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Carbon storage, biodiversity and species composition of Miombo woodlands in recovery trajectory after charcoal production and slash and burn agriculture in Zambia's Copperbelt

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Carbon storage, biodiversity and species composition of Miombo woodlands in recovery trajectory after charcoal production and slash and burn agriculture in Zambia's Copperbelt

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Abstract

There is increasing awareness of the United Nations Reduction of Emissions from Deforestation and Forest Degradation (UN-REDD) programme in tropical woodland management. This programme aims to increase Carbon (C) storage and conserve biodiversity, linking them to livelihood improvement. The objective of this study was to provide an integrated understanding of aboveground (AG) C storage, structural and floristic composition in charcoal and agriculture fallows in Miombo woodland of Zambia. We conducted ecological surveys; measuring tree diameters and assessing species composition on twenty-four 0.25 ha plots in undisturbed woodlands, and fifty-eight plots re-growing after agriculture (5-58 years) and charcoal production (5-44 years). Undisturbed Miombo stored 39.6 t C ha^{-1} AG, while after clearance, C stocks accumulated at 0.98 and 1.42 t C ha^{-1} yr^{-1} in agriculture and charcoal fallows respectively. There were no significant differences in C stocks between woodlands and ≥ 20 year old fallows, implying that in terms of AG C storage, woodlands sufficiently recover after 20 years. Stem densities were significantly higher in charcoal than agriculture fallows but the difference reduced with fallow age. Importance values (IVI) show low presence of less fire resistant tree species such as *Uapaca kirkiana* Müll. Arg. in the initial regrowth of post agriculture fallows, while higher dominance was observed in charcoal fallows. Shannon diversity indices showed high diversity in both woodlands and fallows though the Jaccard similarity coefficient indicated low species similarities, suggesting that though Miombo recover relatively fast in terms of forest structure and C storage, species composition takes longer to recuperate. Findings show that agriculture and charcoal fallows hold an enormous management potential for emerging C-based payments for ecosystem services. Forest management should consider managing fallows for C sequestration and biodiversity restoration through natural succession in Miombo systems. In view of the uncertainty of species recovery, mature Miombo woodlands should be conserved for continued ecosystem functioning and supply of ecosystem services.

Keywords: Above ground Carbon; Forest biodiversity; Fallow; slash and burn; Miombo woodland

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Felix Kalaba is in the final stage of a PhD in Environment at the Sustainability Research Institute, University of Leeds. He has a BSc and MSc in Forestry. Felix's research interest focuses on the links between livelihoods and natural resources particularly forests in a developing country context. Prior to his PhD, Felix had work experience at World Wide Fund for Nature (WWF) under Poverty Reduction and Environmental Management (PREM) project, and at the Copperbelt University in Zambia.

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Andy Dougill is Professor of Environmental Sustainability with expertise as a dryland environmental change researcher who has developed research approaches that integrate a range of disciplines including soil science, ecology, development studies and environmental social sciences. He has over 20 years of experience in leading the design and implementation of inter-disciplinary 'problem-based' research projects focused on sustainability issues at range of scales predominantly in southern African drylands.

1 Introduction

Forests are one of the most important terrestrial biomes contributing immensely to carbon (C) sequestration and storage, and regulating other climate related cycles (Gibbs et al., 2007, Nasi et al., 2002). There is growing interest in understanding C stocks in ecosystems in developing countries (Walker and Desanker, 2004). Understanding the capacity of forest ecosystems to sequester and store C is fundamental in quantifying the contribution of trees to climate mitigation because they indicate the amount of C that can be offset (Ditt et al., 2010). Forests have great potential to provide financial resources through C-based payment for ecosystem services (PES) (Baker et al., 2010), but their functions as dynamic C-pools in biogeochemical cycles is largely unknown (Schongart et al., 2008). The C cycle in Miombo and other tropical woodlands is comparatively understudied (Bombelli et al., 2009, Williams et al., 2008). There are very few empirical studies that have attempted to estimate C stocks in Miombo woodland and in African drylands as a whole (Gibbs et al., 2007, Grace et al., 2006, Williams et al., 2008). In southern Africa, there is relatively scarce knowledge of growth rates and wood biomass in natural woodlands due to the focus on fast growing exotic plantations which have been prioritized by governments (Grundy, 1995), thereby making the total carbon stores in woodlands uncertain (Bryan et al., 2010). Despite the high rates of deforestation in Miombo, the woodlands are able to recover once disturbances cease (Malambo and Syampungani, 2008, Malimbwi et al., 1994, Syampungani et al., 2009). How land use affects forest recovery and C stores in the recovery trajectory remains understudied (Mwampamba and Schwartz, 2011). Estimating carbon stocks is a global challenge that needs to be addressed in woodlands (Bryan et al., 2010). Understanding C stores, the rates and extent to which forests recover from disturbances and how C-stores change in this recovery trajectory has important implications in the emerging C-based PES schemes (Mwampamba and Schwartz, 2011) which are taking centre- stage in United Nations Framework Convention on Climate Change (UNFCCC) climate negotiations for the post-2012 climate regime after the expiry of the Kyoto Protocol commitment period. Quantifying C under different land use scenarios will help in making future land use decisions to ensure optimal land use benefits (Ditt et al., 2010), hence informing forest conservation and sustainable management (Schongart et al., 2008) especially in developing countries which have high poverty levels, and where people’s livelihoods often depend on the forest resource.

An integrated understanding of C storage, and the structural and floristic composition of trees in succession stages, is important in understanding forest restoration processes and in designing forest management strategies in different forest disturbance regimes (Gutiérrez and Huth, 2012). Our objective was to quantify the aboveground (AG) C contained in selected sites of the Miombo woodlands and to assess species composition and forest biodiversity richness in undisturbed woodlands and regrowth sites after slash and burn and charcoal abandonment at various successional stages. This provides an indication of the AG biomass capacity to store C and therefore whether C-based PES are possible livelihood options in the study area. It also provides the empirical data for changes in woodland flora (structure and biodiversity) during forest recovery. Further, the quantification of C in the recovery trajectory of Miombo from the main drivers of forest loss (charcoal production and slash & burn agriculture) will provide insights into how fallow fields can be used in the emerging C markets as a way of maximizing rural people's benefits from the ecosystem. In doing so, it could increase the contribution of forest resources to rural livelihoods, and conserve biodiversity, which is fundamental to the provisioning of ecosystem services. This is both timely and important due to global interest among policy makers on C-based PES as a way of incentivizing reductions in carbon loss from deforestation and degradation (Baker et al., 2010). It is envisaged that this study will provide new understanding of the opportunity that C-storage can bring to increasing financial gains from ecosystem services in local communities who practice slash & burn cultivation and charcoal production, once the AG C stores changes in the recovery trajectory are established and monitoring schemes initiated.

In this paper, we first present an overview of Miombo woodlands. This is followed by unpacking of how biodiversity and floristic composition change in regrowth. The research design and methodology are then outlined, after which the results are presented, discussed and conclusions drawn from the research findings.

1.1 Overview of Miombo woodlands; extent, types and use in livelihoods

Miombo woodland is the most extensive dry forest formation in Africa, with an estimated area of 2.7million km^2 (Frost, 1996, White, 1983). It is classified into wet and dry types divided by the 1000mm mean annual rainfall isohyets (above is the wet, and below is the dry subtypes respectively) (White, 1983) dominated by tree species of the genera *Brachystegia*, *Julbernadia*, and *Isoberlinia*. The woodland is rich in plant diversity, with about 8500 species of higher plants of which 54% are endemic (Chirwa et al., 2008), making the woodlands one of the world's high-biodiversity hotspots (Mittermeier et al., 2003). Miombo hold the bulk of the Earth's biomass (about 43% of the world's tropical dry forests) and are one of the last remaining megafaunal assemblages (Mittermeier et al., 2003). Though tropical forests have been reported as C-dense (Malhi and Grace, 2000), the vast areas covered by the Miombo woodland of Africa shows that it stores substantial amounts of C. The total Earth's surface comprises 11.7% and 18.8% tropical rainforests and tropical woodlands respectively (Grace et al., 2006). Miombo woodlands provide various provisioning ecosystem services which are important for the day-to-day living of their inhabitants. Provisioning services are, "those services supplying tangible goods, finite though renewable, that can be appropriated by people, quantified and traded" (Maass et al., 2005:7). They are a source of foods such as mushrooms (Syampungani et al., 2009), edible insects (Mbata et al., 2002), indigenous fruits (Kalaba et al., 2010, Leakey and Akinnifesi, 2008), seeds, wild vegetables, honey and oils (Shackleton and Gumbo, 2010), and provide medicines , construction materials and energy (Chidumayo and Gumbo, 2010). To the local people, “Miombo woodlands are a pharmacy, a supermarket, a building supply store and a grazing resource” (Dewees et al., 2010:61).

