

**EFFECTS OF FIRE DISTURBANCE AND GRASS REMOVAL ON TREE GROWTH IN  
MESIC SAVANNAS**

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## ABSTRACT

# **SAVANNA APPLICATIONS: INVESTIGATING THE RELATIONSHIP BETWEEN FIRE, GRASS, AND TREES IN MESIC SAVANNAS**

By

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The coexistence of trees and grasses in the savanna landscape has confounded scientists for decades. In contemporary research, the mechanisms for codominance are usually explained through two different hypotheses depending upon annual precipitation level: rooting-niche-based and disturbance-based. The rooting-niche hypothesis states that grasses and trees have different root depths, and thus occupy different competitive niches. The disturbance-based hypothesis argues that life disturbances, such as fire, are what limit tree growth and survival, thus maintaining the grass-tree coexistence. A key principle in savanna ecological models is that the fire regime is a major determinant of vegetation cover where late-dry season fires are more damaging to trees than early fires. This is indirectly linked to the tree-grass coexistence, as less grass would provide less competition for resources, meaning small trees may grow faster in order to escape fire. Thus, such disturbances as fire and grass removal may have the ability to alter the growth rate and height of juvenile and adult trees. This study analyzed the effects of fire disturbance and grass elimination on tree growth in a mesic savanna in Mali, West Africa, for a period of two years. This study divided two study sites into eight experimental plots, four were subjected to different grass reduction treatments (clipping, hoeing, herbicide, and grazing), three were subjected to fire treatments (early, middle, and late-season), and one plot was set as the control variable. This study addresses three key hypotheses:

1. Grass removal will increase the rate of juvenile tree growth in terms of height and growth rate.
2. Late fires will slow tree growth rate the most, followed by mid-season and then early season fires.
3. Tree deaths will be highest on late-season fire plots followed by mid-season and then early-season.

All trees under grass removal disturbances and early-season fires had an increased growth rate as compared to the control trees. Late-season fires may trigger some trees to grow faster to overcome flames, but they also cause the most tree deaths. Trees of each size class—small juvenile, large juvenile, and adult—grow more rapidly with grass removal. The findings suggest that early burning and grass reduction practices have applications for increasing woody vegetation in savanna landscapes, which would increase carbon sequestration. However, it must be noted that local savanna inhabitants rely on grass cover for animal fodder and other uses; thus, converting grasslands to woodlands may have negative side effects.

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# CHAPTER 1

## INTRODUCTION

The savanna ecosystem is defined by a mixture of grasses and a discontinuous or grouped layer of trees and/or shrubs, where the canopy does not close. Scientists have long debated the reasons for which trees and grasses coexist on the landscape and the phenomenon is not completely understood, earning the title “the savanna conundrum” (House et al. 2003, 1763). In contemporary research, the mechanisms for codominance are usually explained through two different hypotheses: the rooting-niche-based hypothesis and the disturbance-based hypothesis. The rooting-niche-based hypothesis describes how trees and grasses avoid competition for water through different rooting niches in soil layers, as grasses acquire nutrients from the surface soil layers and trees acquire nutrients from the deeper soil layers (Walter 1971). Although this theory holds true in many arid savanna landscapes, it does not account for how tree seedlings become established; as such, it does not account for juvenile trees that have not yet penetrated the deeper soil layers and have roots within the upper soil layers where they compete directly with grasses. By contrast, the disturbance-based hypothesis argues that life disturbances, such as fires or herbivory, are major factors that limit tree growth and survival, including juveniles, thus maintaining the grass/tree coexistence (Higgins, Bond, and Trollope 2000; Sankaran, Ratnam, and Hanan 2004). While many studies have analyzed adult tree-grass competition from a rooting-niche-based perspective (Holdo, Nippur and Mack 2018; February and Higgins 2010; Kulmatiski et al. 2010), others have conducted analysis from a disturbance-based perspective (Laris and Dembele 2012; Staver, Archibald, and Levin 2011; Sankaran et al. 2005; Van Langevelde et al. 2003; Higgins, Bond, and Trollope 2000). However, few studies have focused on the competition between juvenile trees and grasses, specifically in a mesic savanna where

there is enough water available to support a dry forest. This relationship is an important component to understanding the coexistence of trees and grasses in savannas, which ultimately benefits the understanding of our changing environment and climate action plans.

Very few studies have looked at how grass competition affects juvenile tree growth rates in savannas. *In situ* tree height data are one of the best ways to analyze this relationship because it allows direct measurements of tree height and documentation of tree stage of life. In addition, a key principle in savanna ecological models is that the fire regime is a major determinant of vegetation cover where late dry season fires are more damaging to trees than early fires. Thus, this study combines a study on the relationship between fire disturbances and tree growth, and a study of grass elimination in an effort to provide valuable data for addressing the current knowledge gap.

Previous research regarding tree and grass competition primarily focused on adult tree growth with or without grass competition, but Riginos (2009) is one of the few studies to analyze juvenile tree and grass competition in savannas. By clipping and spraying herbicide on grasses around trees of three size classes (sapling, juvenile, adult) and measuring tree growth rates, Riginos (2009) observed that trees, regardless of demographic size, grow faster without grass competition. While their experimental design focused on different tree demographics, one shortcoming of the study was that it took in a dry savanna (precipitation = ~500 mm/yr) where theory suggests that trees and grasses should be in competition for resources. This thesis adopts a similar methodological approach to that of Riginos (2009) but in a mesic savanna (precipitation > 750mm/yr) where theoretically there is sufficient water for both grasses and trees and, as such, disturbances are considered necessary to prevent tree canopy closure. Additionally, this study

incorporates different methods of grass removal (clipping, hoe-farming, herbicide, and grazing) to analyze how different practices affect competition.

While many studies have focused on the relationship between burning regimes and different tree demographics growth rates, most of these studies focus more on how fire affects tree growth without including the effects of tree-grass competition. Werner (2011) looked at how trees of five size classes (small juveniles, large juveniles, small saplings, large saplings, poles) grow with different fire season burns, observing that grassy/non-grassy understories played an important role for how much a tree can grow after fires. One shortcoming of this study is the experiment did not take into account how the fire season affects grass growth; for example, early-season fires may eliminate grass competition for water thus allowing trees to receive more water longer into the dry season. This thesis differs from Werner's work in that it incorporates fire disturbances and grass removal to examine how combinations of disturbances can affect tree growth and savanna stability.

The tree-grass-fire relationship in savanna systems is very complex and may differ from one savanna to the next depending on location, annual precipitation amount, tree species, and tree stage of life. This thesis provides additional information about tree-grass competition in mesic savannas by analyzing tree growth rates through different grass elimination techniques and different fire season burn regimes. Four types of grass treatments were chose either to represent common anthropogenic practices for managing grass growth (hoe-farming, livestock grazing) or to represent other grass removal methods used in similar studies (clipping, herbicide) (Riginos 2009). The primary goal is to test whether grass removal matters for tree-grass competition in savannas through different grass elimination disturbance methods. This study also intends to document tree growth and dieback rates for trees that have gone through early, middle, or late-

season fire disturbances, as well as investigating how the disturbances may affect the growth of smaller, juvenile trees differently than taller, adult trees.

Understanding the relationship between trees, grasses, and disturbances is also important for the current social and political pressure towards afforesting grasslands and savannas in order to sequester more carbon (Bond et al. 2019). This issue has created a need for better understanding of the dynamic relationship of tree-grass competition in savannas, especially considering the importance of both grasses and trees as resources for local farmers and herders and the increase in existing projects intending to afforest/reforest grasslands (Brottem 2016; Bond et al. 2019).

There are four specific research questions:

1. How do the different types of grass removal disturbances (grass-clipping, hoe-farming, animal-grazing, and herbicide) affect growth rate/height in a mesic savanna?
2. How does burning in different seasons affect tree height/growth rate?
3. How does burning in different seasons affect tree deaths?
4. Do juvenile trees have better success escaping the fire trap without grass competition?

This study addressed these questions by collecting *in situ* tree growth data from village sites in a mesic savanna of Mali, West Africa. The data were studied using statistical analysis to better understand the relationships between grass/fire disturbances and tree growth, particularly for juvenile trees. The next chapter provides a literature review of the research on tree-grass competition and fire disturbances for savannas. Additionally, the chapter addresses the importance of this study in the context of existing afforestation plans implemented to reduce greenhouse GHG emissions versus usage of savannas by local inhabitants of the region.

## CHAPTER 2

### LITERATURE REVIEW

#### Tree/Grass Coexistence

As noted, explanation for the coexistence of trees and grasses in the savanna landscape has eluded scientists for decades with the rooting niche dominating early theorization and the disturbance hypothesis emerging in recent decades. The rooting-niche hypothesis states that grasses and trees have different root depths, and thus occupy different competitive niches (Scholes and Archer 1997). Scholes and Archer (1997) review the model used in the rooting-niche-based hypothesis created by Walter (1971). In this model, Walter suggests that trees have roots in the deep soil layers and top soil layers but use very little water in the top soil layers; however, grasses have roots only in the top soil layers. Because the woody and herbaceous plants are not acquiring water from the same layers, there would be no competition for nutrients between the two, thus creating a stable equilibrium with grass/tree codominance.

Walter (1971) originally intended the hypothesis to be valid only for dry savannas, as he reasoned that more humid savannas were controlled by other factors and disturbances, such as fires (Ward, Wiegand, and Getzin 2013). This hypothesis could help explain why mesic savanna patterns of root distribution are similar for trees and grasses, because both trees and grasses take water from the top soil layers. Holdo, Nippur and Mack (2018) verified this claim by assessing rooting depths of grasses and trees in an area of a savanna that extended from 450 to 750 millimeters (mm) of annual rain and found that trees had shallower roots at sites with more rain. This finding implies that Walter (1971) is correct that the rooting-niche hypothesis would be valid for dry savannas but does not carry over for mesic savannas. However, based on a content analysis of 36 core studies investigating the rooting-niche model, Ward et al. (2013) found that

the hypothesis is applicable even in some mesic savannas. By taking root measurements of trees and grasses, known as vertical resource partitioning, their study showed mixed results supporting the rooting-niche hypothesis for moist savannas, with six studies showing no support, four studies showing complete support, and three studies showing partial support (Ward et al. 2013, 623). These findings could result from other disturbance factors playing a greater role than rain does in limiting resources for trees in this environment.

Many scientists have tried directly testing the rooting-niche hypothesis and the findings of several cast doubt on the rooting niche hypothesis, often finding mixed results when testing in the field (February and Higgins 2010; Higgins, Bond, and Trollope 2000; Kulmatiski et al. 2010). February and Higgins (2010) tried measuring rooting systems of trees and grasses in an arid savanna and found that both had roots in the top soil layers and bottom soil layers. Additionally, Kulmatiski et al. (2010) injected deuterated water into the soil of trees and found that the trees do not absorb large amounts of water past 20 centimeters below the surface, even in non-rainy seasons. Together these findings suggest that these trees receive most of their nutrients from the top soils and would be in competition with grasses, which goes against the rooting-niche hypothesis.

An important factor of the hypothesis is whether or not young juveniles and tree saplings compete more strongly with grasses in the upper soil layers. Few studies have looked at the rooting-niche hypothesis specifically for juvenile tree and grass competition. In particular, it is uncertain whether there is more competition for juveniles and grasses in areas with high amounts of nutrients, such as mesic savannas. Competition could be critical given that smaller trees, that are below or within the flame zone in mesic areas, must grow fast enough to escape the fire trap.

As such, competition for water between grasses and juveniles takes on new meaning when rapid growth is required for juvenile tree success in fire prone environments.

The disturbance-based hypothesis argues that life disturbances limit tree growth and survival, thus maintaining savanna grass-tree coexistence. To test the validity of the disturbance-based hypothesis, Higgins, Bond, and Trollope (2000) developed a model that concentrates on disturbances to control the dominance of trees and grasses in savannas rather than different rooting depth. The results suggest that grasses and trees coexist through disturbances such as fire and herbivory. Fire intensity can produce variation in recruitment rates, and grazing is also shown to influence grass and tree dominance. These disturbances make it harder for trees to establish and grow, as fires burn trees while cattle and browsing animals can trample or eat young tree sprouts. On the other hand, grazing can also reduce fuel loads for fires, thus benefiting trees.

Unlike the rooting-niche model, the disturbance-based hypothesis assumes that the grass/tree codominance is unstable, and grasses and trees are only coexisting in savannas through disturbances that stop trees from overtaking grasses. As the theory goes, without disturbances, savanna landscapes would convert into forests as trees gradually shade-out sun-loving grasses. In support of this hypothesis, Sankaran et al. (2005) found that savannas with a mean annual precipitation above 650 mm would be considered unstable because the water levels are enough for trees to outcompete grasses and become the only dominant vegetation. Similarly, Staver, Archibald, and Levin (2011) found that only fire differentiates between a savanna and forest in areas with an annual rainfall of 1,000 to 2,500 mm annually. Both researchers conducted broad-scale studies using remotely-sensed data on fire regimes and tree cover.

Some scientists say that disturbances have the potential to create positive feedback loops that ultimately overthrow the grass/tree coexistence (Van Langevelde et al. 2003; Scholes and Archer 1997). For example, altering methods of grass or tree removal affects fuel loads and fire intensity, which can lead to one plant form dominating over another. In places where livestock has been allowed to graze in savannas for long periods, there has been a noticeable increase in tree dominance (often referred to as shrub or tree encroachment (Scholes and Archer 1997; Bassett and Zuéli 2000). Van Langevelde et al. (2003) created a model of the tree-grass relationship based on the effects of fire and herbivory and found it plausible to explain tree encroachment in savannas. Disturbances such as grazing and hoe-farming have the ability to cause a decrease in herbaceous vegetation which would decrease fuel loads and cause less intense fires, thereby increasing tree survival and growth rates (Laris and Dembele 2012). Conversely, disturbances such as browsing may decrease woody biomass thus facilitating grass growth creating a higher fuel load, which can cause grasses to dominate trees (Van Langevelde et al. 2003, 345).

