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THE IMPACT OF FIRE ON TREE SPECIES DIVERSITY AND NATURAL REGENERATION IN MIOMBO WOODLANDS IN CHONGONI FOREST RESERVE, MALAWI

By

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ABSTRACT

The role of fire in Miombo forest ecosystems as an important forestry management tool for sustaining tree biodiversity has increasingly been recognized. In Malawi, there has been a limited or lack thereof of research conducted to date on the role of fire in Miombo forests, in particular focusing on the tree species diversity and natural regeneration. This study was conducted at Chongoni Forest Reserve (14 ^{0} 15 ^{1} S, 34 ^{0} 12 ^{1} E) to determine the effects of forest fire occurrence on tree species diversity and natural regeneration. Data on species occurrence, diameter at breast height, height, and regeneration were collected from a cluster of 3 concentric circular plots of sizes 0.01 ha, 0.04 ha, and 0.16 ha, assessed from 20 plots each in burnt and unburnt area. The data collected on the number of individual species and diameter at breast height were analyzed for tree species diversity, richness, and structure. The study has established that tree species in the reserve are differentiated in both burnt and unburnt plots in their richness. However, less significant differences are encountered on tree species and regenerants diversity values. More so, further analysis on frequency distribution among regenerants shows that relatively higher numbers are found in unburnt sites than in burnt sites. However, the existence of regenerants in both sites further indicates the resilient nature of Miombo woodland species to disturbances. In addition, Brachystegia floribunda and Uapaca kirkiana were identified as the two-key species in the forest. The study concluded that fire occurrence in the forest reserve has a role in maintaining the tree and regenerants species diversity and richness. Thus, the study recommends fire management, including monitoring routines to maintain species diversity.

Keywords: Forest fires, tree species diversity, natural regeneration, Miombo woodlands

DECLARATION

I, Pilirani Tendai Khoza, declare that the thesis/dissertation, which I hereby submit for the degree MSc in Forest Management and Environment at the University of Pretoria is my work and has not previously been submitted by me for a degree at this or any other tertiary institution.

Signature: P.T. Khoza

Date: 25/11/2021

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CHAPTER ONE: INTRODUCTION

1.1 BACKGROUND

The world's forests cover about 40 percent of the land surface on earth (FAO, 2014) and are habitant to more than 70 percent of all land-dwelling plant and animal species (Chao, 2012). In Africa, the miombo woodlands make up the most extensive tropical seasonal woodland and dry forest formation, with an estimated coverage of 2.7 million km2, largely dominating parts of Sub-Saharan Africa (Thompson and Helmschrot, 2021). Forests provide for a wide range of benefits in areas of ecological, social, and economical benefits; in particular, tropical forests have high protective and productive value, with an estimated 10-30 million different species living in very complex ecological communities found in them (Bengtsson, et al., 2000). Forests have such a significant role in the world, and they are undeniably a source and major contributor to the livelihoods and functioning of individuals, and communities; they are as well an important part of the ecosystem that sustains life in them (both plant and animal life) (Chirwa et al., 2008; Chirwa et al., 2017). Additionally, forests are an essential source of a variety of products and services, which include among others: food, fuelwood, timber wood, genetic pools, and regulating services such as carbon storage, watershed protection, and cultural services (Kozlowski, 2002).

Interestingly, tropical forests including miombo woodlands are diverse, and they comprise a vast majority of plant and animal genetic material; providing habitat to millions of species of plants and animals (Ramachandra, 2000). Miombo woodlands are habitant to diverse life forms, including plants, mammals, reptiles, amphibians, birds, et cetera (Chidumayo, 2010; Godbless et al., 2019). Despite a low faunal species diversity and richness, the Miombo woodlands are estimated to contain about 8500 higher plant species, of which more than 54% are endemic (Chidumayo, 2010, Dewees et al., 2011). According to a study by Morgan et al., (2001), more than 2.5 million people live in surrounding and bordering areas of tropical forests such as miombo woodlands, relying greatly on the forest resources for food, water, fuel, and others. Forests also provide beauty to the scenic environment, as well as other opportunities such as education, health, and recreation.

Unfortunately, forests in Malawi, Africa, and across the world have continually been threatened by disturbances occurring naturally (i.e., naturally-induced drought, windthrow, flooding, and disease outbreaks) and human-induced (i.e. fire, mining, charcoal production, and agriculture) (Kozlowski, 2002). In particular, forest fires are a commonly occurring phenomenon and have occurred in large areas of forests worldwide. According to Chidumayo (1997) and Chuvieco (2009), essentially, forest fires are uncontrolled fires that grow out of control, which occur accidentally, naturally, or by intentional causes. Forest fires are a major hazard and potentially cause significant loss of human, cultural, economic and ecological resources, cultural, economic and human resources (Morgan et al., 2001; Chuvieco et al., 2008). Gonzalez et al., (2005) reported that more than 220,000 forest fires occur every year worldwide resulting in an estimated 6 million hectares of forests burnt. However, despite this devastating and grooming picture, fire is an important and critical natural part of some ecosystems of forests and has been used for a long time as a land management tool (SCBD, 2001). The use of fire as a forest and land management tool is not a new phenomenon; fire has been used by individuals across the world (Chidumayo, 1997b). For instance, in miombo woodlands, fire has been established to influence the regeneration of tree species (Mapaure, 2001; Zolho 2005). Granström, (2001) also reported that the disturbance in forests and their associated landscapes are necessary for the maintenance of diversity of living things and processes in them. Study's by Mapaure, (2001), Zolho, (2005), Dayamba, (2010), Hollingsworth, (2015) have also reported that forest fires are among the important disturbances in ecosystems. According to Malmström, (2006), the knowledge and understanding of the effects of fire have in recent years grown in popularity among the academia. Malmström, (2006) indicated that more evidence has shown that fire as a disturbance is integral in ecosystems conservation management, affecting the structure, composition, and vegetation pattern in a landscape (Botkin, 1990; Morgan et al., 1994).

Several studies on forest fire assessment and management have been conducted, however, forest fire research has been pioneered by a few countries, including the US, Canada, Australia, and Russia (Malmström, 2006). Many researchers have since concentrated their efforts on gaining a better understanding of fire processes in various ecosystems. Such research on forest fires has led to the development of management systems and risk forecast systems in time and space, which many governments and other non-state actors have been trying to advance. Malawi, on the other hand, has made slow progress. The country's situation is similar to that of other miombo woodland ecoregions. Although reports indicate an increase in fire occurrence, loss of biodiversity, deforestation, and increased encroachment, the majority of which are happening in protected areas. Forest fires have been overlooked and given insufficient attention, limiting the amount of effort available. Indicating a clear need for evidence to develop guidelines and programmatic interventions to address the problem of forest fires. However, in the last decade, the context in which management and conservation of Malawi's forests have changed considerably. Forest management and environmental sustainability have received increased attention in Malawi, as they have in much of Sub-Saharan Africa, and have sparked a lot of policy debate. However, it requires an understanding of the many factors that contribute to forest fire regimes and their consequences, but Malawi has done very little analytical research on this topic. Irrespective of this, a large number of people continue to earn a living from miombo forests using fires, and demand for forest resources is increasing every day (Millington and Kaferawanthu, 2004). Furthermore, issues of climate change further exacerbate the risk of fires in the forests. This suggests therefore the majority of the country's miombo forests, as well as those in all other countries with miombo, are still at risk of fire. It is clear that neglecting the issues of forest fires which account for a large portion of the leading causes of forest resource destruction, will have negative consequences in the not-too-distant future.

1.2 FOREST FIRES AND WORLDS FORESTS AND ECOSYSTEMS

Based on the fossil record, wildfires first appeared about 420 million years ago shortly after the evolution of terrestrial plants (Glasspool et al. 2004, Bowman et al. 2009). From that time, forest fires have played a critical role in molding the evolution and distribution of flora and fauna across the globe (Keeley and Rundel 2005; Beerling and Osborne, 2006). According to Chidumayo (1997) and Godbless et al., (2019), fire is a key process that has shaped the structure and function of many of the world's ecosystems. More recently, forest fires from anthropogenic origins have played a significant role in shaping the structure and function of modern global ecosystems(Stephens, 1999). Noteworthy, the forest fires that have affected large tracts of land across the dry parts of North America, the Mediterranean, Africa, and Australia and have been a key process shaping the evolution of flora and fauna in these regions (Keeley and Rundel, 2005, Beerling and Osborne,2006). For instance, wildfires can have devastating effects on biodiversity via the removal of vegetation, habitat, and food sources and subsequently increase the vulnerability of surviving animals to predation (Kodandapani et al., 2008). According to McKenzie et al. (2004), it is predicted that larger, more intense and more frequent wildfires could occur in many parts of the world in the future due to climate change. Smaller and controlled fires have been used as a tool to manage vegetation. Such fires are prescribed to clear vegetation, improve the forage value of pastures, reduce wildfire hazards (Letnic et al., 2004, Valentine and Schwarzkopf, 2008), and, more controversially, promote the conservation of biodiversity (Noble et al., 1997). The mammalian diversity may be unaffected, positively affected (Masters, 1996), or negatively affected by fire (Letnic et al., 2004), and additional factors such as rainfall and grazing that interact with fire also influence biodiversity (Yarnell et al., 2007).

1.2.1 Causes of fires

In Africa, the causality of fires includes natural and human-made (FAO, 2005). Natural fire causes include lightning (FAO, 2005), however, a major portion of forest fires in Africa, are caused by anthropogenic activities (Chidumayo,1997). According to FAO, (2005) fire and early humans were instrumental in the evolution and development of Africa's forests, and are a major ignition source. Such anthropogenic causal activities include prescribed fires, arson, and negligence.

1.3 FORESTS IN MALAWI, STATUS, DEGRADATION, AND CONSERVATION

Malawi is rich in forests and biodiversity, comprising forest reserves, national parks, and plantation forests, among others. A large part of Malawi's forests is made up of miombo woodlands (Mawaya et al., 2011; World Bank, 2019). However, the forests and biodiversity in the country are declining substantially (GoM, 2015; USAID, 2019). Malawi forest areas have provided habitat to plants and wildlife, but it currently has the highest rate of deforestation in Southern Africa (GOM, 2015; USAID, 2019). According to FAO-FRA (2010), by 1975 in Malawi about 47 percent of the land surface area was classified as forest, since then, however, 36 percent of the land surface area is now classified as forests, of which 15 percent is under natural woodlands on customary land, 11 percent under game reserves and national parks and 10 percent under forest reserves and protected hills slopes (Knoema, 2021).

Malawi's forest cover decline has been attributed to several causes, most of which are anthropogenic and some natural. An estimated 1 percent of forest coverage is lost every year due to human activities, which include among others rampant deforestation, clearing of land for settlement, farming, and other economic activities (GOM, 2010). Largely, deforestation has been on the rise due to population growth, poverty, infrastructure development, economic activities, agriculture (CURE, 2010). Forest cover per district also varies from district to district, and forest cover percentages have declined since the 1990s (Munthali and Muriyama, 2012).

Many communities are at risk of losing forests and other environmental endowments, which threaten the sustainability and management of forests and forest resources. With the realization of an increasing need to conserve forests, the government of Malawi and development practitioners, are at work trying to develop, and implement various interventions that can: slow degradation, and increase conservation of these forests and their resources. Some of the cited reasons for conserving forests include: (1) to maintain essential ecological processes and life support systems, (2) to preserve genetic diversity; and (3) to ensure the sustainable utilization of species and ecosystem (Phiri, 2013).

For Malawi, the government has gazetted forest reserves to promote watershed and catchment conservation, and the provision and regulation of environmental services including soil and biodiversity conservation, aesthetic and cultural value among others (GOM, 2010). However, there are still other pertinent issues and anthropogenic activities that threaten the status of these forest reserves and other forest-protected areas. For instance, human encroachment for cultivation and settlement is one major problem facing forest reserves. CURE, (2010) reported that about 2.6 percent representing 23,012 hectares out of 870,052 hectares of forest reserves had encroached at 571 locations across Malawi.

According to Kambewa et al., (2007), the illegal production of charcoal is also compromising the of these protected areas. As charcoal is burned to supply urban areas with low-cost fuel energy. FAO-FRA (2010) reiterates in their report that fire remains one of the biggest problems that has affected the management of natural woodlands, industrial softwood plantations, and fuelwood and poles plantations in Malawi.