1.2 Vegetation recovery and biodiversity

Slash and burn agriculture and charcoal production are the major causes of forest loss in Miombo woodlands (Chidumayo, 1991, Malambo and Syampungani, 2008, Stromgaard, 1987), and have been linked to huge losses of C and biodiversity of forest systems (Kotto-Same et al., 1997), though changes in woodland flora in post-disturbance growth in slash and burn agriculture and charcoal production sites remains understudied. Miombo biodiversity and species composition are important to people whose livelihood depends on various tree products and services, some of which are species specific (Wilson, 1989). Tree species composition is further vitally important for symbiotic relationships between fungal species (mainly ectomycorrhizae) and specific Miombo tree species which produce mushrooms (Hogberg, 1982). These are an important source of household income. Decline in woodland structure has been associated with a corresponding decline in mushroom populations (McGregor, 1995). Altering tree species diversity (species composition, species richness, species evenness and interactions among species) has functional consequences on ecosystems, reducing productivity and ultimately undermining ecosystem functioning (Chapin et al., 2000). Vegetation structure and species compositional changes in recovery have been discussed by several authors mainly in post -slash and burn agriculture abandonment sites in tropical rainforests (Denslow and Guzman, 2000, Ferreira and Prance, 1999, Guariguata et al., 1997), with a few studies in African woodlands (Syampungani et al., 2010, Williams et al., 2008). Among the factors affecting forest biodiversity recovery and subsequent floristic composition are the proximity of propagule sources and intensity of land use preceding abandonment (Connell and Slatyer, 1977, Guariguata and Ostertag, 2001, Peters, 1994, Saldarriaga et al., 1988), biology of tree species, and climate and edaphic factors (Guariguata and Ostertag, 2001). Most studies have focused on specific successional stages (see review by Guariguata and Ostertag, 2001), with the exception of a few that have studied the processes of recovery at various stages (e.g Denslow and Guzman, 2000, Williams et al., 2008). Plant species richness rapidly increases under light-moderate pre-abandonment disturbances (Guariguata and Ostertag, 2001, Syampungani, 2009, Williams et al., 2008), though their floristic composition remains contested. Some studies (e.g Kappelle et al., 1996, Stromgaard, 1985, Syampungani, 2009) have reported the presence of dominant tree species of old-growth on young (i.e. < 10 years-old site) slash and burn regrowth sites, while others have reported absence of old-growth dominant species in regrowth of the same age (Saldarriaga et al., 1988, Williams et al., 2008). The time taken for floristic composition to recover varies. In the tropical rainforests for example, Saldarriaga et al (1988) reported 60 years of abandonment after slash and burn agriculture. Further, in regrowth of 40 years old, Ferreira and Prance (1999) found differences in tree species richness compared with primary forests. Some studies have suggested it takes centuries for forest to return to primary forest species composition and further argue that forests may not return to their original composition after severe disturbances (Jacobs et al., 1988, Meng et al., 2011). In comparing species richness between old-growth forests and recovering sites in central Panama , Denslow and Guzman (2000) reported no differences in species richness between young regrowth sites and old-growth when seedlings were considered. In the Miombo of southern Africa, Williams et al (2008) found tree species richness to increase asymptotically with the age of fallow, while Syampungani (2009) reports higher species richness in re-growing Miombo at different stages of recovery, which is comparable to old-growth. Further, fire- tolerant tree species have been reported to dominant early stages of recovery, as fire is important in breaking seed coat imposed dormancy (Orwa et al., 2009, Stromgaard, 1984). The amount of species richness observed in recovering forests is affected by the methodological approaches used too. For example Saldarriaga et al. (1988) in a study in Colombia and Venezuela found that it took 40 years for species richness of regrowth with diameter at breast height (DBH) >10 cm to be similar with the primary forest, and 10 years if DBH > 1 cm. Similar observations were obtained in Costa Rica in comparing 16-18 year regrowth sites and primary forests (Guariguata et al., 1997).

2 Research design and methodology

2.1 Study area

This case study was conducted in the Miombo woodlands of Copperbelt Province of Zambia (12˚ 49´ S to 13˚ 36´S and 28˚ 22´ to 28˚ 42´ E, and elevation of 1292 - 1300m above sea level). The Copperbelt province is bordered by the Democratic Republic of Congo on the north and east, and lies on the central African plateau (Figure 1). It is a high rainfall area (average 1200mm per annum), and experiences three weather seasons that are distinguished based on rainfall and temperature, namely; hot dry (September –November), rainy season (December –March) and the cold dry season (April-August) (Chidumayo, 1997).

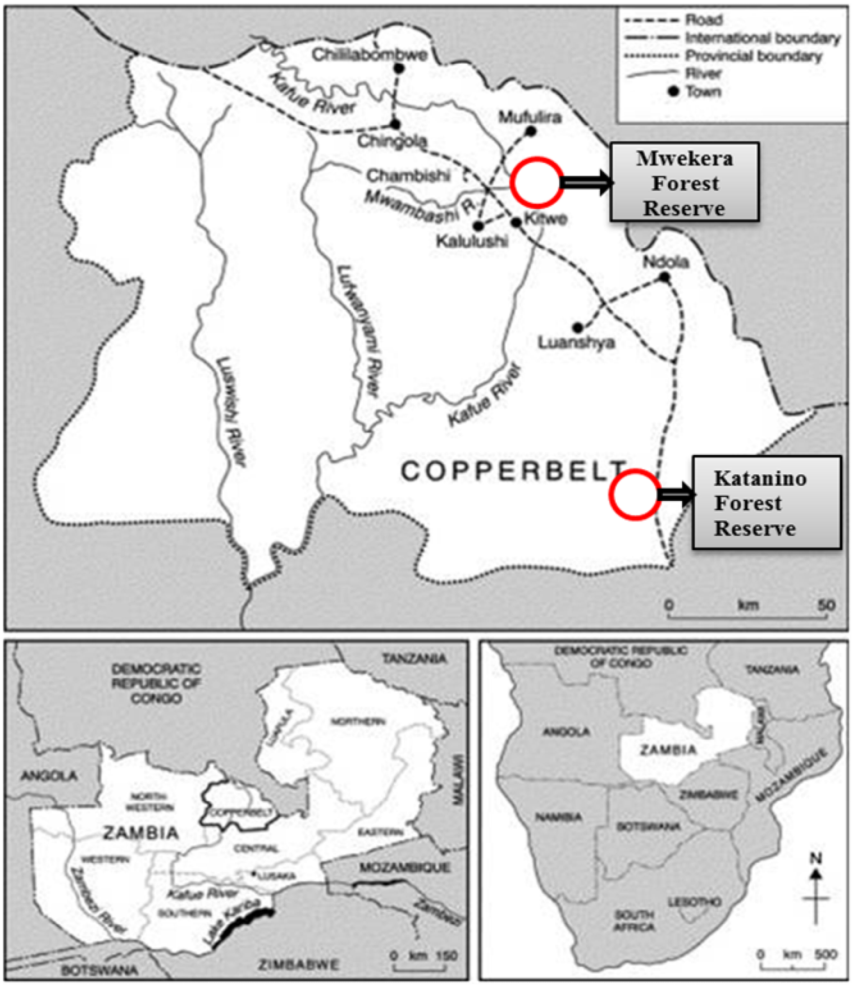

Figure 1: Location of study sites

Source: modified from von der Heyden and New (2004)

In the entire Miombo eco-region , Zambia has the highest diversity of trees and is the centre for endemism for *Brachystegia* tree species (Rodgers et al., 1996) which is one of the Miombo's key species. In Copperbelt province, Miombo represents 90% of the total vegetation (GRZ, 1998). The Copperbelt Miombo is an interesting case in that it is an area of biological significance as it is rich in plant diversity, some species of which are endemic (Chirwa et al., 2008, Rodgers et al., 1996). The miombo woodland lies on the Congo-Zambezi watershed (Chidumayo, 1987) which supports fauna and flora and hydrological cycles in southern Africa. Further, the forests are a source of livelihood and income for their inhabitants in a region characterised by high poverty (73%) and deforestation levels (PRSP, 2002), and is an area often referred to as the 'world's most -income poor region' (Fisher et al., 2011:161).