Recently, scholars have postulated that the mechanism determining tree/grass coexistence varies according to annual precipitation. Savannas can be categorized by the amount of rain the area receives annually. Arid and semi-arid savannas receive 100 to 650 millimeters of rain a year while sub-humid (mesic) and humid savannas receive more than 650 millimeters of rain a year (Ward, Wiegand, and Getzin 2013, 620). According to this new approach, trees in arid savannas are thought to be limited by precipitation, whereas trees in mesic savannas are thought to be disturbance limited because, theoretically, there is ample precipitation to support a closed canopy woodland (Sankaran et al. 2005; Staver, Archibald, Levin 2011).

Although there remains debate over the specific limits of each theory, it is important to note that supporters of both hypotheses agree that rainfall benefits trees more than grasses (Greve 2013, 168). High rainfall theoretically allows more water to percolate into the deeper soil layers for trees, so they do not need to compete for water with grasses in the top soil layers. A longer rainfall season would also create fewer fires due to the moisture in the air and vegetation, thus creating a “high end” to the limit of disturbance-driven savannas. However, more recent research challenges this assumption. February et al. (2013) tested the effects of rainfall manipulation in both arid and mesic savannas found that an increase in rainfall had no measurable impact on tree growth. They did find, however, that reduced grass cover resulted in increased tree growth rates. The authors concluded that both grasses and trees benefit from higher rainfall amounts, which, therefore, *increase* the competitive pressure exerted by grasses on trees. As Scholes and Archer (1997) have argued, perhaps no single model is able to explain completely the different phenomena at all savanna locations. That said, careful review of the literature on the two hypotheses sheds some additional light.

### **Tree-Grass Competition**

Research to observe grass-tree competition has been conducted by removing grass in savannas for long periods, usually years. Riginos (2009) tested the effects of grass competition on tree recruitment with a grass removal experiment on trees of different heights in an arid (precip = ~500 mm/yr) East African savanna. To remove the grass, Riginos clipped the grass and sprayed it with herbicide, while covering the smaller trees for protection. The study was carried out for over two years, and she found that trees of all heights in plots with grass removed doubled in growth, suggesting that grass competition can substantially reduce the growth of trees

regardless of demographic stage. However, her study took place in a dry savanna where trees and grasses may be more in competition for resources, as water is limited.

Looking at grass-tree competition from a disturbance-specific approach, Laris and Dembele (2012) examined how hoe-farming, grazing, and burning affected tree mortality and growth on clay and sandy soils. This work was done on experimental plots that had previously been hoe-farmed for eight years prior to being fallow. The plots were assigned two treatments, burning and grazing. Some plots received only one treatment while other plots received both. Hoeing grass was found to increase tree recruitment because hoe-farming provides ideal conditions for seedling establishment by removing deep rooted perennial grass and breaking soil crust. Additionally, short grasses burn in early-season and at low intensity, so trees are more likely able to survive such fires in the area (p.18). These results suggest that indirect human activities may ultimately be affecting the tree-grass coexistence, and farming may play a key role in grass-tree coexistence.

Some studies use a modeling approach when determining grass-tree competition. Bertram and Dewar (2013) used statistics to analyze tree-grass existence. Using maximum entropy techniques, they modelled tree-grass ratios in savannas based on assumptions about the evapotranspiration requirements of the different components of savannas. For mesic savannas, their results do not agree with the disturbance-based hypothesis, in which disturbances such as fire keep trees and grass in coexistence. This is because their results suggest that in areas of high rainfall, random processes alone, such as tree dispersal and migration, were sufficient to predict savanna existence, and disturbances are not necessary for keeping grass-tree codominance.

## **Tree Mortality and Seedling Survival**

Understanding mechanisms of fire provides information for tree mortality and tree-grass coexistence. Fire can limit seedling recruitment and prevent the transition of juvenile trees to the canopy (Bond 2008, 648). It was originally thought that trees must either grow tall, raising apical or stem tip buds out of the flames, or wide, providing a buffer around the center of the tree in order to survive a fire (Lawes et al. 2011, 2). Lawes et al. (2011) measured the stem diameter of varying trees to define the role of bark thickness. They found that although some trees with a bark thickness of 8-9 mm ensured avoidance of topkill, other eucalypt trees were able to protect themselves from fires by growing tall enough to get out of topkill and protect their buds (Lawes et al. 2011, 5-6). This “safe zone” is known as escape height, because the trees are able to avoid topkill by growing tall; this height is usually between 2-4 meters but varies by tree species and environment (Bond 2008; Higgins, Bond and Trollope 2000). Once escape height is reached, a tree will not be reset to the base after a fire and can survive fire for many years. Tree saplings may be trapped in the flame zone because their stems are repeatedly burned by fires or browsed by animals, so they cannot reach escape height. These trees are known as “gullivers,” (Bond and van Wilgen 1996, 160). The time it takes for a tree to reach escape height varies depending on disturbances and other biotic variables, but Gignoux et al. (2009) estimated that this process may take at least ten years.

Seedling establishment may also be suppressed by fires and grazers (Bond 2008; Midgley, Lawes, and Chamillé-Jammes 2010). Browsers and grazers may consume seedlings on the ground or trample them, which negatively affects their establishment rate, and fires can also prevent seeds from surviving after a fire (Midgley, Lawes and Chamillé-Jammes 2010, 4). Others found that seedlings may acquire the ability to resprout within the first growing season in

frequently burned areas (Midgley and Bond 2001). This observation would suggest that areas that go through a frequent burn regime have less seedling establishment than other places. However, seedlings may be killed by fire in arid savannas because they are less tolerant of burning (Bond 2008 648). The timing of a fire may affect the rate at which young trees grow as well. Research has suggested that more juvenile trees grow faster and surpass the gulliver stage when fires are burned early (Laris and Dembele 2012, 19). More research needs to be done to determine how much of an effect grazing has on reducing tree recruitment.

While fire and grazing prevent seedling establishment, grass removal techniques have been shown to increase the chance of seedling survival, most likely by removing competition in the upper soil layers (Riginos 2009; Laris and Dembele 2012). When grass is removed, even low rates of seedling establishment have been shown to produce tree recruitment in grasslands over a large period of time (Bond 2008, 648). However, less is known about sapling survival and growth rate in competition with grasses. Once recruitment is established, adult trees are able to persist for decades or centuries, but eventually die because they are not entirely immune to fires or other large disturbances (Bond 2008, 650). Due to the critical importance of fire in savanna environments, a number of long-term burning experiments have been established to quantify the effects of various factors on tree growth and survival (Lawes et al. 2011; Riginos 2009). In addition, other research has attempted to understand and map anthropogenic burning practices in savannas (Laris and Dembele 2012).

### **Burning Experiments**

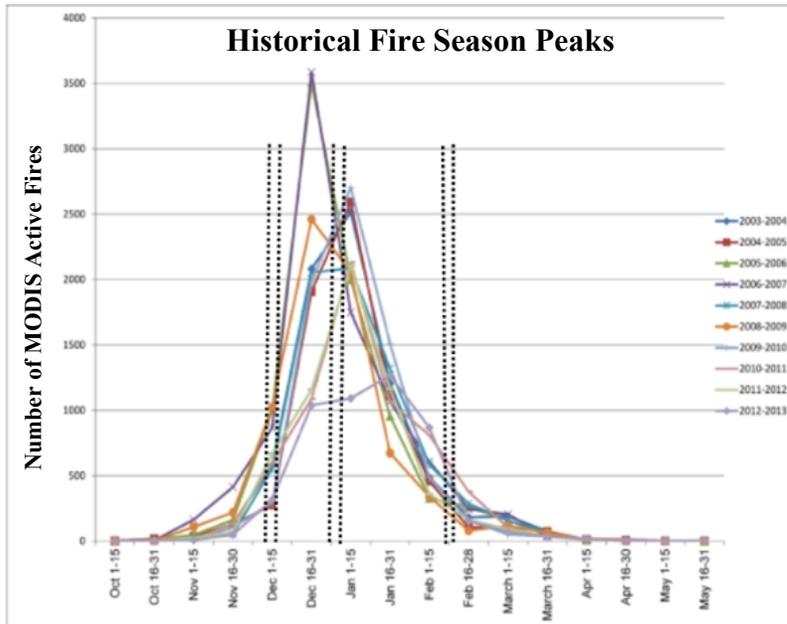
Fires can be a selective process that can restructure habitats or cause a species to dominate over another (House et al. 2003). Fire research in the savannas attempts to document the human practices, environmental factors, landscape effects, and causal factors of the

anthropogenic fires on the landscape. Fires in the savanna are often a function of the burning practices of local inhabitants, who set fires with varying frequencies depending on location and weather patterns (Laris 2002, Mistry 2000). Variation in fire intensity during peak fire season can be attributed to variations in grass standing biomass and moisture content, which vary with species and season as well as with land use practices (Higgins, Bond, and Trollope 2000, 215). Because grass standing crop, or living grass, and moisture content decrease as the dry season progresses, fire intensity is, in part, dependent on seasonality and is often most intense during the late dry season when grasses are the least moist. This fact forms the basis for a fundamental principle in savanna fire ecological models—the fire regime is a key determinant of vegetation cover in a savanna, where late dry season fires burn more intensively and are more damaging to trees than early fires (Laris 2016; Aubréville 1953).

That fire timing (or seasonality) is a critical factor determining tree cover in a savanna was first popularized by the well-known French scientist, André Aubréville, who had a large influence on savanna fire research. To study the impact of fire timing on savanna trees, Aubréville established a methodological design whereby fires were set early and late to grasses in a savanna landscape. Aubréville's dichotomous experimental design was so influential that it was repeated in nearly all savanna environments including African, Australian, and Latin American areas. This practice continues today (Furley et al. 2008; Prior et al. 2010, Russel-Smith et al. 2013).

A key weakness of Aubréville's approach was that the dates he chose for the categorization of early fires and late fires do not reflect the burning practices of West African inhabitants, but rather the extreme ends of fire seasons (Laris et al., 2017). Recent work shows that most of the fires occur in mid-dry season, with researchers recommending an expansion of

the early/late fire dichotomy to include a middle season of West African fires in order to include the bulk of fires. As Laris et al (2017) argue, Aubréville’s experiments cannot tell us the precise impacts of fires on the savannas because most fires occur in the middle season and are thus not covered by the experimental data (figure 1).



**Figure 1. Number of active fires for the study area in south-west Mali detected by MODIS smoothed over one-week intervals for multiple years (Laris et al. 2015). Note the high degree of regularity and the peak in late December/early January. Dashed lines show approximate dates of our experimental fires.**

### Importance of the Study

Timing and usage of prescribed savanna fires have been under considerable debate in recent years due to public attention to GHG emissions, carbon sequestration, and deforestation. Lipsett-Moore, Wolff, and Game (2019) propose that burning more often in the early dry season more often may lead to fewer GHG emissions as fires are smaller and less intense, and seventeen African Countries, including Mali, may reduce GHG emissions by as much as 37% (4). One flaw in this proposal is Lipsett-Moore et al. employ an oversimplified two-season fire model failing to

account for the fact that the bulk of fires are set in “mid-season” (Laris 2020). In addition, their model does not account for the fact that many grasses are not dry enough to burn in the early months (Laris 2020). Finally, the proposed changes in burning may affect the ratio of trees to grasses, shifting to a more wooded biome and thereby affecting local farmers and herders in the area who prefer grasses, such as perennials, for livestock fodder and other uses.

Along with proposals to shift burning to the early dry season, efforts are currently being made to reforest landscapes in countries across the globe, with the goal to sequester more carbon from GHG emissions. (<https://afr100.org/>, <https://www.bonnchallenge.org/>, <https://redd.unfccc.int/>). Carbon sequestration is the process in which carbon is captured in long-term storage systems such as plants, soils, oceans, etc. (Lal 2008). These projects, particularly The Bonn Challenge and AFR100, typically consist of restoring previously forested landscapes by 2030. Mali has specifically pledged to forest 10 million hectares of land by 2030 in AFR100; however, the maps erroneously assume that low tree cover areas in climates that can support forests are actually deforested and degraded (Bond et al. 2019, 963). The grassy landscape supports a valuable ecosystem, which, as noted, is used by local farmers and herders to support livelihoods. Typical farming in West Africa consists of livestock corridors, which are delineated pathways between pastoral resources, such as water points and grazing areas (Brottem 2014, 639). Because of variability in rainfall throughout the year, livestock herders practice transhumance herding, defined as seasonal north-south movements between rainy and dry season grazing areas (Brottem, 2016, 55). Removing grasses may increase farmer-herder conflicts that have already risen due to resource struggles caused by changes in climate (Raleigh 2010; Hendrix 2012). It is also unknown whether afforestation will work to cool the climate (Bond et al. 2019, Smith and Torn 2013). Some researchers argue that grassy biomes may be better than

forests at conserving carbon because grasslands can sequester high amounts of carbon belowground (Bond et al. 2019). For all of these reasons, it is currently unknown whether shifting the ratio of savannas to forests would be feasible or even preferable in the regions studied.

This thesis aims to shed further light on the factors that affect tree growth and establishment in mesic savannas, by studying the impacts of a variety of grass manipulations on trees of different ages. It also examines how fire regimes for three different burning dates, including the one most commonly employed by local populations, affect tree growth and survival.

