignited fire became the Poplar WFU (Figure 3.5). Following several weeks of calm fire behavior, a high wind event occurred, increasing fire behavior dramatically in the Poplar Fire. Suppression actions were initiated on the Poplar WFU following active crown fire. Nearby the Rose Fire, a WFU, ignited in October, was also allowed to burn for a total of 1409 ha (Figure 3.5).

Sampling Design

Plots were stratified by burn severity (low or high), burn entry (unburned, first, second), year of last burn, and forest type (ponderosa pine with white fir encroachment or dry mixed conifer) (Table 3.1). Due to the limited duration of this study and the nature of wildland fire we did not collect pre-fire data, relying instead on adjacent unburned areas for comparison. Unburned areas were located outside of all fire perimeters recorded by GCNP since the 1980's. Prior to the 1980's the majority of fires were suppressed before reaching a size over 1 hectare. Burn severity maps were provided by GCNP, obtained through the Monitoring Trends in Burn Severity program. Using ArcMap v9.3.1 (ESRI 1999-2009) 15 plots were randomly located in each strata using minimum burn severity patch sizes of 8100 m^2 (nine 30 x 30 m pixels) to account for error in the Landsat imagery and GPS locations. An edge buffer of 33.75 m was used to locate plots away from patch edges. Plots were then located in the field using coordinates and a handheld GPS unit; each plot was then field verified for burn severity and forest type prior to installation. Plots were located a minimum of 183 m from any road to avoid areas potentially hand thinned pre- or post-fire. Plots in the Tower and Dragon fires tended to be clumped due to the limited amount of encroachment forest type burned at low severity, as the majority of those fires burned through pure ponderosa pine forests (Figures 3.1-3.5).

Field Methods

All field data were collected during the summers of 2009 and 2010 on 0.04 ha circular plots. Aspect and elevation were recorded at plot center with overstory and seedling data collected in the plot and fuels data collected along three transects radiating from plot center. The 0.04 ha circular plot was established to record all live and dead overstory trees (>10 cm diameter at breast height (DBH)). Species, DBH, height, crown base height, live crown ratio, status (live, sick, dead), and damage were recorded for all live trees. A sick tree was defined as any tree exhibiting abiotic or biotic stress that would lead to mortality within 3-5 years. Damages included visible biotic and/or abiotic damage. For stand structure descriptions, canopy class and strata (Smith et al. 1997) were also recorded for all live trees. Distance to nearest seed tree (ponderosa pine and white fir) was recorded if no seed trees were present within the plot (maximum 100 m from plot perimeter). For dead trees, species, DBH, decay condition class (1-5) and mortality agent were recorded.

We recorded species, diameter class, height class and status for each sapling on 0.004 ha circular plots located at the same point center as the larger 0.04 ha plots. Four smaller seedling plots (0.0004 ha) were located at point center and at 9.14 m along each fuel transect. We recorded species, height class ((1): 0.0 – 15.0 cm; (2): 15.1 – 30.0 cm; (3): 30.1 – 60.0 cm; (4): 60.1 – 100.0 cm; (5): 100.1 – 200.0 cm), and status (live, sick, dead) for each seedling. If regeneration was overabundant (i.e. aspen and/or New Mexico locust (*Robinia neomexicana*) sprouts > 7500 stems ha^{-1}) then we randomly selected two of the four seedlings plots to measure.

Two measurements of duff and litter depth were taken along each of the three fuel transects at 9.14 m and 15.24 m. We visually estimated percent cover of shrubs, forbs, grasses, litter, rock, bare soil and moss in a 0.0004 ha plot located at 9.14 m along each transect; these were the same plots as the seedling plots described above.

Along each transect, percent canopy cover was recorded using vertical densitometer readings every 1 m for a total of 10 measurements per transect; these readings were then averaged across all three transects.

Analyses

To quantify stand structure and composition we calculated live and dead overstory, sapling and seedling densities, and the following for overstory only: Basal Area (BA), Quadratic Mean Diameter (QMD) and Stand Density Index (SDI). We calculated BA using the following equation: $(\Sigma d_i^{\,2} \text{x } 0.00007854)/n$ where d_i is the diameter at breast height of an individual tree and n is the total number of trees per plot. We calculated QMD by diameter class using the following equation: QMD = $\sqrt{}\ (\Sigma d_i^2)/n$ where d_i is the diameter at breast height of an individual tree and n is the total number of trees per hectare. We calculated

SDI using the summation method (Long and Shaw 2005): SDI = $(D_q/25.4)^{1.6}$ x TPH where D_q = is the quadratic mean diameter in inches at breast height, 1.6 is the constant related to the self-thinning rule (Reineke 1933), and TPH is the number of trees per hectare. We then calculated percentage of maximum SDI using 450 as maximum SDI (Long and Shaw 2005). Seedlings <15 cm height were excluded from all analysis due to high mortality rates in this size class (Shepperd et al. 2006; Johnson 2011). Seedlings exhibiting damage that would lead to mortality were also excluded from analysis. Aspect was transformed for analysis (Beers et al. 1966).

We tested for significant differences between strata using permutational multivariate analysis of variance (PerMANOVA) (Anderson 2001) in PC-ORD v5.10 (McCune and Mefford 2006) due to non-normal data. Pair-wise comparisons were made following significant PerMANOVA results. Univariate analyses used Euclidean distance and 4999 permutations. Vegetation and substrate data was analyzed using Pearson's correlation coefficient to determine positive or negative correlation between variables. Significance was set at $\alpha \leq 0.05$ for all analyses.

Results

Ponderosa pine with white fir encroachment

<u>*Overstory*</u>

A significant difference in total live overstory densities was seen in a second-entry burn compared to unburned and first-entry burned areas (Figure

3.6a). Total live tree densities in second-entry burns were only 52 and 63% of that found in unburned and first-entry burns, respectively (Table 3.2). No significant differences were found between standing dead trees by species in the control, first- or second-entry (Figure 3.6b). However, first- and second-entry burns had over twice the amount of total standing dead tree densities compared to the unburned, a significant difference. There was a significant difference in white fir overstory densities and an apparent change in the diameter distribution in a second-entry burn, as compared to first entry and unburned (Figure 3.6a, 3.7, 3.8). Live ponderosa pine densities were similar across treatments; however, the diameter distributions appeared different in a second-entry burn (Figure 3.7). Live aspen was only a minor component of the overstory, ranging in density from 1.7 - 11.7 stems ha^{-1} in all three treatments, with the least amount of aspen found in a second-entry burn (Figure 3.6a). Live basal area was similar in first- and second-entry burns, at about 70% of unburned live basal area (Figure 3.9a; Table 3.2). Ponderosa pine comprised 86.1%, 81.0%, and 93.2% of the total basal area for unburned, first and second entry (Table 3.2). Total SDI was different in both first- and second-entry burns, compared to unburned at about 69% of maximum SDI to about 50% for both first- and second-entry burns (Table 3.3). Percent canopy cover was significantly different in the unburned and second-entry burns but not between unburned and first entry (Figure 3.9a).

Seedlings and saplings

No white fir seedlings were found on any second-entry plots, a significant difference compared to unburned and single entry (Figure 3.10a). White fir seedling densities in the first-entry burn were 46% of the total found in unburned areas; however it was not a significant difference (Figure 3.10a, Table 3.4). There was no significant difference in ponderosa pine seedling densities by burn entry due to the high variability between plots (Figure 3.10a). However, the first- and second-entry burns had 417 and 1100% of ponderosa pine seedlings compared to densities in unburned areas (Table 3.4). There was about 7.5 times the amount of aspen seedlings/suckers in the second-entry burn compared to the unburned and first entry (Table 3.4). There was no significant difference in sapling densities by species by burn entry (Figure 3.10b). We found no white fir saplings in the second-entry burn and few ponderosa pine saplings.

Vegetation, litter, duff and bare soil

There was a significant difference in duff depth and percent litter cover in the first- and second-entry burns compared to unburned areas (Figure 3.11). Duff depths were half the amount of the unburned in the first- and second-entry burns. Percent litter cover in the first- and second-entry burns was 25 and 59% of the cover in the unburned area (Figure 3.11). Ponderosa pine regeneration decreased ($r = -0.33$, $p = 0.03$) and white fir regeneration increased ($r = 0.31$, $p = 0.04$) with an increase in litter cover. There was a significant difference in bare soil in both first- and second-entry burns ($f_{2,44} = 7.6$, $p \leq 0.01$) (Figure 3.11). There was no significant difference in percent cover of grass and forbs among treatments. Grass ($r = -0.35$, $p = 0.02$) and forb cover ($r = -0.35$, $p = 0.02$) decreased with an increase in canopy cover. There was a significant difference in forbs in a second-entry burn ($f_{2,44} = 3.3$, $p = 0.04$) compared to unburned and first entry. Forbs tended to decrease with an increase in litter cover ($r = -0.52$, $p < 0.01$).

Mixed conifer

Overstory

As expected (and confirmed during plot establishment), no live overstory trees remained in high severity burned areas. Total live overstory densities and aspen and white fir overstory densities differed in the unburned area and areas that burned in 2003 at low severity (Figure 3.12a). There was a significant difference in aspen overstory densities in the low-severity burns compared to the unburned. The unburned area had over six times the aspen compared to the low-severity burned areas (Table 3.5). Total live tree densities in the 2003 low-severity burn were half as much as in the unburned areas and about a third of the totals in the 2000 low-severity burn (Table 3.5). We found no significant difference in standing dead tree densities in unburned compared to low-severity burn (Figure 3.12b). Basal area was different in both the 2000 & 2003 low-severity fires compared to the unburned area (Figure 3.9b). Canopy cover in the 2000 and 2003 low-severity burns was about 75% of the cover then in the unburned areas (Figure 3.9b). In all, low-severity burns seem to have little effect on the overstory structure and composition.

Seedlings and saplings

White fir seedling densities were not significantly different in low-severity burns with densities of 250 and 750 stems ha^{-1} in the 2000 and 2003 low-severity burns compared to densities in unburned areas (Figure 3.13; Table 3.6). No white fir seedlings were found in high-severity burns. Ponderosa pine regeneration was absent in the unburned areas as well as in areas that burned at low and high severity in 2000 (Figure 3.13a; Table 3.6). Ponderosa pine seedling densities were 72% higher in the 2003 low-severity burn compared to the 2003 high-severity burn (Figure 3.13b). Aspen densities were significantly different in the high-severity burns compared to the unburned and low severity; with the majority of stems reaching sapling size 6+ years post-fire (Table 3.6). Aspen sapling densities in the 2000 and 2003 high-severity burns were 574 and 2220% of the densities in the low-severity burns, a significant difference (Figure 3.13b). Aspen seedling and sapling densities in the unburned area were 1458.3 and 266.7 stems ha^{-1} (Table 3.6). There was no significant difference in conifer sapling densities in low-severity burns compared to the control (Table 3.6). We found no conifer saplings in the high-severity burns.

Vegetation, litter, duff and bare soil

Duff depths in the low- and high-severity burns were half the depths found in the unburned area (Figure 3.11). There was a significant difference in litter cover in a high-severity burn, but there was no difference in litter cover in a low-severity burn compared to the unburned areas (Figure 3.11). The amount of exposed bare mineral soil was significantly different in high-severity burns especially in the 2003 high-severity burn (Figure 3.11). As litter cover increased the percentage of grasses ($r = -0.39$, $p < 0.01$) and forbs ($r = -0.40$, $p < 0.01$) decreased. As percent canopy cover increased duff depth ($r = 0.38$, $p < 0.01$)

and litter cover ($r = 0.49$, $p < 0.01$) increased. As canopy cover decreased New Mexico locust regeneration densities increased ($r = -0.27$, $p = 0.02$). Grass cover was significantly different in the low and high-severity burns compared to the unburned areas ($f_{4,74} = 2.70$, $p = 0.03$).