As indicated earlier, several research studies () have looked at the effects of fire in miombo woodland forests, as well as its impact on diversity and regeneration variables, but there is a lack of knowledge in research on forest fire effects on Malawi's miombo woodland ecoregions. Although there are programs in Malawi that study fire regime, few have focused on how these forests respond to the effects of fires on diversity and regeneration.

1.4 PROBLEM STATEMENT

Malawi is primarily represented by the central Zambezian miombo woodland ecoregion, which covers most of the western side of Lake Malawi in the north, and small areas to the southwest and southeast represented by the Southern and Eastern Miombo Woodlands, which differ primarily in rainfall, being drier woodlands (Millington and Kaferawanthu, 2004). Being the most common woodland in the country, however, has meant that these woodlands have been primarily affected by fire, some of which have been induced by government forestry officials as a management tool, and others which have been triggered by anthropogenic activities that have gotten out of hand. In particular, in 1995 in Chongoni Forest Reserve, some 5,550ha (about 36%) of the forest were destroyed by forest fires caused by anthropogenic activities such as charcoal production, honey harvesting, hunting, and livestock grazing, which resulted in smoke haze, pollution, loss of seedlings and biodiversity; more recent fires have been reported (MoMNRE, 2006), According to Pastro (2011), the effect of fire on plant diversity can vary depending on factors such as habitat and geographic location, and such effects can be both long-lasting and short-lasting in some cases. Think Hazard (2021), reports have indicated that Malawi is classified as having a high wildfire hazard, and numerous other studies and reports have revealed that fire is one of the most common occurrences in the country's forests (Munthali et al., 2012; GOM, 2008, 2014). A study by USAID(2005) indicated that approximately 45 percent of forest areas in Malawi experienced fires; and the causalities varied from natural to unnatural, as well unintended and intended (prescribed) fires. Despite that destructive forest fires are a common incident in Malawian forests, there is little information on the impact that these fires have had on the affected forest tree species diversity and regeneration.

It is also important to note that most forest areas are adjacent to or are surrounded and bordered by rural areas than in more urban vicinities. Additionally, forest interaction has been disproportionate between rural and urban areas. In rural areas, due to the interaction between bordering and surrounding communities, such incidences of wildfires are common. However, there has not been any study on forest fires in Malawi (FAO-FRA, 2010). Furthermore, most studies have only considered focusing on the importance of community forests and the roles of community structures in their management (Ngwira and Watanabe, 2019). This is indicative that forest fires and their impact on forests are not widely understood in the country. According to Ryan and Williams (2011), and Ratnam et al., (2014) there is a lack of data in the less-studied miombo ecosystem, particularly on the impacts of fire. It is against this background that there is a need to establish the impact of forest fires on plant species diversity and regeneration. This will contribute to understanding forest fires and the conservation of forests and biodiversity.

1.5 RESEARCH OBJECTIVES

1.5.1 Main Objective

To investigate the effects of forest fire on tree species diversity and regeneration of Miombo woodlands in Chongoni forest reserve.

1.5.2 Specific Objectives

- To determine the effect of forest fire on stand and tree species composition and structure
- To determine the impact of fire on tree species diversity and richness
- To determine the effect of fire on tree species natural regeneration

1.5.3 Research Hypothesis

- There is no significant difference in stand and tree species composition and structure in burnt and unburnt areas
- There is no significant difference in tree species diversity and richness
- Forest fires do not significantly affect tree species natural regeneration

1.6 SIGNIFICANCE OF THE STUDY

The study of the effects of forest fires on tree species diversity and regeneration is of immerse relevance, considering that it documents the species diversity and regeneration in the country. It also assists in bringing forward empirical data, evidence, and significant knowledge on the effects of forest fires in forest areas in the country, especially that this study makes a unique contribution by studying one of the ecological regions of the miombo woodlands (Central Zambezian Miombo Woodland). Knowledge on these effects of fires contributes to and strengthens forest and biodiversity conservation interventions delivery for protected forest areas, like Chongoni Forest Reserve. Additionally, the information generated in the study will help policymakers and other development partners on how they may formulate and implement appropriate policies that will promote forest and biodiversity conservation in Malawi, and other countries in general.

The study contributes to the existing research data on forest fires, more especially in miombo woodland in Malawi and Africa. It contributes to the few studies and body of knowledge on forest fires and tree species diversity and regeneration.

1.7 THESIS LAYOUT

The thesis is composed of five chapters. Chapter 1 is an introduction chapter and Chapter 2 is a literature review that covers the relevant literature. Chapter 3 outlines the methodology adopted for the study. Chapters 4 and 5 give a presentation of results and discussion of the findings as they related to other literature and make a conclusion and recommendation.

CHAPTER TWO: LITERATURE REVIEW

2.1 DEFINITIONS AND CONCEPTS

2.1.1 Fire

Barnes et al., (2004) define fire as a manifestation of burning involving fuels, air, and heat that produces light and heat and often smoke. According to Barnes et al., (2004) for fire to exist critical three elements must be present (i.e., Oxygen, Heat, and Fuel). Mysterud et al., (1997) indicate that fires vary extremely and can lead to everything from sparse influences on the ground layer to severe impact on an entire ecosystem. Other studies have indicated that humans have altered the natural fire regimes by changing the frequency and intensity of fires in ecosystems (Ganz et al., 2003; Sanders 2003). However, other natural causes have also been contributing factors to fires in forest ecosystems (Mysterud et al., 1997). Fire has played and will continue to play, an important role in forest ecosystems by affecting the distribution, survival, and regeneration of trees (Ganz et al., 2003; Sanders, 2003; Rowell and Moore, 2000; Mysterud et al., 1997).

Additionally, Mysterud et al., (1997) make mention that fires are often classified according to the location of the fire namely: forest fires, heather fires, and grass fires. Sometimes, fires are referred to as catastrophic fires; based on the intensity and severity of prescribed fires which are hazard-reducing fires that are ignited as part of risk management. According to Mysterud et al., (1997), it is purposeful during surveys to separate fires into some main categories based on their background and causal conditions. Suggested categories include the following; natural fires, anthropogenic fires, and wildfires. Barnes et al., (2004) fire types include domestic fires; industrial fires; wildfires or bushfires/forest fires; and prescribed fires.

2.1.2 Fire Management

Karki, (2002) indicates that fire management is essentially about incorporating efforts to maintain a fire within the desired fire regime. Thus, a fire management system enables the identification and assessment of needs. Many models and approaches about fire management systems exist and have evolved worldwide (Ratnam et al.,2014). According to Karki, (2002), fire management systems attempt to balance the requirement for an effective approach tuned to national and local needs and resources.

Fire management as defined by FAO (2009) is the discipline of using fire to achieve land management and traditional use objectives, together with the safeguarding of life, property, and resources through the detection, prevention, control, restriction, and suppression of fire in the forest and other vegetation in rural areas.

2.3 IMPORTANT ROLES OF FIRE IN FOREST ECOSYSTEMS

Forest fires have an important role in many forest ecosystems (Brown and Smith, 2000). According to Reinhart et al., (2016) fires are a fundamental part of most forests, and are largely impacting ecosystems and soils. Forest fires have both social and ecological effects. In their study, Trapnell et al. (1976), reported that fire has positive effects on soil nutrients status, which improved due to the long-term effects of burning in comparison with complete protection from fire. Adding to this, Ohl' s (2005) study of post-regeneration low fire frequency in Spain reported that species richness increased in the first years after the fire due to the availability of a good supply of nutrients and light. Burning is beneficial for fire-dependent species and for invasive grasses that can be used for grazing by cattle (Mayer, 2002), however, it can be destructive in fire-sensitive. For fire-sensitive species, that are not adapted to survive fire, moderate and ecologically appropriate fire is required to create specific habitats and successional processes, other protracted and frequent fires which may change the species composition to more flammable species (for example, miombo woodlands may be converted to shrublands or woodlands) (Chidumayo,1988; Chidumayo, 1997; Hollingsworth et al., 2015).

Both plants and animals depend on the occurrence of fires as a necessary disturbance for the structural diversity it generates (Lindenmayer, 2019). According to Barlow and Peres, (2008) forest fires can be a constructive force where it is used and responsible for the maintenance of the health of certain firedependent ecosystems. The study goes further to indicate that, where prescribed fires can be used as an integral component of ecosystems and management due to their (fire) ecological role in mediating and regulating the functionality of ecosystems. Chen (2006) explained that there is still a need for more knowledge on the temporal and spatial variations of fire effects through long-term experimental monitoring and modeling. In his study, Huston (1994) mentioned that where fire creates space and makes the light available, it has the potential to enhance diversity more in productive plant areas where aboveground competition is high.

2.3.1 Fires and natural regeneration

After the occurrence of a fire, sprouting is one way in which plants recover. Plant shoots may originate from dormant buds on plant parts located above the ground surface or from various levels within the litter, and soil layers (Flinn and Wein, 1977). Fire also can induce the germination of dormant seeds of some species which leads to an abundance of seedlings of such species in the first post-fire year (Mwansa, 2018). Also, in areas where there are perennial seedlings, they may flourish depending on their inherent longevity, as long as the site has its specific environmental requirements (Mwansa, 2018). Sometimes, they may persevere only as seeds (Lykke, 1998). In regards to miombo woodlands, Chidumayo (1997) indicated that miombo woodlands are to some extent resilient to fire. Missanjo et al., (2014) additionally mention that most miombo tree species are tolerant to fire and are believed by Ecologists to have evolved with fire. Seedlings establishment in miombo woodlands is not common amongst the tree species due to sporadic fruit production, high levels of pre-dispersal seed predation, limited seed dispersal distance, very low field survival due to drought stress, and early and late fires (Malaisse,1978; Chidumayo, 1992; Chidumayo 1997; Mwabumba et al., 1999). Other studies have reported plant persistence and regeneration that have evolved in fire-dependent communities. These however range from fire-dependent plant species (for example, shrubs that solely rely on fire to break dormancy) or fire-enhanced (for example, annuals that benefit from the exposure of mineral soils for germination) to those inhibited by fire (for example, shallow root herbs whose structure are consumed by fire) (Brown and Smith, 2000). Chidumayo and Gumbo (2010), Kennard et al, (2002), and Mwansa (2018) have concluded that the regeneration abilities in trees species, tend to differ among species and decline with age, size, frequency, and severity of disturbance.

2.3.2 Fires and tree species diversity

According to Kindt and Coe (2005), and Cunningham et al., (2005) tree species diversity is the number of species that can be differentiated (richness) and their relative abundances in each species (evenness). Malaisse, (1978), and White (1983) indicate that miombo woodlands are floristically rich, especially in the three genera of Brachystegia, Julbernardia, and Isoberlinia. Donagh et al., (2001), found that anthropogenic activities are negatively impacting the spatial distribution of tree species in miombo woodlands. Other studies have observed that frequent cutting, burning, and grazing in woodlands increases species diversity as well as abundance (Lowore et al., 1995; Gondwe, 2011). Chidumayo (1997b) reported that regrowth stands (after clearing) usually have higher species diversity than the oldgrowth stands they replace in miombo woodlands. Much as these studies have shown that anthropogenic activities alter the appearance and size class profile of Miombo woodlands, other studies have shown that there has been no significant decrease in the number of woody plant species (Vermeulen, 1996; Malimbwi et al., 2005; Banda et al., 2006; Gondwe, 2011).

Halpern (1988), Halpern and Spies (1995) reported that the severity of fire can have a large effect on understory response. They further point out that at low severity, plant mortality is minimal, but fire dependent species (for example, seed-banking shrubs) will less likely be able to establish. Thus, the effects on abundance, richness, and composition are likely small (Dayamba, 2010). During high severity fires, the mortality rates of fire-inhibited species are high, while fire-dependent or fire-enhanced species are more likely to establish (Wang and Kemball, 2011). Where the effects on abundance, richness, abundance, and composition are greater they, in turn, facilitate greater diversity of plant functional groups, leading to a composition of species with different environmental requirements and sensitivities to fire (Bourgoin, and Reymondin, 2019), However, Wan et al., (2001) in their study, they mention that the consequences for other ecosystem attributes (for example, understory structure and diversity) have not been evaluated critically. The study further mentions that it is implicitly assumed that the restoration of structures and reduction of microclimate (for example, light and temperature) and soil resources (moisture and nutrient) result from reductions in tree density or consumption of coarse woody debris or forest-floor litter (Fites-Kaufman et al., 2007). According to (Metz and Dindal,1975) the relative importance of such direct and indirect effects can be mediated by the frequency, spatial heterogeneity, and severity of burning. Importantly, plants having different growth forms, life (annuals/biennials, herbaceous perennials, and shrubs), and sensitivity to burning will differ in their response to fire and its variation in frequency or severity.