## **CHAPTER 3**

### **METHODS**

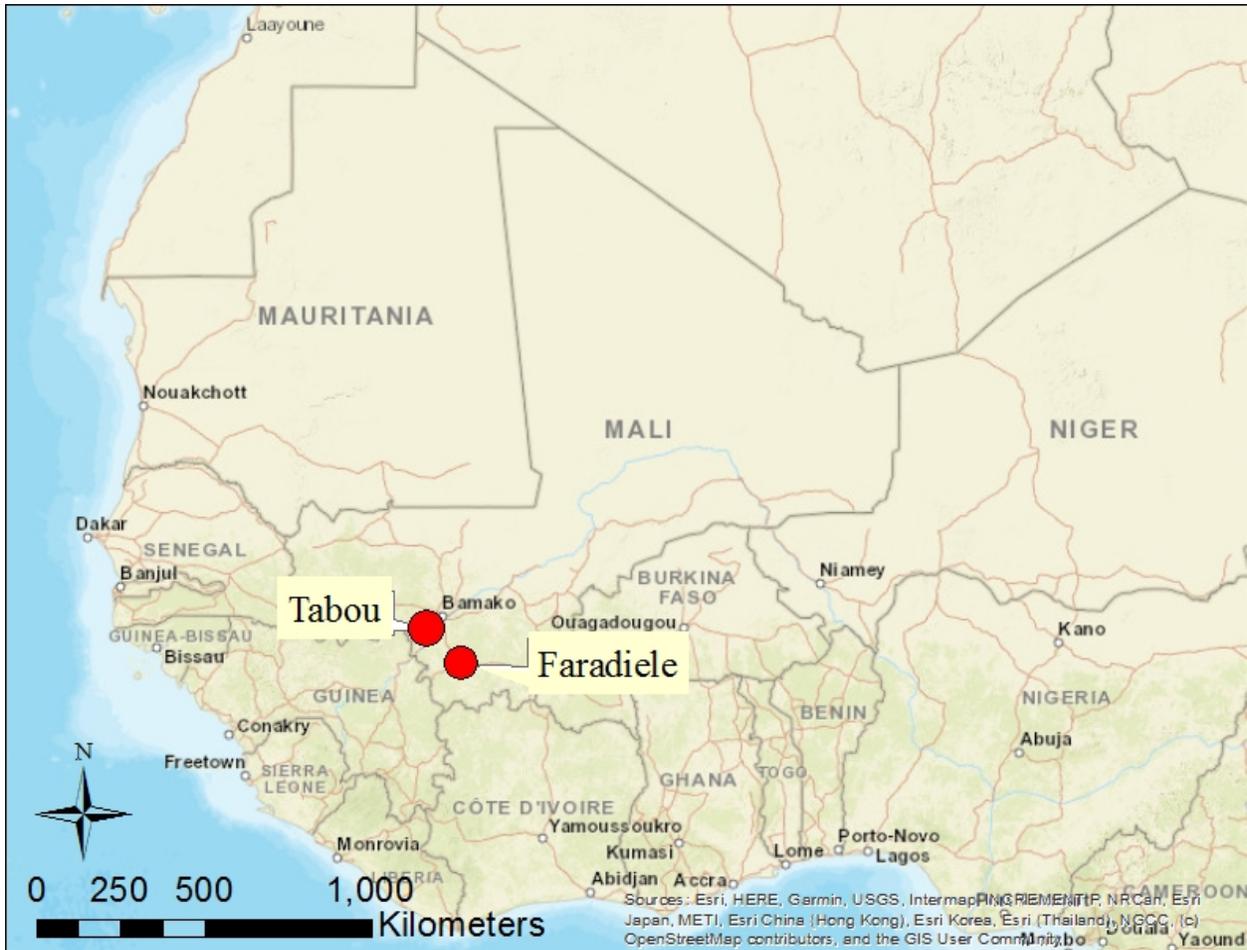
#### **Study Area**

This study took place in the southwestern region of Mali, West Africa. The region, which falls within the broad belt of southern Sudanian savanna (White 1983), is defined by wooded-savanna vegetation comprising a complex assortment of trees and grasses (Laris 2008). Short grasses tend to dominate areas of hard-pan and gravelly soils, while more fertile soils are covered in tall perennial grasses with a wide variety of trees and shrubs. Thickly forested savanna woodlands can form in very moist areas and there are closed canopy riparian forests along stream channels (Laris et al. 2011; Duvall 2011).

The climate can generally be divided into three seasons: a cool, dry season from October to February; a hot and dry season from February to June; and a warm and rainy period from June to October. The average annual precipitation varies from about 900 mm in Tabou in the north to approximately 1200 mm in the south (Henry 2011), categorizing the region as a mesic savanna.

Two study sites were established in the uncultivated areas of the villages known as Tabou and Faradiele. The people of Tabou and Faradiele are mostly smallholder farmers of the Bambara and Malinke ethnic groups (Laris, Foltz, and Voorhees 2015). Both savanna sites are considered mesic: Faradiele receives about 1200 mm of precipitation per year whereas Tabou receives about 1000 mm per year (Laris, Foltz, and Voorhees 2015). Significant amounts of land in both villages remain fallow annually due to the shifting agriculture patterns. Farming practices in southern Mali traditionally rotate land use patterns at multiple spatial and temporal scales, resulting in a patchwork of utilized, fallow, and unfarmed plots (Laris and Dembele 2012). Recently fallowed lands tend to have the most grazing as well as the earliest burns in the fire

season. About 43 percent of the land in Tabou is cropped annually while only 33 percent is cropped annually in Faradieie (Laris, Foltz, and Voorhees 2015).



**Figure 2. Locations of experimental burning areas. The study sites, Tabou and Faradieie, are marked as red circles**

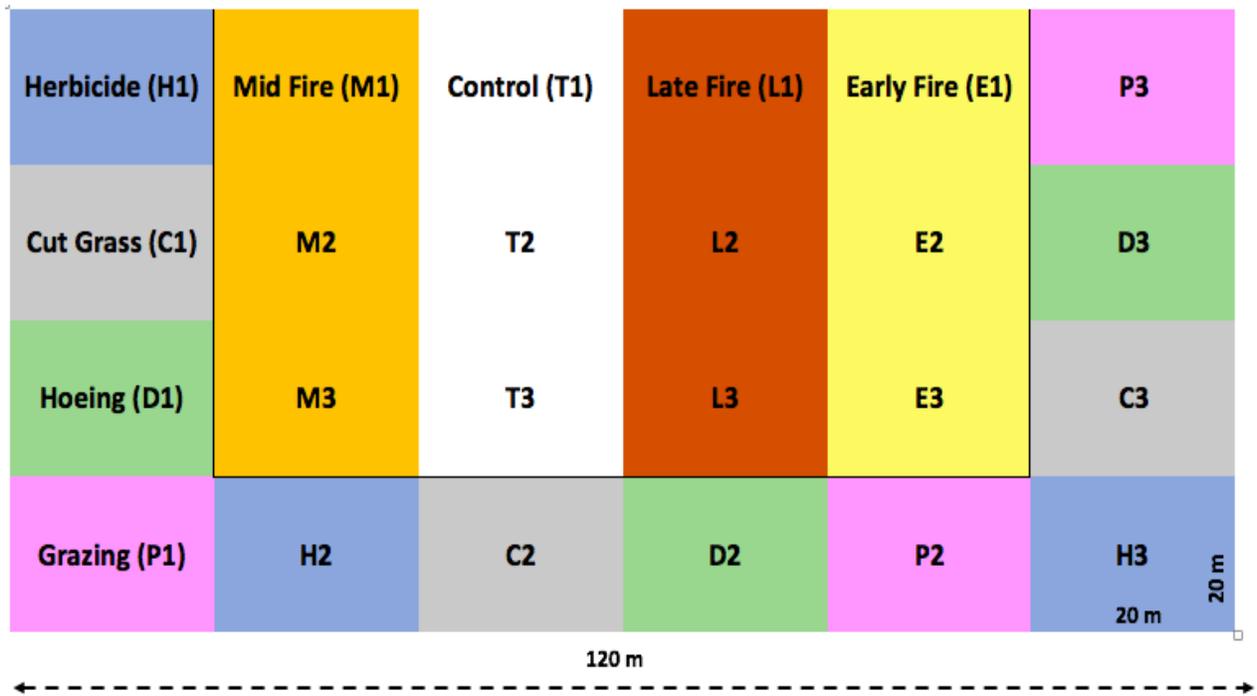
Both villages contain a variety of African trees native to the Sudanian and Guinean savannas (Arbonnier 2004). The main trees used for this study in the Tabou site belong to the *Terminalia* tree group; these include *Terminalia avicennioides* Guill. & Perr., *Terminalia macroptera* Guill. & Perr., and *Terminalia laxiflora* Engl. These trees all grow approximately 7-12 meters high with short boles, and flower during the second half of the dry season (Arbonnier 2004, 271-276). In Faradieie, the main tree species accounted for in this study include *Senegalia*

*dudgeonii* (Craib.) Kyal. & Boatwr., *Annona senegalensis* Pers., *Detarium microcarpum* Guill. & Perr., *Terminalia engleri* Gere & Boatwr.), *Terminalia macroptera* Guill. & Perr all contain straight boles and grow 6-10 meters high (Arbonnier 2004). Unlike the other trees, *Annona senegalensis* is a shrub that grows to only about 4 meters high with an open crown and grows a small fleshy berry commonly known as the wild custard apple (Arbonnier 2004, 155).

The fire season begins soon after the rainy season ends and generally runs from November to March with the bulk of the burning occurring in December and January (Figure 1) (Laris and Dembele 2012). As sufficient rain falls to support a closed canopy forest for these mesic savannas (Staver, Archibald, and Levin 2011), regular disturbances such as grazing, farming, or prescription burning maintain the landscape as a savanna (Laris et al. 2017).

### **Data Collection**

The data for this study were collected as part of a field campaign conducted by Drs. Paul Laris and Fadiala Dembele and a team of Malian students as part of a long term research project. Identical field experiments were established at each savanna site. Eighty trees of each size class were located at each study area and used to measure tree growth throughout each year of the experiment. Each site was divided into multiple plots to test fire and grass removal treatments. Experimental plots, measuring 20 meters by 20 meters each, were established on unfarmed plots that contain mostly perennial grasses. Each plot received a different grass or fire treatment except for the three control plots at each site. No plots received both a grass treatment and fire treatment. This method of isolating plots of field in the study area allowed for the comparison of different effects of grass and fire treatments on the trees located on experimental plots, and to analyze how the treatment plots differ from the control plots.



**Figure 3. Diagram of experimental plots (the dark line represents a fenced area)**

Grass treatment types consisted of grass clipping, hoeing, herbicide, and grazing. Hoe-farming was designed to eliminate grasses by replicating the traditional cultivation technique (which also breaks the soil crust) using a small hand-held hoe. To remove grasses for the different treatments, hoe farming was done once at the beginning of the rainy season for each year. Since herbicide is commonly used by local farmers in the region during the farming season, glyphosate, an herbicide purchased locally, was applied once when grasses had sprouted. On alternating plots, smaller trees were wrapped in plastic during the herbicide spraying to prevent the herbicide from affecting all juvenile trees. Grasses in clipping plots were clipped when they reached a height of 20 centimeters. Grazing by animals was also restricted on control and burned sites using a chain-link fence.

Fire treatments were applied according to the long-term fire regime for the region. The timing of the fire treatment was categorized into early, middle, or late-season fires, an approach proposed by Laris (2017) for savanna fire research. Early-season fires took place in November

and December, middle season fires were set in early January, and late-season fires were set in mid-February when grasses were dry. All fires were ignited into the direction of the wind, known as back fires, forcing them to move into the wind. Fire breaks were cut to prevent control sites from burning. Each fire and grass treatment type ensured 30 trees from each study site, amounting to 420 experimental trees. Combined with the 60 control trees, there were a total of 480 trees used within the experiment.

Following Riginos (2009), tree growth was measured in two ways: change in height and change in stem diameter. Change in stem diameter was measured at 5 cm above ground level. Trees were tagged, measured, and treated in June of each year of the experiment prior to the growing season. The tree names were recorded in both local and scientific nomenclature.

To determine how different disturbances impact tree growth rates for different life-cycle phases, trees were divided into three tree classes: (1) small, young juveniles ( $\leq 2$  m); (2) large juveniles (2-3 m); and (3) tall, adult trees ( $\geq 3$  m). The tree class sizes were selected based on their height with respect to the flame zone; the small and large juveniles are within the flame height and adult trees are above the flame height. The tree size class was determined by the height of the tree during the start of the experiment in 2016. Throughout the experiment, some trees died due to disturbances and other natural causes not controlled by the study, such as termite infestation. Dead trees were accounted for each year and excluded from the statistical analysis portion of the study on tree growth.

### **Statistical Analysis**

The data were organized into two Microsoft Excel spreadsheets: one for the grass-disturbance data and the other for the fire-disturbance data. These data sets were not combined

for any statistical inquiry. All statistical analyses were performed and processed through IBM Statistical Package for Social Science 24 software (SPSS).

The total growth of each tree was calculated by taking the height of the tree the last year of the experiment (2018) and subtracting it from the height of the tree during the start of the experiment (2016). In order to compare how grass and fire treatment affect tree growth and height, growth data for each of the metrics were analyzed using a one-way analysis of variance (ANOVA) with treatment type as a fixed factor. Grass treatment and fire treatment were not used within the same one-way analysis and were instead split into two ANOVAs, one analysis using tree growth as the measurement variable with grass treatments as the nominal variable, and the other using tree growth as the measurement variable with fire treatment as the nominal variable. The analysis will test the treatments together on whether the means of the measurement variable are the same among different groups (grass or fire treatment). This is done by calculating the mean observation value within each group and then comparing the variance among these means to the average variance within each group (McDonald 2014).

For the one-way ANOVAs, the statistical null hypothesis for this study states that the mean of the tree growth rate is the same for each fire or grass treatment group. An alpha value, or significance level, of 0.1 was used rather than the standard  $\alpha=0.05$  because of the exploratory nature of this test in order not to miss a potentially important factor. The use of a lower alpha value, however, increases the risk of a “Type II” error, in which we fail to reject an invalid null hypothesis or a “false negative.” This error would be more detrimental in this study than a “Type I” error, which is when we reject a true null hypothesis or a “false positive” (McDonald 2014 19).

The one-way ANOVA test also assumes that the observations within each sample are normally distributed and are homoscedastic (McDonald 2014). Homoscedasticity was important to maintain as the sample sizes between the groups were not even after accounting for tree deaths. In order to follow this assumption, each sample was transformed using a base-10 log operation in order to meet normality. After the data had undergone the base-10 log transformation, the samples were analyzed through a Shapiro-Wilks test to ensure that the data are normally distributed. Histograms of each group were also created to ensure a bell-shaped curve, which is a distinguishing feature of normal distributions (McDonald 2014)

To analyze how the various grass or fire treatment methods differ in tree growth rate, a Tukey-Kramer test was utilized after the one-way ANOVA to compare different pairs of means in order to see which means were significantly different from one another. This differs from the one-way ANOVA in that the test does not analyze the means as an ensemble but rather compares the mean values in groups of two (McDonald 2014), such as examining how the grass-clipping mean differs from the herbicide mean, or how the early-season fire treatment mean differs from the control mean. Each treatment was compared against one another and the control group for a total of twelve fire treatment pairwise comparisons and twenty grass treatment pairwise tests.