Discussion

The densification and shift of species in the frequent-fire ponderosa pine and mixed conifer forests from grazing and over 100 years of fire exclusion has cascading effects including: reduced seral tree regeneration rates, increased water and nutrient stress among trees, reduced nutrient cycling, reduced herbaceous understory growth, changes in soil chemistry and changes in future fire behavior (Covington and Moore 1994b; Abella et al. 2007). Grand Canyon National Park, similar to several other National Parks, has an active fire management program compared to other public agencies. While fire monitoring plots are placed across the landscape to monitor trends and determine if management objectives are being met, the plots are not able to successfully address many specific research questions (USDI 2003). The pseudoreplication inherent in the sampling design and typical of landscape-level wildland fire research limits definitive conclusions regarding differences in stand structure, composition and tree regeneration resulting from the effects of fire. Our research has helped address questions related to post-fire forest response and interactions between fire severity and number of burn entries and we have made inferences on the effects of fire through differences or similarities seen in unburned areas (control) compared to burned areas (treatment)

Ponderosa pine with white fir encroachment

<u>*Overstory*</u>

First-entry prescribed fires typically have little effect on overstory structure (Stephens and Moghaddas 2005; Fulé et al. 2006; Youngblood et al. 2006) as seen with similar white fir and ponderosa pine densities in a first-entry burn compared to densities in unburned areas. As we hypothesized, the effect of fire on total overstory tree densities was seen only in a second-entry burn and only in white fir densities; ponderosa pine densities remained high. Johnson (2011) also noted the inability of first- and second-entry, low-severity fire to reduce ponderosa pine overstory densities in Zion National Park in Utah. We found tree densities of 336.5 stems ha^{-1} in the second-entry burn comparable to reconstructed historical densities of 383.3 trees ha^{-1} by Fulé and others (2003a); although the authors' reconstruction included only trees ≥ 2.5 cm dbh while our totals include all trees ≥ 0.1 cm dbh.

The low aspen densities found in all three treatments was surprising but not unexpected. This area is in the lower latitudinal range for aspen in North America (Jones 1985) and in the middle elevation range for aspen in Arizona (Fairweather et al. 2007). If we combine sapling and overstory densities, the combined density of 118.4 trees ha^{-1} is more similar to reconstructed densities of 142.1 aspen trees ha^{-1} in the higher elevation ponderosa pine forests on the North Rim (Fulé et al. 2003a).

Seedlings and saplings

Although ponderosa pine seedling densities were different in the unburned, first- and second-burn entries, high levels of variation contributed to the lack of significance therefore we were unable to verify our hypothesis. Compared to Mast and Wolf's (2004) study which found little ponderosa pine establishment since the 1930's, we did find ponderosa seedlings and saplings in the unburned areas. Due to high canopy closure it is uncertain whether the ponderosa pine regeneration will successfully replace the aging cohort however. Bailey and Covington (2002) suggested a rate of 3.6 trees ha^{-1} per decade in Arizona as sufficient to sustain pre-Euro-American densities. The current regeneration densities following a second-entry burn are much higher and will require additional fire events to reduce densities to more appropriate levels (Bailey and Covington 2002). Without frequent recurring fire, ponderosa seedlings will eventually become fire-resistant as Bailey and Covington (2002) found no mortality in ponderosa pine saplings over 2 m in height following a prescribed burn. If fire alone is used to reduce seedling densities it must be used relatively frequently as Shepperd (2006) estimated from seedling growth in Colorado that it takes over 22 years for a ponderosa pine to reach 1.4 m in height and thus become more fire-tolerant. The relatively high levels of white fir seedlings and saplings following a single-entry, low-severity burn are also indicative of the need for repeated burning; especially since we found no white fir seedlings or saplings in a second-entry, low-severity burn.

We hypothesized that there would be a difference in aspen regeneration in a first-and second-entry burn, however we only saw a difference in seedling densities in a second-entry burn. The difference in aspen seedlings/suckers in the second-entry burn is likely due to a combination of reduced canopy cover and abundant suckering following mortality of the overstory aspen. Repeated burnings may be detrimental to aspen success, as aspen is relatively fire-intolerant (Baker 1925). Aspen suckers are more drought-tolerant than aspen seedlings, due to the established root system (Graham et al. 1963), and if fire is excluded around the aspen suckers, they might even thrive. Even if the aging overstory aspen are not replaced, the abundant aspen found in the higher elevation forests may provide a refuge for the species and allow aspen to survive through changing climatic conditions (Shaw and Davis 2001).

Mixed conifer

Overstory

Pre-Euro-American mixed conifer forest reconstructions are lacking in many areas due to a lack of "reference forests" (Vankat 2011) and the associated species higher rates of decay compared to ponderosa pine (Dieterich 1983; Arno et al. 1995). Several recent reconstructions in Arizona and southwest Colorado show the variation in species composition and densities across different mixed conifer forests (Cocke et al. 2005; Heinlein et al. 2005; Fulé et al. 2009). Two mixed conifer forest reconstructions from 1880 on the North Rim, found total tree densities of 242.8 and 245.7 trees per ha^{-1} (Fulé et al. 2002, 2003a). Although these reconstructions did not differentiate between the different types of mixed conifer forests, the combined sapling and overstory densities we found in the low-severity burns are still approximately 200 to 765% greater than historical densities. When we compared unburned areas to areas experiencing low-severity fire we found differences in live tree densities by species but not with standing dead tree densities by species. Since the dead overstory densities were similar following low-severity fire, which typically has little impact on larger trees, we can infer that the stands differed prior to fire. It is probable that GCNP began prescribed burning and managing lightning-ignited fires in the least dense forests first, with the remaining unburned areas having higher tree densities and fuel accumulations (Dave Robinson 2011, personal communication). Several studies have shown that single-entry, prescribed burns do not reduce stands to more historic levels of overstory tree densities and basal area (Stephens and Moghaddas 2005; Fulé et al. 2006; Youngblood et al. 2006; North et al. 2007) or species composition (van Mantgem et al. 2011) and our research suggests the same.

Seedlings and saplings

Finding no ponderosa pine regeneration in the oldest fire was unexpected but understandable. The Outlet 2 Fire (2000) occurred following and during the extreme drought years of 1996 and 2000 (www.wrcc.dri.edu; Erickson 2011) and this may explain the lack of ponderosa pine regeneration. The increase in needle cast and litter fall following fire quickly covers the bare mineral soil exposed from fire (Sackett 1980; Johnson 2011), if the soil was exposed at all. If there was not a good seed crop or favorable climatic conditions (i.e. ample summer moisture), then the window of opportunity for establishment is quickly missed as duff, litter and competing vegetation can increase as time since fire increases (Passovoy and Fulé 2006; Johnson 2011). Sackett and others (1996) found satisfactory ponderosa pine regeneration following prescribed fires in Arizona where 19% bare mineral soil was exposed, which is higher than amounts found in this study, although our data was collected between 6 and 10 years post-fire. Ponderosa pine can regenerate in the absence of fire but requires the presence of other soil-exposing disturbances, such as heavy grazing, which has not occurred on the North Rim since the 1930's.

Similar to the first-entry burn in the encroachment and as hypothesized, we found that white fir densities were not different in a low-severity burn in the dry mixed conifer forest type. More moderate-severity fire may be required to reduce white fir seedling and sapling densities and return the forests to a more historical and resilient structure. While managers at GCNP are open to more moderate-severity fire across the landscape (Chris Marks 2011, personal communication) the acceptable amount is up for question as the acceptable 30% moderate-high to high-severity fire in the mixed conifer forest type is not based on historical evidence.

The abundant aspen regeneration following high-severity fires confirmed our hypothesis and results from other studies (Bailey and Whitham 2002; Keyser et al. 2005). Although the tree densities were lower than in other regions (Brown and DeByle 1989; Bartos et al. 1994; Keyser et al. 2005) our study separated living aspen by live and sick (Table 3.7) and we included the sick aspen in the dead totals for a more conservative estimate of future densities. The lack of ungulate grazing pressure on the North Rim likely indicates that even as aspen self-thins, these numbers are sufficient to support a thriving aspen forest. While Keyser and others (2005) found aspen regeneration rates similar in low-severity burns and unburned areas in South Dakota, we were surprised that aspen was regenerating at all in the unburned areas as so much literature has focused on aspen decline and the lack of aspen regeneration in recent years (Romme et al. 1995; Fairweather et al. 2007; Worrall et al. 2008). As patches open in the canopy, aspen may be able to enter the overstory without fire (Kurzel et al. 2007).

Conclusion

Reintroducing the frequent-fire regime into the ponderosa pine with white fir encroachment and dry mixed conifer forest types is imperative to ensure a resilient forest in the face of climate change. Reducing white fir densities in both the understory and overstory will allow ponderosa pine to regain its competitive advantage. Compared to more mesic species, ponderosa pine is better suited to withstand a warming and drying climate as it is more drought-tolerant and fire-resistant than other tree species found on the North Rim.

While many studies have focused on the success (or not) of prescribed fire in restoring historical conditions, ours is the first study that we know of to examine the effects of a second-entry burn on reducing white fir densities in a ponderosa pine with white fir encroachment forest type. Our research shows that live white fir densities are significantly different in a second-entry burn compared to densities in unburned and first-entry burned areas, yet there is no difference in ponderosa pine overstory densities. The density of pole sized trees in second-entry burns meets GCNP's fire management objectives, yet these objectives were written for first-entry burns. We also show the ability of ponderosa pine regeneration to increase tremendously in first- and second-entry burns and the need for repeated frequent burning to reduce seedling densities to a more historic level. As second-entry burns seem ineffective in reducing ponderosa pine overstory densities, this may be of concern to managers as the overstory may remain susceptible to bark beetle attack (Negrón et al. 2009) and future stand-replacing fire.

While the Outlet and Poplar fires displayed the immense amount of forest that can quickly be consumed following a high wind event, the mixed conifer forests of the North Rim seem to be resilient following high-severity fire. The abundant aspen regeneration following high-severity fire will ensure that a forest will propagate and the conversion of forestland to shrubland is unlikely. The ponderosa pine seedlings found in high-severity burned areas far from seed-trees ensures a future forest composed, at least in part of conifers. Even isolated patches of high-severity fire in the ponderosa pine forests will contribute to a more heterogeneous forest structure across the landscape and not all high-severity fire is detrimental to ponderosa pine regeneration (Ehle and Baker 2003; Savage and Mast 2005). As white fir seedling densities in low-severity burns are similar to densities in unburned areas, more moderate-severity fire may be needed to open up the understory for herbaceous growth and ponderosa pine regeneration. Our study also shows that not every fire was successful in promoting ponderosa pine establishment and that the combination of a good seed crop and exposed bare mineral soil is crucial.

Park managers at GCNP must frequently decide which lightning-ignited fires to allow to burn and which to suppress due to the sheer volume of lightning strikes. Most decisions are based on when the area last burned (Windy Bunn 2010, personal communication) however more thought may be given in the future to creating smaller patches of burned areas. Due to personnel, budget, air shed and view shed constraints most fires on the North Rim burn thousands of acres at a time. Without fire the landscape has become more homogenized and by burning such large areas in relatively short periods of times with low-severity prescribed fires, the homogeneity is further perpetuated. Wildland fires frequently burn over longer intervals and with more mixed severity, thus creating heterogeneity throughout the forest. As most lightning starts occur during peak tourist season, it is difficult to allow all fires to burn considering the consequences if a high wind event occurs. There is only one main access road in and out of the North Rim, subsequently the forests along that road are under complete fire suppression; they are thus the densest and most at risk for stand-replacing fire. The park's plan to reintroduce fire to the exterior portions of the park to provide a buffer for future lightning-ignited fires in the interior is a step in the right direction (Chris Marks, personal communication). However smaller fires may be ignited or allowed to burn throughout the park to create a more historical and resilient heterogeneous forest structure across the landscape.

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Table 3.1. Plot stratification by forest type (Encroachment = ponderosa pine with white fir encroachment, Mixed conifer = dry mixed conifer), burn entry (unburned; 1st; 2nd), burn severity (unburned; low; high), and year (2000; 2003).

Fire Name	Year	Fire Type	Forest Type	Hectares	Burn Severity % Unburned	Low	Moderate	High	# Plots	Plot Strata
Tower	2001	Wildfire	Encroachment	1643	5	81	13	1	15	1st Entry
			Mixed conifer						4	2000 Low
Outlet	1999	Prescribed	Encroachment	1632	1	66	23	10	15	2nd Entry
Dragon	2005	Wildfire		3207						
Outlet	2000	Prescribed & Wildfire	Mixed conifer	5033	9	16	21	54	11	2000 Low
									15	2000 High
Poplar	2003	Wildfire	Mixed conifer	4811	5	46	21	28	13	2003 Low
									15	2003 High
Rose	2003	Wildfire	Mixed conifer	1409	17	59	15	9	2	2003 Low
Unburned	-	-	Encroachment	-	100	-	-	-	15	Control
			Mixed conifer						15	

Table 3.2. Sapling and overstory densities in the ponderosa pine/white fir encroachment. Saplings are ≥ 137 cm in height and < 10 cm dbh. Trees are ≥ 10 cm dbh. Tree densities are trees ha^{-} , BA = overstory basal area (m^2/ha^{-1}), values are means with standard errors in parentheses. pipo = ponderosa pine, abco = white fir, psme = Douglas-fir, potr = quaking aspen.