The Copperbelt Province derives its name from the copper ore deposits in the area which were discovered in the early 1900s. Mining started in the early 1930s and fuelled the migration of people from different parts of Zambia into the Copperbelt province owing to the job opportunities provided by the mining industry. The economic boom in the area gave rise to a sharp increase in population, establishing it as the most densely populated province in Zambia (62.5 persons / km^2) (CSO, 2011). In the early 1990s, the Government of the Republic of Zambia liberalized its economy and embarked on structural changes in the development pattern from public to private investments, leading to privatization of all the copper mining companies. This led to the retrenchment of workers both in mining and other related sectors (PRSP, 2002). The decline in the mining industry in Copperbelt province has led to high unemployment levels in urban areas, forcing people to migrate to rural areas on the outskirts of mining towns in search of agricultural land and natural resources with which to earn a living (Kaoma, 2004), thus exacerbating existing pressures on forest resources (PRSP, 2002). These people directly depend on the forest as a source of agricultural land and as a source of cheaper and readily available energy in the form of charcoal and firewood.

2.1 Site selection and data collection

The study sites were selected using stratified purposive sampling (using the criteria discussed below) (Creswell, 1998). This method is appropriate in selecting a sample on the basis of the knowledge, its elements and the purpose of the study and facilitated comparisons between the stratified elements (Patterson and Coelho, 2009). Data collection was conducted from December 2011 to April 2012.

2.1.1 Undisturbed Miombo

Ground inventories were established in undisturbed Miombo woodland. Plots were not surveyed if they showed any evidence of disturbances or if local farmers or forest officers had any records of the forest being used. Twenty-four 50 m x 50 m (0.25 ha) plots were established (i.e. 16 plots in Mwekera Forest Reserve and 8 in Katanino Forest Reserves). These two sites represented the northern (Mwekera) and Southern (Katanino) parts of Miombo in the Copperbelt Province (see Figure 1). In Katanino, plots were established between Bwengo village and the Katanino Forest Reserve border along a transect line perpendicular to the Oposhi road junction. In Mwekera Forest Reserve, the plots were established along the Mwekera Forest reserve main road from the rail line near *Kamfisa* Prison through the Zambia Forest College to *Mabote* village. Plots were randomly established along the road at distances of at least 100m between them to avoid overlapping. Trees at the centre and corners of the plots were then marked. In the established plots, the tree diameters was measured using a diameter tape at breast height (i.e. 1.3 m above ground) (Ditt et al., 2010, Lawton, 1978, Malimbwi et al., 1994). Trees forking below 1.3 m were measured and recorded separately, while those forking above 1.3m were measured at breast height. Tree species were recorded for all trees within the plots using local names (with the help of traditional botanists), while a botanist from Mwekera Forestry College (engaged as a research assistant) and the authors' knowledge were also used in identifying tree species. For trees that were difficult to identify, leaves, and fruits were taken to the Kitwe Forest Research Herbarium for identification.

2.1.2 Recovering Miombo

The fallow (Fitumbo) ages of sites previously under charcoal production and slash and burn agriculture were estimated using land-use history through informal interviews with local farmers, charcoal producers and traditional councillors (*Ba filolo*). A total of 18 respondents were interviewed following a snowball sampling approach (Patton, 1990). This processes started by holding discussions with the traditional authorities, asking if they knew of any member of the community who had fallows. The leaders provided contact details of possible interviewees. This process was iterative, as participants provided details of other possible interviewees. Fallow areas that local people went to in order to cut trees were not selected, as this study focused on post-disturbance regrowth. Our method is consistent with that of other studies (Mwampamba and Schwartz, 2011, Syampungani, 2009, Walker and Desanker, 2004). This is considered a reliable method as an earlier study by Robertson (1984) on fallow plots in Malawi used this approach and the information obtained from interviews tallied with that obtained from time series aerial photographs. Local people recalled the age of the fallows using various events ranging from family experiences such as the birth, marriage or death of a family member or relative, to natural events. One man recalled that he cleared trees for charcoal on a day when it became dark in the afternoon during the total eclipse. Such events were confirmed with secondary literature. To probe fallow age, the researcher asked several question to triangulate the ages given. The questions were broad and cut across several issues such as the political situation (e.g. the name of the Republican President), biographic data (e.g. age of the farmer/charcoal producer, records of births, marriages and deaths in respondents' families) and the national economic conditions at the time the land was left fallow.

The vegetation survey in recovering Miombo employed double stratified random sampling. The sites were first stratified according to pre-abandonment land use (i.e. slash and burn agriculture or charcoal), and then age of fallows, after which plots were established at random locations within the identified age categories. Slash and burn recovering fallows ranged between 5-58 years, and plots were all rain-fed. No tilling was mechanised as the farmers used hand hoes. 24 plots were established with 4 plots in each identified age class. The ages of charcoal Fallows ranged from 5-44 years, in a total of 34 plots. These age ranges represented the available fallow land in the study area which had undisturbed portions after abandonment. In these sites, 10 m X 20 m plots were established (Chidumayo, 1997, Munishi and Shear, 2004). Diameters of trees was measured at breast height (1.3 m aboveground level) for all trees (trees defined as woody plants more than 2 m (Frost, 1996)) and tree species were recorded as outlined in the undisturbed woodlands. The use of smaller plots in regrowth plots is due to the many species and high density of these plots which makes the use of larger fixed plots time consuming (Syampungani et al., 2010). Plot sizes are determined by a number of factors including study objectives, time and financial resources (Chidumayo, 1997). At least 4 plots were surveyed in recovering Miombo for each identified Fallow age. These fall within the plot numbers used in similar studies (Syampungani, 2009, Williams et al., 2008). We recorded a total of 5 310 stems in 60 plots of recovering Miombo.

2.2 Data analysis

2.2.1 Floristic indices and biodiversity

To describe the tree species composition and vegetation structure of the plots, this study used the Importance Values Index (IVI) (Curtis and McIntosh, 1951), which has been widely used to describe vegetation structure and species composition of forests (Ferreira and Prance, 1999, Johnson and Skousen, 1990, van Andel, 2001). IVI is a summation of the relative density, dominance and frequency of species, i.e. *IVI= (Relative frequency + relative basal area + relative density)/3*, where

$$Relative\ frequency = \frac{\text{Number of plots in which species is present x 100}}{\text{Total number of plots recorded}}$$

$$Relative\ density = \frac{\text{Number of stems recorded for the species x 100}}{\text{Number of stems recorded for all species}}$$

$$Relative\ basal\ area = \frac{\text{Basal area of a species in a plot x 100}}{\text{Total basal area for all species in the plot}}$$

The Jaccard similarity index (J) was used to estimate the species composition similarity between different age classes of the two management regimes, as it is useful in determining the extent of overlap of tree species between communities. It measures similarity based on species composition and hence was used to compare composition overtime. It was used to compare disturbed and undisturbed sites.

J was calculated using the formula:

$$J = \frac{A}{A + B + C}$$

Where A = number of species found in both age classes, B = species in age class A and not in B, C = species in age class B but not in age class A. The Jaccard index varies between 0 and 1. When all the species are similar between the compared groups, J is equal to 1 and 0 when there are no overlapping species.