Removal of the dead trees, following Riginos (2009), led to a decrease in observations used in the statistical analysis, as dead trees were not accounted for in tree growth rate, the measurement variable used in the one-way ANOVA tests. However, dead trees were analyzed in conjunction with the tree growth rates of trees subjected to fire disturbances to determine whether there is increased tree dieback from burning. The latter was done by performing a chi-squared test of independence to see whether the counts of the dead/alive tree groups were different for the different values of the treatment group (McDonald 2014). The number of dead

trees were also cross-compared with the amount of tree deaths with the average number of tree growth per group.

## **CHAPTER 4**

### **RESULTS**

#### **Tree Growth Analysis**

At the completion of the study period, analyses of tree growth and tree death counts were conducted on each plot type. Of the 240 grass-treatment trees, 79 trees (32.9%) grew at least one meter taller than their initial height during the course of the study, with 46 of them being large and small juvenile trees. Of those 45 trees, 14 were in the grass-clipping plots, 13 were in the herbicide plots, 15 were in the hoe-farming plots, and 3 were in the grazing plots. Fifty small and large juveniles were able to reach escape height (3 meters) in the grass removal plots. Nineteen came from grass-clipping, 11 came from herbicide, 15 came from hoe-farming, and 5 came from grazing.

**Table 1. Growth of Juvenile Trees on Plots with Grass Treatment**

	<b>Total Juveniles</b>			<b>218</b>
	# Juveniles deaths			47
<b>Total</b>	# of juveniles who grew $\geq$ 1 m	48	# Attained escape height	37
			# Did not attain escape height	11
	# Juveniles who grew < 1 m	123	# Attained escape height	16
			# Did not attain escape height	107
	<b>Total Clipping Juveniles</b>			<b>49</b>
	# Juveniles deaths			3
<b>Clipping</b>	# of Juveniles who grew $\geq$ 1 m	14	# Attained escape height	13
			# Did not attain escape height	1
	# Juveniles who grew < 1 m	32	# Attained escape height	6
			# Did not attain escape height	26
	<b>Total Herbicide Juveniles</b>			<b>37</b>
	# Juveniles deaths			3
<b>Herbicide</b>	# of Juveniles who grew $\geq$ 1 m	13	# Attained escape height	7
			# Did not attain escape height	6
	# Juveniles who grew < 1 m	21	# Attained escape height	4
			# Did not attain escape height	17
	<b>Total Hoeing Juveniles</b>			<b>50</b>
	# Juveniles deaths			11
<b>Hoeing</b>	# of Juveniles who grew $\geq$ 1 m	15	# Attained escape height	13
			# Did not attain escape height	2
	# Juveniles who grew < 1 m	24	# Attained escape height	2
			# Did not attain escape height	22
	<b>Total Grazing Juveniles</b>			<b>40</b>
	# Juveniles deaths			23
<b>Grazing</b>	# of Juveniles who grew $\geq$ 1 m	3	# Attained escape height	2
			# Did not attain escape height	1
	# Juveniles who grew < 1 m	14	# Attained escape height	3
			# Did not attain escape height	11
	<b>Total Control Juveniles</b>			<b>42</b>
	# Juveniles deaths			7
<b>Control</b>	# of juveniles who grew $\geq$ 1 m	3	# Attained escape height	2
			# Did not attain escape height	1
	# Juveniles who grew < 1 m	32	# Attained escape height	1
			# Did not attain escape height	31

Forty-three (23.9%) of the 180 trees in the fire-treatment plots grew over a meter taller than their initial height. Only 26 (14.4%) of these were in the large or small juvenile trees size

class. Twelve trees came from the early-season plots, 9 from the middle season plots, and 5 from the late-season plots. Thirty-two small and large juveniles reached escape height, with 14 of them coming from the early-season plots, 11 from the middle season plots, and 6 from the late-season plots.

**Table 2. Growth of Juvenile Trees on Plots with Fire Treatment**

<b>Total Juveniles</b>		<b>173</b>		
# Juveniles deaths		44		
<b>Total</b>	# of Juveniles who grew $\geq$ 1 m	29	# Attained escape height	22
			# Did not attain escape height	7
	# Juveniles who grew < 1 m	100	# Attained escape height	13
			# Did not attain escape height	87
<b>Total Early Juveniles</b>		<b>41</b>		
# Juveniles deaths		3		
<b>Early</b>	# of Juveniles who grew $\geq$ 1 m	12	# Attained escape height	9
			# Did not attain escape height	3
	# Juveniles who grew < 1 m	26	# Attained escape height	5
			# Did not attain escape height	21
<b>Total Middle Juveniles</b>		<b>50</b>		
# Juveniles deaths		11		
<b>Middle</b>	# of Juveniles who grew $\geq$ 1 m	9	# Attained escape height	6
			# Did not attain escape height	3
	# Juveniles who grew < 1 m	30	# Attained escape height	5
			# Did not attain escape height	25
<b>Total Late Juveniles</b>		<b>40</b>		
# Juveniles deaths		23		
<b>Late</b>	# of Juveniles who grew $\geq$ 1 m	5	# Attained escape height	5
			# Did not attain escape height	0
	# Juveniles who grew < 1 m	12	# Attained escape height	1
			# Did not attain escape height	11
<b>Total Control Juveniles</b>		<b>42</b>		
# Juveniles deaths		7		
<b>Control</b>	# of juveniles who grew $\geq$ 1 m	3	# Attained escape height	2
			# Did not attain escape height	1
	# Juveniles who grew < 1 m	32	# Attained escape height	1
			# Did not attain escape height	31

Dead trees were removed from one-way ANOVA portion of the study following Riginos (2009). They were, however, included for the chi-squared test of independence to analyze whether the dead tree relative frequencies differed between fire seasons. Unfortunately, an intense, accidental fire burned the plots of the control trees in the Tabou study site in year 2017, killing numerous trees and stunting the growth of others. To account for the accidental tree deaths by fire in the control plots, the number of dead trees in the control plot for Tabou in 2016 was doubled to estimate of the actual number of dead trees from the two-year study period. The growth rate for the Tabou control trees was also negatively affected, dropping to -0.37 meters for the 2017-2018 period. This was mitigated by substituting the growth rates for the 2016-2017 year period. While these mitigation techniques, while *ad hoc*, were validated by comparing the dead trees and growth rate values to the second study site, Faradieie. Faradieie had a total of 5 control trees die during the study period while the correction for the Tabou site amounted to be 2 dead trees. The total growth rate for Faradieie's control trees was 0.542 meters. The growth rate for Tabou control trees using the 2016-2017 growth period was 0.482 meters. We feel confident that the estimates for the death rate and growth rate in the control plots in Tabou are realistic. Indeed, the new values for the Tabou control trees were close to the values of the Faradieie control trees giving us further confidence in our method to account for the accident.

After removing the dead trees, the dataset for the grass removal disturbances included fifty-seven trees subjected to grass-clipping, forty-nine trees to hoe-farming, fifty-seven trees to herbicide, and thirty trees to grazing. The dataset for the fire disturbances totaled to fifty-six trees subjected to early-season burns, twenty-eight to middle season burns, and thirty-three to late-season burns. Of the control trees using the mitigated method, fifty-two trees were alive at the end of the experiment. The numbers of trees differ from group to group due to tree deaths, as

some disturbances (or other factors) caused more tree mortality on certain plots than others.

After applying a base-10 log transformation on the tree growth data, the standard deviations were homogeneous among the different groups, ranging from 0.27 from the grazing treatment to 0.34 from the herbicide treatment, thus implying homoscedasticity.

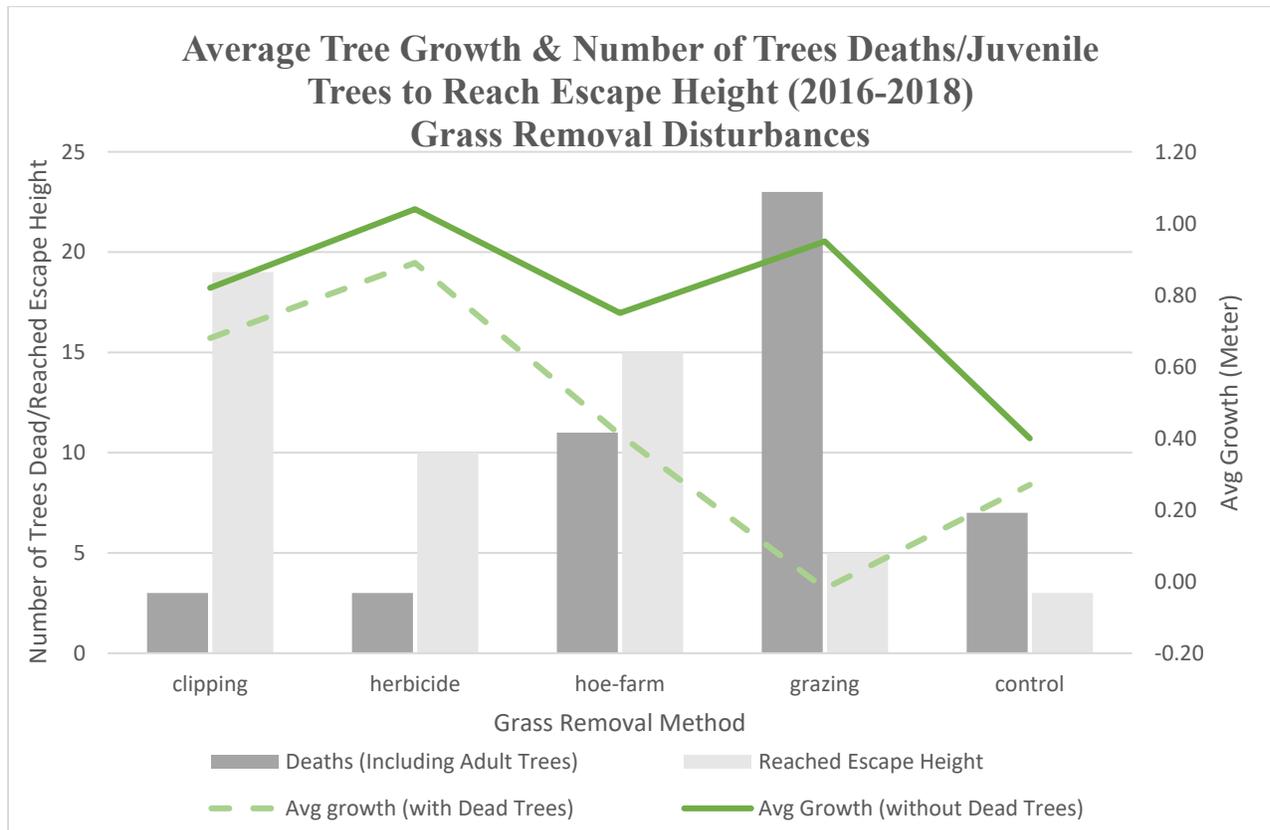
The Shapiro-Wilks test proved all groups except for herbicide and grazing treatment to be normally distributed with p-values above  $\alpha = 0.05$  level, rejecting the null hypothesis of a non-normal distribution, thus meeting the assumptions for a one-way ANOVA (Table 3). Despite failing the test for normality, it was decided to move forward with the one-way ANOVA for the herbicide and grazing treatments, as histograms for these groups did not appear to be severely non-normal, and an ANOVA is not very sensitive to moderate deviations from normality (McDonald 2014, 151). However, a higher false positive rate for these groups will be taken into consideration.

**Table 3. Shapiro Wilks Test for growth rate by all treatment variables. \* indicates the variables passed the test for normality at an alpha level of 0.05.**

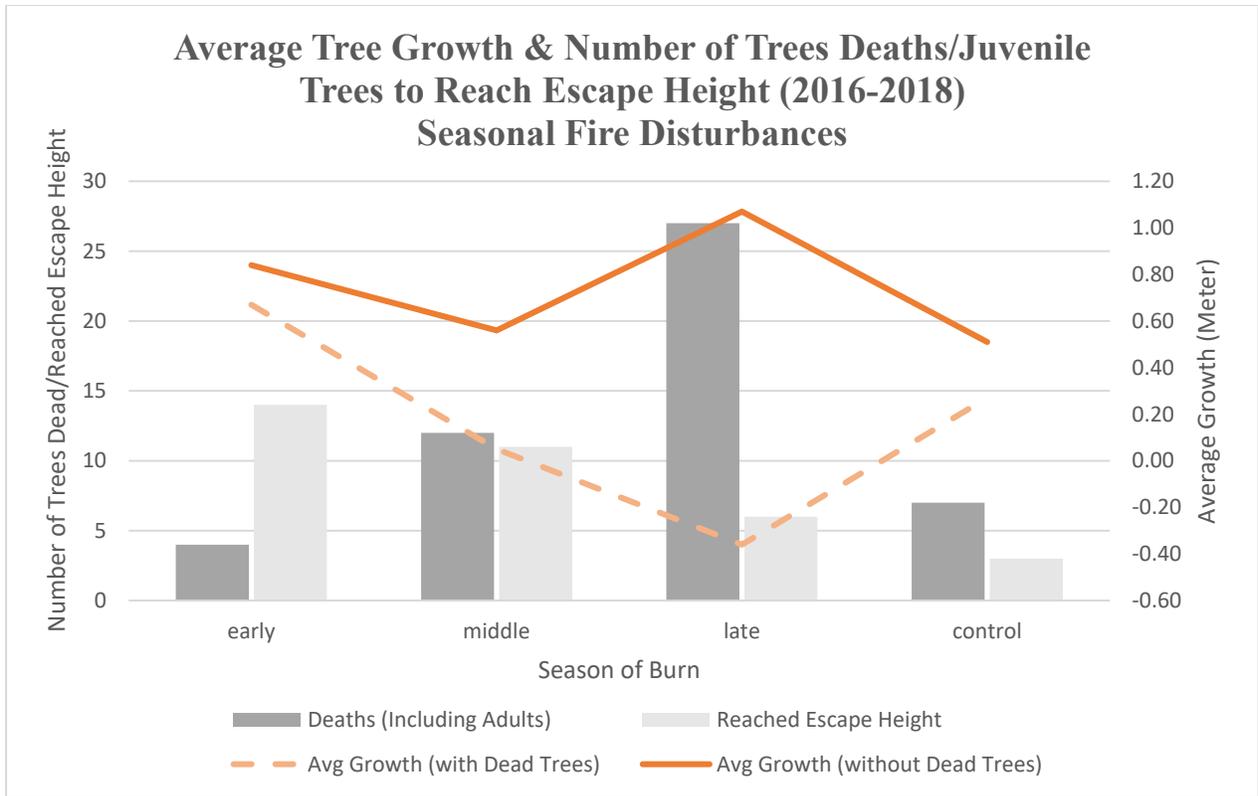
Shapiro-Wilk Test			
	Statistic	df	Sig.
Early	0.960	51	0.085*
Late	0.974	33	0.599*
Middle	0.957	41	0.127*
Clipping	0.986	56	0.757*
Hoe-farm	0.957	41	0.124*
Herbicide	0.940	55	0.008
Grazing	0.878	33	0.002
Control	0.957	51	0.065*

All grass removal trees grew almost twice as much as the controls after a two-year period (Figure 4). All fire disturbance trees also grew more than the control trees; however, early-season and late-season fire trees grew the most (Figure 5). The one-way ANOVA proved to be significant for both the fire treatments and grass treatments (fire,  $F_{3,155} = 7.087, p < 0.001$ ; grass,

$F_{4,214} = 7.247, p < 0.001$ ). From this, we reject the null hypotheses and conclude that there is a significant difference for at least one of the groups for both fire and grass removal disturbances.