Burn entry & year of fire(s)		Total	pipo	abco	psme	potr
Control (Unburned)	Sapling density	466.7 (184.0)	133.3 (87.5)	250.0 (111.8)	0 (0)	83.3 (52.7)
	Overstory density	478.3 (48.2)	263.3 (53.8)	193.3 (45.6)	10.9 (5.9)	11.7 (7.3)
	BA m^2/ha^{-}	50.8 (7.3)	43.7 (7.8)	6.1 (1.3)	0.2 (0.2)	0.7 (0.5)
First Entry (2001)	Sapling density	866.7 (554.1)	366.7 (349.2)	366.7 (180.7)	0 (0)	13.3 (7.3)
	Overstory density	395.0 (54.2)	268.3 (42.4)	113.3 (33.1)	0 (0)	0.8 (0.6)
	BA m^2/ha^{-1}	34.3 (3.3)	27.7 (3.5)	5.7 (1.3)	0 (0)	0.8 (0.6)
Second Entry (1999/2005)	Sapling density	133.3 (80.4)	16.7 (16.7)	0 (0)	0 (0)	116.7 (72.6)
	Overstory density	250 (37.7)	220.0 (38.6)	28.3 (9.7)	0 (0)	1.7 (1.7)
	BA m^2/ha^{-}	36.6 (2.9)	34.1 (3.4)	2.2 (0.9)	0 (0)	0.3 (0.3)

Table 3.3. Quadratic mean diameter (QMD) and stand density index (SDI) by size class. DBH = diameter at breast height.

	Ponderosa pine / white fir encroachment						Mixed conifer transition					
	Control		1st Entry		2nd Entry		Control		2000 Low Severity		2003 Low Severity	
DBH Size Classes (cm)	QMD	SDI	QMD	SDI	QMD	SDI	QMD	SDI	QMD	SDI	QMD	SDI
10 - 29.9	18.0	184.5	20.1	258.2	19.9	83.2	17.9	171.0	19.8	156.9	20.3	78.8
30.0 - 49.9	39.9	165.4	38.2	232.9	39.5	128.1	38.2	193.7	38.6	130.1	37.4	101.6
50 - 69.9	60.0	203.1	59.7	73.7	60.7	179.3	59.8	101.0	58.7	111.1	60.0	128.6
70.0 +	79.2	242.4	81.6	99.6	80.7	163.2	81.6	132.9	76.8	80.3	85.5	142.9
Average/Total	36.8	795.4	33.2	664.4	43.2	553.8	32.5	598.5	32.9	478.3	41.9	451.9

Table 3.4. Seedling densities in the ponderosa pine with white fir encroachment forest type. Seedlings are ≥ 15 cm & < 137 cm in height. Seedling densities are trees ha^{-} . Values are means with standard errors in parentheses. pipo = ponderosa pine, abco = white fir, potr = quaking aspen, rone = New Mexico locust.

Burn entry & year of fire(s)	Total	pipo	abco	potr	rone
Control (Unburned)	2000.0 (625.6)	250.0 (118.9)	916.7 (285.2)	375.0 (159.1)	458.3 (458.3)
1st Entry (2001)	2166.7 (818.0)	1041.7 (743.7)	583.3 (187.7)	333.3 (159.8)	208.3 (145.2)
2nd Entry (1999/2005)	6333.3 (1849.9)	2750.0 (1090.7)	0 (0)	2625.0 (1178.3)	958.3 (708.3)

Table 3.5. Sapling and overstory densities in the dry mixed conifer forest type. Saplings are ≥ 137 cm in height and < 10 cm dbh. Trees are ≥ 10 cm dbh. Tree densities are trees ha^{-} , BA = basal area (m^2/ha^{-1}) values are means with standard errors in parentheses. pipo = ponderosa pine, abco = white fir, psme = Douglas-fir, potr = quaking aspen, pien = Engelmann spruce.

Year of fire and burn severity		Total	pipo	abco	psme	potr	pien
Control (Unburned)	Sapling density	983.3 (280.8)	33.3 (22.7)	433.3 (206.3)	16.7 (16.7)	433.3 (228.2)	66.7 (51.6)
	Overstory density	435.0 (41.2)	86.7 (17.4)	153.3 (31.5)	50.0 (16.2)	115.0 (35.3)	30.0 (19.2)
	BA m^2/ha^{-1}	36.1 (4.2)	14.5 (2.9)	9.3 (2.2)	4.0 (1.3)	5.1 (1.5)	3.2 (2.1)
2000 Low Severity	Sapling density	1533.3 (848.3)	0 (0)	700.0 (535.7)	16.7 (16.7)	816.7 (716.5)	0 (0)
	Overstory density	335.0 (61.1)	171.7 (37.9)	101.7 (34.3)	13.3 (5.4)	18.3 (9.3)	30.0 (23.3)
	BA m^2/ha^{-1}	25.2 (4.5)	21.9 (4.2)	3.2 (1.1)	< 0.1 (< 0.1)	< 0.1 (< 0.1)	< 0.1 (< 0.1)
2003 Low Severity	Sapling density	266.7 (104.8)	0 (0)	16.7 (16.7)	0 (0)	250.0 (106.3)	0 (0)
	Overstory density	216.7 (34.6)	140.0 (30.6)	51.7 (17.5)	11.7 (8.4)	11.7 (4.8)	1.7 (1.7)
	BA m^2/ha^{-1}	29.9 (3.7)	23.8 (3.6)	4.3 (1.8)	0.9 (0.7)	0.6 (0.3)	0.4 (0.4)

Table 3.6. Seedling densities in the dry mixed conifer forest type. Seedlings are ≥ 15 cm & < 137 cm in height. Seedling densities are trees ha^{-1} Values are means with standard errors in parentheses. pipo = ponderosa pine, abco = white fir, psme = Douglas-fir, potr = quaking aspen, rone = New Mexico locust.

Year of fire and burn severity	Total	pipo	abco	psme	potr	rone
Control (Unburned)	2708.3 (837.8)	0 (0)	750.0 (280.8)	0 (0)	1458.3 (627.0)	500.0 (457.3)
2000 Low Severity	3583.3 (995.7)	0 (0)	708.3 (298.0)	41.7 (41.7)	1958.3 (645.1)	875.0 (597.0)
2003 Low Severity	6958.3 (3235.5)	750.0 (622.6)	250.0 (102.1)	0 (0)	5166.7 (3287.7)	791.7 (450.1)
2000 High Severity	1958.3 (763.9)	0 (0)	0 (0)	0 (0)	1958.3 (763.9)	0 (0)
2003 High Severity	1750 (525.4)	208.3 (78.7)	0 (0)	0 (0)	1541.7 (498.8)	0 (0)

Table 3.7. Live, sick and dead aspen sapling densities in the mixed conifer. Saplings are ≥ 137 cm in height and < 10 cm diameter at breast height. Sapling densities are trees ha^{-1}. Values are means with standard errors in parentheses.

Year of fire and burn severity	Total	Live	Sick	Dead
Control (Unburned)	466.7 (294.7)	266.7 (157.1)	150.0 (116.5)	50.0 (50.0)
2000 Low Severity	866.7 (766.0)	816.7 (716.5)	50.0 (50.0)	0 (0)
2003 Low Severity	716.7 (332.6)	250.0 (106.3)	466.7 (282.3)	0 (0)
2000 High Severity	6783.3 (1988.5)	4700.0 (1642.1)	1250.0 (676.1)	833.3 (350.7)
2003 High Severity	7650.0 (1475.8)	5933.3 (1159.2)	1650.0 (380.9)	66.7 (29.5)

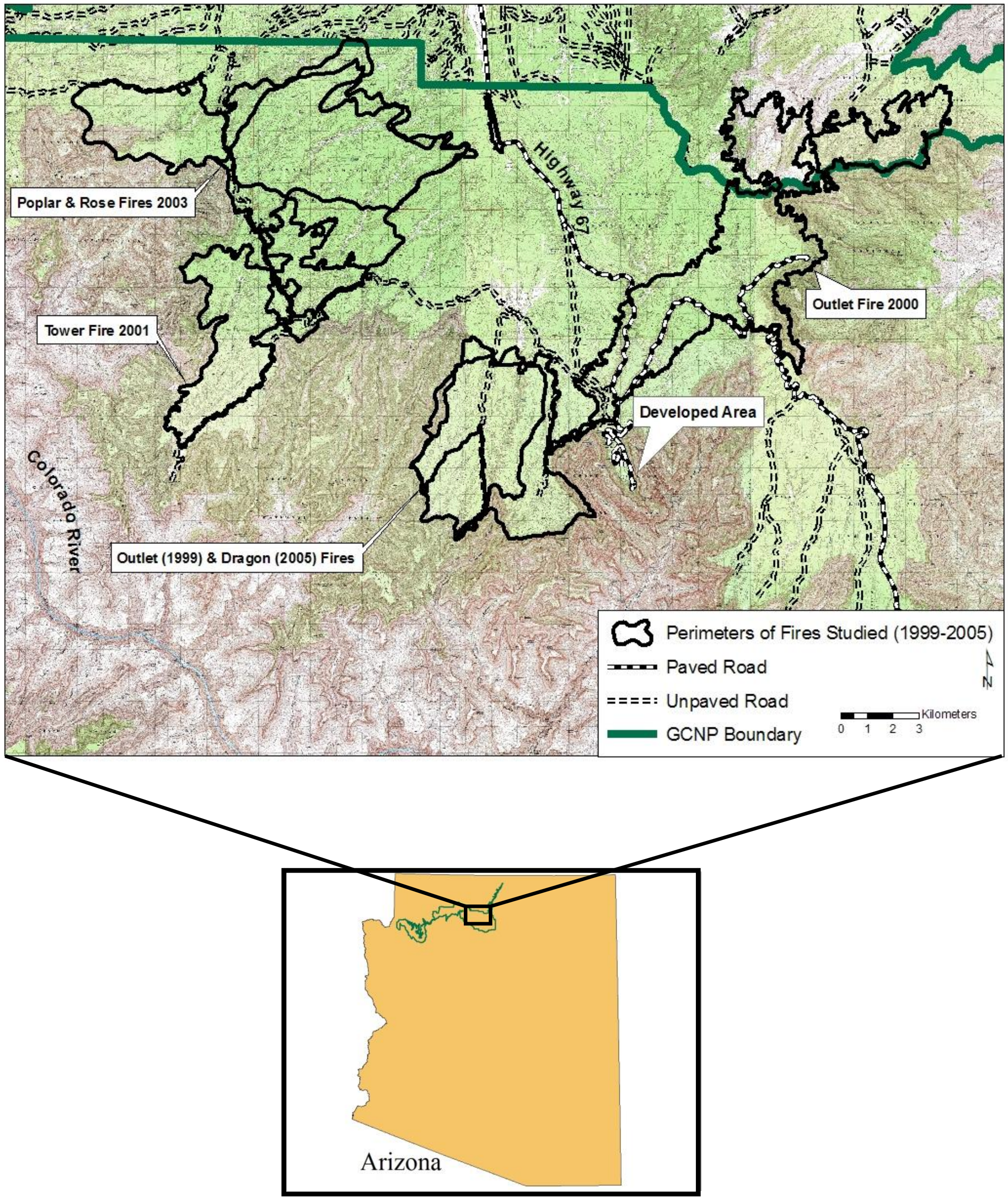


Figure 3.1 Map of North Rim of Grand Canyon National Park with perimeters of fires studied.