4.3 BURNING AS A FOREST MANAGEMENT TOOL

According to Haugaasen et al., (2003) fire and shifts in fire regimes are the major drivers of ecosystems structure and process and they have the potential to cause a decline in species diversity. Chidumayo (1997), points out that fire is an important natural phenomenon that has affected the savannah systems. The occurrence of forest fires has been linked by many authors to be determined by climate (environmental conditions), ignition source, and fuel load (Bond and Van Wilgren, 1996; Chidumayo, 1997). In addition, the increased dependence on forests resources and products by humans for various uses is a leading cause of forest fragmentation that could further exacerbate future fire events in forests (Cochrane and Schultz, 1999). Reside et al., (2012) predicted that forest fires incidences may increase in many areas due to change in climate and land management. From the foregoing, another study reiterated the need to urgently undertake more studies on the patterns of fire effects on ecosystems composition, structure, and functions for the application in fire and ecosystem management (Chen, 2006). A good understanding of the fire effects and its underlying principles are important to the reduction of risk of uncharacteristic wildfires (Chen, 2006), and for the proper use of fire as an effective forest management tool toward management goals.

Goldammer (1990) explains in his study that the character of the fire is determined by the amount, nature, and spatial distribution of ignitable fuel available in any forest. Those, who would want to do prescribed burning must first also consider the biodiversity in them. In his study, Cottle (2007) indicated that fire risk assessment and research should essentially begin at understanding the ecosystems and the effects that may follow if such forest fires are to be used. Fire severity is a term used to describe the ecological effects of a specific fire; It further describes the magnitude of disturbance and reflects on the degree of change in ecosystem components (Slik et al., 2010). Having that fire affects both the aboveground and below-ground components of the ecosystem, the term has also been used to describe the effects of fire on soil and water systems, fauna and flora, and society (Simard, 1991). Other studies and researchers have seldom considered the contributing effects of landscape processes and abiotic pressures such as forest fires in the loss of biodiversity (Buhk et al., 2007).

The occurrence of fire results in a wide diversity of ecological effects which change forest structure and floristic composition (Cochrane and Schultz, 1999; Slik et al., 2010). In other cases, ecosystems may recover from small fire disturbances but frequent and severe fires events may potentially offset the balance which can lead to the inability of an ecosystem to recover causing it to collapse together with the loss of its major ecological functions and diversity (Haugaasen et al., 2003; Slik et al., 2008). Plant tissues may die of the amount of heat a plant tissue receives; such heat received by a plant is determined by the maximum temperatures reached and the duration of exposure of the plant tissue to the fire (Wright and Bailey, 1982). Most plant tissues (cells) die when heated to temperatures between 50 to 55 ^{o}c (Wright and Bailey, 1982).

Nonetheless, the characteristics of the fire, the vegetation, conditions of the fire site, and post-fire weather determines the vegetative response to fire which may vary from plant to plant. Fire effects on plants may vary significantly among different fires and on different areas of the same fire (Balch et al., 2011). The behavior of fire, duration, patterns of fuel consumption, amount of subsurface heating all influence a plant's injury and recovery (Wright and Bailey, 1982). The characteristics of the plant species on the site, their susceptibility to fire, and how they do recover after a fire event also influence the postfire response of a plant (Haugaasen et al., 2003).

4.4 UNDERSTANDING PRESCRIBED FIRES AND WILDFIRES

For most people, fire is an inexpensive and readily available tool that can be utilized to conduct a variety of activities including clearing vegetation for agriculture, and improving pastures in their communities (Finch,1997; Cochrane and Laurance,2008). Such anthropogenic fires are reported to have affected parts of forests around the world (Webster, 2010). The studies by Jones (2005) and Cottle (2007) add to this, as they point out that most tropical forest fires are caused by agricultural fires escaping into the surrounding vegetation.

DeBano et al., (1998) defined prescribed fires as fire burning with certain prescriptions resulting from planned ignition. The other fires that occur as wildfire are defined by Bowman et al., (2009) as fires that are ignited by natural causes such as lightning or unintentionally by human action.

The use of prescribed fires also referred to as prescribed burning, in other regions mainly in the Americas and Mediterranean region have used prescribed burning to control woody debris accumulation (Cochrane and Schultz, 1999; Cochrane, 2003). In other regions, people have used prescribed burning as a traditional forest regeneration method by way of setting fires in clear-cut regions (Hollingsworth et al., 2015). Such methods have particularly been used in areas having thick raw humus layers to improve soil quality as well as to uncover it to promote seed germination and seedling growth. Other studies have reported such use of prescribed burning as aimed at the restoration of animal and plant habitats, provision of great grazing areas for herds, and maintaining open landscapes (Cottle,2007). Others have used prescribed fires to hunt for small animals and large animals in agricultural lands, and thick forests for easy visibility (Hollingsworth, 2015). Although, the use of prescribed burning has lost its popularity in most areas, those who still use it mainly burn small areas (JICA, 2013). For instance, (Granström, 2001) reported that forest companies in northern and central Sweden burn an estimated 4000 hectares per year (which is 5 percent of the dry and mesic forest land areas).

Prescribed fires differ greatly from naturally occurring wildfires. Natural causalities such as those ignited by lightning are rare in most of the regions, occurring in peak lightning seasons and most times occurring in areas of drought (Granström, 2001). Lightening fires are known to burn deep into soil layers and are known to cause severe damage to vegetation (Granström, 2001). Fire incidents stemming from human anthropogenic activities on average, mainly occur during peak agricultural activities and favor drier areas, and are usually less severe (Buhk et al., 2007). Prescribed fires are normally intended to burn less than wildfires; their severity is usually between low to moderate. According to DeBano, et al., (1998), the low severity fires normally are intended to burn the ground cover and spread to herbaceous plants, while leaves, stems, and tops of the trees are left largely unburnt. On the other hand, sometimes wildfires can get to be much harder and intense;they burn more heterogeneously as some areas end up getting burnt more than once.

4.5 INFLUENCE OF FOREST MANAGEMENT ON PLANT SPECIES DIVERSITY AND REGENERATION

Forests that are sustainably managed stand to have multiple environmental and socio-economic utilities at global, national, and local scales, and are vital in the achievement of sustainable development. In Malawi, the government is a signatory to voluntary guidelines that help in the promotion of forest conservation. As already indicated above, Malawi using the provisions in the forestry act 1997, has gone further to gazette several forest areas into protected forest reserves, wildlife reserves, and national parks, as well as setting village forest areas and forest management units (FMUs).

Diversity on various levels(genetic, individual, population, species, and ecosystem) is affected differently by the forest management practices (Ledig, 1992; Finkelday and Ziehe,2004; Ratnam et al.,2014). Additionally, Spiecker (2003) points that management practices have contributed largely to the shaping of the current state of forests around the world and further reiterates that future management practices must be considered to sustainably maintain the genetic pool diversity and increase the resilience of forests. There exist a lot of different types of forest management practices, which are determined by the cultures of local communities, climate change, forest types, and site conditions. Several studies have been conducted to evaluate the impacts of forest management on the diversity of species of various stands (Konnert and Hosius, 2010; Danusevicius,2016), and most studies have generally focused on certain plant species or certain management practices. Such studies, however, have generated contradictive results, (Fisher and Binkley, 2000); for instance, some studies have found that genetic structure and variation were not associated with the management practices, whereas other studies have supported that the increased genetic diversity is due to the different forest management practices (Schaberg et al., 2008). Kavaliauskas et al., (2018), however, points out that such studies have failed to capture the long-term effects of forest management practices on genetic diversity processes, but rather only give and consider a snapshot of these changes.

Studies by Vos et al., (2000) and Corona et al., (2011) are of the view that the understanding of past disturbance dynamics and their relationship to human activities and management practices are essential for the prediction of forest ecosystems.

Forest management practices also affect the regeneration of plant species. Despite that regeneration can be natural, naturally combined with assisted natural regeneration (artificial by-planting, or artificial regeneration by sowing or planting); the diversity and genetic structures of naturally regenerated areas may depend on a variety of factors (for example, by the numbers and spatial distribution trees, the flow of pollen, dispersal of seed) (Chidumayo, 1992; Geburek, 2005). According to Konnert and Hosius (2010), natural regeneration generally results in no loss of genetic diversity if the number of trees within the area that essentially participate in the reproduction process is adequately large. Savolainen and Kärkkäinen, (1992), however, add that natural regeneration can differ from adult trees in their genetic variability and structures when, for instance, population sizes are significantly reduced through severe removal of reproductive trees and only a few numbers of parent trees contribute to seed production for the next generation.

CHAPTER THREE: METHODOLOGY

3.1 STUDY AREA

The study area for this research was Chongoni Forest Reserve. The forest reserve is situated in Dedza District in the central region of Malawi. In terms of vegetation, the reserve is predominantly miombo and is dominated by the common canopy species of *Brachystegia floribunda* and *Faurea saligna* and the dominant sub-canopy species is *Uapaca kirkiana* (Gondwe, 2011; Missanjo et al., 2014). The reserve is 120.39 km^2 and was first gazetted in 1924 and re-gazetted in 1948 (Gondwe, 2011). The geographical position is 14° 15' S, 34°12' E, and 1600 m above sea level (Missanjo et al., 2014). The area receives rainfall that ranges from 1200 mm to 1800 mm per annum, with an average rainfall of 1100 mm, and temperatures range from 7°C to 25°C, with an average temperature of 17°C (Malmer, 2007; Gondwe, 2011; Missanjo et al., 2014;). The base rock is of late Precambrian with synenitic and perlithic basement complexes (Lowore et al., 1994; Gondwe, 2011). Miombo woodland soils are ferruginous and ferrallitic in nature; this is why these soils are infertile (Lowore et al, 1994; Gondwe, 2011; Missanjo et al., 2014).

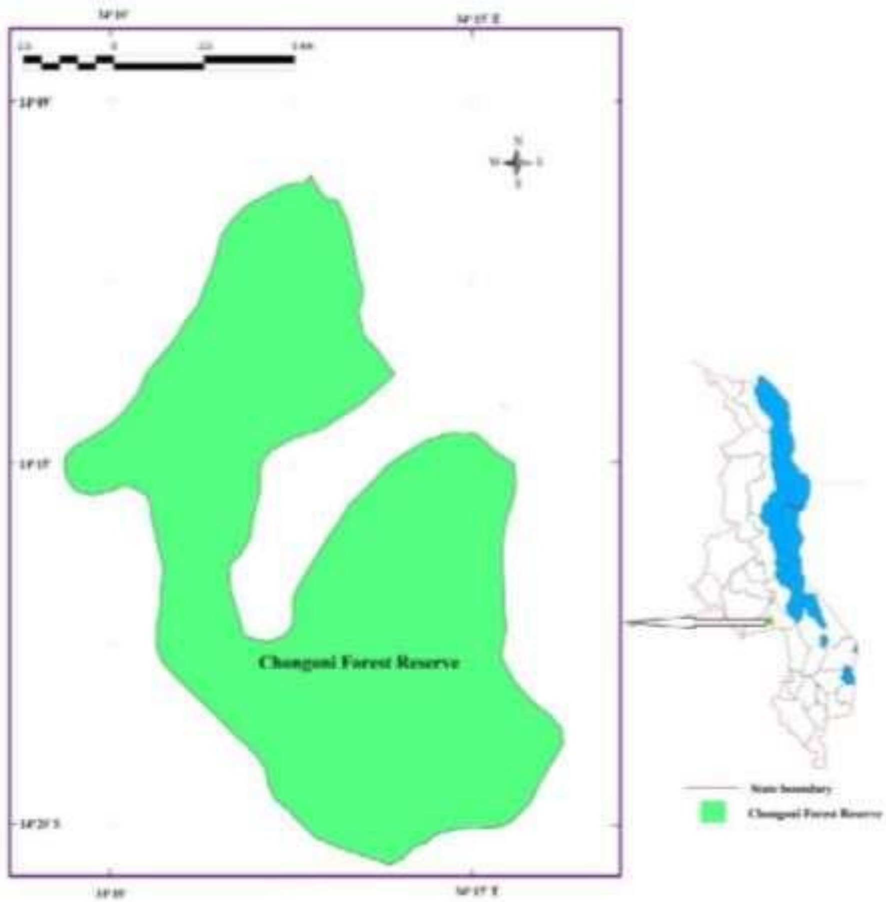


Figure 1: Location of Chongoni Forest Reserve in Malawi.