**Figure 4. Average growth rate by grass disturbance and dead trees in each plot. Grazing seems to cause a large number of tree deaths due to tree trampling and browsing.**



**Figure 5. Average growth rate by fire disturbance next to number of dead trees in each plot.**

While all treatments proved to be significantly different from the ensemble as a whole, the pairwise comparison tests from the Tukey-Kramer analysis were not homogeneous. All of grass-removal treatments were significantly different from the control trees, but none of the grass-removal treatments were significantly different from each other, suggesting that any type of grass-removal disturbance will cause trees to grow taller. For the fire-treatment trees, only the late and early-season trees were significantly different from the control trees (late,  $p < 0.0001$ ; early,  $p = 0.001$ ). The middle-season trees were significantly different from the late-season trees ( $p = 0.051$ ), but they were not significantly different from the early-season trees ( $p = 0.173$ ) or control trees ( $p = 0.355$ ). The late-season trees grew on average 1.071 meters during the two-year period, taller than both the middle-season growth rate (0.563 m) and early-season growth rate (0.844 m). Based on these results, late-season fires cause trees to grow the most (although with the highest

mortality), followed by early-season fires (low mortality). However, middle season fires were similar to prescribing no burns at all (0.510 m) in terms of tree growth and death.

**Table 4. & Table 5. Tukey-Kramer results for grass disturbance and fire disturbance. Means with the same letter are not significantly different from each other.**

<b>Grass Disturbance</b>	<b>Mean Tree Growth (m)</b>	<b>Tukey-Kramer</b>
Clipping	0.822	a
Herbicide	1.044	a
Hoe-Farm	0.752	a
Grazing	0.947	a
Control	0.510	b

<b>Fire Season</b>	<b>Mean Tree Growth (m)</b>	<b>Tukey-Kramer</b>
Early	0.844	a,c
Middle	0.563	b,c
Late	1.071	a
Control	0.403	b

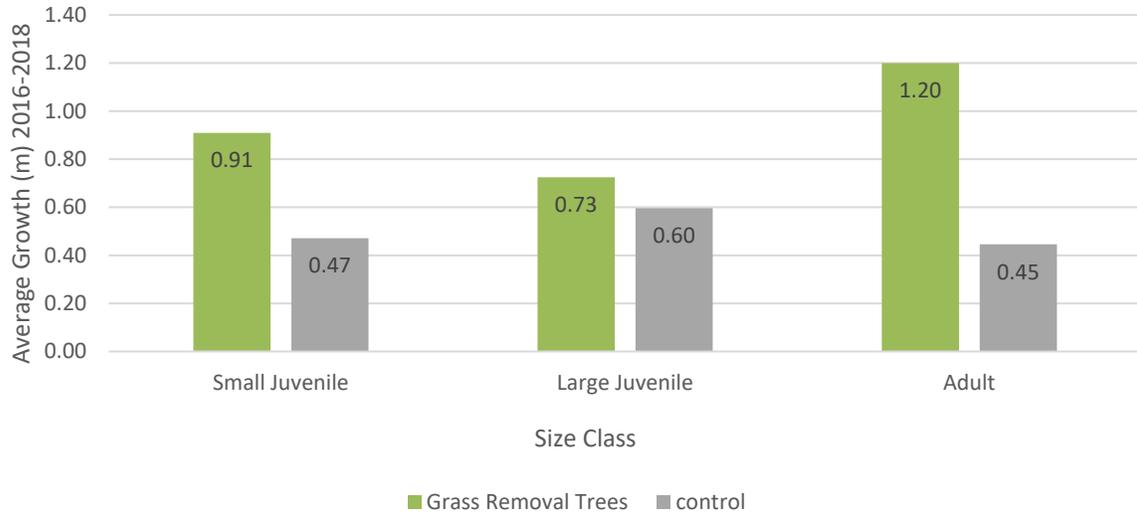
The Shapiro-Wilks test for the tree-size ANOVA determined the small-juvenile and adult trees to be normally distributed (adult,  $p = 0.097$ ; small-juvenile,  $p = 0.059$ ), but large-juvenile trees did not pass ( $p = 0.003$ ). However, the histogram showed only moderate deviations, so the one-way ANOVA was performed with consideration to higher false positives. The tree-size analysis was determined to be significantly different from tree growth rates with grass removal treatments ( $F_{2,182} = 5.532$ ,  $p = 0.005$ ). The adult trees grew significantly more than the small and large-juvenile trees ( $p = 0.002$  and  $p = 0.053$ , respectively), but the small and large juveniles were not significantly different from each other ( $p = 0.624$ ). Of the trees with grass removal treatments, the adults grew on average 0.7 centimeters more than the large-juveniles, and the large juveniles grew only 0.3 centimeters more than the small juveniles.

**Table 6. Shapiro Wilks Test for growth rate by tree stage of life. \* indicates the variables passed the test for normality at an alpha level of 0.05**

<b>Shapiro-Wilk Test</b>			
	Statistic	df	Sig.
Large Juvenile	0.957	99	0.003
Small Juvenile	0.967	70	0.059*
Adult	0.960	49	0.097*

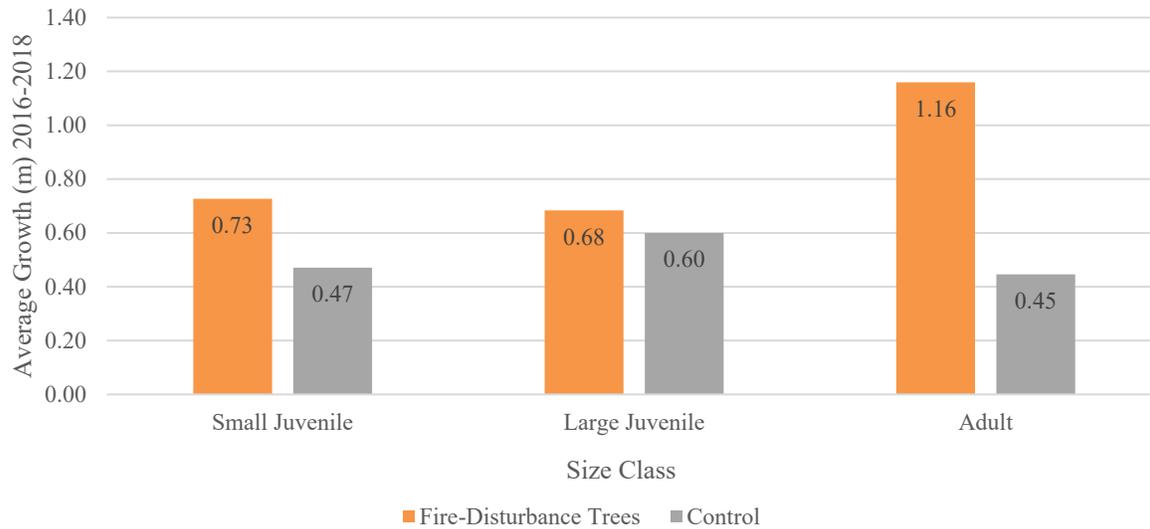
Compared to the control trees, each size class within a grass removal plot grew more than the control trees of the corresponding size class. The small juvenile trees grew 0.44 meters more than their control counterparts while the large juvenile trees grew only 0.13 meters more. The small juveniles growing more than the large juveniles with grass removal disturbances may suggest that small juveniles are under more grass competition than large juveniles, possibly due to rooting depths. The adult trees grew 0.75 meters taller than the adult control trees, more than both large and small juveniles. Trees that had undergone fire disturbances also increased in growth rate in a similar pattern as the grass removal trees. Small juvenile trees grew 0.26 meters more than non-burnt small juveniles while large juveniles only grew 0.08 meters more. Adult, burnt trees also grew more than both small and large juveniles, reaching 1.16 meters more than unburnt adults.

### Tree Growth by Size Class (Grass Removal Trees Only)



**Figure 6. Average growth rate (m) of grass disturbance trees by size class. All tree sizes grew more when grass competition was removed.**

### Tree Growth By Size Class (Fire-Disturbance Trees only)

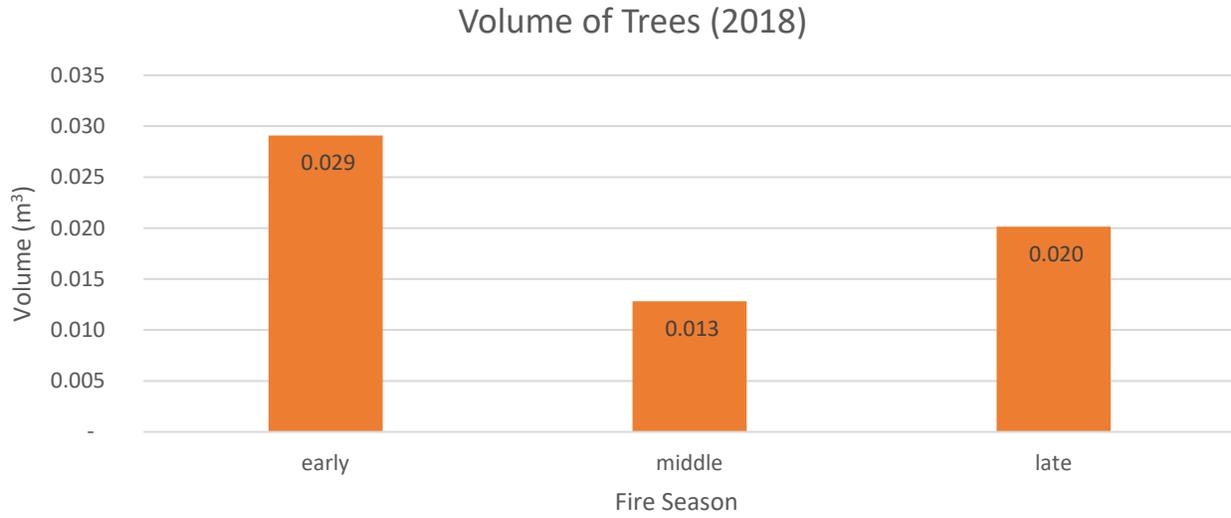


**Figure 7. Average growth rate (m) of fire disturbance trees by size class. All tree sizes grew more when burned.**

Although the late-season fire trees had the tallest growth out of the fire-treatment trees, they also included the most tree deaths (Figure 5), which resulted in a lower tree volume (Figure 8). Of the original 60 trees in each group, 5 trees died in the control group, 4 trees died in the early-season fires, 12 trees died in the middle season fires, and 27 trees died during the late-season fires. Based on the chi-squared test of independence with the corresponding p-value for this test statistic being  $p = 0.000$ , there is an association between the dead trees and tree size. Of the trees that died, 22 trees were classified as small juveniles, 19 were classified as large juveniles, and 7 were classified as adults, with 4 of the adults having died in the late-season fires. This shows that although late-season trees grew the most, out of the fire treatment trees, these trees were also the most likely to die, and late-season fires were the most likely to kill trees that have reached escape height.

**Table 7. Dead Tree Count for Fire Disturbance Plots**

		# Dead Trees
<b>Early</b>	Adult	1
	Large Juvenile	2
	Small Juveniles	1
	<b>Total</b>	<b>4</b>
<b>Middle</b>	Adult	1
	Large Juvenile	4
	Small Juveniles	7
	<b>Total</b>	<b>12</b>
<b>Late</b>	Adult	4
	Large Juvenile	10
	Small Juveniles	13
	<b>Total</b>	<b>27</b>
<b>Control</b>	Adult	1
	Large Juvenile	3
	Small Juveniles	1
	<b>Total</b>	<b>5</b>
<b>Total</b>	Adult	7
	Large Juvenile	19
	Small Juveniles	22
	<b>Total</b>	<b>48</b>



**Figure 8. Total volume (m<sup>3</sup>) of trees in fire season plots.**

## CHAPTER 5

### DISCUSSION AND CONCLUSION

#### **Grass and Fire Disturbances' Effect on Tree Growth**

The goal of this study was to document the effects of grass and fire disturbances on tree growth to better understand the relationship of tree-grass coexistence in a mesic savanna. Using tree growth rates from burning and grass removal experiments, the statistical analysis evaluated the differences in tree growth under various grass and fire conditions and explored the role of grass competition with juvenile trees.