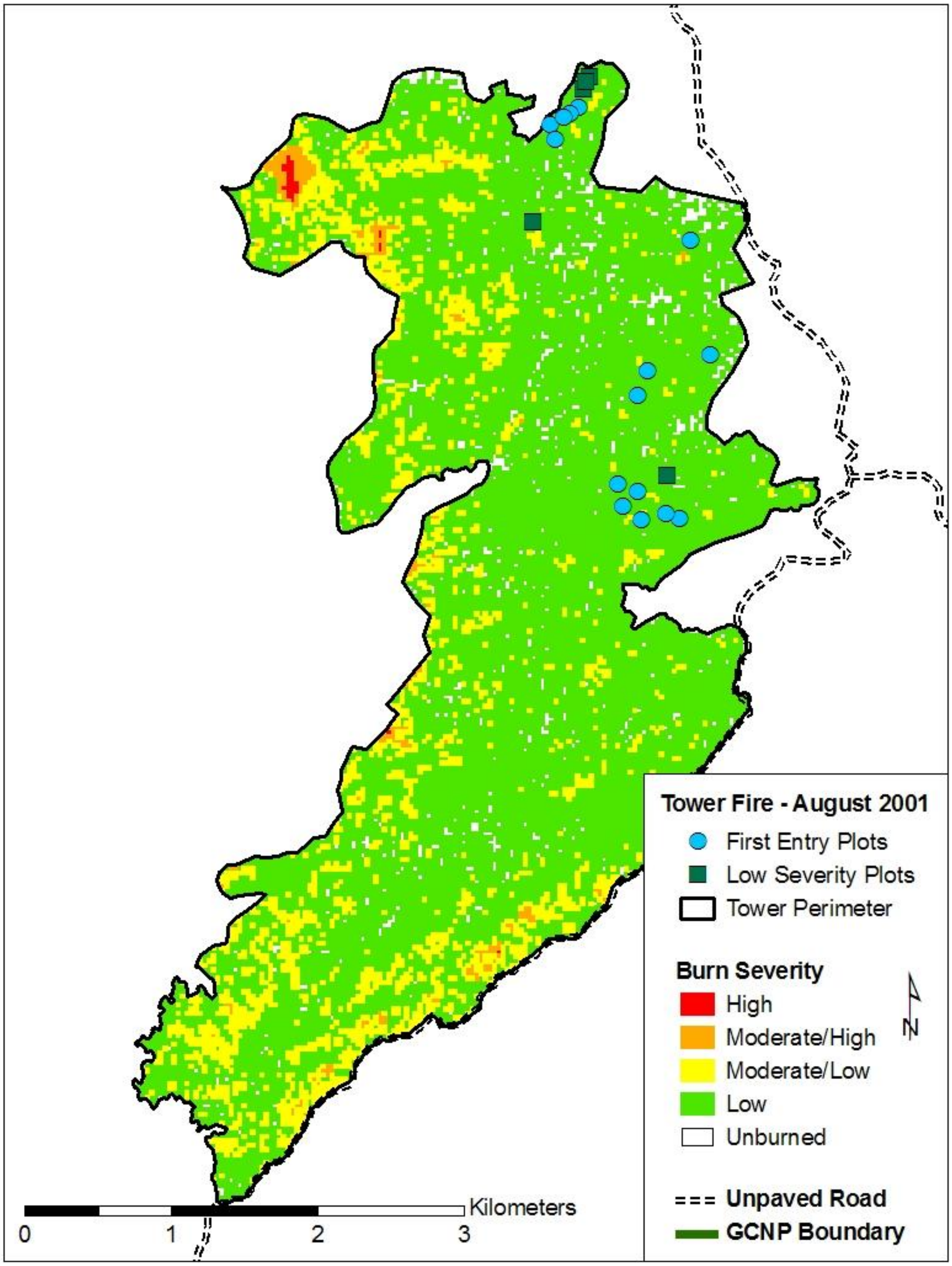


Figure 3.2. Tower Fire burn severity (August 2001). Circles represent first-entry, low-severity plots in the ponderosa pine with white fir encroachment forest type. Squares represent low severity plots in the dry mixed conifer forest type. Plots measured summer 2010.

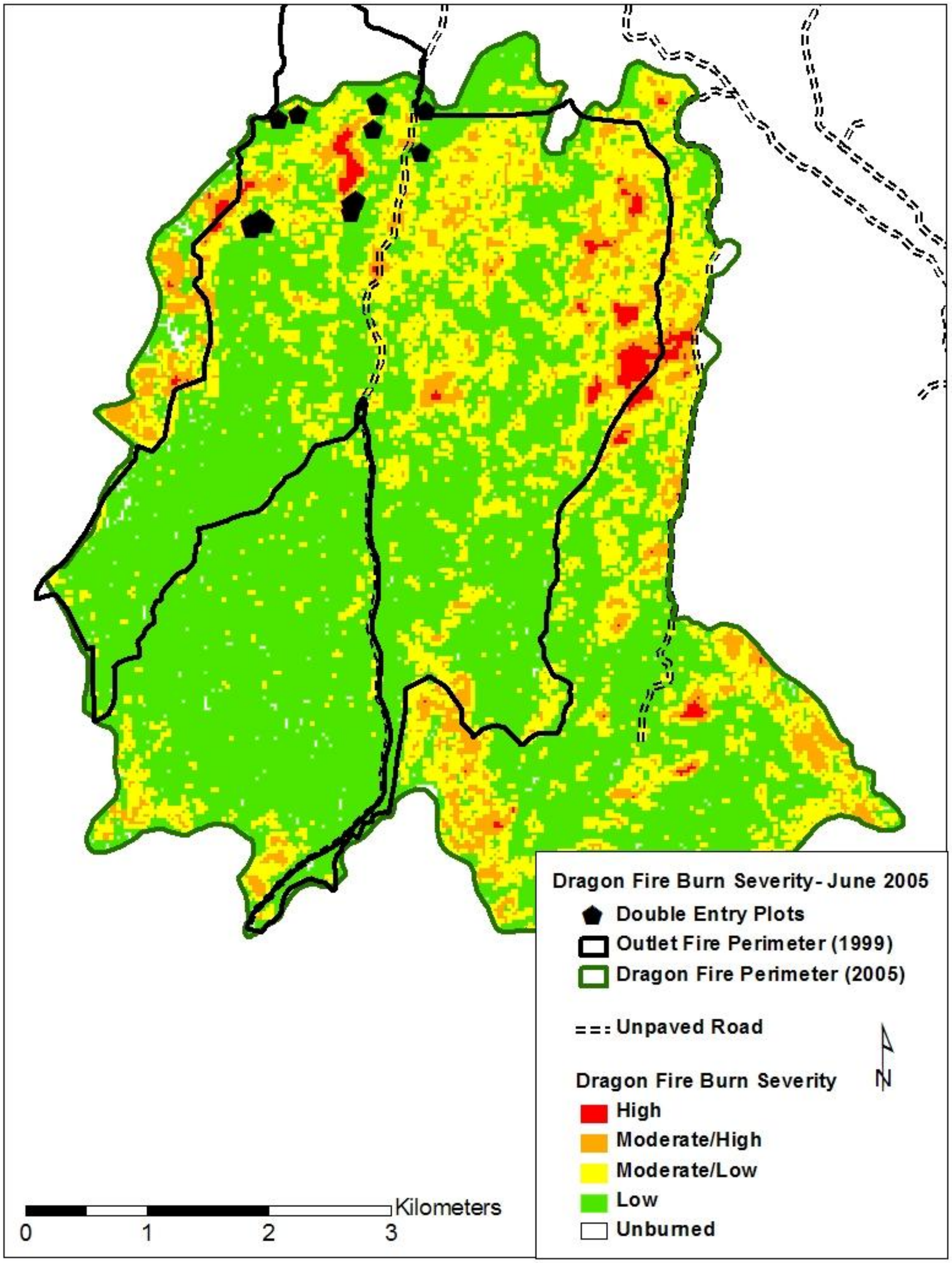


Figure 3.3. Dragon Fire (June 2005) burn severity and Outlet Fire (October 1999) perimeter. Hexagons represent second-entry, low-severity plots in the ponderosa pine with white fir encroachment forest type. Plots measured summer 2010.

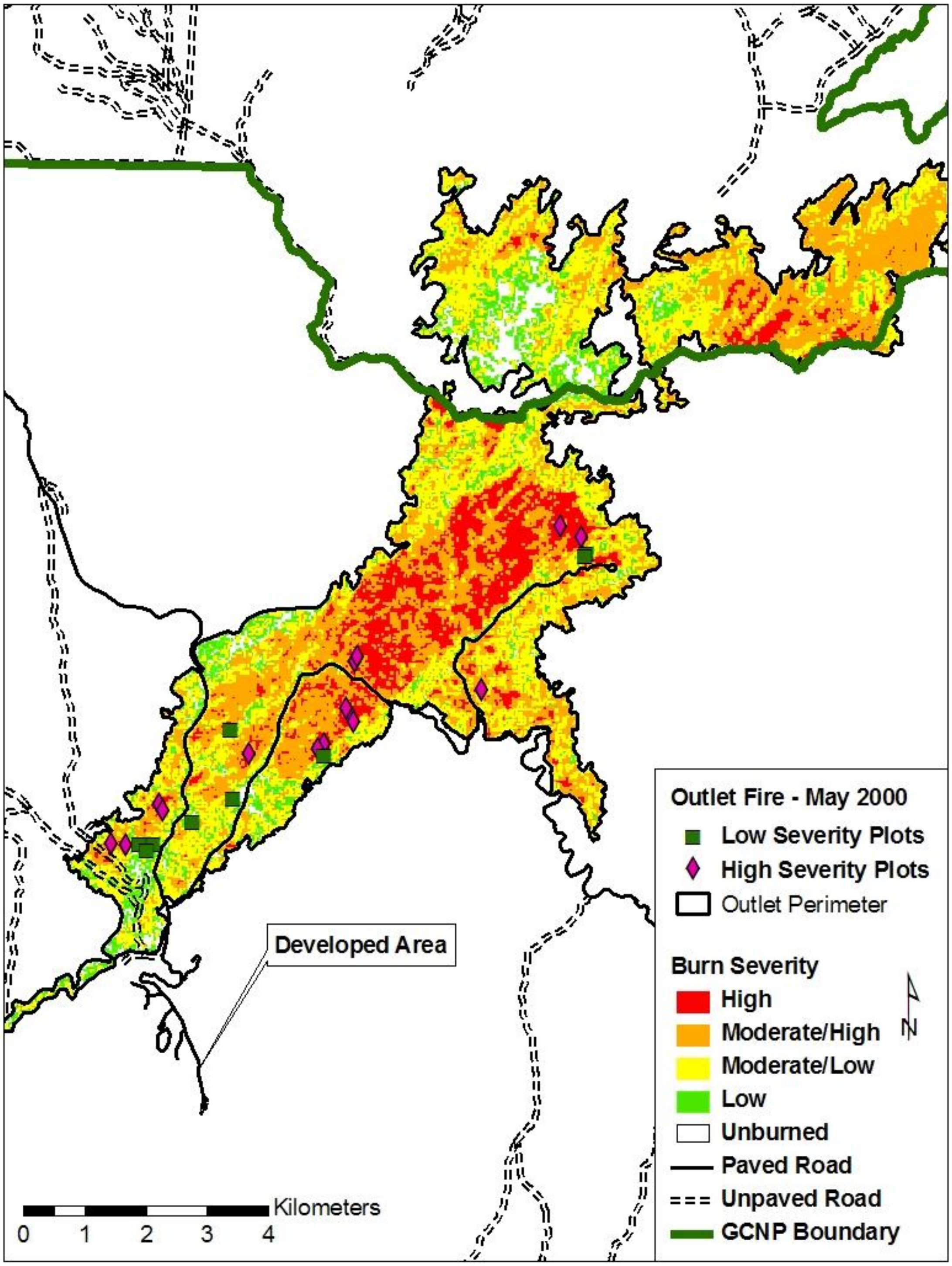


Figure 3.4. Outlet Fire burn severity (May 2000). Squares and diamonds represent low-severity and high-severity plots in the dry mixed conifer forest type. Plots measured summers 2009/2010.