Figure 1: Map showing the location of Chongoni Forest Reserve in Malawi

Source: Missanjo et al., (2014)

3.2 SAMPLING

3.2.1 <u>Sampling method and sample size</u>

The study used a combination of stratified and systematic sampling methods. Stratification is carried out based on major and usually very obvious variations within the area under study (Kent and Cooker,1996). Thus, the area was first stratified into burnt and unburnt areas. Then a systematic random sampling was used to collect data from the selected sites (Gebrehiwot, 2003; Jayaraman, 1999). This sampling strategy is known for being efficient when sampling vegetation cover and other natural resources (Aune-Lundberg and Strand, 2014).

According to Jayaraman (1999), the use of systematic random sampling is necessary because of its ability to increase sampling accuracy, a valid test of significance, and keep the random error at the lowest possible level; as well as being highly suitable for sampling large areas, flexible for heterogeneity, and response to changes in a quick way (Brown et al., 2013; Mwansa,2018). Furthermore, this sampling strategy is known for being efficient when sampling vegetation cover and other natural resources (Aune-Lundberg and Strand, 2014).

Trangmar (1985) indicated that the sample size in a study area is based on the objective of the study and the cost of sampling and measurement and the accuracy desired. Time availability is also a major factor, according to Mganga, and Lyaruu, (2016). A sample size of 40-50 is statistically acceptable (Nangendo,2000). For this study, 40 plots were used (20 for burnt areas, and 20 for unburnt areas).

3.2.2) **Sampling design**

Kuru et al., (2015) and Kabajani (2016) justify the use of concentric circular plots as being relatively better than square plots because they reduce the potential of skewing of boundary lines by obstacles. The circular plots were used in this study because they allow the same plot center to be used for all three groups which included: trees, saplings, and seedlings (Gondwe, et al., 2020). Although there are no hard rules for plot sizes, it is precedent to have a large enough size to cover the variation in species within a locality and it should relate to the size of the vegetation under study. Additionally, Kent and Coker (1992) have suggested a range of plot sizes (quadrants) for the different classes of vegetation that may be studied including woodland canopies at 20 m × 20 m and 50 m × 50 m. However, for this study, plot sizes of 0.01 ha (5.65 m radius) for the regenerants, 0.04 ha (11.28 m radius) for poles, and 0.16 ha (22.60 m radius) for the large trees as suggested by Nangendo (2000), were used (Figure 2).

Google Earth was used to locate the specific plots of interest (both burnt and unburnt) in the Chongoni forest reserve. Thereafter, a 200 m grid was measured to demarcate the distance between plots. From there, a cluster of 3 plots of different sizes was measured. Figure 2, shows the plot layout.

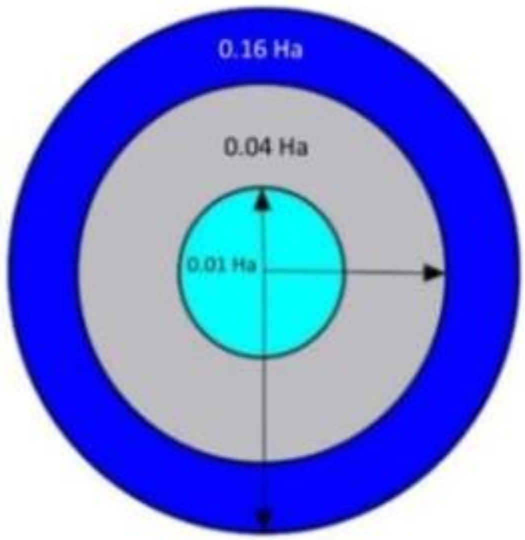


Figure 2: Plot layout

For specific objectives 1, 2, and 3 (as described in chapter 1), primary data were sourced from the field survey which was conducted in Chongoni Forest Reserve. The field survey comprised of a crew of 4 people, the researcher, a taxonomist, and two (2) line cutters. A taxonomist, well versed with tree species, was identified and was responsible for the species identification and measuring of the Diameter at Breast Height (DBH). Two (2) line cutters who are well versed with the area and able to take measurements and record correctly with minimum supervision, were also used in the data collection.

3.2.2 <u>Measurement data</u>

Trees diameter was measured at breast height (at 1.3m). The study also measured Stand Density (SD) and the Basal area for the tree species. Stand density is the quantitative measurement of a stand in terms of a square meter of basal area, the number of trees, or volume per unit area (Husch et al., 1982). The basal area is used to indicate the extent to which the available area in the ground has been utilized (Loetsch and Haller, 1964).

Measurements by plot sizes

- In the plots of 5.65 m radius, stems with less than 4.9 cm dbh were counted
- In the plots of 11.28 m radius, stems with a 5.0-29.9 cm dbh were measured
- In the plots of 22.60 m radius, stems larger than 29.9 cm dbh were measured

For tree species identification, a species identification key was used. The presentation of species name adopted the naming according to a standard format where the first four letters of the genus name and first three letters of the species name (Geldenhuys, 2009).

3.3 DATA ANALYSIS

The data were compiled and organized using R software, and Microsoft Office Excel 2019 package to produce both statistical and descriptive inferences from the data for interpretation.

Diversity and Regeneration measures.

According to Magurran (1998) and Magurran (2004), diversity measurements consist of two components: species richness (the number of species in the area) and abundance (number of individuals for each species). In this study, the researcher used the Shannon-Wiener diversity index (H) to measure the diversity and regeneration in the study sites. The researcher also employed the Shannon (E) evenness which measures the composition and richness of areas to complement the study. Magurran (1998) further indicates that no index has received backing in the field, and there is little consensus on the best diversity measure.

The Shannon-Wiener diversity index (H) was calculated using the following formula as suggested by Magurran (2004).

$$H' = -\sum_{1=i}^{s} p_{i\ln}\ p_i$$ ………………………… Equation 1

Where

S = The number of species

P_i = The proportion of individuals or the abundance of the *i*th species expressed as a proportion of total cover

Ln = Log base n

Considering the species evenness (Shannon E) equality of abundant species, was calculated using the ratio of observed (H'), to maximum diversity H_{max}.

$E = H' / H_{max} = H' / \ln S$

Where,

E = The Shannon evenness index in percentage,

H′ = Shannon′s diversity index, and

H_{max} = The number of total species.

Ln = Log base n

S = The number of species

Tree species composition

The researcher also calculated the Jaccard Similarity Index to define tree species composition similarity among study plots (Jaccard, 1900; Jaccard, 1912; Nyirenda et al., 2019). The index was used to compare shared species among the burnt and unburnt areas and was calculated using the following formula as suggested by Jaccard (1912) and Chao et al., (2005).

$Sj = a/(a + b + c)$ …………………… Equation 2

Where

Sj = Jaccard Similarity Index

(%); a = species count in both areas b = species count in burnt areas

c = species count in unburnt areas.

Species dominance

The study used the Importance Values Index (IVI) to measure species dominance in the forest, this is a summation of the relative values of frequency, density, and dominance of species as suggested by Curtis and McIntosh, (1951) below. According to Gonçalves et al., (2017), this IVI is used to describe the floristic structure and composition of forests, and has been frequently used in the Miombo woodland systems (Kalaba et al., 2013; Jew et al., 2016).

IVI = (RF + RD + RDo) , ……………………………..Equation 3

where

IVI = Importance Value Index;

RF = Relative frequency;

RD = Relative density; and

RDo = Relative dominance.

According to Gonçalves et al., (2017), the relative frequency is a statistical parameter that reflects the spread of a species in a given area; the relative density of a species reflects the abundance of a species in a given community; the relative dominance is the area occupied by the basal area of a species per plot.

Basal area for tree species was calculated using the formula as used by (Wiafe, 2014).

$g = \pi\,(\mathrm{dbh})^2/4$ …………………………. Equation 4

Where

g = basal area in meter squared (m^2)

π = 3.142

dbh = diameter at breast height (m)

Basal area for the study sites (burnt and unburnt plots) was calculated using the following formula (Mwansa, 2018).

$G = \Sigma g / A$ ……………………………… Equation 5

where

G = mean basal area (m^2) per hectare,

Σg= mean basal area (m^2) per individual tree

A = plot area

To calculate relative frequency, density, and dominance of the species, the study used the standard formulas as suggested by Mueller-Dumbois and Ellenberg (1974), Kent and Coker (1992), and Freitas and Magalhaes (2012) below.

Relative frequency

$RF = (AF/TF) \times 100$, …………………………………… Equation 6

Where

RF= Relative frequency of species;

AF= Absolute frequency of the species;

and TF= Sum of absolute frequencies of all species.

Relative density

$RD = (AD/TD) \times 100$, …………………………………… Equation 7

Where

RD = Relative density of species;

AD = Absolute density of species (per ha);

and TD = Total density of trees (per ha).

Relative dominance

$RD_o = (AD_o / TD_o) \times 100$, ………………………………… Equation 8

Where

RD_o = Relative dominance of species;

AD_o = Absolute dominance (or basal area) of species;

and TD_o = Total dominance (or basal area) of all species

To calculate relative frequency, density, and dominance of the species, the study used the standard formulas as suggested by Mueller-Dumbois and Ellenberg (1974), Kent and Coker (1992), and Freitas and Magalhaes (2012) below.

RVI = Relative $Frequency_1$ + Relative $Density_1$, ……Equation 9

Where

RVI= Relative Importance Value Index

Relative $Frequency_1$ = (Number of individuals of a species)/ (Total number of Species) x 100, ……………………………………Equation 10

Relative $Density_1$ = (Number of individuals of a species) / (Total number of individuals) x 100, ………………………………………………. Equation 11

Stand and Diameter Class Distribution

Tree growth and species distribution relate to the stand structure of species within a forest (Gonçalves et al., 2017; Shumi et al., 2020), and researchers have focused on both horizontal and vertical stand structures. For this study, the researcher focused on the horizontal stand for analysis and measured the dbh for tree stems.

CHAPTER FOUR: RESULTS

This chapter is a presentation of the results of the data analysis performed on measurement data collected from burnt and unburnt areas in Chongoni Forest Reserve. The research objectives are used as a basis for analysis and results are presented in tables and figures.

4.1 TREE SPECIES

4.1.1 Trees Composition and structure

The results on species composition and structure show that there are relative similarities between the burnt and unburnt areas. From the results, 30 tree species were recorded in both burnt and unburnt plots. A more detailed analysis is presented in Tables 1 and 2 on tree species in burnt plots and unburnt plots, respectively.

On the observed trees species in the burnt plots, Table 1 shows that 20 species were recorded, where *Brachystegia floribunda* (47.08%) was the most occurring followed by *Uapaca kirkiana* (20.44%) and *Brachystegia spiciformis* (14.59%). In contrast, in the unburnt plots, 26 trees species. *Uapaca kirkiana* (39.99%) was observed to be dominant followed by *Brachystegia floribunda* (31.47%), and *Brachystegia spiciformis* at (8.24%) (Table, 2).