To measure diversity, the Shannon index (*H´*) was calculated for the mature undisturbed forests and all the regrowth plots.

$$H' = \sum_{i=1}^{s} p_{\mathrm{i}} \ln p_{\mathrm{i}}$$

Where $p_i = n_i / N$; n_i is the number of individual trees present for species i, N is the total number of individuals, and S is the total number of species (Chidumayo, 1997, Shannon, 1948). The Shannon index measures rarity and commonness of species within a sampled community. H'=0 when only one species is present in a population with no uncertainty of what species each individual can be in a population. H' usually ranges between 1.5 - 3.5 and often does not exceed 4 (Margalef, 1972). The Shannon index has been widely used in vegetation studies (Magurran, 2004, Munishi et al., 2008, Shirima et al., 2011, Syampungani, 2009), albeit its narrow range of values which may affect interpretation of results (for a detailed critique see Magurran, 2004). Further, the Shannon index is sensitive to small sample sizes. The current study complemented the Shannon index with the Simpson index which is a useful index for relatively small samples as the Shannon index is not sensitive to small sample sizes (Magurran, 2004). This was important in getting a better informed evidence of the biological diversity of trees, measured using two different diversity indices.

2.2.2 Quantifying aboveground C

We used allometric equations to estimate tree biomass (Table 1). These equations are applicable to the study area owing to the climatic, edaphic, geographic and taxonomic similarities between the study area and the locations in which the equations were developed. In their paper on errors of forest biomass estimates, Chave et al (2004) suggest that errors arise from using equations beyond their range of applicability (such as those developed in different forest ecosystems). According to Brown et al. (1989) local equations are more suitable for accessing forest biomass. Using more than one equation provided us with a good estimation of biomass. Research shows that species-specific allometric equations are not necessary to generate reliable estimates of carbon stocks in Miombo (Gibbs et al., 2007, Malimbwi et al., 1994). Generalized allometric equations for all species types in specific broad forest types and ecological zones are effective in determining the forest carbon stocks, as DBH alone explains more than 95% of the variations in aboveground forest carbon stocks (Brown, 2002, Gibbs et al., 2007). Studies conducted in the Miombo have shown a strong correlation between woody biomass and DBH (Chidumayo, 1991, Chidumayo, 1997, Malimbwi et al., 1994). We restricted our biomass estimations to trees with DBH ≥ 5 owing to the DBH ranges in which the equations were developed .This helped us to avoid error in our biomass estimates (see Chave et al., 2004). Carbon stocks in the plots were calculated by multiplying biomass by 0.5, owing to the fact that 50% of biomass is carbon (Brown and Lugo, 1982, Bryan et al., 2010, Williams et al., 2008).

Table 1: Biomass allometric equations

Reference	**Equation(s)**	**Source country**	**Notes**
Chidumayo (1997)	$B=3.01D-7.48$ $B=20.02D -203.37$	Zambia	for trees <0.1 m DBH for trees >0.1 m DBH
Malimbwi et al (1994)	$B= \exp(2.516 \ln(dbh)-2.462)$	Tanzania	Aboveground
Brown et al (1989)	$B=34.47-8.067D + 0.659D^2$	Dry tropics	Developed in dry tropics and therefore not Miombo specific

Where: B is biomass; D is diameter at breast height.

3 Results

3.1 Vegetation structure and floristic composition

3.1.1 Vegetation structure

In mature woodlands, a total of 2,761 trees were measured over a total survey area of 6 ha. The mean stand density was 592 ± 28.01 stems $ha^{-1.}$ Stems ranged from 308-736 stems ha^{-1}. The mean diameter was 16.57 ± 0.21 cm, with the majority of trees being found within the smaller diameter classes, with 88.2% of stems with diameter ≤ 30 cm, thus showing an inverse J –shaped size class (Figure 2). The mean basal area was estimated at 14.34 ± 0.52 m^2 ha^{-1}, and in the plots ranged from 10.48 to 18.8 m^2 ha^{-1}. The species density was 22 ± 1.2 species ha^{-1}, while species density ranged from 11-33 among the plots.

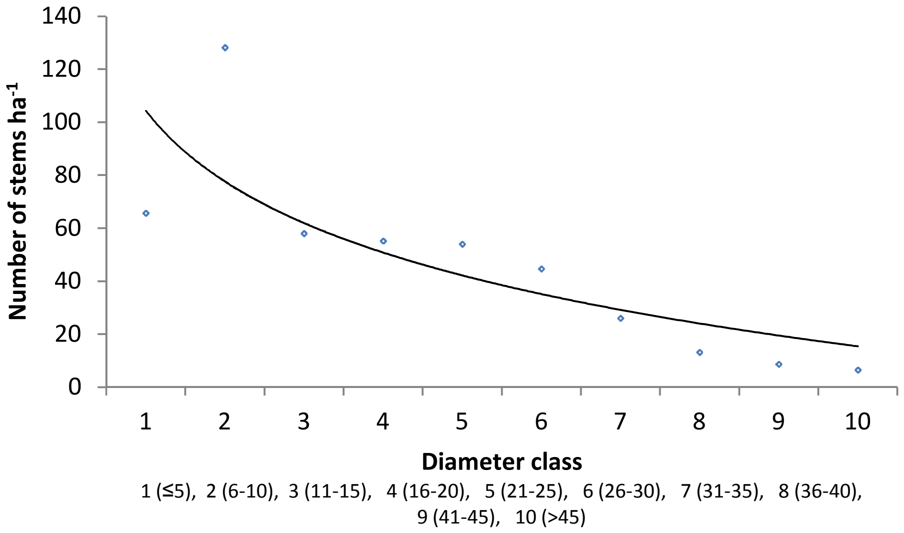

Figure 2: Diameter distribution showing inverse j- shaped size classes

In slash and burn Fallows, the stem stocking density at 5 years was 1,075 stems ha^{-1}. The stem density steadily increased, peaking at around 20 years, after which stocking density declined (Figure 2). A third-order polynomial fitted to the data explained 93% of the observed variability (Figure 3).

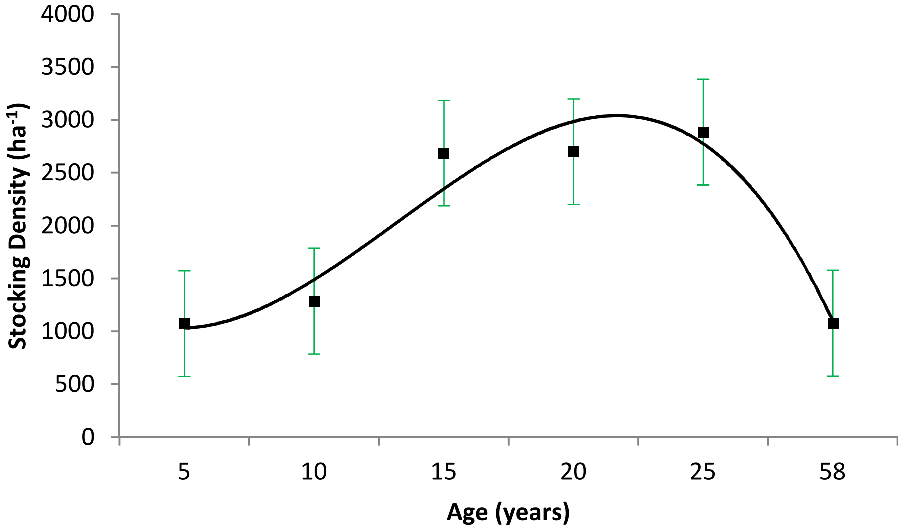

Figure 3: Stocking density (stems per ha) of slash and burn Fallows plotted against age of plots

Stocking density=$1595.3-1286t+823.8t^2-103.9t^3$; t is the time in years.

In charcoal fallows, the stem density at 5-6 years was 1638 ha $^{-1}$ and reached a peak at 12-18 years, then later steadily declined (Figure 4). A third-order polynomial fit to the data was able to explain 45% of the variability.