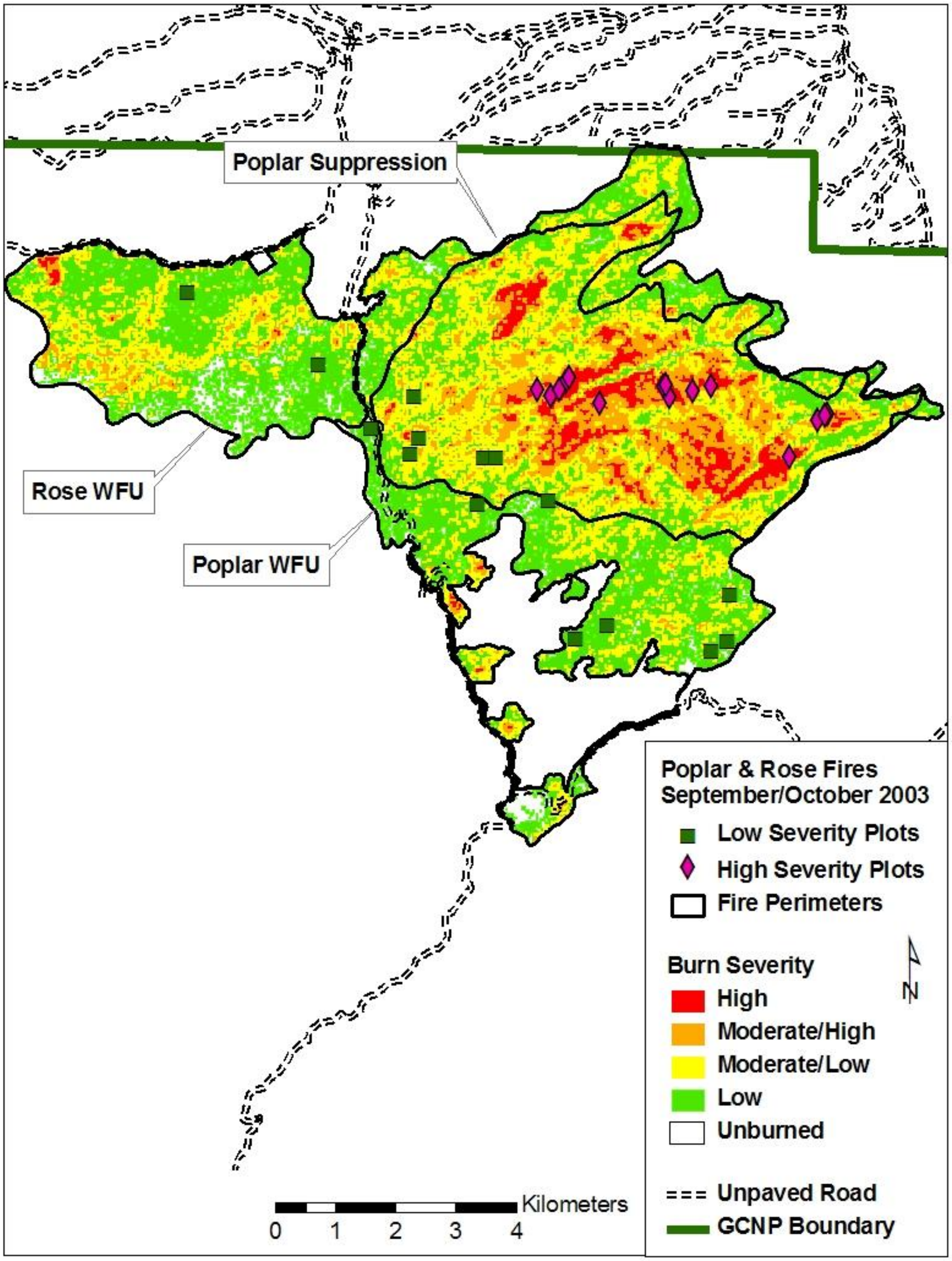


Figure 3.5. The Poplar Complex (September 2003) and Rose Fire (October 2003) burn severity. Squares and diamonds represent low-severity and high-severity plots in the dry mixed conifer forest type. Plots measured summers 2009/2010.

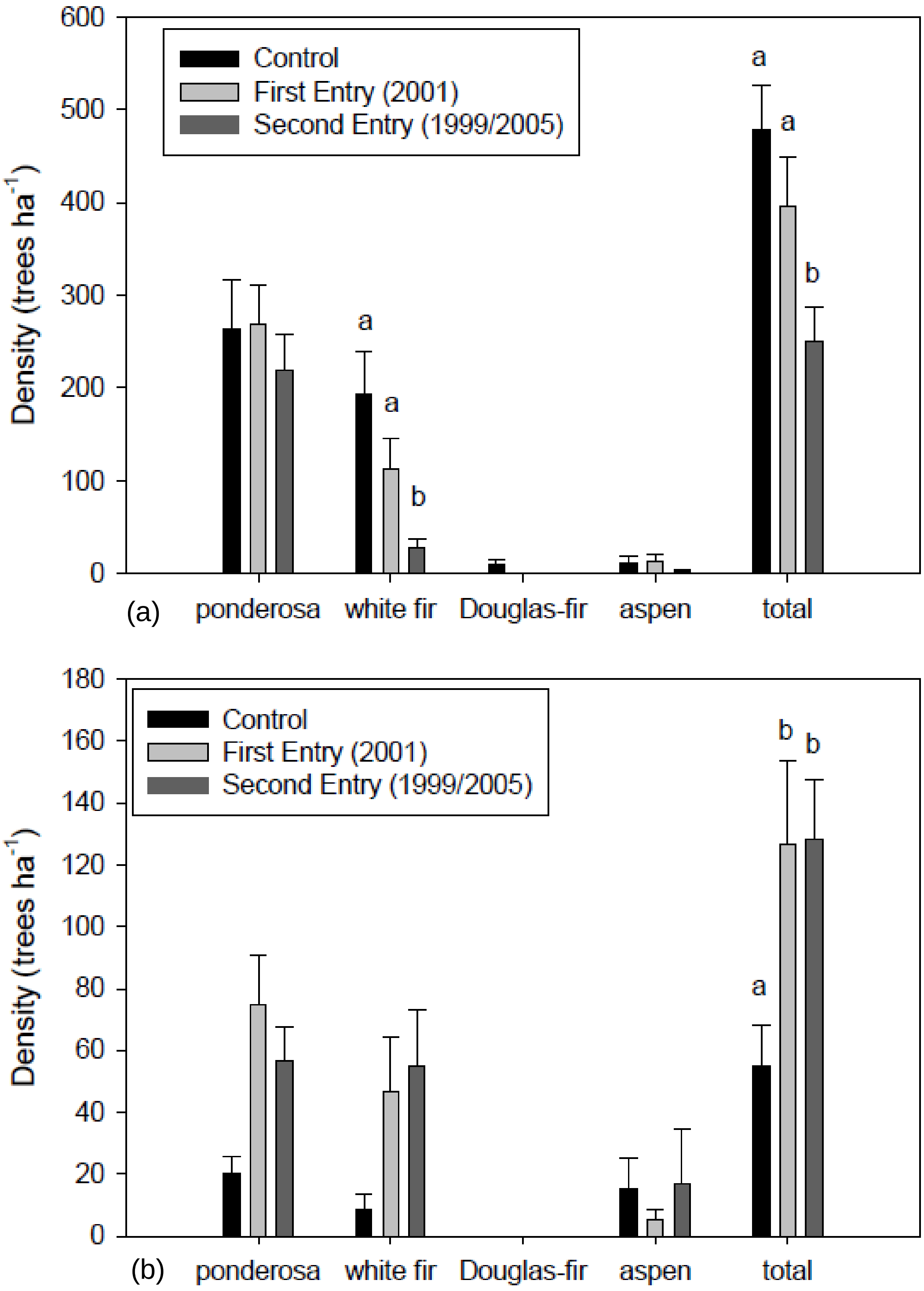


Figure 3.6. Live (a) and standing dead (b) tree densities by species and burn entry in the ponderosa pine with white fir encroachment forest type. Columns and bars represent means and standard errors. Letters denote a significant difference in means within each category; bars without letters indicate no significant differences.

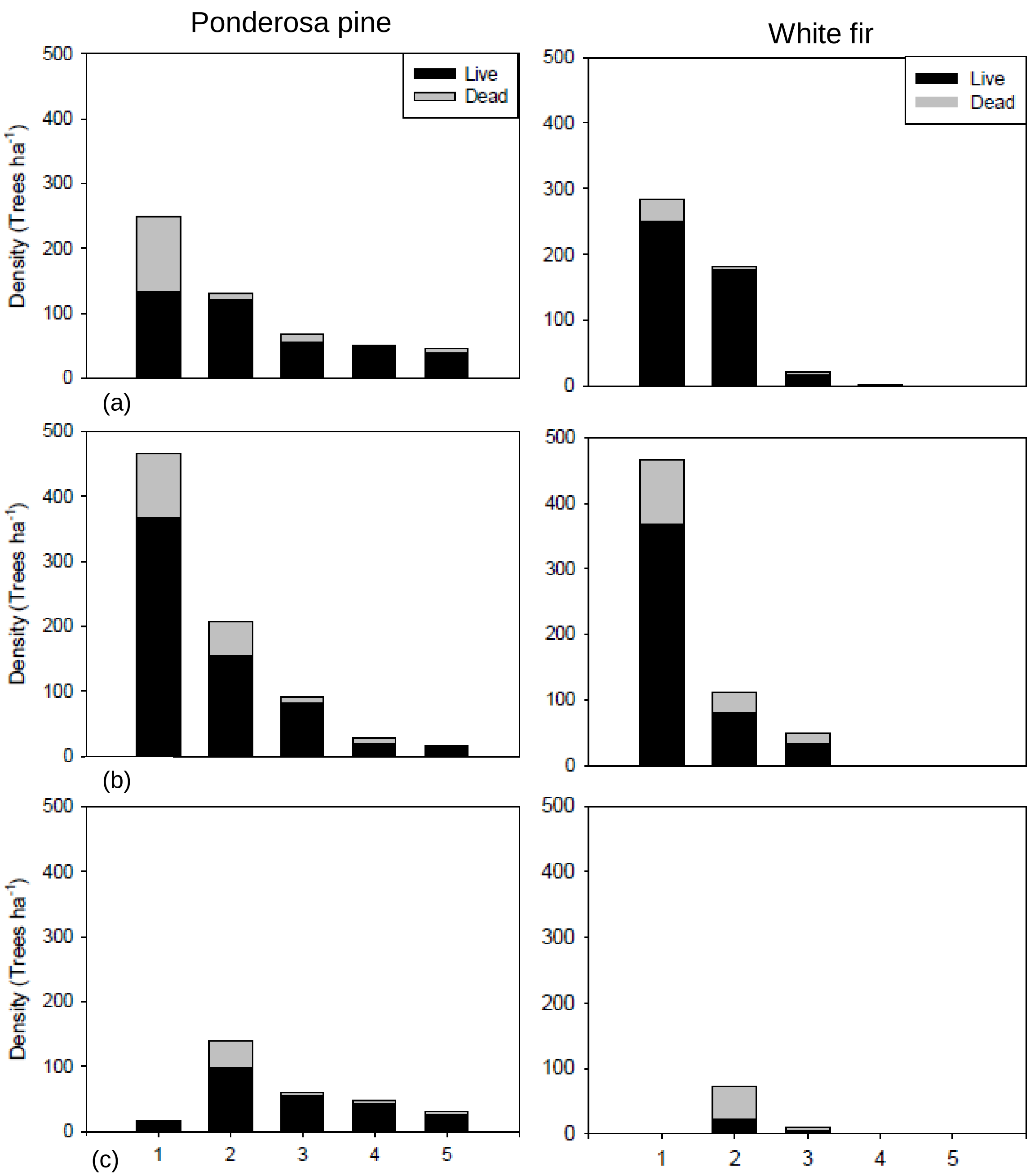


Figure 3.7. Live and dead ponderosa and white fir by DBH size class in unburned (a), first entry (b), and second entry fires (c) in the ponderosa pine with white fir encroachment forest type. DBH (diameter at breast height) size classes are: 1 = 0.0 – 9.9 cm; 2 = 10.0 – 29.9 cm; 3 = 30.0 – 49.9 cm; 4 = 50.0 – 69.9 cm; 5 = 70.0 cm +.