Table 1: Tree species in burnt areas in Chongoni Forest Reserve

Name of Species	Stems Count	Pole Count	Total Species Count	Percentage
Brachystegia boehmii	3	1	4	1.46
Brachystegia floribunda	105	24	129	47.08
Brachystegia spiciformis	33	7	40	14.59
Bridelia micrantha	3	0	3	1.09
Byrsocarpus orientalis	0	1	1	0.36
Cussonia arborea	5	1	6	2.19
Combretum mole	5	2	7	2.55
Combretum triforium	3	0	3	1.09
Curatellifolia parinari	3	0	3	1.09
Faurea cafra	0	2	2	0.72
Faurea saligna	0	3	3	1.09
Faurea rochetiana	0	1	1	0.36
Ficus natalensis	1	0	1	0.36
Phyllus guieensis	1	1	2	0.73
Rhus natalesis	1	1	2	0.73
Syzygium cordatum	8	1	9	3.28
Trema orientalis	2	0	2	0.73
Uapaca kirkiana	54	2	56	20.44
Vangueria infausta	1	1	2	0.73
Ximenia caffra	0	1	1	0.36
Grand Total	**225**	**49**	**274**	**100.00**

Table 2: Tree species in unburnt areas in Chongoni Forest Reserve

Name of Species	Stems	Poles	Total Species	Percentage
Boscia spp	1	1	2	0.59
Brachystegia boehmii	3	1	4	1.18
Brachystegia floribunda	104	3	107	31.47
Brachystegia spiciformis	20	8	28	8.24
Bridelia micrantha	6	3	9	2.65
Byrsocarpus orientalis	1	3	4	1.18
Cussonia arborea	3	1	4	1.17
Combretum molle	2	0	2	0.59
Combretum triforium	1	0	1	0.29
Curatellifolia parinari	5	1	6	1.76
Faurea saligna	1	0	1	0.29
Ficus cronata	4	0	4	1.18
Ficus natalensis	6	0	6	1.76
Ficus sycomorus	1	0	1	0.29
Flacourtia Indica	1	0	1	0.29
Khaya anthotheca	2	0	2	0.59
Lannea discolor	1	1	2	0.59
Motes Africanus	1	6	7	2.06
Multidentia crass	0	3	3	0.88
Psorospermum febrifugum	1	0	1	0.29
Sapium ellipticum	0	4	4	1.18
Syzygium cordatum	99	0	9	2.64
Uapaca kirkiana	108	19	127	39.99
Vangueria infausta	3	0	3	0.88
Ximenia caffra	1	1	2	0.59
Grand Total	**285**	**55**	**340**	**100.00**

4.1.2 Trees Species Diversity and Richness

Importance Value Index (IVI) for tree species in burnt areas in Chongoni Forest Reserve

Table 3 presents the results on tree species dominance indices in the burnt areas of the forest reserve including tree species relative frequency, tree species relative density, tree species relative dominance and Importance Value Index (IVI). .

Species relative density: The results (Table, 3) indicate that *Brachystegia floribunda* had relative density of (16.99%) and was the most abundant followed by *Uapaca kirkiana*, with relative density of (8.73%), *Brachystegia spiciformis* (4.85%), and *Brachystegia boehmii* (1.33%). The other species had relative densities below (1%) and had low relative density. The total relative density of the study area is 37.094.

Frequency distribution: The results (Table, 3) show that the frequency distribution of all tree species encountered in the burnt areas is varied and dependent on the type of tree species. Thus, there are some, which had registered a high frequency, while others registered a low frequency. From the result, the total frequency for all the tree species encountered in the study site is 228. *Brachystegia floribunda* had 69, and was the most abundant followed by *Uapaca kirkiana* at 48. Most other species had a frequency distribution of less than 10.

Species dominance: The results (Table, 3) show that for the burnt areas, *Brachystegia floribunda* had the Importance Value Index of 94.57, and is the most dominant species. This is followed by *Uapaca kirkiana* at 39.44. Species such as *Bridelia micrantha* (7.26), *Cussonia arborea* (7.06), and *Brachystegia boehmii* (5.73). The other species had low IVI.

Table 3: Tree species Dominance indices in burnt areas in Chongoni Forest Reserve

Tree species	Frequency	Relative Frequency	Relative Density	Relative Dominance	IVI
Brachystegia boehmii	7	2.899	1.333	1.500	5.732
Brachystegia floribunda	69	28.985	16.991	48.596	94.572
Brachystegia spiciformis	37	15.942	4.854	17.059	37.856
Bridelia micrantha	10	4.348	0.485	2.432	7.265
Curatellifolia parinari	3	1.449	0.162	0.345	1.956
Cussonia arborea	10	4.348	0.809	1.899	7.056
Combretum molle	9	4.348	0.809	5.224	10.381
Combretum triforium	3	1.449	0.485	2.150	4.084
Curatellifolia parinari	7	2.899	0.324	1.126	4.348
Ficus natalensis	3	1.449	0.162	0.470	2.082
phyllus guieensis	3	1.449	0.162	0.078	1.689
Syzygium cordatum	9	4.347	1.294	8.406	14.048
Trema orientalis	7	2.899	0.324	0.114	3.336
Uapaca kirkiana	48	20.290	8.738	10.417	39.445
Vangueria infausta	3	1.449	0.162	0.182	1.793
Totals	228		37.094		

Importance Value Index for tree species in unburnt areas in Chongoni forest reserve

Table 4 shows the results of the tree species used to estimate the Importance Value Index (IVI) - Species dominance in unburnt areas.

Species relative density: The results (Table, 4) indicate the density of each tree species encountered in the study site. *Uapaca kirkiana*, had relative density of 37.89% and was the most abundant. It was closely followed by *Brachystegia floribunda*, with relative density of (36.49%). *Brachystegia spiciformis* (7.02%), *Syzygium cordatum* (2.80%), *Bridelia micrantha* (2.11%), *Curatellifolia parinari* (1.40%), and *Brachystegia boehmii*

(1.05%). The other species had relative density below (1%) and had low relative density. The total relative density of the study area is 90.88.

Frequency distribution: Similarly, to the burnt areas the results in Table 4 show that the frequency distribution of all species encountered in the unburnt areas is varied and also dependent on the tree species where some had registered a high frequency while others had low frequency. The total frequency for all the tree species encountered in the study site is 217. *Uapaca kirkiana* was observed to be 64 and the most abundant followed by *Brachystegia floribunda* at 48, and *Brachystegia spiciformis* at 37. Most other species had a frequency distribution of less than 10.

Species dominance: The results (Table, 4) show that for the unburnt areas, *Brachystegia floribunda* had the Importance Vale Index of 105.00, and is the most dominant species. This is followed by *Uapaca kirkiana* at 86.88. The other species such as *Brachystegia spiciformis* (34.00), *Syzygium cordatum* (14.155), and *Bridelia micrantha* (9.64) are seen to have low IVI.

Table 4: Tree species dominance indices for unburnt areas in Chongoni Forest Reserve

Name of species	Frequency	Relative dominance	Relative frequency	Relative density	IVI
Boscia spp	2	1.006	1.075	0.351	2.432
Brachystegia boehmii	8	1.433	3.226	1.053	5.712
Brachystegia floribunda	48	50.230	18.280	36.491	105.000
Brachystegia spiciformis	37	13.012	13.978	7.018	34.008
Bridelia micrantha	17	1.084	6.452	2.105	9.640
Byrsocarpus orientalis	2	0.030	1.075	0.351	1.456
Cussonia arborea	5	0.814	2.151	0.702	3.666
Curatellifolia parinari	11	1.439	4.301	1.404	7.143
Ficus coronata	5	0.104	2.151	0.702	2.956
Syzygium cordatum	10	7.087	4.301	2.807	14.155
Uapaca kirkiana	64	23.762	24.731	37.894	86.88
Totals	217			90.878	

Stand and Diameter Class Distribution

With regards to the tree species dbh, Table 5 shows that the mean dbh for all tree stems in burnt plots was 24.208 cm, and the maximum mean dbh was found to be for *Syzygium cordatum* at 39.50 cm. For the unburnt plots, the mean dbh for all stems in unburnt plots was 23.09 cm, and the maximum mean dbh was found to be for *Syzygium cordatum* at 53.30 cm.

Table 5: DBH of tree species in burnt and unburnt plots in Chongoni Forest Reserve

Name of species	Number of Stems	Mean DBH
Burnt plots	**225**	**24.208**
Brachystegia boehmii	3	27.800
Brachystegia floribunda	105	26.294
Brachystegia spiciformis	30	26.350
Bridelia micrantha	3	34.167
Curatellifolia parinaric	1	23.700
Cussonia arborea	5	23.600
Combretum molle	5	28.540
Combretum triforium	3	33.567
Curatellifolia parinaric	2	21.500
Ficus natalensis	1	27.500
Phyllus guieensis	1	11.800
Rhus natalensis	1	11.300
Syzygium cordatum	8	39.500
Trema orientalis	2	10.800
Uapaca kirkiana	54	16.179
Vangueria infausta	1	17.500
Unburnt plots	**125**	**23.091**
Brachystegia boehmii	1	34.500
Brachystegia floribunda	53	26.333
Brachystegia spiciformis	10	32.300
Bridelia micrantha	1	30.000
Byrsocarpus orientalis	1	5.000

Cussonia arborea	1	26.000
Curatellifolia parinaric	4	13.800
Ficus coronate	2	4.650
Syzygium cordatum	2	53.300
Uapaca kirkiana	45	18.020
Grand Total	**350**	

It is also noted that the diameter class distribution pattern in the burnt and unburnt plots shows that there are more species in the smaller diameter class as compared to the larger diameter classes (Figures 3 and 4). Both the burnt and unburnt plots show an inverted J-shape distribution pattern. However, relatively lower numbers are observed in the burnt plots for the 7.5 cm to 17.5 cm diameter class than those observed in the unburnt plots.