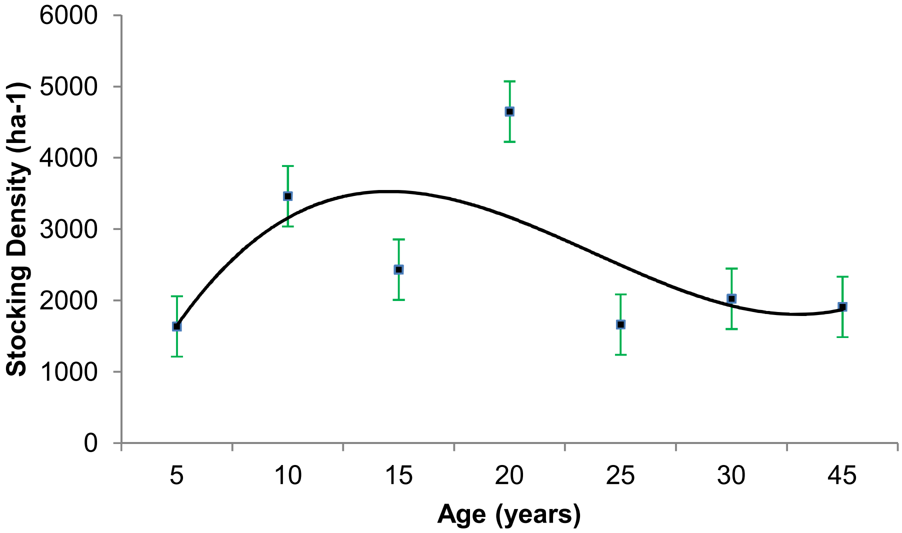

Figure 4: Stocking density of charcoal Fallows plotted against age of plots

Stocking density $=-1414.9+3977.1t -983.7t^2 +68.9t^3$, where t is the age after abandonment.

The stocking density at 5 years after abandonment was not significantly different between slash and burn and charcoal Fallows (P > 0.05), but later became significantly higher in charcoal Fallow*s* at 10 – 15 years. Tree density then later reduced with age for both regimes, with tree density differences narrowing as fallow age increased. The vegetation structure of fallows ≥ 20 years showed a diameter distribution with more trees in lower diameter classes, following a J-inverse shape as observed in mature woodlands.

Basal area for both slash and burn and charcoal were significantly correlated with time since abandonment (P < 0.001) as basal area increased with age of plots (Figure 5). In slash and burn sites, basal area per hectare along the surveyed chronosequence ranged 5.6-26.8 $m^2ha^{-1,}$ and increased at an average of 0.58 m^2ha^{-1}. In charcoal regrowth sites, basal area was higher (ranging between 8.3 - 27.5 m^2ha^{-1} along chronosequence of recovery), increasing at an average rate of 0.73 m^2ha^{-1}

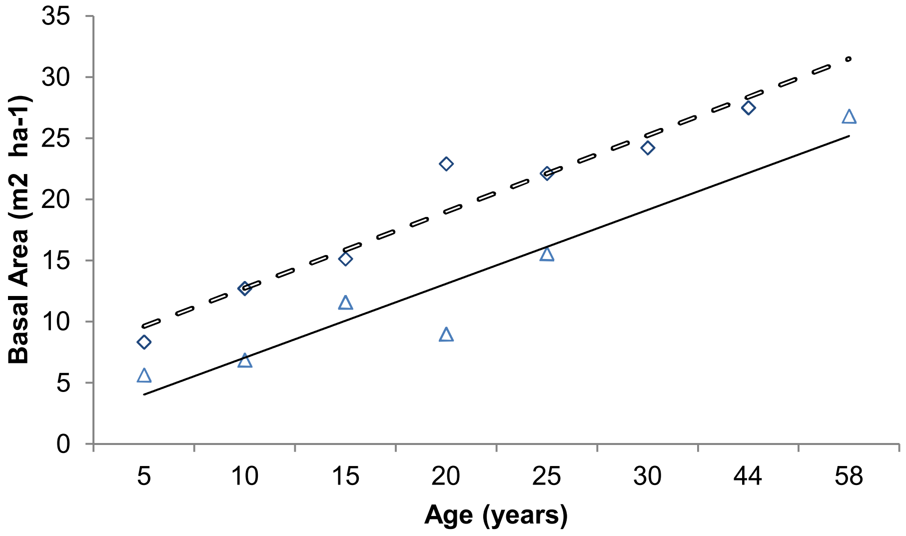

Figure 5: Basal area plotted against age of abandonment for Slash and burn (diamonds, and dash line) and charcoal (triangles, and solid line)

Regression parameters for charcoal are y= 3.12x +6.49; r^2 =93%, and slash and burn y=3.02x + 0.99; r^2=92%, where t in both equations is the time after abandonment.

3.1.2 Floristic composition

The total number of species identified in the mature woodlands was 83 species in 53 families. The original mature Miombo consisted of little understory, with layers of litter on the forest floor. In terms of IVI, the most important species in mature woodland are *Julbernadia paniculata* Troupin, *Marquesia macroura* Gilg, and *Diplorhynchus condylocarpon* (Müll. Arg.). The twenty frequently occurring tree species in descending order are summarized in Table 2. These species are typical of the wet Miombo (Stromgaard, 1985, Vinya et al., 2012).

Table 2: Tree species composition of mature Miombo woodland ranking by IVIs

Rank	Tree species	Relative density (%)	Relative frequency %	Relative Basal area %	IVI %
1	*J. paniculata*	20	91.7	41.6	51.1
2	*M. macroura*	9.1	75	11.0	31.7
3	*D. condylocarpon*	5.3	87.5	1.0	31.3
4	*Parinari curatellifolia* Planch ex Benth.	2.4	83.3	3.0	29.5
5	*Pericopsis angolensis* (Baker)	2.1	79.2	1.2	27.5
6	*Isoberlinia angolensis* (Welw. ex Benth.)	5.5	66.7	8.4	26.9

7	*Brachystegia speciformis* Benth.	3.5	70.8	6.2	26.8
8	*Pseudolachnostylis maprouneifolia* Pax.	2.8	75	1.0	26.3
9	*Monotes africanus* A.DC.	2.5	66.7	1.1	23.4
10	*Brachystegia longifolia* Benth.	3.3	62.5	3.7	23.1
11	*Albizia antunesiana* Harms	3	62.5	1.5	22.3
12	*Syzygium guineense* (Willd) DC.	1.7	62.5	0.3	21.5
13	*Ochna pulchra* Hook	1.6	58.3	0.2	20.1
14	*Phyllocosmus lemaireanus* (De Wild & T. Durrand).	4.3	54.2	0.6	19.7
15	*Brachystegia boehmii* Taub.	3	50	6.0	19.7
16	*Uapaca kirkiana* Müll. Arg.	2.4	54.2	1.0	19.2
17	*Anisophyllea boehmii* Engl.	2.9	54.2	0.6	19.2
18	*Pterocarpus angolensis*(Kiaat)	1.0	54.2	0.3	18.5
19	*Baphia bequaertii* De Wild.	1.9	50	0.5	17.5
20	*Brachystegia floribunda* Benth.	1.5	37.5	0.5	13.2

The floristic composition of regrowth plots differed according to the pre-disturbance land uses and the age of the fallows (Tables 3 and 4).

The study results show that in the early recovering plots (5-10 years), *D. condylocarpon* dominated slash and burn followed by *I. angolensis*, *Securidaca longepedunculata* Fres., *Bridelia micrantha* (Berth) *and B. bequaertii* (Table 4). Most of these species also dominated charcoal regrowth sites of the same age class (Table 5) except the fruit trees *U. kirkiana* which were restricted to charcoal regrowth plots.