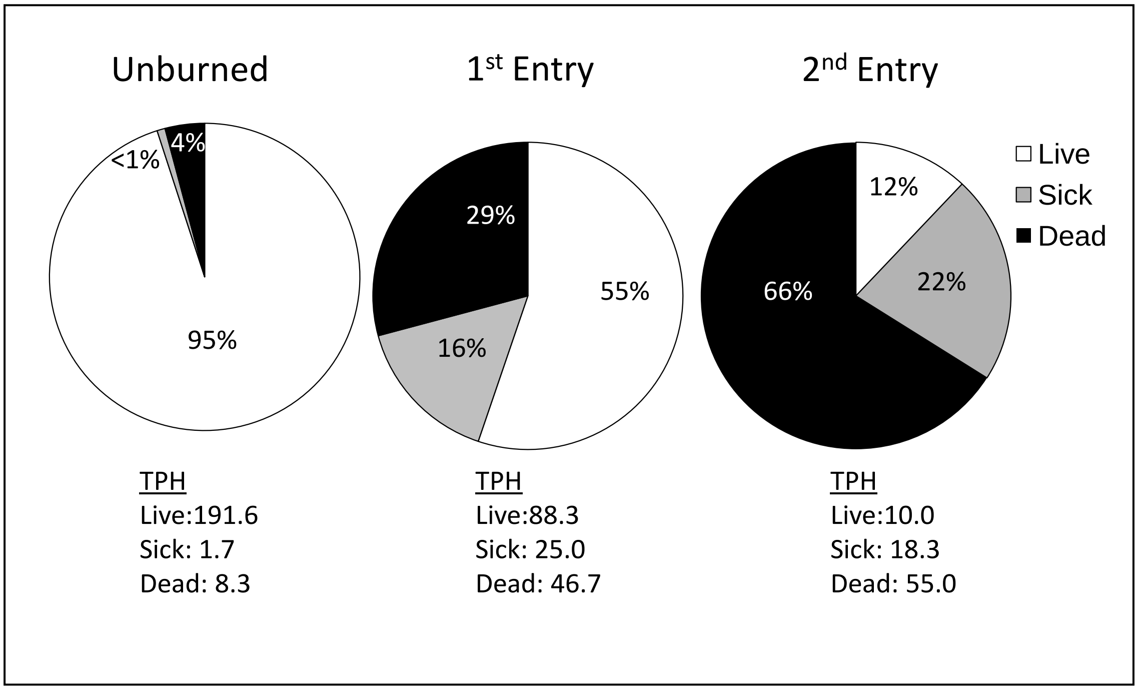


Figure 3.8. Live, sick and dead white fir overstory densities by burn entry in the ponderosa pine with white fir encroachment forest type. Sick tree is defined as any tree exhibiting signs of abiotic or biotic stress which will lead to mortality within 3-5 years. TPH = trees ha^{-1}.

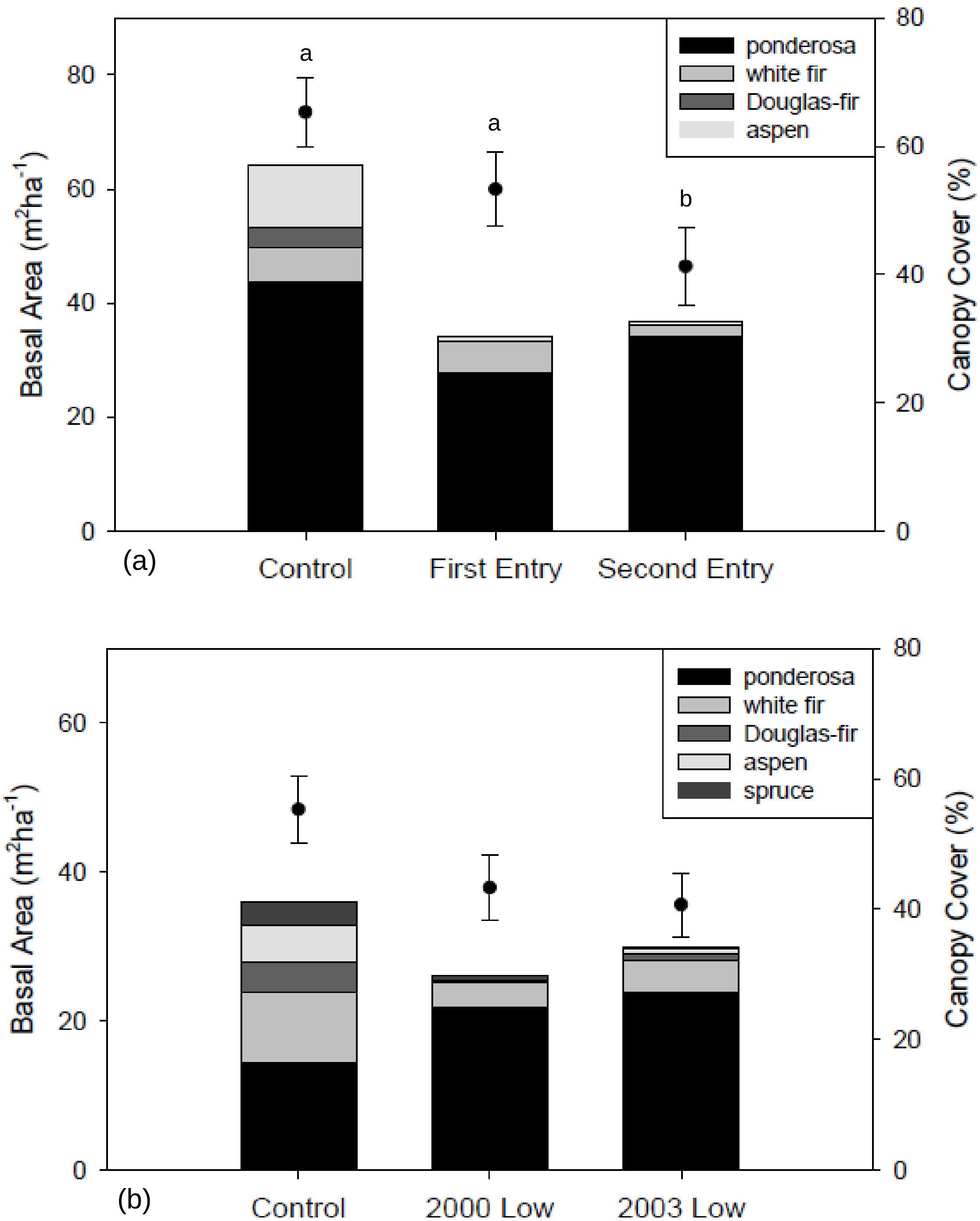


Figure 3.9. Basal area by species and percent canopy cover and standard error by treatment in the ponderosa pine with white fir encroachment (a) and in the dry mixed conifer (b) forest types. Stacked bars represent mean basal area by species. Circles and lines represent mean percent canopy cover and standard errors. Letters denote a significant difference in means in canopy cover; bars and circles without letters indicate no significant differences.

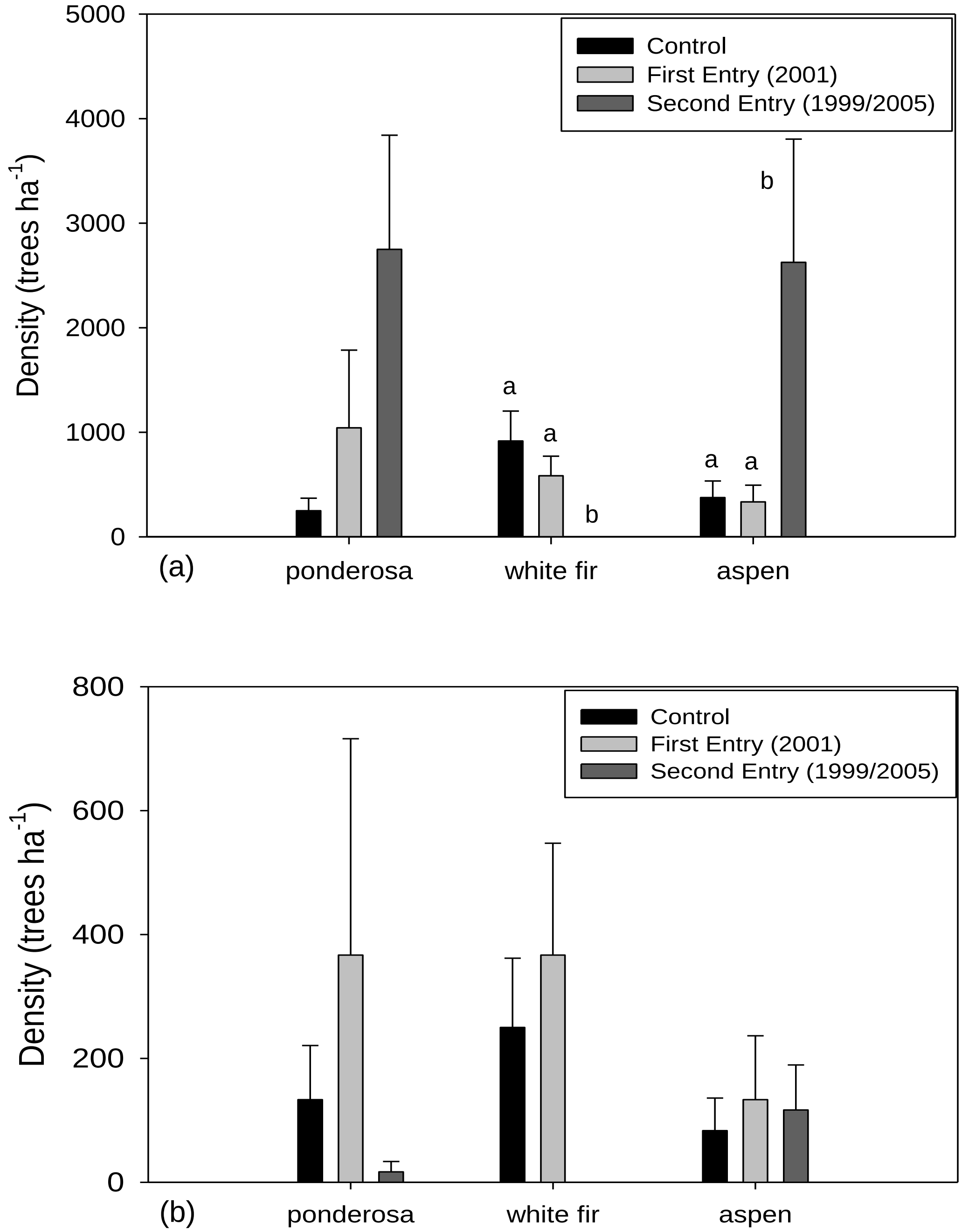


Figure 3.10. Seedling (a) and sapling (b) densities by species and burn entry in the ponderosa pine with white fir encroachment forest type. Columns and bars represent means and standard errors. Letters denote a significant difference in means within each category; bars without letters indicate no significant differences. A seedling is any tree ≥ 15 cm and < 137 cm in height. A sapling is any tree ≥ 137 cm in height and < 10 cm diameter at breast height (dbh).