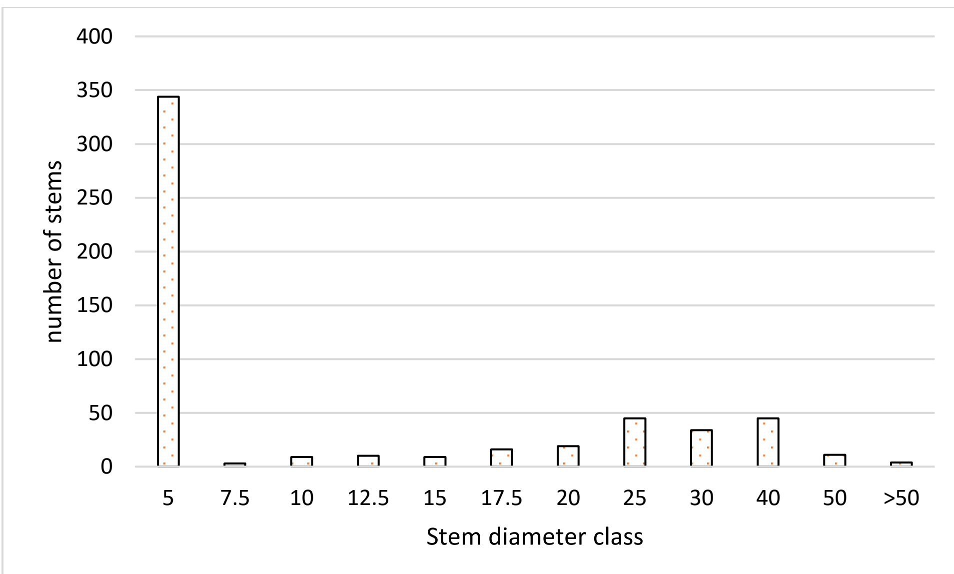


Figure 3: Stand stem diameter distribution in burnt plots

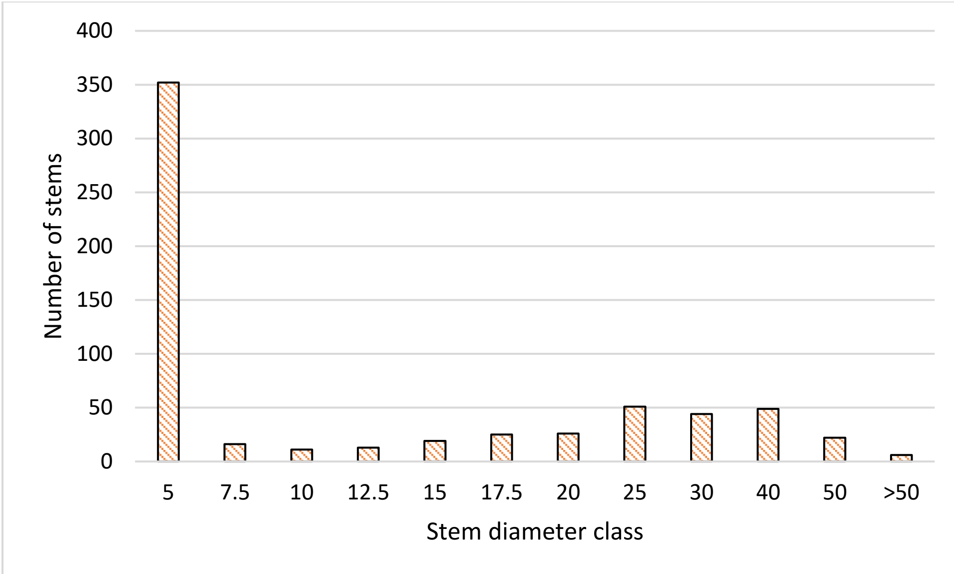
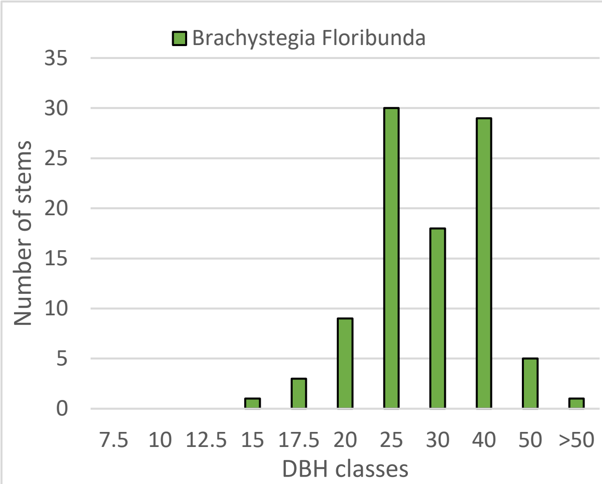
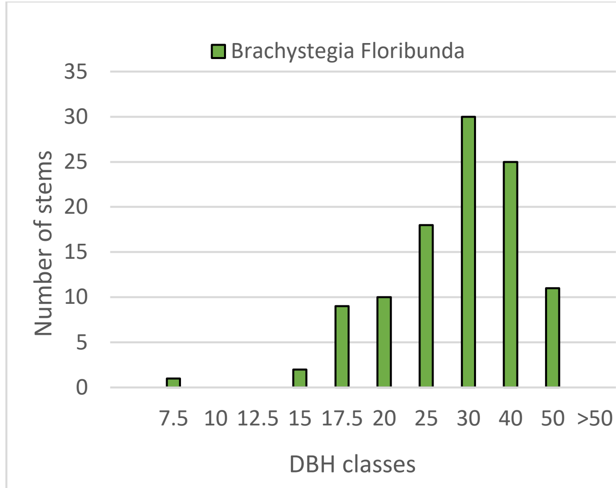


Figure 4: Stand Stem diameter distribution in unburnt plots

Further analyses of the key species diameter size class structure from both the burnt and unburnt plots are shown in Figures 5 through 14. These figures show the comparison of size classes between 5 key species as identified by the study using the IVI values calculated for both the burnt and unburnt plots and include: *Brachystegia floribunda*, *Brachystegia spiciformis*, *Bridelia micrantha*, *Syzygium cordutum* and *Uapaca kirkiana*.

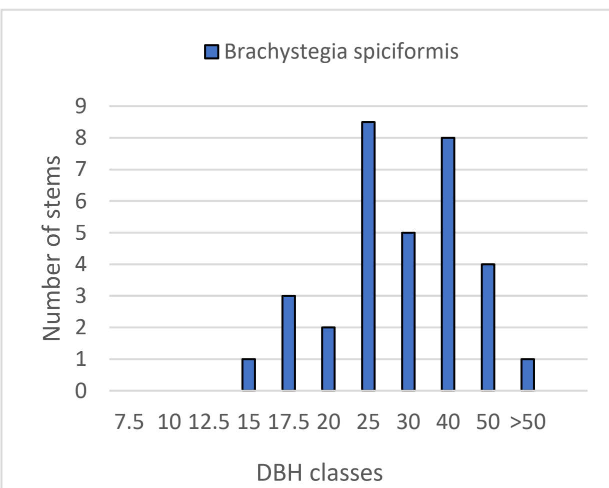
For *Brachystegia floribunda* it is observed that both the burnt and unburnt plots indicate a bell-shaped distribution pattern (Figures 5 and 6). That is to say, the frequency of stems increases in the mid-dbh values and decreases on the lower and higher dbh values. However, for the unburnt plots it is also observed that there is a slightly higher frequency in the lower size classes (<25 cm dbh) than in the burnt plots.

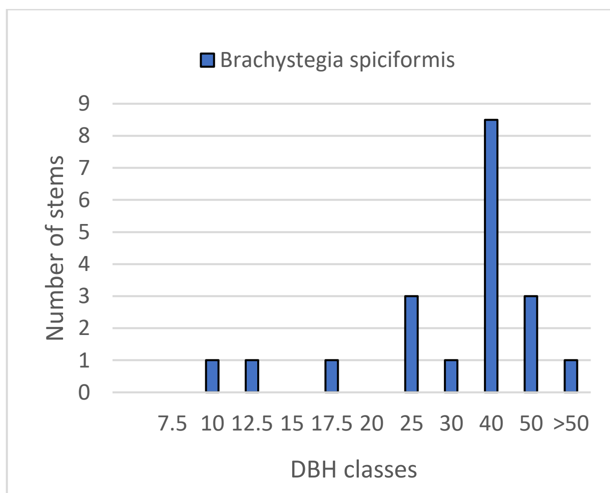




Figures 5 and 6: *Brachystegia floribunda* size classes in burnt (left) and unburnt (right) plots

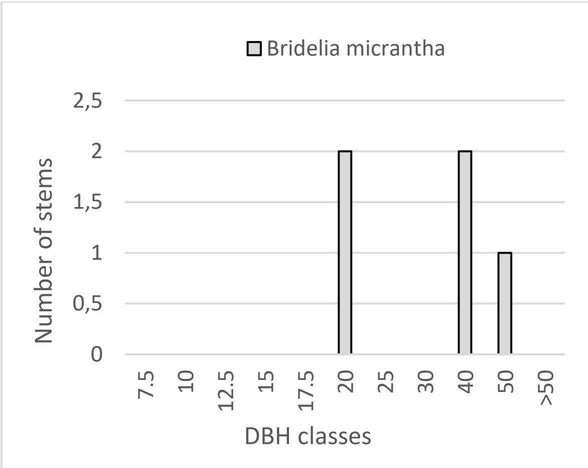
A similar pattern is observed for *Brachystegia spiciformis*, in which the frequency of stems in the mid-dbh size classes increases and decreases in the lower and higher size classes for both burnt and unburnt plots (Figure 7 and 8). Noteworthy, a slightly higher frequency of *Brachystegia spiciformis* is observed for the lower size classes in burnt plots than in burnt plots. Overall, the frequency for all the size classes in burnt plots is relatively higher than those observed in unburnt plots.

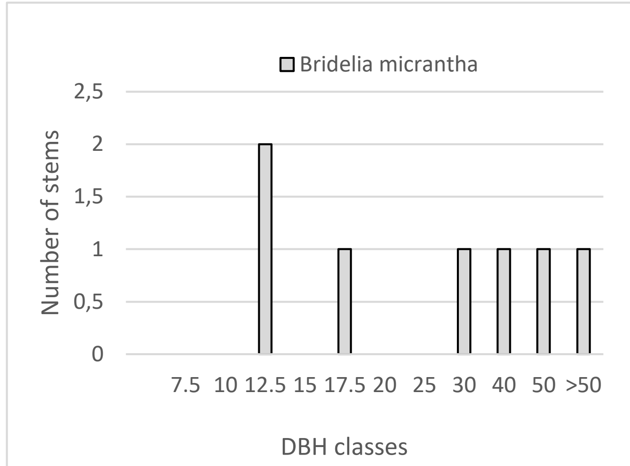




Figures 7 and 8*: Brachystegia spiciformis* size classes in burnt (left) and unburnt (right) plots

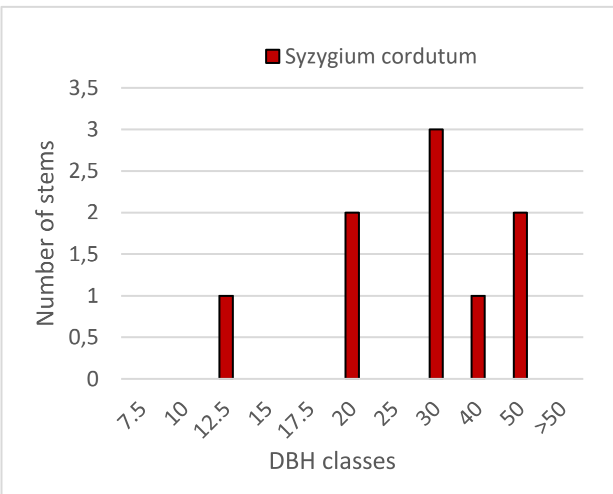
For *Bridelia micrantha,* stem distribution frequency is observed to be higher in all size classes in the unburnt plots than in burnt plots (Figures 9 and 10).

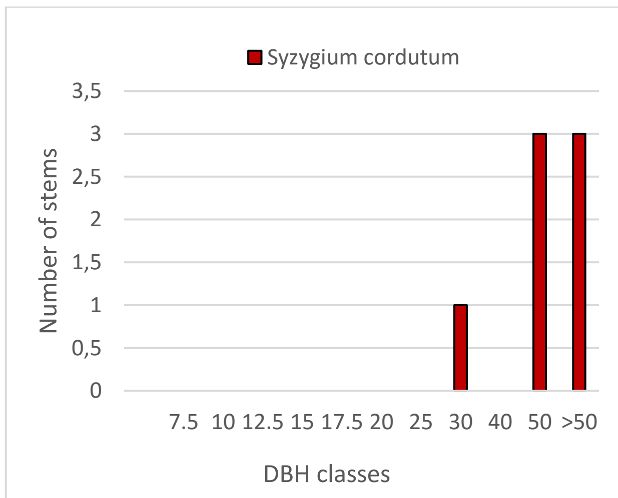




Figures 9 and 10*: Bridelia micrantha* size classes in burnt (left) and unburnt (right) plots

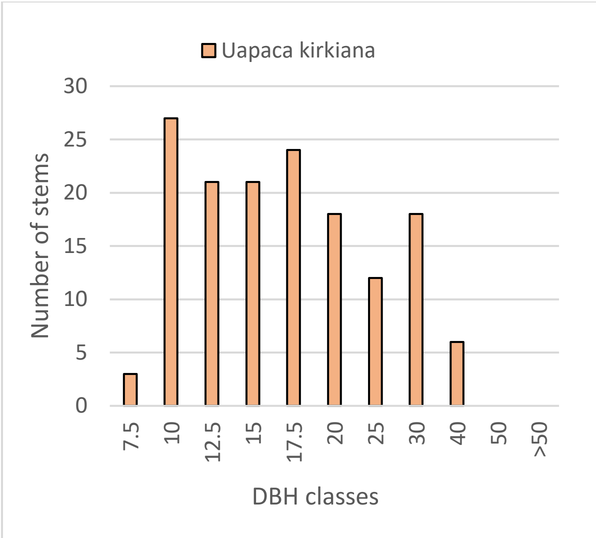
In the burnt plots, *Syzygium cordutum* frequency in size classes lower than 50 cm dbh is observed to be higher than in unburnt plots (Figures 11 and 12).

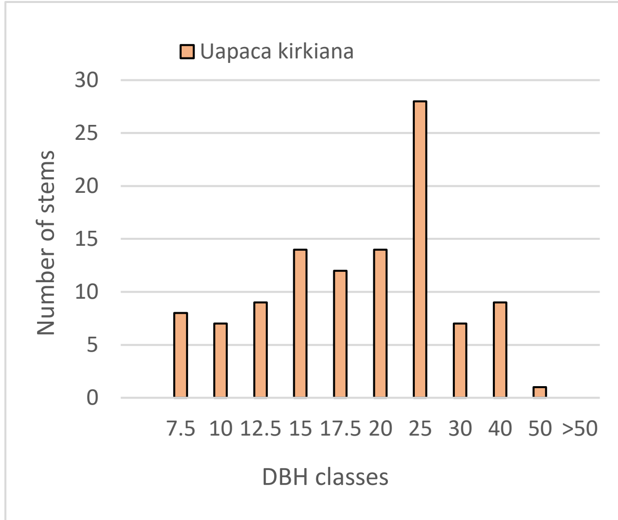




Figures 11 and 12: *Syzygium cordatum* size classes in burnt (left) and unburnt (right) plots

With respect to Uapaca *kirkiana*, medium trees (7.5 cm - 29.9 cm dbh) frequency is observed to be higher in burnt plots than in the unburnt plots (Figures 13 and 14).