Table 3: The ten most dominant species, ranked by IVI (in parenthesis) in each age class of abandoned slash and burn fallow, species richness and Jaccard similarity coefficient

Rank	**5 year**	**10 years**	**15 years**	**20 years**	**25 years**	**58 years**
1	*D. condylocarpon* (42.0)	*I. angolensis* (40.1)	*B. longifolia* (53.4)	*I. angolensis* (58.0)	*J. paniculata* (39.2)	*J. paniculata* (46.1)
2	*I. angolensis* (38.8)	*O. pulchra* (40.0)	*J. paniculata* (44.4)	*B. boehmii* (38.5)	*I. angolensis* (37.7)	*I. angolensis* (41.7)
3	*S. longepedunculata*	*B. bequaertii* (39.2)	*B. speciformis*	*O. pulchra* (38.2)	*Swartzia madagascariensis* Desv.	*B. floribunda* (37.9)

	(36.3)		(38.4)		(37.3)	
4	*B. micrantha* (36.1)	*D. condylocarpon* (39.0)	*Uapaca nitida* Müll. Arg. (36.5)	*J. paniculata* (24.9)	*B. bequaertii* (37.1)	*P. lemaireanus* (37.7)
5	*B. bequaertii* (34.2)	*P. curatellifolia* (37.3)	*O. pulchra* (36.1)	*Strychnos spinosa* Tincture. (23.9)	*Dichrostachys cinerea* (L.) Wight & Arn. (36.9)	*S. madagascariensis* (36.2)
6	*A. boehmii* (31.2)	*J. paniculata* (32.7)	*A. antunesiana* (35.8)	*S. cocculoides* (23.2)	*B. boehmii* (36.3)	*S. guineense* (35.3)
7	*A. antunesiana* (28.4)	*B. floribunda* (30.0)	*Strychnos cocculoides* Baker. (33.9)	*Vitex doniana* Sweet. (12.6)	*B. floribunda* (30.3)	*Lannea discolour* (Sond) Engl. (35.3)
8	*B. floribunda* (28.2)	*B. speciformis* (27.0)	*Strychnos pungens* Soler. (26.8)	*U. kirkiana* (12.6)	*P. maprouneifolia* (27.8)	*A. antunesiana* (28.4)
9	*P. lemaireanus* (28.2)	*A. antunesiana* (19.3)	*P. angolensis* (20.6)	*S. guineense* (12.6)	*Hymenocardia acida* Tul. (27.3)	*O. pulchra* (27.6)
10	*Ekebergia benguelensis* Welw ex C.DC (27.0)	*S. longepedunculata* (19.0)	*M. africanus* (18.1)	*D. condylocarpon* (12.2)	*S. guineense* (27.6)	*B. boehmii* (27.0)
Species richness	19.5 ± 1.2	16.3 ± 1.9	14.8 ± 1.5	10.0 ± 3.7	19.5 ± 1.2	23.0 ± 0.41
J	0.35	0.36	0.26	0.19	0.32	0.37

At 15 years after slash and burn abandonment, the tree canopy was open and consisted of a high proportion of light demanding species (e.g. *Uapaca, Strychnos*, and *Albizia* spp). Some of the Miombo dominant trees species such as *J. paniculata* were present while others (such as *D. condylocarpon, P. curatellifolia*) had few individuals. After 20 years, the forest canopies closed up, with most species found in mature woodland becoming dominant.

In all charcoal fallows we observed high IVI for fire intolerant species such as *A. antunesiana* and *U. kirkiana,* while some Miombo defining species (e.g. *J.*

paniculata, I. angolensis, B. floribunda) were observed in the first 5 years and through the chronosequence (Table 4).

Table 4: The ten most dominant species, ranked by IVI (in parenthesis) in each age class of abandoned charcoal fallow, species richness and Jaccard similarity coefficient

Rank	5 year	10 years	15 years	20 years	25 years	30 years	44 years
1	*B. floribunda* (42.5)	*D. condylocarpon* (55.0)	*B. boehmii* (50.6)	*U. kirkiana* (42.8)	*A. antunesiana* (48.2)	*M. macroura* (51.9)	*I. angolensis* (43.0)
2	*I. angolensis* (41.2)	*U. kirkiana* (48.6)	*O. pulchra* (41.0)	*J. paniculata* (42.6)	*B. boehmii* (41.3)	*U. kirkiana* (43.5)	*B. boehmii* (40.6)
3	*A. boehmii* (39.8)	*B. boehmii* (41.0)	*P. curatellifolia* (38.7)	*I. angolensis* (41.9)	*J. paniculata* (39.5)	*J. paniculata* (41.3)	*J. paniculata* (38.0)
4	*J. paniculata* (39.3)	*S. guineense* (39.6)	*J. paniculata* (38.4)	*O. pulchra* (37.7)	*B. floribunda* (37.1)	*I. angolensis* (38.7)	*P. angolensis* *(36.1)*
5	*A. antunesiana* (34.6)	*I. angolensis* (39.2)	*D. condylocarpon* (35.7)	*A. antunesiana* (36.8)	*P. lemaireanus* (34.5)	*B. boehmii* (29.4)	*B. floribunda* (36.0)
6	*P. lemaireanus* (36.5)	*B. bequaertii* (37.6)	*Diospyros batocana* Hiern (34.1)	*B. boehmii* (36.5)	*I. angolensis* (28.4)	*P. curatellifolia* (19.5)	*Uapaca benguelensis* Müll. Arg. (35.8)
7	*S. madagascariensis* (28.2)	*P. curatellifolia* (38.3)	*A. antunesiana* (25.4)	*A. boehmii* (36.3)	*D. condylocarpon* (27.9)	*A. antunesiana* (19.0)	*P. maprouneifolia* (34.2)
8	*S. guineense* (28.1)	*B. floribunda* (36.6)	*P. maprouneifolia* (25.2)	*B. speciformis* (35.9)	*D. cinerea* (26.7)	*P. angolensis* (19.0)	*Albizia antunesiana* (34.2)
9	*U. kirkiana* (26.8)	*P. lemaireanus* (36.2)	*M. africana* (25.0)	*M. africanus* (35.0)	*P. maprouneifolia* (26.4)	*O. pulchra* (18.7)	*P. lemaireanus* (28.8)
10	*B.*	*M.*	*Brachystegia*	*P.*	*A. boehmii*	*Dalbergia*	*U. kirkiana*

	speciformis (26.7)	*africana* (35.5)	*manga* De Wild. (24.3)	*lemaireanus* (34.9)	(26.1)	*nitudula* Welw ex Baker. (18.4)	(27.5)
Species richness	17.3 ± 2.1	23.3 ± 0.9	15.0 ± 1.2	27.0 ± 1.8	20.5 ± 1.2	18.8± 2.2	20.8± 0.9
J	0.33	0.45	0.26	0.44	0.39	0.26	0.33

The Jaccard similarity coefficient for comparing species composition between slash and burn regrowth sites and mature woodlands ranged from 0.19 to 0.37, and was highest in the oldest regrowth site (Table 3). In charcoal regrowth sites, the Jaccard coefficient ranged from a minimum of 0.26 to a maximum of 0.44 (Table 4). The study revealed that there was a relatively higher similarity with mature woodlands in charcoal (0.35 ± 0.03) than slash and burn regrowth sites (0.31 ± 0.03), though the difference was not statistically significant (t =1.04, *P* = 0.32). A comparison of the dominant leguminous tree genera in mature woodlands and the oldest regrowth sites is summarized in Figure 6.

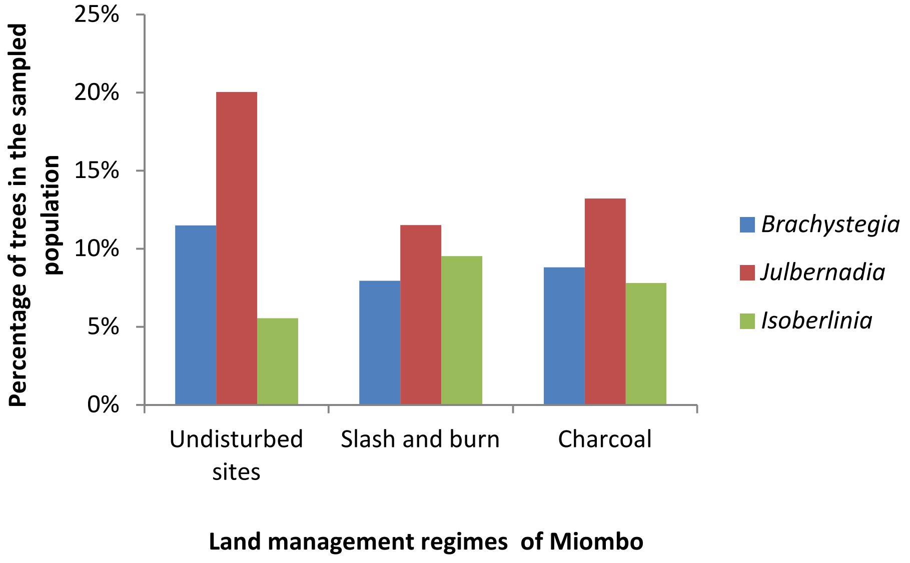

Figure 6: Distribution of dominant leguminous genera in undisturbed site and oldest regrowth sites.