This study finds that for the case of the mesic savannas studied, grass competition suppresses the growth of trees for multiple demographic stages. Trees with grass removal treatments grew more than the control group regardless of the treatment used. The results show that in areas where rainfall levels exceed 900 millimetres per year, trees in their early-life stages compete with grasses for water and that this competition significantly affects juvenile tree growth rates. This finding is critical in that it suggests that the dichotomous view of resource-based versus disturbance-based theories is oversimplified. Specifically, grass competition slows juvenile tree growth, thus reducing the likelihood of a tree reaching the disturbance escape height. As such, it is the combined effects of disturbance and competition that maintains the tree and grass ratio of in a mesic savanna. This thesis extends the applicability of the findings of Riginos (2009) (who worked in a more arid savanna where the model suggests that grass competition would limit tree establishment and growth) into the mesic zone.

Grass removal was shown to increase the growth rate (above the control level) of all trees regardless of method used with most of the juveniles reaching escape height in grass-clipping and hoe-farming plots. This striking finding suggests that a two-year treatment of grass-clipping

or hoeing can shift the balance of a mesic savanna from grasses to trees by providing a pathway for juvenile trees to escape the flame height. In addition, few trees (11 or less) died in most grass removal treatment plots, the exception being for grazing plots, where 23 trees died in total. It is probable that cattle cause additional tree stress by trampling or grazing on small trees while also compacting the soil and potentially causing an increased death rate. Indeed, of the trees that died in the grazing plots, 21 were small juveniles and only 2 were large juveniles, but none were adult sized.

In terms of fire and fire timing impacts, trees subjected to early and late-season fires grew twice as tall as the control trees, while middle season fires had little effect on tree growth. It is worth noting that the trees in the plots studied were regularly burned annually in the middle fire season prior to this study (Laris 2011). Not surprisingly, late-season fires resulted in the largest number of tree deaths, especially juveniles, limiting the overall number of trees as numerous other studies have shown (Laris 2008; N'Dri et al. 2018; Prior et al. 2006; Werner 2011; Werner and Prior 2013). There are several, well-documented reasons late fires cause an increase in tree death. These include higher fire intensity and higher flame heights due to lower fuel moisture, increased tree moisture stress as well as the fact that many savanna tree species push new leaves in the late-season leaving them depleted of reserves and vulnerable to death or die-back (West 1965; Murphy, Russel-Smith, and Prior 2010,).

This study also found that burnt trees demonstrated a higher growth rate than unburned ones, regardless of tree size or seasonal timing. However, the study found that the season with the theoretically most intense fires, late-season, did not reduce growth rate but rather killed more trees—especially small juveniles, reducing overall volume of trees (Figure 8) and reducing the number of juveniles that reached escape height (Table 2). One potential explanation for the faster

growth rates of trees burned in the late-season involves tree the life-history and fire disturbances. As all trees in this research were frequently burned prior to the study, they may have developed traits that encouraged them to tolerate the impacts of fires better (Higgins et al. 2012). It is possible that the “shift” in fire regime—from middle to early or late—signalled to trees there was an opportunistic moment to grow rapidly and escape the fire trap. Surviving juvenile trees of a late-season fire may have responded by an increased effort to grow tall quickly as opposed to withstanding a more moderate mid-season fire, for example. A second possibility is that the death of multiple small trees reduces the tree-to-tree competition for water and nutrients allowing remaining trees to grow more quickly. A life-history response to fires would also explain why adult trees were less prone to the impacts of the fires. The adult trees were the most unaffected by fire disturbances in terms of tree growth or tree death, as only one adult tree died in the middle season fires and two died in the late-season fires.

Early-season fires, which are normally the mildest in terms of fire severity and intensity led to a very high growth rate (0.844 m) and low death rate (4 trees). The explanation for the increased growth rate of trees subjected to early fires likely differs from that for late fires. Specifically, early-season fires burn grasses, killing them in the case of annuals and dramatically reducing the biomass, in the case of perennials. As such, small trees are able to capitalize on increased soil moisture and nutrient access in the early-dry season (Prior et al. 2006, Werner 2005). Surviving trees may also benefit from a “fertilization effect” causing trees to quickly recover from fires by rapidly releasing nutrients stored in a below ground carbon storage (Prior et al. 2006, Werner 2011).

While some other studies also found that early-season fires have a positive effect on tree growth rate (Werner 2011, Prior et al. 2006, Werner 2005) others found the opposite to be true.

For example, Murphy, Russell-Smith, and Prior (2010) found that mild or early-season fires had a clear negative effect on trees (p. 339) while Prior (2006) found that early fires had a much greater effect on the growth of juvenile trees (<1.5 meters), though not on saplings or adults. It should be noted that these studies were undertaken in Australian savannas where some of the dynamics—specifically the competition between *Eucalyptus* trees and grasses, are somewhat different (Bond, Cook, and Williams 2012).

All early-season fire trees in this thesis were at least 1.5 meters tall, except for one tree that was 1.25 meters; thus, it is possible that these trees had enough height to survive in the milder early-season fires. Additionally, all fires prescribed in this experiment were backfires, which are known to burn with lower intensity (Laris et al 2020).

The results of the study clearly demonstrated that all tree size classes were able to grow faster in plots with grass eliminated than in plots with grasses. Werner (2011) found similar results; trees that were greater than 3 meters tall with a non-grassy understorey grew in height after a late-dry season fire with only 20% of those trees suffering die-back to a smaller class size. Although all tree size classes were able to grow more with reduced grass competition, small juveniles grew more than large juveniles as compared to trees of their size class with grass competition (figure 7). This finding suggests that small juveniles are in more competition with grasses than large juveniles, which makes sense theoretically given the root development of juvenile trees.

### **Applications: The Savanna as a Means of Carbon Sequestration**

With grass removal and early fire disturbances both facilitating a more wooded landscape, it is important to consider whether or not a shift to a forest from a savanna is preferred by the local and/or scientific community. With increased annual mean temperatures predicted for

the upcoming decades due to global climate change, much public attention has been focused on how to mitigate the problem (Arguez et al. 2020).

Current research has looked at savannas as a means of carbon sequestration to mitigate excess carbon emittance (Bond et al. 2019; Laris 2020; Lipsett-Moore, Wolff, and Game 2018). Trees are natural carbon sinks and are often used to sequester carbon and offset emissions produced by greenhouse gas emitting facilities. Savannas are naturally carbon sinks due to the number of trees and grasses, but burning mesic savannas has shown to decrease the amount of carbon that savannas can store (Lipsett-Moore, Wolff, and Game 2018; Williams et al. 2004; Murphy, Russell-Smith, and Prior 2010). Murphy, Russell-Smith, and Prior (2010) found that mild fires may reduce net primary production of trees by  $0.19 \text{ t C ha}^{-1}\text{y}^{-1}$  while severe fires can reduce net primary production of trees up to  $1\text{-}2 \text{ t C ha}^{-1}\text{y}^{-1}$ . These results suggest that reducing burns, particularly late-season burns, in mesic savannas would not only cause a shift to a more forested biome, but it would also allow for more carbon to be sequestered in the area due to an increase in the number of trees permitted to grow.

Although the idea of sequestering more carbon would be appealing to most people, shifting the biome from savanna to forest in these areas may not be as appealing to the local farmers of the villages, and losing the savanna may even be detrimental to livestock and means of living. This is especially important because agriculture is viewed as a key role in reducing poverty in sub-Saharan Africa (Brottem 2014, 638). It is also unknown whether burning more fires in the early-season will actually sequester more carbon, as emissions factors have not been taken into consideration in these models.

Perennial grasses, like the ones found in the savanna of Faradiele and Tabou, provide the livestock with a significant source of protein, which is not found in annual grasses (Brottem,

2016, 554). As perennial grasses deplete more in the central zone of transhumant herding and droughts become more common, seminomadic livestock herders have become numerous in subhumid zones such as southern Mali, where Faradiele and Tabou are located (Brottem 2016, 549). Thus, losing the perennial grasses and turning the mesic savanna into a forest could cause detrimental damage to the transhumant herders, as there would be less grass for livestock when herders are travelling through the area.

The timing of transhumant herders arriving in the southern zone is also dictated by the fire burns of the season. This is because the herders want to arrive in time for the livestock to graze perennial grasses that resprout about 10-15 days after burning when soil moisture is sufficient, which is normally only in the early-dry season (Brottem 2016, 554). Altering the fire season to stimulate maximum tree growth may cause herders to stay for longer in the central zone, where they have the most farmer-herder conflicts due to depleting resources and crop damage.

Theories that support early burnings to reduce emissions also do not take into consideration emissions factors increasing due to high fuel moisture content. The emissions factor for methane increases by 50-400% in early-season fires, due to higher fuel moisture content, and decreases closer to the dry season as fuels cure (IPCC 2000; Hoffa et al. 1999; Korontzi, Justice, and Scholes 2003, Korontzi 2005; Laris 2020). Emission factors for methane and nitrous oxide are also highly dependent on vegetation type and fuel size class (Meyer et al. 2012) that these studies do not take into consideration.

## **Conclusions**

The observations in the present study on the relationship between trees and grasses in savannas are important for characterizing fire regimes and demonstrating the ways in which

specific grass treatments and fire disturbances can significantly change tree growth rates at specific stages in a tree life cycle in a mesic savanna. On average, tree growth rate increased by 0.381 m with grass removal disturbances. Herbicide, grazing, and clipping had the highest growth rate averages, but grazing also had the highest number of tree deaths (23 deaths) for the grass treatment plots. Thirty-eight percent of juveniles within the clipping plots were able to reach escape height, the highest percentage for any of the grass treatment groups. In conclusion, this study finds that grass removal treatments of all kinds result in increased annual tree growth and, thus, supports the notion that tree-grass competition has significant impacts on the chances of juvenile trees escaping the fire trap and reaching adulthood. The study found that that grass clipping had the greatest impact on juvenile tree growth followed by hoeing, herbicide and grazing, but that cattle grazing also had a significant and negative impact on tree survival especially for the smallest tree class.

This study found that early-season fires and grass removal techniques stimulate tree growth in a mesic savanna regardless of demographic stage of life. The average growth rate of trees under late-season increased by 0.668 m and 0.442 m for early-season fires. However, due to the large amount of tree deaths caused by the late-season fires, the total volume of trees for early-season fires was 45% larger than late-season. The total volume of trees for early-season fires increased by 45%, and the early-season fire trees also grew on average 0.442 m more than trees without fires prescribed. As expected, late-season fires were most detrimental to trees and contributed the most to tree death (27 trees). Interestingly, mid-season fires were most closely aligned with the control (no fire/no grass removal).

This study also finds that although disturbance by fire has clear impacts on tree growth and survival rates, the precise impacts on both tree and savanna growth rates depends upon the

timing of the fire. Importantly, given that most fires in this region occur in mid dry season, this study finds the mid-season fires to have distinct impacts on trees. Mid-season fires had little impact on tree growth rates, which did not differ significantly from the control, while early fires resulted in an 0.344 m increase and late fires a 0.561 m increase. In terms of tree survival rates, mid-season fire tree deaths were between that of early fire (93.3%) and late fire (55.0%)

One important finding is that a two-year grass treatment by clipping or hoeing results in a greater fraction of trees (38.8% - 32.0%, respectively) reaching the flame escape height. This suggests that a combination of grass treatment plus early burning would result in a significant increase in tree cover. According to savanna demographic models (Higgins, Bond, and Trollope 2000), once trees reach escape height, they usually mature and live long lives as fire is no longer a major factor. In addition, as trees mature, they shade out grasses thereby reducing the fuel load, further increasing tree survival chances.

Although some argue that savannas hold great potential as a carbon sink, others point out that savannas are not, in fact, degraded woodlands and it is not certain that shifting savannas to forests would effectively increase carbon sequestration (Bond et al. 2019). While this thesis finds that there are “treatments” that could effectively shift mesic savannas to a woodier trajectory, we caution that this is an inappropriate use of the findings. Any decision to take measures to dramatically shift the composition of a savanna landscape should be that of a local population. This is because trees and grasses are key sources of livelihood for both human societies and wild animals. Mesic savannas have already been seen as an unstable system that can easily be turned into a forest without the use of fire disturbances, and the use of grass removal techniques and early-season burns may only increase the speed of turning mesic savannas into forests. Although some may advocate switching these biomes to forests to increase carbon sequestration, doing so

may not be easy. For example, transhumant herders could lose a large amount of pasture for livestock if grasslands were converted to woodlands. Herders from the more arid interior, in Sahelian countries, such as Mali and Burkina Faso, already send their herds farther and farther south to seek forage in the dry season as pasturelands are converted to agriculture and disappear. as they move from grassy savannas to forested conditions, as may happen to Malian herders forced to move from northern to southern Mali as a result of afforestation. Efforts to shift the savanna biome to a forest biome will need to focus on local community issues that may arise from the reduction of perennial grasses. Decreasing resources would increase already elevated levels of herder-farmer conflicts. Policymakers and researchers interested in shifting the biome will need to focus on agro-pastoral resource governance and conflict mediation in order to make things work at the local community scale. Overall, investigating the relationship between trees and grasses will allow scientists, herders, farmers, and policymakers to understand better the consequences of action. The future of this research will need to look into the growth rates of different tree and grass species in mesic savannas of West Africa.