Figures 13 and 14: *Uapaca kirkiana* size classes in burnt (left) and unburnt (right) plots

Tree diversity and richness

Table 6 shows the results of tree species diversity indices for the burnt and unburnt plots. The calculated values for Shannon-wiener and Evenness for unburnt plots were -0.81 and - 0.24 respectively, whereas for the burnt plots the Shannon-wiener and evenness indices were -1.04 and -0.34, respectively. This clearly indicates that there is diversity among tree species in both burnt and unburnt plots. However, there is no significant difference between the plots. As regards species richness, the unburnt plots had a richness value of 29, whereas the burnt plots had 21.

Table 6: Various indices for tree species in burnt and unburnt plots

Diversity Indices	Unburnt plots	Burnt plots
Shannon-wiener	-0.81	-1.04
Evenness	-0.24	-0.34
Richness	29	21

Similarity index

A further analysis on species composition established that the species Jaccard Similarity Index was 59.49%. Thus, the study found that the number of species common to burnt and unburnt plots was 22, number of species unique to burnt plots was 5, and the number of species unique to unburnt plots was 10. This is an indication that more than half of the number of species is common to both areas.

4.2 NATURAL REGENERATION

4.2.1 Natural regeneration composition

Tables 7 and 8 show results of the analysis on regenerants composition in burnt and unburnt plots, respectively. As presented in the tables, 30 regenerants species are recorded, where 24 were recorded in burnt plots and 26 in unburnt plots.

For the regenerants in the burnt plots 24, it was established that the most occurring species is *Brachystegia floribunda* (27.9%), followed by *Uapaca kirkiana* (20.06) and *Cussonia arborea* (9.01%) (Table, 7). Whereas, in the unburnt plots (Table, 8) *Brachystegia spiciformis* (35.79%) was the most occurring, followed by *Uapaca kirkiana* (23.03%) and *Brachystegia floribunda* (8.01%).

Table 7: Natural regeneration in burnt areas in Chongoni Forest Reserve

Name of Species	Number of Seedlings	Number of Saplings	Total number of Species	Percentage
Brachystegia boehmii	5	0	5	1.45
Brachystegia floribunda	88	88	96	27.9
Brachystegia spiciformis	23	4	27	7.84
Bridelia micrantha	4	0	4	1.16
Byrsocarpus orientalis	2	0	2	0.58
Curatellifolia parinari	3	1	4	1.16
Cussonia arborea	17	14	31	9.01
Combretum triforium	4	0	4	1.16
Faurea cafra	9	1	10	2.91
Faurea saligna	2	1	3	0.87
Faurea rochetiana	1	0	1	0.29
Ficus natalensis	1	2	3	0.87
Lannea discolour	1	1	2	0.58
Multidentia crass	7	0	7	2.03
Phyllus guieensis	2	1	3	0.87
Psidium guajava	1	0	1	0.29
Rhus natalensis	10	0	10	2.91
Sapium ellipticum	16	0	16	4.65
Syzygium cordatum	12	0	12	3.49
Trema orientalis	9	0	9	2.62
Uapaca kirkiana	65	4	69	20.06
Vangueria infausta	0	2	2	0.58
Ximenia caffra	6	2	8	2.33
Ziziphus mucronata	22	0	22	6.40
Grand Total	**304**	**40**	**344**	**100.00**

Table 8: Natural regeneration in unburnt areas in Chongoni Forest Reserve

Name of species	Number of Seedlings	Number of Saplings	Total Number of Species	Percentage
Brachystegia boehmii	37	1	38	4.76
Brachystegia floribunda	50	14	64	8.01
Brachystegia spiciformis	266	20	286	35.79
Bridelia micrantha	26	2	28	3.50
Byrsocarpus orientalis	7	1	8	1.00
Cussonia arborea	6	2	8	1.00
Combretum cordatum	2	0	2	0.25
Combretum molle	2	0	2	0.25
Combretum triforium	1	2	3	0.38
Curatellifolia parinari	16	8	24	3.00
Faurea Cafra	26	2	28	3.51
Faurea saligna	8	1	9	1.13
Faurea rochetiana	1	0	1	0.13
Ficus cronata	6	0	6	0.75
Lannea discolour	1	4	5	0.63
Motes Africanus	3	0	3	0.38
Multidentia crass	0	1	1	0.13
Pseudolachnostylis maprouneifolia	1	0	1	0.13
Psorospermum febrifugum	4	4	8	1.00
Rhus natalensis	8	8	16	2.005
Sapium ellipticum	42	1	43	5.38
Syzygium cordatum	0	11	11	1.38
Trema orientalis	4	0	4	0.50
Uapaca kirkiana	168	16	184	23.03
Vangueria infausta	14	0	14	1.75
Ximenia caffra	1	1	2	0.25
Grand Total	**700**	**99**	**799**	**100.00**

4.2.2 Diversity and Richness of Natural Regeneration

Dominance indices

Relative Importance Value Index (RVI) in burnt areas in Chongoni Forest Reserve

Regarding the regenerants species dominance in burnt areas, Table 9 shows the results of the regenerants as used to estimate their Relative Importance Value Index (RVI).

Species relative density: The results in Table 9 indicate the density of each regenerants species encountered in the burnt areas. From the analysis, it is established that *Brachystegia Floribunda*, had a relative density of (27.90%) and was the most abundant, which is followed by *Uapaca kirkiana*, with a relative density of (20.06%), *Cussonia arborea* (9.01%), *Ziziphus mucronata* (6.40%), *Sapium ellipticum* (4.65%). The other species had relative density below (4%) and indicated low relative density. The total relative density of the study area is 100.00.

Frequency distribution: Similar to the tree species, the results (Table, 9) show that the frequency distribution of all regenerants encountered in the burnt areas is also varied; there are some which had high frequency while others had low frequency. The total frequency for all the regenerants encountered in the burnt study sites is 344. According to the results, *Brachystegia floribunda* had a frequency distribution of 96 and was the most abundant followed by *Uapaca kirkiana* at 69. Most other species had a frequency distribution of less than 10.

Species dominance: The results (Table, 9) show that for the burnt areas, *Brachystegia floribunda* had the Relative Importance Value Index of 358.94, and is the most dominant species. This is followed by *Uapaca kirkiana* at 257.99. The lowest RVI were observed for regenerants of *Lannea discolor* (7.48), *Vangueria infausta* (7.48), Byrsocarpus *orientalis* (7.48), and *Psidium guajava* (3.74).

Table 9: Dominance indices for burnt areas in Chongoni Forest Reserve

Name of species	Frequency	Relative Frequency	Relative Density	RVI
Brachystegia boehmii	5	17.241	1.453	18.695
Brachystegia floribunda	96	331.034	27.906	358.942
Brachystegia spiciformis	20	68.966	5.814	74.780
Bridelia micrantha	4	13.793	1.163	14.956
Byrsocarpus orientalis	2	6.897	0.581	7.478
Curatellifolia parinari	4	13.793	1.163	14.956
Cussonia arborea	31	106.897	9.012	115.908
Combretum triforium	4	13.793	1.163	14.956
Faurea cafra	10	34.482	2.907	37.390
Faurea saligna	4	13.793	1.163	14.956
Ficus natalensis	3	10.345	0.872	11.217
Lannoa discolor	2	6.897	0.581	7.478
Multidentia crass	7	24.138	2.035	26.173
Phyllus guieensis	3	10.345	0.872	11.217
Psidium guajava	1	3.448	0.291	3.739
Rhus natalesis	10	34.483	2.907	37.390
Sapium ellipticum	16	55.172	4.651	59.824
Syzygium cordatum	12	41.379	3.489	44.868
Trema orientalis	9	31.034	2.616	33.651
Uapaca kirkiana	69	237.931	20.058	257.989
Vangueria infausta	2	6.897	0.581	7.478
Ximenia caffra	8	27.586	2.326	29.912
Ziziphus mucronata	22	75.862	6.395	82.257
Totals	344		100	

Relative Importance Value Index (RVI) in unburnt areas in Chongoni Forest Reserve

Table 10 shows the results of the regenerants species used to estimate the Relative Importance Value Index (RVI) to determine species dominance in unburnt areas.

Species relative density: The results (Table, 10) indicate the density of each tree species encountered in the unburnt areas. *Brachystegia spiciformis* had relative density of (64.21%) and was the most abundant, and followed by *Uapaca kirkian*a with a relative density of (9.94%), *Brachystegia boehmii* (5.40%), *Faurea cafra* (4.83%). The other species had relative density below (4%) and had low relative density. The total relative density of the study area is 100.00.

Frequency distribution: The results (Table, 10) show that the frequency distribution of all regenerants species encountered in the unburnt areas is varied; there are some which had high frequency while others had low frequency. The total frequency for all the regenerant species encountered in the study site is 352. The most abundant was established to be *Brachystegia spiciformis* which a frequency distribution of 226, followed by *Uapaca kirkiana* at 35. Most other species had a frequency distribution of less than 10.

Species dominance: The results (Table, 10) show that for the unburnt areas, *Brachystegia spiciformis* had the Relative Importance Value Index of 1319.76, and is the most dominant species. This is followed by *Uapaca kirkiana* at 204.39. The lowest RVI was observed for species of *Bridelia micrantha* (11.68), *Combretum molle* (11.68), *Sapium ellipticum* (11.68), *Byrsocarpus orientalis* (5.84), *Faurea saligna* (5.84), *Pseudolachnostylis maprouneifolia* (5.84), and *Rhus natalesis* (5.84).

Table 10: Dominance indices of natural regeneration for unburnt areas in Chongoni Forest Reserve

Name of species	Frequency	Relative Frequency	Relative Density	RVI
Brachystegia boehmii	19	105.556	5.398	110.953
Brachystegia Floribunda	19	105.556	5.398	110.953
Brachystegia spiciformis	226	1255.556	64.205	1319.760
Bridelia micrantha	2	11.111	0.568	11.679
Byrsocarpus orientalis	1	5.556	0.284	5.840
Cussonia arborea	6	33.333	1.705	35.038
Combretum mole	2	11.111	0.568	11.679
Curatellifolia parinari	4	22.222	1.136	23.359
Faurea cafra	17	94.444	4.830	99.274
Faurea saligna	7	38.889	1.989	40.878
Faurea rochetiana	1	5.556	0.284	5.840
Ficus coronata	6	33.333	1.705	35.038
Pseudolachnostylis maprouneifolia	1	5.556	0.284	5.840
Rhus natalesis	1	5.556	0.284	5.840
Sapium ellipticum	2	11.111	0.568	11.679
Uapaca kirkiana	35	194.444	9.943	204.388
Vangueria infausta	3	16.667	0.852	17.519
Totals	352		100.00	

Regenerants diversity and richness

With regards to regenerants diversity, seedlings and sapling in unburnt plots had a slightly higher diversity having a Shannon-wiener value of -1.58, and -0.43 compared to -1.54, and -0.32 in the burnt plots (Table 11). Similarly, the same is observed for evenness values which are slightly higher in the unburnt plots than in the burnt plots. From the results, it is also seen that higher values of richness are found in the unburnt plots, having saplings richness at 20 compared to 14 in the burnt plots. Seedlings richness for unburnt plots was observed to be 29, whereas in burnt plots the seedling richness was 26. Thus, there is diversity among both regenerants in the unburnt and burnt plots; however, it is the unburnt plots that show more diversity than the burnt plots. More individuals are also observed to be in the unburnt plots, as according to the richness values.

Table 11: Various indices for natural regeneration in burnt and unburnt plots

	Unburnt plots		**Burnt plots**	
Diversity indices	**Saplings**	**Seedlings**	**Saplings**	**Seedlings**
Shannon-wiener	-0.43	-1.58	-0.32	-1.54
Evenness	-0.14	-0.47	-0.12	-0.47
Richness	20	29	14	26

CHAPTER FIVE: DISCUSSION

This chapter presents a discussion of the results under the different objectives. Thus, the discussion is on the composition of the stand, species diversity and richness and regeneration, respectively.