3.2 Diversity of tree species

The results of this study show that species richness in regrowth sites in the two management regimes was significantly different from mature woodlands (F= 4.65, P = 0.01), as undisturbed mature sites had higher species richness. The study showed no significant differences between slash and burn and charcoal regrowth sites (t= -0.18, P=0.86, equal variances assumed) though generally charcoal regrowth sites had more species (17.9 ± 6.5 and 17.6 ± 4.9 respectively). Species richness significantly positively correlated with age of abandonment in slash and burn regrowth ($P < 0.05$), and not in charcoal plots. There were no statistically significant differences in mean species richness between regrowth sites of slash and burn and charcoal of 20 years and above and mature woodlands ($F = 1.48$, $P = 0.24$). Our results on species diversity as measured by the Shannon index (*H'*) in slash and burn plots ranged from 2.1 to 2.8 as diversity increased along the chronosequence. In charcoal regrowth plots, *H'* values ranged from 2 to 2.7 with diversity increasing with age. In mature woodlands, the mean *H'* was 2.8 ± 0.1. Species diversity was not significantly different between mature woodlands and the sampled regrowth sites ($F= 0.61$, $P = 0.55$). The Simpson index of diversity further confirmed the diversity of regrowth with ranges of 0.8-0.92 and 0.78-0.91 in slash and burn and charcoal sites respectively.

3.3 Aboveground C storage

Using the mean of 3 allometric equations, in the mature woodlands, the mean ± S.E. estimated C as 39.6 ± 1.5 Mg C ha^{-1}, ranging from 28.7 to 52.8 Mg C ha^{-1}. Results from the slash and burn fallows showed that along the chronosequence of recovery, carbon storage ranged from 5.4 ±1.1 Mg C ha^{-1} at between 5-6 years, to 61.7 ± 18.1 Mg C ha^{-1} in trees that were approximately 58 years old. Using a weighted mean of the three equations, carbon accumulation was estimated to be 0.98 Mg C ha^{-1} year^{-1}. The range was from 0.84 to 1.21 Mg C ha^{-1} year^{-1}. The recovery trajectory of charcoal fallows contained 10.5 ± 2. 7 Mg C ha^{-1} at the age of 5 years, and the storage was estimated at 64.3 ± 10.1 Mg C ha^{-1} in the oldest plots (44 year old plots). The average accumulation of C was estimated to be 1.42 Mg C ha^{-1} year^{-1}. The sequestration rate was highest in the initial regenerating phase (up to 2.1 Mg C ha^{-1} in the first 5 years), and lowest in the oldest plots i.e. over 25 years (0.89 Mg C ha^{-1} year^{-1}).

Comparing C storage in slash and burn and charcoal fallows, the results show that in the first 5 years, C storage was higher in charcoal than slash and burn plots, though not significantly different (t =-1.76, P =0.16). The study found that at 10 years after abandonment, charcoal fallows had statistically significant higher C storage (19.2 ± 2.6 Mg C ha^{-1}) than slash and burn regrowth (9.6 ± 2.0 Mg C ha^{-1})($t= -3.23$, $P= 0.02$). Statistically significant results were also observed at 15-16 years, while no significant differences were observed in C storage between the two management regimes after 20 years (Table 5).

Table 5: Comparisons of carbon stocks between slash & burn and charcoal regrowth at different age classes of abandoned fallows

Age of Plot	Mean C stocks		t-value	P Value
	Slash and burn	Charcoal		
5 years	5.4 ± 1.1	10.5 ± 2.7	-1.76	0.16
10 years	9.6 ± 2.0	19.2 ± 2.6	-3.23	0.018*
15 years	15.7 ± 2.4	24.1 ± 1.7	-2.63	0.046*
20 years	22.0 ± 7.6	32.9 ± 3.7	-1.30	0.24
25 years	26.5 ± 3.9	44.9 ± 17.6	-1.019	0.35
30 years		51.9 ± 11.8	X	X
44 years		64.3 ± 10.1	X	X
˜58 years	61.7 ±1 8.1		X	X

**Significant at 0.05

X: t was not computed as at least one of the management regimes did not have plots corresponding with the age.

A one-way ANOVA showed that there were no statistically significant differences (F = 2.22, P = 0.12) in C estimates between mature woodlands and regrowth stands ≥ 20 for both slash and burn and charcoal fallows. C estimates in regrowth stands were positively and significantly correlated to the age of fallow (*P* < 0.001).

4 Discussion

4.1 Vegetation structure and floristic composition

4.1.1 Vegetation structure

In mature woodlands, the inverse J-shaped size classes showing more trees in the smaller size classes is an indicator of a steady and expanding population, which according to Peters (1994) is a self-maintaining population, in which young trees will eventually replace the older trees. Other studies within the Miombo have reported a similar size class distribution (Chidumayo, 1997, Munishi et al., 2008, Shirima et al., 2011). In this size class profile, young trees continue to regenerate under the canopies of more mature trees indicating that they are shade tolerant, as well as resistant to fire in fire prone areas such as the dry woodlands of southern Africa (Peters, 1994). When the forest canopy closes, some seedlings are stunted as some Miombo species require high light intensities for growth (Chidumayo and Frost, 1996). The diameter distribution obtained at 25 years old regrowth and older is indicative that the Miombo is able to achieve its mature vegetation structure after 25 years of abandonment. Our findings demonstrate that Miombo develops primary forest characteristics within 2-3 decades of fallow (both charcoal and agriculture), similar to observations by Chazdon (2003) in slash and burn regrowth sites in tropical rainforests.

The basal area obtained in charcoal fallows was higher than the slash and burn fallows. This can be attributed to the fact that after charcoal production, most Miombo trees grow from coppices, and thus grow faster than on slash and burn sites. The stem density per hectare declines with age of regrowth (Chidumayo, 1988a, Chidumayo, 1988b) due to inter-shoot competition. The basal area annual increment obtained in this study compares with that of other studies on regrowth forests within the Miombo eco-region (Stromgaard, 1985, Williams et al., 2008).

4.1.2 Floristic composition

The Miombo floristic structure changed at various stages in the chronosequence. The vegetation composition of regrowth sites suggests that pre-disturbance land use affects the vegetation composition in recovery. After disturbances, increases in sunlight reaching the forest floor due to removal of canopies during tree cutting provides favourable germinating conditions and thus triggering regeneration of light demanding species (Peters, 1994). The tree species that grow earlier are those whose seeds are available in the soil before disturbance or the sprouting of the cut adults (Connell and Slatyer, 1977).

This study shows that in early regrowth, after slash and burn, fire tolerant species e.g. *D. condylocarpon*, *B. bequaertii*, *I. angolensis*, *J. Paniculata*, *B. boehmii* and *B. floribunda* were dominant (see Lawton, 1978, Strang, 1974). These findings are consistent with the findings of Peters (1994) and Stromgaard (1984) which have reported dominance of fire and drought tolerant species in the early stages of recovery after slash and burn agriculture. Our findings show a high concentration of less fire-resistance species (such as *B. speciformis*, *S. guineense*, *and U. kirkiana*) in early charcoal regrowth sites. These species' successful establishment in early stages under slash and burn regrowth sites is hampered by fire (Orwa et al., 2009), though fire can be later used for management after establishment. At about 15 years regrowth, sites were still associated with light demanding pioneer species growing in open canopies (such as *U. kirkiana*, *O. pulchra*, *and A. antunesiana*); this is the case until the canopy begins to close after 25 years. These trees are eventually replaced by species which are also dominant in mature woodlands (e.g. *I. angolensis*, *J. paniculata* and *Brachystegia* spp). Our study shows varying diameters of key Miombo species in regrowth sites of different ages with higher proportions observed in charcoal sites. Our findings contradict the findings of Williams et al (2008) in

Mozambique who did not find any Miombo defining species in regrowth from slash and burn among the top five dominant species in all the re-growing plots sampled. We suspect the difference may be partly attributed to responses of Miombo species being different between wet and drier regions, or the differences in proximity of regrowth sites to mature Miombo woodland which was further in their study. The changes in species dominance along the chronosequence may be explained by the fact that succession species enhance growth conditions and hence, enhance invasion of later succession species, based on the understanding that later species can only grow when the pioneer species have modified the environment by ameliorating soil conditions e.g. by increasing nitrogen (Connell and Slatyer, 1977). Though some species dominate in the early stages due to their rapid dispersal ability and fire tolerance, and occupy the 'empty area', these species are shade intolerant and cannot continue to grow under their own shade. Therefore, they start reducing with the age of the forest stand (Connell and Slatyer, 1977, Saldarriaga et al., 1988) thus explaining the changes in species dominance.