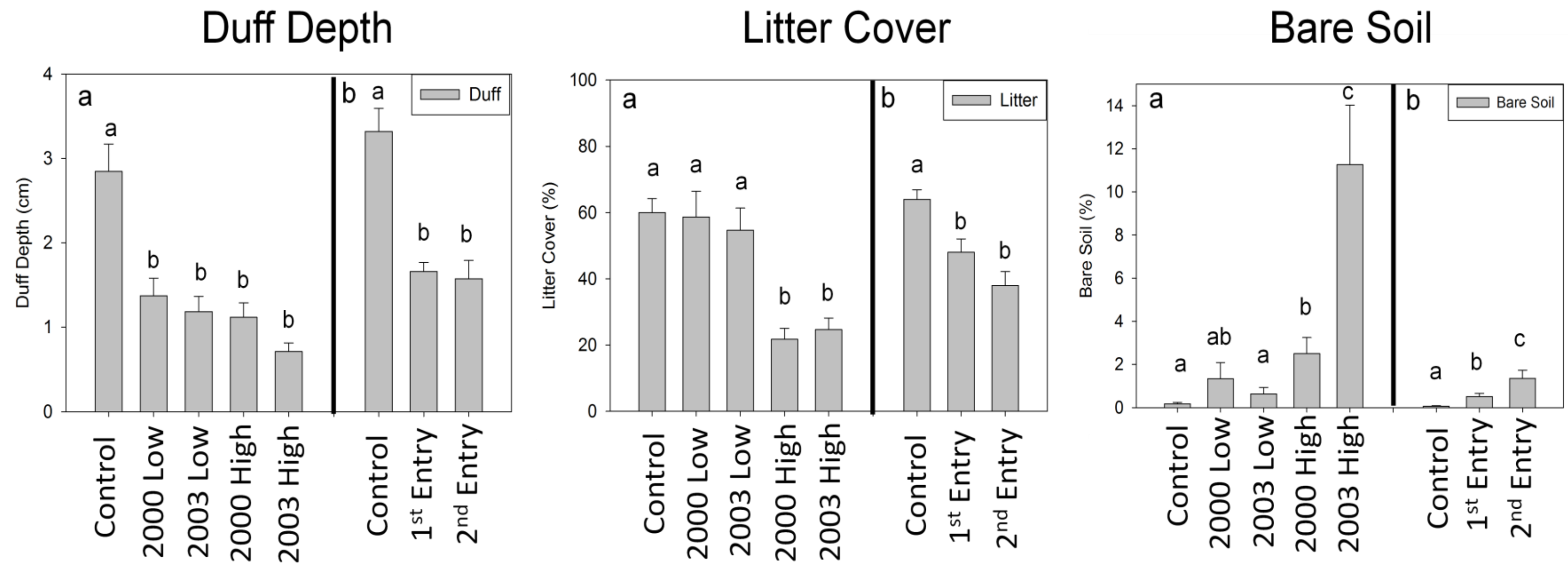
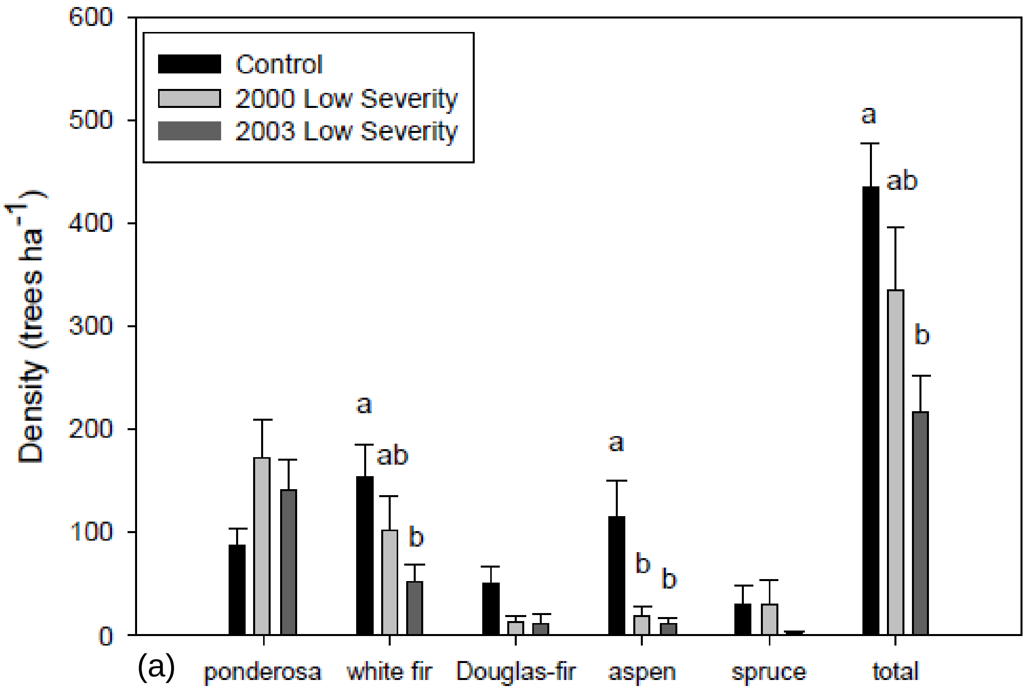


Figure 3.11. Duff depth, percent litter cover, and percent bare soil by treatment in the mixed conifer transition (a) and ponderosa pine/white fire encroachment (b). Columns and bars represent means and standard errors. Letters denote a significant difference in means.



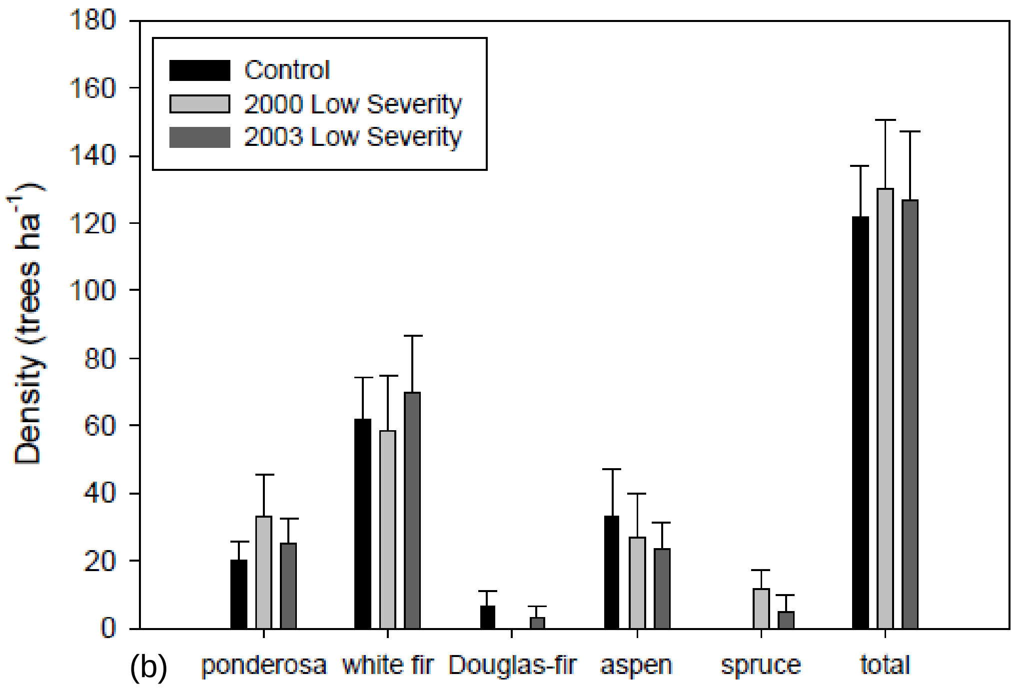
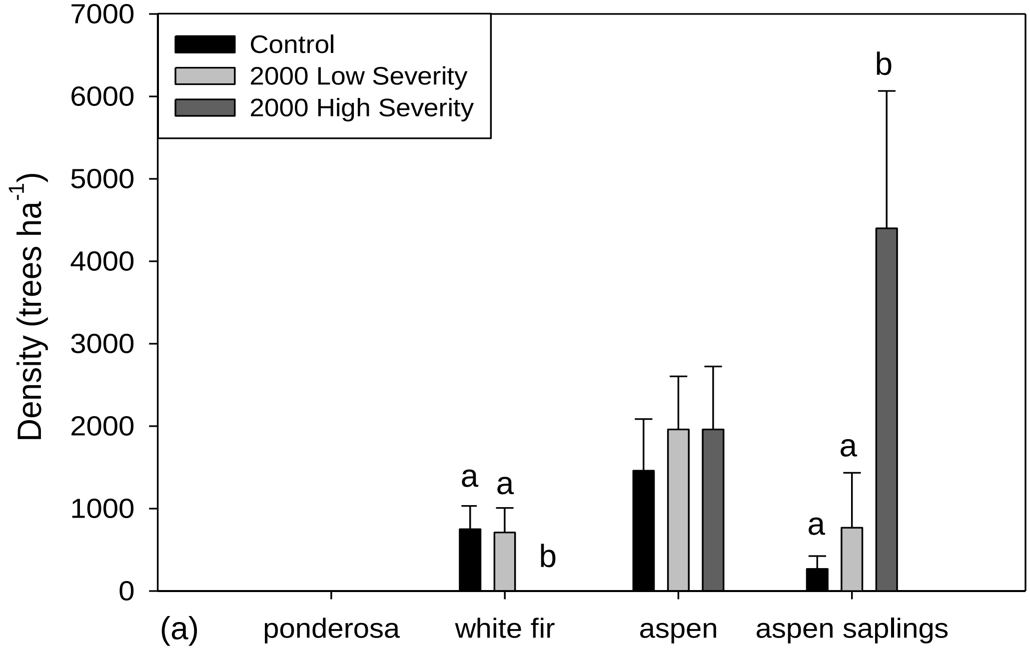


Figure 3.12. Live (a) and standing dead (b) overstory tree (≥ 10 cm dbh) densities by species by year of low-severity burn in the dry mixed conifer forest type. Columns and bars represent means and standard errors. Letters denote a significant difference in means within each category; bars without letters indicate no significant differences.



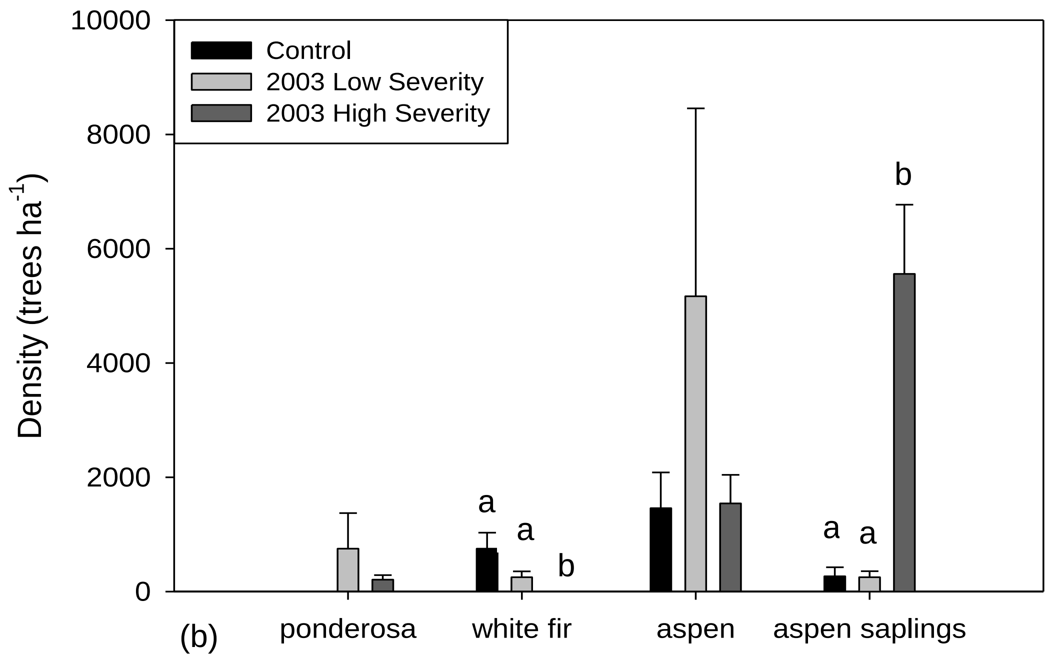


Figure 3.13. Tree regeneration (conifer seedlings; aspen seedlings and saplings) densities in unburned areas and in low and high-severity burns in 2000 (a) and 2003(b) in the dry mixed conifer forest type. Columns and bars represent means and standard errors. Letters denote a significant difference in means within each category; bars without letters indicate no significant differences.

CHAPTER FOUR

Management Conclusions

Grand Canyon National Park's (GCNP) fire management program has been active in managing fire (both prescribed and more recently wildland) for over 30 years. The historic forest structures and fire regimes of the ponderosa pine and mixed conifer forests in the Park are well documented (Wolf and Mast 1998; Fulé et al. 2003a,b) and GCNP has a successful monitoring program of fire effects (USDI 2003) in the ponderosa pine forests. As more fire is planned in forest types other than ponderosa pine, more scientific understanding is needed of fire effects and tree regeneration to predict future forest conditions and fire behavior. Our study compared conifer and aspen regeneration success across two forest sub-types (ponderosa pine with white fir encroachment; dry mixed conifer), burn-entry (unburned; first-entry burn; second-entry burn) and burn-severity (low; high).

Many frequent-fire forests in the western US have experienced species shifts and densification associated with over 100 years of fire exclusion (Abella et al. 2007). In the mixed conifer forests, species shifts include downward movement of less fire-tolerant, later successional species (Keeling et al. 2006; Fulé et al. 2009). In the ponderosa pine forests, densification of the existing forest has created conditions where forests are more susceptible to insect outbreaks and uncharacteristic stand-replacing fire (Covington et al. 1997, Allen et al. 2002). Many forests have also experienced recent aspen mortality and decline or aspen overstory replacement from conifer encroachment (Smith and Smith 2005; Fairweather et al. 2007; Worrall et al. 2008). All of these changes have led managers to apply different treatments across the landscape to reduce tree densities and fuel loading to promote resiliency.