## **APPENDICES**

**APPENDIX A**  
**ANOVA DETAILS FOR FIRE AND GRASS DISTURBANCE**

## Shapiro Wilks for Fire Disturbances Case Processing Summary

Fire treatment		Valid		Cases Missing		Total	
		N	Percent	N	Percent	N	Percent
2016-2018	Early	51	85.0%	9	15.0%	60	100.0%
	Late	33	55.0%	27	45.0%	60	100.0%
	Middle	41	68.3%	19	31.7%	60	100.0%
	Control	51	85.0%	9	15.0%	60	100.0%

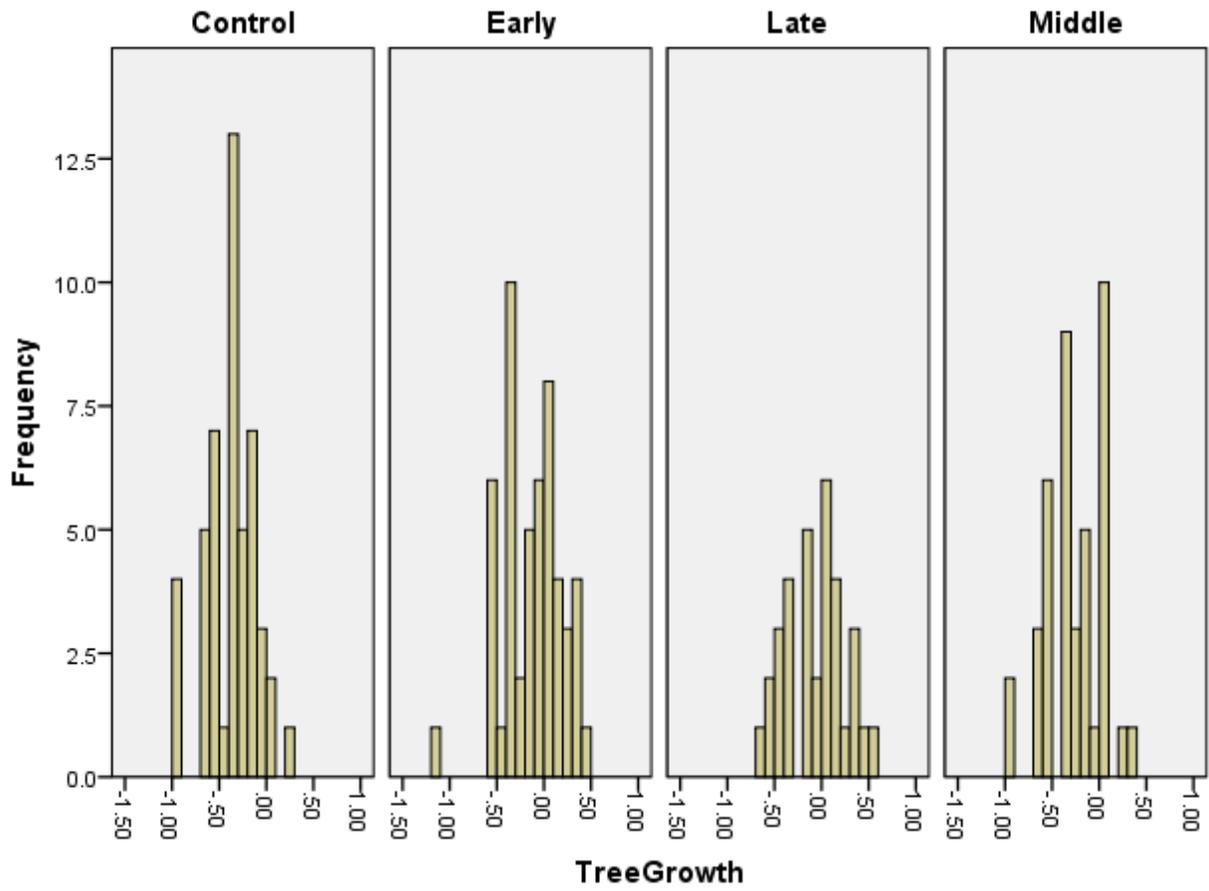
2016-2018	Early	Mean		-0.1274	0.04299
		95% Confidence Interval for Mean	Lower Bound	-0.2138	
			Upper Bound	-0.0410	
		5% Trimmed Mean		-0.1210	
		Median		-0.0969	
		Variance		0.094	
		Std. Deviation		0.30704	
		Minimum		-1.15	
		Maximum		0.43	
		Range		1.59	
		Interquartile Range		0.38	
		Skewness		-0.539	0.333
		Kurtosis		1.070	0.656
			Late	Mean	
95% Confidence Interval for Mean	Lower Bound			-0.1838	
	Upper Bound			0.0371	
5% Trimmed Mean				-0.0729	
Median				-0.0223	
Variance				0.097	
Std. Deviation				0.31149	
Minimum				-0.70	
Maximum				0.51	
Range				1.20	
Interquartile Range				0.52	
Skewness				-0.073	0.409
Kurtosis				-0.799	0.798
	Middle			Mean	

	95% Confidence Interval for Mean	Lower Bound	-0.3565		
		Upper Bound	-0.1662		
	5% Trimmed Mean		-0.2490		
	Median		-0.2218		
	Variance		0.091		
	Std. Deviation		0.30136		
	Minimum		-1.00		
	Maximum		0.30		
	Range		1.30		
	Interquartile Range		0.52		
	Skewness		-0.585	0.369	
	Kurtosis		0.157	0.724	
	Control	Mean		-0.3689	0.04516
		95% Confidence Interval for Mean	Lower Bound	-0.4596	
Upper Bound			-0.2782		
5% Trimmed Mean			-0.3738		
Median			-0.3979		
Variance			0.104		
Std. Deviation			0.32253		
Minimum			-1.00		
Maximum			0.40		
Range			1.40		
Interquartile Range			0.37		
Skewness			-0.072	0.333	
Kurtosis			0.380	0.656	

### Tests of Normality

Fire treatment		Shapiro-Wilk		
		Statistic	df	Sig.
2016-2018	Early	0.960	51	0.085
	Late	0.974	33	0.599
	Middle	0.957	41	0.127

### Tree Growth Histograms - Fire Disturbance



## Shapiro Wilk for Grass Disturbances Descriptives

Grass treatment			Statistic	Std. Error	
2016-2018	Clipping	Mean	-0.1655	0.03865	
		95% Confidence Interval for Mean	Lower Bound	-0.2430	
			Upper Bound	-0.0880	
		5% Trimmed Mean	-0.1607		
		Median	-0.1399		
		Variance	0.084		
		Std. Deviation	0.28926		
		Minimum	-1.00		
		Maximum	0.54		
		Range	1.54		
		Interquartile Range	0.40		
		Skewness	-0.283	0.319	
		Kurtosis	0.420	0.628	
		Hoe-Farming	Hoe-Farming	Mean	-0.1089
95% Confidence Interval for Mean	Lower Bound			-0.2162	
	Upper Bound			-0.0016	
5% Trimmed Mean	-0.0899				
Median	-0.0706				
Variance	0.116				
Std. Deviation	0.33989				
Minimum	-1.00				
Maximum	0.43				
Range	1.43				
Interquartile Range	0.42				
Skewness	-0.674			0.369	
Kurtosis	0.676			0.724	
Herbicide	Herbicide			Mean	-0.0712
		95% Confidence Interval for Mean	Lower Bound	-0.1633	
			Upper Bound	0.0209	
		5% Trimmed Mean	-0.0510		
		Median	0.0000		
		Variance	0.116		
		Std. Deviation	0.34068		

	Minimum		-1.00	
	Maximum		0.51	
	Range		1.51	
	Interquartile Range		0.48	
	Skewness		-0.891	0.322
	Kurtosis		0.693	0.634
Grazing	Mean		-0.0028	0.04640
	95% Confidence Interval for Mean	Lower Bound	-0.0974	
		Upper Bound	0.0917	
	5% Trimmed Mean		0.0129	
	Median		-0.0458	
	Variance		0.071	
	Std. Deviation		0.26653	
	Minimum		-1.00	
	Maximum		0.48	
	Range		1.48	
	Interquartile Range		0.32	
	Skewness		-1.325	0.409
	Kurtosis		5.061	0.798
Control	Mean		-0.3689	0.04516
	95% Confidence Interval for Mean	Lower Bound	-0.4596	
		Upper Bound	-0.2782	
	5% Trimmed Mean		-0.3738	
	Median		-0.3979	
	Variance		0.104	
	Std. Deviation		0.32253	
	Minimum		-1.00	
	Maximum		0.40	
	Range		1.40	
	Interquartile Range		0.37	
	Skewness		-0.072	0.333
	Kurtosis		0.380	0.656

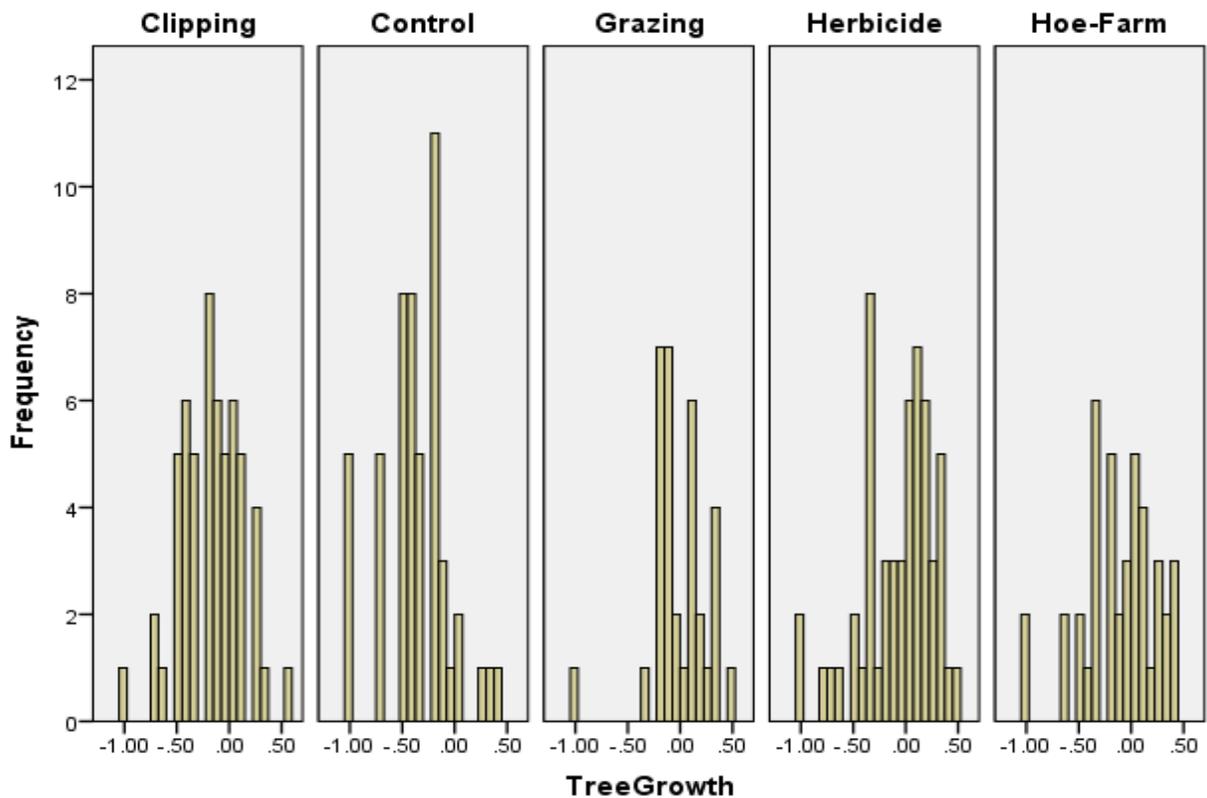
### Case Processing Summary

Grass treatment		Cases					
		Valid		Missing		Total	
		N	Percent	N	Percent	N	Percent
2016-2018	Clipping	56	93.3%	4	6.7%	60	100.0%
	Hoe-Farm	41	68.3%	19	31.7%	60	100.0%
	Herbicide	55	91.7%	5	8.3%	60	100.0%
	Grazing	33	55.0%	27	45.0%	60	100.0%
	Control	51	85.0%	9	15.0%	60	100.0%

### Tests of Normality

Grass treatment		Statistic	Shapiro-Wilk		Sig.
			df		
2016-2018	Clipping	0.986	56		0.757
	Hoe-Farm	0.957	41		0.124
	Herbicide	0.940	55		0.008
	Grazing	0.878	33		0.002
	Control	0.957	51		0.065

### Tree Growth Histograms- Grass Disturbance



**ANOVA and Tukey Test for Fire Disturbances**  
**Descriptives**

	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Minimum	Maximum
					Lower Bound	Upper Bound		
Early	51	-0.1274	0.30704	0.04299	-0.2138	-0.0410	-1.15	0.43
Late	33	-0.0733	0.31149	0.05422	-0.1838	0.0371	-0.70	0.51
Middle	41	-0.2613	0.30136	0.04706	-0.3565	-0.1662	-1.00	0.30
Control	34	-0.3723	0.29353	0.05034	-0.4747	-0.2699	-1.00	0.24
Total	159	-0.2031	0.32075	0.02544	-0.2533	-0.1528	-1.15	0.51

**ANOVA**

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	1.961	3	0.654	7.087	0.000
Within Groups	14.295	155	0.092		
Total	16.255	158			

**Multiple Comparisons**

(I) Fire treatment		Mean Difference (I-J)	Std. Error	Sig.	90% Confidence Interval	
					Lower Bound	Upper Bound
Early	Late	-0.05408	0.06785	0.856	-0.2109	0.1027
	Middle	0.13394	0.06370	0.157	-0.0132	0.2811
	Control	.24491*	0.06724	0.002	0.0895	0.4003
Late	Early	0.05408	0.06785	0.856	-0.1027	0.2109
	Middle	.18802*	0.07102	0.044	0.0239	0.3521
	Control	.29899*	0.07421	0.001	0.1275	0.4705
Middle	Early	-0.13394	0.06370	0.157	-0.2811	0.0132
	Late	-.18802*	0.07102	0.044	-0.3521	-0.0239
	Control	0.11097	0.07044	0.396	-0.0518	0.2737
Control	Early	-.24491*	0.06724	0.002	-0.4003	-0.0895
	Late	-.29899*	0.07421	0.001	-0.4705	-0.1275
	Middle	-0.11097	0.07044	0.396	-0.2737	0.0518

\*. The mean difference is significant at the 0.1 level.

## Homogenous Subsets

Tukey HSD<sup>a,b</sup>

Fire Treatment	N	Subset for alpha = 0.1		
		1	2	3
Control	34	-0.3723		
Middle	41	-0.2613	-0.2613	
Early	51		-0.1274	-0.1274
Late	33			-0.0733
Sig.		0.379	0.217	0.863

Means for groups in homogeneous subsets are displayed.

a. Uses Harmonic Mean Sample Size = 38.568.

b. The group sizes are unequal. The harmonic mean of the group sizes is used. Type I error levels are not guaranteed.