5.1 STAND AND SPECIES COMPOSITION, STRUCTURE AND DISTRIBUTION

According to the analysis of species stand and structure in the study, it is established that the tree stands in the forest are differentiated in both burnt and unburnt plots. However, as has been revealed, both the burnt and unburnt plots show closer relationships in their characteristics of species stand and composition. From the results, 37 species are found to occur in both the burnt and unburnt plots. Additionally, from the analysis is established that the most occurring or dominant tree species for both the burnt and unburnt plots were *Brachystegia floribunda* (47.08%) and *Uapaca kirkiana* (20.44%), and *Uapaca kirkiana* (39.99%) and *Brachystegia floribunda* (31.47%) respectively. This agrees with findings by Gondwe, (2011) and Missanjo et al., (2014) who report that the dominant tree species in the reserve are *Brachystegia floribunda* and *Uapaca kirkiana*.

In both burnt and unburnt plots, it has been established that there are higher frequencies in stem counts for lower dbh classes (<5 cm dbh) than for higher dbh classes (>5 cm dbh); notably so, these frequencies gradually decrease with increasing dbh size classes but increase slightly at >30cm dbh size classes. However, here it cannot be said as to what the influence of fire is as both the burn and unburnt plots are showing relatively similar size classes. A further analysis of key species has shown that the number of lower dbh size classes are higher in burnt plots than in unburnt plots. The opposite is observed for higher dbh classes, where the frequency of stems size classes is slightly higher in unburnt plots than in burnt plots. Additionally, in the burnt plots, the minimum mean dbh mean was shown to be from *Phyllus guieensis* (10.80 cm), whereas the maximum mean dbh was shown to be from *Syzygium cordutum* (39.50). Similarly, in the unburnt plots *Ficus coronate* (4.65 cm) showed the minimum mean dbh while as *Syzygium cordutum* (53.30 cm) showed a maximum mean dbh. This demonstrates the dominance of *Syzygium cordutum* in the reserve. This domination of the species in the two sites can be attributed to its fast-growing nature, as well as its succession and regenerative pattern, even after the occurrence of fire. Further analysis, however, demonstrates that there is an almost consistent mean dbh among most of the encountered species in burnt plots. For instance, *Brachystegia boehmii* (27.80 cm), *Brachystegia floribunda* (26.29 cm) and *Brachystegia spiciformis* (26.35 cm) have closely consistent mean dbh values. This suggests that the tree species can withstand fire disturbance this finding agrees with results reported by Mwansa, (2018) who reported having found no statistically significant difference among tree species in burnt sites with different burn intensities. Bowers (2017) indicates that for miombo woodlands, stem growth and mortality do not occur wholly at random, and evidence has suggested that stem damage and stem size alter tree population dynamics. Additionally, the stem diameter distribution has also revealed that the burnt plots have a lower frequency of trees above the 30 cm dbh threshold. As much as this cannot be explained by the influence of fire, other anthropogenic activities such as illegal felling of the trees in the reserve may be the contributing factor. However, the analysis found that *Uapaca kirkiana* (Figure, 15) showed a higher frequency in the lower size classes (<30 cm dbh). This also agrees with findings by Chidumayo (1997b) and Gondwe (2011) who highlighted the dominance of *Uapaca kirkiana* and its ability to colonize disturbed areas, due to its thick bark which allows it to survive fires. Chidumayo (1992) also noted that anthropogenic disturbances, including forest fires, affect tree stands in miombo woodlands.

The IVI for tree species in both burnt and unburnt plots shows that the most dominant species is *Brachystegia floribunda*, followed by *Uapaca kirkiana* (Tables, 5 and 6). According to Gondwe (2011), the common canopy species in the reserve are *Brachystegia floribunda* and *Faurea saligna,* whereas the dominant sub-canopy species is *Uapaca kirkiana*. The RVI for regenerants shows that, in the burnt plots, the most dominant was *Brachystegia floribunda,* followed by *Uapaca kirkiana* (Table 9), whereas for the unburnt plots, the most dominant was *Brachystegia spiciformis* followed by *Uapaca kirkiana* (Table,8). Furthermore, in both burnt and unburnt plots, it is appreciated that some tree species have low IVI values, which indicates that the majority of these species are rare in the forest (Kacholi, 2014). According to Chidumayo (1988), there is a difference between young or regenerating miombo which has a high floristic diversity compared with mature miombo. Since DBH measurements were taken >5cm, it is indicative that canopy species that have corresponding vertical stand height of up to 12 m out-compete most understory species for water and nutrients (Lowore, 1997b). Furthermore, other studies including Kikula (1986) attribute canopy closure due to species dominance, as well as in terms of basal area to reduce the prevalence of other species in miombo woodlands.

From these findings, it can be inferred that the occurrence of fire in some areas of the forest, may not necessarily contribute to the differences in species dominance as well as alterations in the composition (Ryan and Williams, 2011). However, focusing on the regenerants, the presence of lower RVI values in the burnt plots, it can be inferred that fire had stimulated dormant seeds to generate. But, this is so because the majority of the regenerants in the burnt plots have higher densities of regeneration capacities.

5.2 DIVERSITY AND RICHNESS

The study has shown that there is high diversity in the species available in Chongoni forest. The study has established about 37 miombo species in the forest. This finding is much larger than the one reported by Gondwe (2011) and Missanjo et al., (2014) who reported 26 species between 1992 and 2008, and 22 tree species respectively. According to Misanjo et al., (2014), the reduced diversity of miombo species was a result of anthropogenic activities and poor management practices. However, from the results, it can be implied that the diversity in the reserve has increased due to the addition of other tree species. This is, however, still small compared to other miombo woodland forests, which have recorded above 80 species (Muvengwi et al., 2020).

The study has also shown some variation in terms of diversity among burnt and unburnt plots in the forest. Regarding the richness of species, both burnt and unburnt have 29 and 31 respectively. Further analysis on mature trees, shows a count of 21 species for burnt areas and 29 species count for unburnt areas. The Jaccard similarity index was 59.49%. This is indicative that there is indeed some variation in species found in the two fire regime areas. Chidumayo (1988), indicates the existence of fire-hardy and resistant miombo species: some miombo woodland species may survive fires of medium intensity (Lawton, 1998). In this case, *Uapaca kirkiana* is a fire-tolerant species hence its dominance in the burnt plots. *Brachystegia spiciformis* which is considered to be fire tender and associated with high mortality by fire occurrence is also seen to have a higher frequency in the unburnt plots than the burnt plots. According to (Luoga et al., 2002; Luoga et al., 2004), over time, the dominance of fire-hardy species reduces while the other key miombo species increases in dominance and abundance. This can be seen from the unburnt plots, which also record a diversity of species. Thus, this phenomenon may also be contributing to the observed differences in the reserve. The evenness index shows differences in the abundance of species, with 0.65 and 0.69 for the burnt and unburnt plots respectively. Thus, it is indicative that the species abundance is also not very differentiated between the burnt and unburnt plots.

5.3 NATURAL REGENERATION

Concerning species regeneration, the study established that the seedlings Shannon diversity values were -1.58 and -1.54 for unburnt and burnt plots respectively. This finding indicates that both unburnt and burnt plots are almost similar in their diversity, with unburnt plots being slightly more diverse. Additionally, the study established that unburnt plots have species richness slightly higher than burnt plots at 29 and 26 respectively. Most importantly, as indicated above, the availability of high frequencies of seedlings and saplings in both areas is an indication of the forest's regeneration potential. Noteworthy, as evidenced in the results, there is a higher frequency in seedlings in key species of *Brachystegia floribunda* and *Uapaca kirkiana* in burnt plots than in unburnt plots. Thus, it can be implied that the disturbance of fire occurrence, has resulted in the increased numbers of seedlings of some species in burnt areas.

According to other studies on miombo tree species, the regeneration of miombo woodlands has been established to occur largely by coppice regrowth and root suckers than through seeds (Trapnell, 1959; Robertson, 1984; and Chidumayo, 1992). According to Bognounou et al., (2010), Immaculada and Yadvinder, (2016), and Teketay et al., (2018), sexual regeneration (by seeds), is considered the primary forest regeneration method, however for miombo tree seedlings, this is dependent on the seasonality of precipitation, and abiotic factors such as fire (Ribeiro et al., (2017). Additionally, other studies have established that seedling establishment is not common among tree species in miombo woodlands due to early and late fire (Mwabumba et al., 1999), sporadic fruit production (Chidumayo, 1997), high level of pre-dispersal seed predation (Chidumayo, 1992; Chidumayo, 1997). Thus, it can be hypothesized that in the reserve the relatively higher frequencies of regenerants in the unburnt plots, are because of reduced disturbance from fire.

With regards to the species diversity values, the Shannon-wiener index values for both burnt and unburnt plots indicate an almost similar higher diversity for seedlings in both plots, which is closely followed by trees and saplings which also show an almost similar diversity. The findings reported by Ryan and Williams (2011) and Suspence et al., (2016) agree with this study's findings and they report that there were no clear effects on miombo species susceptibility to fire among trees and saplings. Suffice to mention that from the study's findings, the similarity in species diversity and richness may be a result of the fire characteristics in the forest (i.e. size, frequency, duration, and intensity). According to Godbless et al., (2019), miombo species regeneration is seen to be effective where fire has not occurred in an area for more than 2 years. This allows seedlings to mature to a certain size that may be resistant to fire effects. In fire frequent areas, seedlings must first establish and grow to fire-tolerant sizes within the short period between burns to allow successful species sexual regeneration (Hoffman, 1998).

CHAPTER SIX: CONCLUSIONS

The study was motivated to establish the effects of fire on tree species diversity and natural regeneration in miombo woodland forest using Chongoni forest reserve as a case. Studies on diversity and regeneration are an important assessment on the status of forests and inform decision-making. Forest management decisions can be made based on these findings to properly plan conservation programs.

With respect to the effect of forest fire on tree species stand, composition, and structure, it is established that there is no significant difference at stand level. However, there was a closer relationship at the individual species level concerning composition and structure between the burnt and unburnt plots. The occurrence of fire in the reserve seem not to have significantly affected or altered the species composition. This was demonstrated by the IVI values of *Brachystegia floribunda* and *Uapaca kirkiana*, which were consistently the dominant species in both burnt and unburnt plots. This could also help show that the occurrence of the other species in the reserve is suppressed by these dominant species, making them be rare species. Thus, there may be tendencies of dominance by some species such as *Uapaca kirkiana* once the reserve has been disturbed (Lowore, 1997). Furthermore, the study has revealed that fire had no impact on the diversity of tree species. Notwithstanding, in the unburnt plots, it was observed that there is a slightly higher diversity than burnt plots. Additionally, evenness values show that both plots have relatively the same species abundance. This implies that there are still tree species, which have similar occurrences regardless of the fire regime. On the other hand, unburnt plots had a richness of 26 tree species compared to 20 tree species in the burnt plots.

Concerning natural regeneration, the study does establish a significant difference in the composition of regenerants due to the effect of fire. However, as has been established from the study results, the diversity measurements on regenerants in unburnt plots are relatively higher compared to those registered in the burnt plots. Adding to this, the total individual regenerants counts show that there is high frequency among regenerants in unburnt than burnt plots. The possible reason for these differences among the plots is the occurrence of fires. Most importantly, fire is seen to have significantly influenced the regeneration of different species in the reserve including *Uapaca kirkiana* whose frequency distribution is higher in burnt plots than in the unburnt plots. The study by Gondwe (2011) also reported a higher number of regenerants of *Uapaca kirkiana a*fter silvicultural disturbances in the reserve. Thus, it can be indicated that the characteristic resilience to disturbances such as fire is shown in the reserve.

In general, for Chongoni Forest, species diversity and regeneration are contributed by several factors, however, from the established parameters it is obvious that fire occurrence plays an important role in influencing species diversity, frequency, and abundance. It is further established that the forest is in its recovery stage, and no fires have occurred in the forest in the past two to three years, which may have a bearing on the regeneration diversity among species. These findings also agree in some part to findings by Chidumayo (1988), who reported that young or regenerating Miombo have a high floristic diversity compared with mature Miombo, whose dominant canopy species have achieved heights up to 12m and out-compete their undergrowth for water and nutrients (Lowore, 1997).

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