Aspen, one of the most widely distributed trees in North America (Perala 1991), is an important tree for wildlife and biodiversity. In Arizona, the recent mortality in the overstory coupled with a lack of regeneration from fire exclusion or grazing pressure (Fairweather et al. 2007) has caused an increased interest in aspen restoration projects. While GCNP is not managing for aspen specifically it is important to understand the response of aspen to fire in both the ponderosa pine with white fir encroachment ('encroachment') and dry mixed conifer ('mixed conifer') forest types. In the encroachment type, of the 11 overstory aspen trees found in the 15 second-entry, low-severity plots only one remained alive and was classified as sick. There was evident aspen in the understory, most notably in the seedling size class. This new cohort of aspen likely established or sprouted following the second-entry burn and might reach the overstory if there is sufficient canopy opening and a fire-free period. In the low-severity burned areas of the mixed conifer forests there was also more standing dead aspen than live in the overstory. Similar to the encroachment type, there appeared to be sufficient aspen regeneration to replace the dying cohort, with the caveat of sufficient light from canopy openings to allow release from the understory. The largest percentage of healthy live to standing dead aspen (85%) was found in the unburned plots in the mixed conifer forest. The percentage of healthy to standing dead aspen in the encroachment type was notably lower (44%). As pre-fire data was not available we cannot definitively conclude that the abundant aspen mortality in the first- and second-entry, low-severity burns was fire-related. Other studies have noted a higher percentage of dead aspen in ponderosa pine forests compared to mixed conifer forests in Arizona (Ganey and Vojta 2011; Zegler in review).

In the encroachment forest type repeated low-severity fire can propagate extensive ponderosa pine regeneration. Additional fire may be needed to reduce densities to more desirable levels. Average densities of 3000 ponderosa pine seedlings ha^{-1} following a second-entry burn are far more than needed for overstory replacement (Bailey and Covington 2002). As the regeneration grows larger, individual trees will become less susceptible to fire mortality while the dense thicket becomes more susceptible as a whole to stand-replacing fire events (Lentile at al. 2006). Fire repeated more frequently than every 20 years, when seedlings and saplings are the most fire susceptible, may keep regeneration numbers at more sustainable levels for desired future forest structure. However, frequent prescribed fire should be implemented to enhance spatial heterogeneity and avoid uniform burns that cause extensive mortality of regeneration across large areas.

Due to the impending wilderness designation for much of GCNP's North Rim (USDI 1995) it is improbable that future mechanical treatments will occur in the ponderosa pine forests, thus fire will remain the most utilized treatment for reducing tree densities. Frequent burning, especially in areas visited by humans, can create conditions where non-native invasive species can colonize and out-compete native vegetation. Due to this risk an argument may be made that repeated entries should occur as infrequently as possible to reduce the risk of non-native plant invasions (Keeley 2006), while still meeting the objectives of reducing fuel loads and restoring forest structure to a more resilient state. Prevention is the most cost and time effective method in the battle against invasive plants and future care should focus on preventing fire-regime changers, such as cheatgrass (*Bromus tectorum*), from invading recently burned sites (Keeley and McGinnis 2007). The fire return interval in the ponderosa pine and mixed conifer forests historically varied dependent on vegetation, topography and climatic conditions (Fulé et al. 2003a). Allowing a variable fire return interval will be beneficial to the forests as it promotes heterogeneity within stands and across forest types.

Ponderosa pine is the most drought-tolerant tree found in the mixed conifer forests of the North Rim, yet it has suffered reduced regeneration rates due to fire exclusion and increased canopy closure (Mast and Wolf 2004). As artificial regeneration isn't a practical option, it is necessary to understand the regeneration response of ponderosa pine to recent fires with the dry mixed conifer type. Not all fires in the mixed conifer forest have promoted ponderosa pine seedling establishment. Future monitoring in the mixed conifer forest will allow managers to note which fires fail to propagate ponderosa pine regeneration. Aimed with that knowledge managers may decide to give priority for repeated, frequent-fire in areas lacking ponderosa pine regeneration over areas with successful regeneration. Fire is effective in preparing seedbeds for ponderosa pine germination and establishment; hence, fire should be repeated until successful establishment occurs.

Past extreme droughts throughout the West have generated concern about the ability of forests to withstand changing climatic conditions. Recent tree mortality in Arizona from a variety of causal factors is correlated with drought conditions (Fairweather et al 2007; Negrón et al. 2009; Ganey and Vojta 2011). Increased tree mortality is only one concern of a drier climate; fire behavior is directly linked to climatic conditions as well. Fulé and others (2003) found that in the ponderosa pine forests of the North Rim, historic fires occurred during years with an average Palmer drought severity index (PDSI) of -1.89. Widespread fires at all elevations occurred during extreme drought years with an average PDSI of -3.28. Historic fire reconstructions suggest past fires covered 5000 ha during regional fire years with the potential to cover 24,000 ha in more extreme fire years (Fulé et al. 2003). With predictions for the Southwest to become warmer and drier (Seager et al. 2007), and the size of wildland fires increasing in Arizona (www.nifc.gov), concern is growing over the amount of high-severity fire in ponderosa pine and mixed conifer forests. Uncharacteristically large stand-replacing fire in ponderosa pine forests is troublesome due to the loss of ecosystem services including but not limited to: carbon sequestration (Dore et al. 2008), water infiltration rates (Robichaud 2000), soil retention (Agee 1993), biodiversity (Savage and Mast 2005) and cultural resources (DeBano et al. 1998).

The recent high-severity fire in the mixed conifer forests on the North Rim caused an immediate shift in forest type from mixed conifer forests to mostly pure aspen forests (Vankat 2011). Aspen are relatively drought-intolerant so not all of the new aspen forests will flourish (see Table 3.7) and conifer encroachment will occur at varying rates across the landscape. Although not statistically proven from this research, visual evidence suggests that aspen on drier aspects, or at lower elevations, are more susceptible to injury and/or death related to moisture stress. The large patches of even-aged aspen forests created from stand-replacing fire will potentially become susceptible to damage, decline or future stand-replacing fire at similar times upon reaching maturation. Managers might consider creating patches within extensive even-aged aspen forests to create a mosaic of age structures. Pure aspen stands do not readily burn (Jones and DeByle 1985) yet it may be possible to introduce fire post leaf-fall during drought conditions.

In addition to addressing our post-fire research objectives, we summarized burn severity across the GCNP landscape using GIS and fires that burned between 2000 and 2009. We summarized the fires by both fire type (prescribed, wildland fire use / wildland, and suppression) and forest type (ponderosa pine, mixed conifer, spruce-fir and all others). We also analyzed high-severity patches across the landscape and across three forest types: ponderosa pine, mixed conifer and spruce-fir. Prior to analysis, we removed all high-severity patches smaller than 0.5 ha.

The total acreage burned from 2000-2009 in wildland fire use (WFU) or wildland fire was nearly double the acreage burned in prescribed fire (Table 4.1). The total percentage of high-severity fire was greater in wildland fires as compared to prescribed fires yet still only accounts for 16% of the total acreage burned in GCNP in the last 10 years. The majority of fires have occurred in the ponderosa pine forests with the majority of high-severity fire occurring in the mixed conifer forests (Table 4.2).

The largest high-severity patch occurred in the Outlet fire in 2000 and covered 1296.2 ha (Table 4.3). The majority of this patch (86%) occurred in the spruce-fir forest type. The historical disturbance regime of spruce-fir forests is typically characterized by infrequent, stand-replacing fire. Research on the size and frequency of fire in spruce-fir forests is not as extensive as in ponderosa pine and mixed conifer forests, yet it is widely accepted by researchers and managers that stand-replacing fire is within the historical range of variation (Agee 1993). Between 2000 and 2009 only 10 high-severity patches exceeded 100 ha in size. The mean high-severity patch size was 6.3 ha with the median patch size 1.2 ha and a total of 808 high severity patches greater than 0.5 ha across GCNP (Table 4.3). If GCNP wishes to minimize the amount of high-severity fire across the landscape, suppressing fires during extreme drought years will help prevent large, stand-replacing fire events. During moderate drought years stand-replacing fire in the spruce-fir forests might be allowed to create a mosaic of varying aged stands to prevent future large, continuous, high-severity fires.

In Yosemite National Park, Collins and Stephens (2010) found larger high-severity patch sizes in fir-dominated forest types compared to smaller high-severity patches in pine-dominated forest types. They also found more high-severity fire during extreme fire behavior periods and found that the shrub type saw more frequent yet smaller high-severity patches. Another study in Yosemite by Collins and others (2007) found that increasing time since last fire and different temperatures contributed to changes in burn severity in vegetation dominated by Jeffrey pine (*Pinus jefffryi*), red fir (*Abies magnifica*), juniper (*Juniperus spp.*), and meadows. In vegetation dominated by white fir (*Abies concolor*), lodgepole pine (*Pinus contorta*) and shrubland, burn severity was correlated with wind speed and slope. In the future, GCNP may wish to delineate mixed conifer forest to aspen or shrub conversions post stand-replacing fire so that future high-severity fire in the aspen or shrub types is not counted towards the 30% maximum high-severity fire allowed in the restricted Mexican spotted owl habitat.

The large number of small high-severity patches is beneficial for creating landscape heterogeneity. It appears that repeated low-severity fire in the encroachment forest type has little to no impact on existing tree densities in the overstory. Allowing small patches of high-severity fire within these forests will promote heterogeneity, although thick stands of ponderosa pine may establish in these patches (Ehle and Baker 2003; Savage and Mast 2005) further perpetuating more high-severity fire. If ponderosa pine fails to establish within high-severity patches the resultant shrub fields will provide wildlife habitat and potentially create natural fuel breaks.

Grand Canyon National Park plans to eventually allow a more natural fire regime, which is commendable as most visitors come for the view and not the forests. However, the long fire-free period in the ponderosa pine forests of the North Rim has created dense forests that are more susceptible to insects, disease and stand-replacing fire (Covington and Moore 1994b). Re-establishing the historic fire regime of low-intensity, surface fires will not restore the dense overstory of fire-tolerant ponderosa pine to more historical, and thus resilient, levels (Fulé et al. 2006; Kobziar et al. 2006; Johnson 2011). Continued monitoring should ensure early detection of future mountain pine beetle outbreaks. Mechanical treatments such as hand thinning projects may be necessary to help restore the overcrowded ponderosa pine forests.

The geographical remoteness of GCNP, particularly the North Rim, allows for a robust fire management program. Expanded monitoring in the mixed conifer forests will help guide GCNP’s management actions in the future. Accepting more high-severity fire across the landscape, especially in the spruce-fir forests, may be a necessary component of adaptive management, as both fire and tree mortality continue to increase across the landscape in response to changing climatic conditions. Managers will need to continue with adaptive management and monitoring of all forest types to continue creating resilient forests for the future.

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Table 4.1. Amount of low, moderate and high burn-severity by fire type from 2000-2009. Values are in hectares, percentage of total is in parentheses.

Fire Type	Total	Low	Moderate	High
Prescription	10,504.2	7,310.3 (70%)	2,442.8 (23%)	751.1 (7%)
Wildland Fire Use & Wildland Fire	19,504.6	11,964.4 (61%)	4,709.8 (24%)	2,830.4 (15%)
Suppression	3,050.9	505.8 (16%)	660.2 (22%)	1,884.9 (62%)
Total	33,059.7	19,780.5 (60%)	7,812.8 (24%)	5,466.4 (16%)

Table 4.2. Amount of low, moderate and high burn-severity by forest type from 2000-2009. Values are in hectares, percentage of total acreage burned in that burn-severity class is in parentheses.

Forest Type	Total	Low	Moderate	High
Ponderosa pine	18,055.0 (55%)	13,336.2 (74%)	3,771.8 (21%)	947.0 (5%)
Mixed conifer	9,771.7 (29%)	4,395.7 (45%)	2,868.2 (29%)	2,507.8 (26%)
Spruce-fir	2,004.7 (6%)	240.1 (12%)	345.5 (17%)	1,419.1 (71%)
Other	3,244.0 (10%)	1,817.5 (56%)	831.1 (26%)	595.4 (18%)
Total	33,075.4	19,789.5	7,816.6	5,469.3

Table 4.3. The number of high-severity patches and the mean, median, and maximum size of high-severity patches by forest type from 2000-2009. Values are in hectares and all patches are greater than 0.5 hectare in size.

Forest Type	Total	Mean	Median	Max
Ponderosa pine	398	2.0	0.8	47.7
Mixed conifer	314	7.6	1.1	848.7
Spruce-fir	45	31.0	1.4	1117.8
Total	808	6.3	1.2	1296.2