**ANOVA and Tukey Test for Grass Disturbance  
Descriptives**

	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Minimum	Maximum
					Lower Bound	Upper Bound		
Clipping	56	-0.1655	0.28926	0.03865	-0.2430	-0.0880	-1.00	0.54
Hoe-Farming	41	-0.1089	0.33989	0.05308	-0.2162	-0.0016	-1.00	0.43
Herbicide	55	-0.0712	0.34068	0.04594	-0.1633	0.0209	-1.00	0.51
Grazing	33	-0.0028	0.26653	0.04640	-0.0974	0.0917	-1.00	0.48
Control	34	-0.3723	0.29353	0.05034	-0.4747	-0.2699	-1.00	0.24
Total	219	-0.1388	0.32761	0.02214	-0.1825	-0.0952	-1.00	0.54

**ANOVA**

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	2.791	4	0.698	7.247	0.000
Within Groups	20.607	214	0.096		
Total	23.398	218			

**Homogeneous Subset**

Tukey HSD<sup>a,b</sup>

Grass Treatment	N	Subset for alpha = 0.1	
		1	2
Control	34	-0.3723	
Clipping	56		-0.1655
Hoe-Farming	41		-0.1089
Herbicide	55		-0.0712
Grazing	33		-0.0028
Sig.		1.000	0.122

Means for groups in homogeneous subsets are displayed.

a. Uses Harmonic Mean Sample Size = 41.617.

b. The group sizes are unequal. The harmonic mean of the group sizes is used. Type I error levels are not guaranteed.

## Multiple Comparisons

Dependent Variable:

Tukey HSD

(I) Grass Treatment		Mean Difference (I-J)	Std. Error	Sig.	90% Confidence Interval	
					Lower Bound	Upper Bound
Clipping	Hoe-Farming	-0.05658	0.06378	0.901	-0.2145	0.1013
	Herbicide	-0.09426	0.05891	0.499	-0.2401	0.0516
	Grazing	-0.16266	0.06810	0.122	-0.3313	0.0059
	Control	.20680*	0.06747	0.021	0.0398	0.3738
Hoe-Farming	Clipping	0.05658	0.06378	0.901	-0.1013	0.2145
	Herbicide	-0.03768	0.06403	0.977	-0.1962	0.1208
	Grazing	-0.10607	0.07257	0.589	-0.2858	0.0736
	Control	.26339*	0.07198	0.003	0.0852	0.4416
Herbicide	Clipping	0.09426	0.05891	0.499	-0.0516	0.2401
	Hoe-Farming	0.03768	0.06403	0.977	-0.1208	0.1962
	Grazing	-0.06840	0.06833	0.855	-0.2376	0.1008
	Control	.30107*	0.06770	0.000	0.1335	0.4687
Grazing	Clipping	0.16266	0.06810	0.122	-0.0059	0.3313
	Hoe-Farming	0.10607	0.07257	0.589	-0.0736	0.2858
	Herbicide	0.06840	0.06833	0.855	-0.1008	0.2376
	Control	.36946*	0.07583	0.000	0.1817	0.5572
Control	Clipping	-.20680*	0.06747	0.021	-0.3738	-0.0398
	Hoe-Farming	-.26339*	0.07198	0.003	-0.4416	-0.0852
	Herbicide	-.30107*	0.06770	0.000	-0.4687	-0.1335
	Grazing	-.36946*	0.07583	0.000	-0.5572	-0.1817

\*. The mean difference is significant at the 0.1 level.

**APPENDIX B**  
**ANOVA DETAILS FOR SIZE CLASS ANALYSIS**

## Shapiro Wilks Test

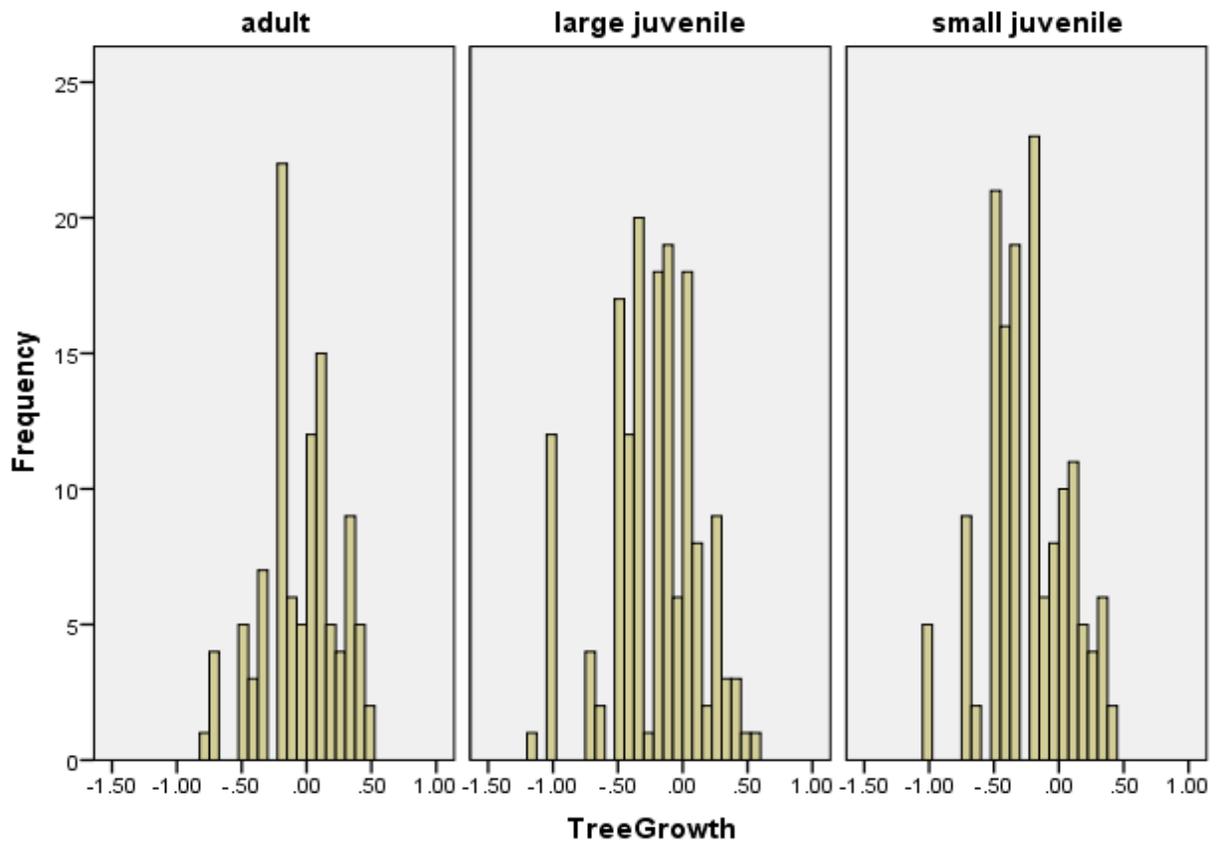
### Case Processing Summary

size class	Cases					
	Valid		Missing		Total	
	N	Percent	N	Percent	N	Percent
2016-2018 Large Juvenile	99	71.2%	40	28.8%	139	100.0%
Small Juvenile	70	70.0%	30	30.0%	100	100.0%
Adult Juvenile	49	81.7%	11	18.3%	60	100.0%

### Tests of Normality

size class		Shapiro-Wilk		
		Statistic	df	Sig.
2016 - 2018	Large Juvenile	0.957	99	0.003
	Small Juvenile	0.967	70	0.059
	Adult	0.960	49	0.097

### Size Class Histograms



## Descriptives

size class		Statistic	Std. Error		
2016-2018	Large Juvenile	Mean	-0.1954	0.03242	
		95% Confidence Interval for Mean	Lower Bound	-	0.2598
			Upper Bound	-	0.1311
	5% Trimmed Mean	-0.1825			
	Median	-0.1549			
	Variance	0.104			
	Std. Deviation	0.32255			
	Minimum	-1.00			
	Maximum	0.54			
	Range	1.54			
	Interquartile Range	0.40			
	Skewness	-0.620	0.243		
	Kurtosis	0.662	0.481		
	Small Juvenile	Small Juvenile	Mean	-0.1618	0.03950
			95% Confidence Interval for Mean	Lower Bound	-
Upper Bound				-	0.0830
5% Trimmed Mean		-0.1482			
Median		-0.0969			
Variance		0.109			
Std. Deviation		0.33047			
Minimum		-1.00			
Maximum		0.43			
Range		1.43			
Interquartile Range		0.49			
Skewness		-0.528	0.287		
Kurtosis		0.081	0.566		
Adult Juvenile		Adult Juvenile	Mean	0.0093	0.04244
			95% Confidence Interval for Mean	Lower Bound	-
	Upper Bound			0.0946	
	5% Trimmed Mean	0.0242			
	Median	0.0414			
	Variance	0.088			
	Std. Deviation	0.29706			
	Minimum	-0.82			
	Maximum	0.51			
	Range	1.33			
	Interquartile Range	0.40			
	Skewness	-0.703	0.340		
	Kurtosis	0.394	0.668		

**ANOVA and Tukey Test for Size Class – Fire Disturbances**  
**Descriptives**

	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Minimum	Maximum
					Lower Bound	Upper Bound		
Large Juvenile	38	-0.1841	0.33600	0.05451	-0.2945	-0.0737	-1.15	0.43
Small Juvenile	48	-0.2448	0.28872	0.04167	-0.3286	-0.1610	-1.00	0.34
Adult	39	-0.0227	0.27990	0.04482	-0.1134	0.0680	-0.70	0.51
Total	125	-0.1570	0.31333	0.02802	-0.2125	-0.1016	-1.15	0.51

**ANOVA**

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	1.101	2	0.551	6.068	0.003
Within Groups	11.072	122	0.091		
Total	12.173	124			

**Multiple Comparisons**

(I) size class		Mean Difference (I-J)	Std. Error	Sig.	90% Confidence Interval	
					Lower Bound	Upper Bound
Large Juvenile	Small Juvenile	0.06071	0.06541	0.624	-0.0748	0.1962
	Adult	-.16140*	0.06867	0.053	-0.3037	-0.0191
Small Juvenile	Large Juvenile	-0.06071	0.06541	0.624	-0.1962	0.0748
	Adult	-.22211*	0.06494	0.002	-0.3567	-0.0876
Adult	Large Juvenile	.16140*	0.06867	0.053	0.0191	0.3037
	Small Juvenile	.22211*	0.06494	0.002	0.0876	0.3567

\*. The mean difference is significant at the 0.1 level.

**Homogeneous Subset**

Tukey HSD<sup>a,b</sup>

size class	N	Subset for alpha = 0.1	
		1	2
Small Juvenile	48	-0.2448	
Large Juvenile	38	-0.1841	
Adult	39		-0.0227
Sig.		0.632	1.000

Means for groups in homogeneous subsets are displayed.

a. Uses Harmonic Mean Sample Size = 41.214.

b. The group sizes are unequal. The harmonic mean of the group sizes is used. Type I error levels are not guaranteed.

## ANOVA and Tukey Test for Size Class – Grass Disturbances

### Descriptives

	N	Mean	Std. Deviation	Std. Error	95% Confidence Interval for Mean		Minimum	Maximum
					Lower Bound	Upper Bound		
Large Juvenile	81	-0.1419	0.29545	0.03283	-0.2072	-0.0766	-1.00	0.54
Small Juvenile	61	-0.1313	0.33519	0.04292	-0.2171	-0.0454	-1.00	0.43
Adult	43	0.0409	0.29212	0.04455	-0.0490	0.1308	-0.82	0.51
Total	185	-0.0959	0.31589	0.02322	-0.1417	-0.0501	-1.00	0.54

### ANOVA

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	1.052	2	0.526	5.532	0.005
Within Groups	17.309	182	0.095		
Total	18.361	184			

### Multiple Comparisons

(I) size class		Mean Difference (I-J)	Std. Error	Sig.	90% Confidence Interval	
					Lower Bound	Upper Bound
Large Juvenile	Small Juvenile	-0.01061	0.05228	0.978	-0.1186	0.0974
	Adult	-.18277*	0.05819	0.006	-0.3029	-0.0626
Small Juvenile	Large Juvenile	0.01061	0.05228	0.978	-0.0974	0.1186
	Adult	-.17216*	0.06141	0.015	-0.2990	-0.0453
Adult	Large Juvenile	.18277*	0.05819	0.006	0.0626	0.3029
	Small Juvenile	.17216*	0.06141	0.015	0.0453	0.2990

\*. The mean difference is significant at the 0.1 level.

### Homogeneous Subsets

Tukey HSD<sup>a,b</sup>

size class	N	Subset for alpha = 0.1	
		1	2
Large Juvenile	81	-0.1419	
Small Juvenile	61	-0.1313	
Adult	43		0.0409
Sig.		0.981	1.000

Means for groups in homogeneous subsets are displayed.

a. Uses Harmonic Mean Sample Size = 57.698.

b. The group sizes are unequal. The harmonic mean of the group sizes is used. Type I error levels are not guaranteed.

**APPENDIX C**

**CHI SQUARE DETAIL FOR DEAD TREES**

## Chi Squared Test Case Processing Summary

	Cases					
	Valid		Missing		Total	
	N	Percent	N	Percent	N	Percent
alive/dead * treatment	180	100.0%	0	0.0%	180	100.0%

### alive/dead \* treatment Crosstabulation

			treatment			Total
			E	L	M	
alive/dead	alive	Count	56	33	48	137
		Expected Count	45.7	45.7	45.7	137.0
		Residual	10.3	-12.7	2.3	
dead	dead	Count	4	27	12	43
		Expected Count	14.3	14.3	14.3	43.0
		Residual	-10.3	12.7	-2.3	
Total		Count	60	60	60	180
		Expected Count	60.0	60.0	60.0	180.0

### Chi-Square Tests

	Value	df	Asymptotic Significance (2-sided)
Pearson Chi-Square	24.994 <sup>a</sup>	2	0.000
Likelihood Ratio	25.910	2	0.000
N of Valid Cases	180		

### Symmetric Measures<sup>a</sup>

	Value
N of Valid Cases	180

a. Correlation statistics are available for numeric data only.

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