4.2 Diversity, species composition and ecosystem functioning

Both the Simpson diversity index and the Shannon index show that the Miombo woodlands have high biodiversity. Our Shannon index results (2.8) show a high diversity as Shannon index values greater than 2 is indicative of medium to high diversity (Barbour et al., 1987). Our study results further shows a higher diversity than other studies in the Miombo region such as in Tanzania were Shannon indices of 1.05 and 1.25 were obtained (Shirima et al., 2011), and from Mozambique's Miombo (Williams et al., 2008), but similar to diversity (2.7) in the landscapes of the west Usarambara (Munishi et al., 2008). These results show that within the Miombo region, our study region is highly biologically diverse at tree species level and could be important for various biogeochemical cycles since diversity often is indicative of better ecosystem functioning (Barbour et al., 1987). Once land is abandoned after slash and burn and charcoal production, tree species diversity remains high in Miombo once the woodland is left to recover without subjecting it to further disturbances. Slash and burn agriculture has been linked to extensive losses of biodiversity (Chidumayo, 1987, Kotto-Same et al., 1997). However, our study shows that in recovery, biodiversity is comparable with mature woodlands. This study has however shown a low similarity in floristic composition of oldest (both charcoal and slash and burn plots, though slightly higher in charcoal fallows) and mature woodlands. Our results therefore show that 58 and 44 years after abandonment for slash and burn and charcoal respectively, the floristic composition is still different from mature woodlands. In their study on species composition after slash and burn agriculture in the Amazon, Ferreira and Prance (1999) suggested that 40 years of re-growth was not sufficient for the species composition of re-growth sites to equal that of primary forests, while in Indonesia, low species similarities were observed between primary forests and 55 year-old secondary forest (Brearley et al., 2004).

According to Jacobs et al (1988), the return to primary forest species composition takes centuries and they warned that as the fallow age increases, regrowth sites closely resemble primary forests to the extent that only a detailed examination of species composition can reveal the dissimilarities .

Miombo dominant species have tree-specific fungi symbiotic relationships (mycorrhizal associations) and termite symbiotic associations important for ecosystem functioning and producing non wood forest products such as indigenous mushrooms that cannot be domesticated (Hogberg, 1982, Munyanziza, 1996), which are important for livelihoods. Further, since Miombo soils are nutrient poor (Trapnell et al., 1976) mycorrhizal associations are needed for effective nutrient uptake and retention, which are important for growth (Hogberg, 1982), and ultimately enhancing productivity of the ecosystem, and other complex relationships among organisms within the Miombo. Changes in tree species composition have the potential to affect the ecological functioning of ecosystems altering nutrient recycling and an array of ensuing ecosystem benefits (Chapin et al., 2000). These changes though often gradual, may eventually cause irreversible large species shifts (see Figueiredo and Pereira, 2011) and affect the resilience and resistance of ecosystems to environmental change (Chapin et al., 2000).

4.3 C–stocks and changes in the recovery trajectory

C storage in mature Miombo woodland estimated in this study (39.6 ± 1.5 Mg C ha^{-1}) is higher than that reported in Tanzania's Miombo by Shirima et al. (2011), Munishi et al. (2010), which suggest 23.3 Mg C ha^{-1} and 19.1 Mg C ha^{-1} respectively. The differences observed with studies in Tanzania may be attributed to human disturbances. Although their studies were conducted in the forest reserve, neither targeted undisturbed or intact plots. Further, the studies measured diameters ≥ 10 (Shirima et al., 2011) and ≥ 6 cm (Munishi et al., 2010) which may have an impact on the measured C storage as some trees are excluded from the measurement. The results from this study are higher than estimates for Mozambique i.e. 19.0± 8.0 Mg C ha^{-1} (Williams et al., 2008) which has more dry Miombo than Copperbelt Zambia. Our estimated carbon storage is lower than estimates from tropical rainforests of Africa i.e. 202 Mg C ha^{-1} (Lewis et al., 2009). The C storage in the Miombo is likely to be higher than estimated as the allometric equations developed for the Miombo use a diameter of about 5 cm, and relatively more trees are found with DBH >5 cm.

In regrowth sites, charcoal abandoned sites had higher carbon storage than slash and burn agriculture sites. This may be attributed to higher regeneration rates on charcoal sites as trees grow from coppices which are new shoots emerging from stumps of cut trees. The ability of the Miombo species to regenerate from coppices has been reported (Boaler and Sciwale, 1966, Chisha-Kasumu et al., 2007, Guy, 1981), Miombo species' main regeneration is through coppice regrowth and root suckers as opposed to seeds (Strang, 1974, Trapnell, 1959). Regeneration after slash and burn agriculture from coppices may be reduced as some plants may be uprooted or die due to injuries sustained during cultivation (Strang, 1974, Syampungani, 2009). The high regeneration in charcoal regrowth increases C storage rapidly after abandonment, until after 20 years when C storage differences between the two management regimes reduce. Recovery of forests is slow after disturbances that affect soil and aboveground vegetation (Chazdon, 2003). This may partially explain why carbon accumulation was higher in charcoal regrowth plots than slash and burn agriculture. Cultivation using hand hoes has the potential to disturb the soil structure, while uprooting of trees in land preparation may further reduce future sources of propagules. This may be compounded by the constant removal of trees regenerating on site (weeds) during the years of agriculture use in a bid to reduce competition with agricultural crops. Further, seedlings may have been left during charcoal production, therefore increasing carbon storage rapidly. In a study in northern Zambia on fresh biomass of 16 year-old regrowth, Stromgaard (1985) found biomass in regrowth vegetation cleared, burned and cultivated was less than half when compared to trees that were cut without land being cultivated (i.e. 15. 8 and 48.3 t ha^{-1}respectively). The fire in slash and burn agriculture has the potential to kill the roots and substantially reduce the seed bank, thereby slowing plant succession after abandonment (Ferreira and Prance, 1999).

The changes in C storage observed in the recovery trajectory of both management regimes in this study provide empirical evidence of the importance of the Miombo in carbon sequestration. The sequestration rates obtained in this study in slash and burn regrowth sites are comparable with those obtained by other studies (Kotto-Same et al., 1997, Stromgaard, 1985, Williams et al., 2008) i.e. 0.7, 0.98 and 0.98 t ha^{-1} year^{-1} respectively. This accumulative evidence, as demonstrated by this study, suggests that tropical woodlands sequester vast amounts of carbon in their various eco-regions spreading across different countries, even with different topographic and edaphic characteristics.

Our study showed higher C storage in the oldest recovery sites (both slash and burn and charcoal) than mature woodlands, though differences were not significant. These results correspond with those from an earlier study on forest chronosequences in Panama which found biomass to reach its peak after 70 years of disturbance, and declining after 100 years to reach the old-growth value (Denslow and Guzman, 2000). We suggest this trend to be applicable to Miombo as demonstrated by our findings. The lack of significant differences in C storage between older regrowth (≥20 year-old) and undisturbed mature woodlands, shows empirical evidence that after abandonment (whether after slash and burn or charcoal production), 20 years is sufficient for C storage to attain that of undisturbed woodland stores. It should however be noted that the extent of disturbances may affect recovery, and therefore results from the study must be understood within the context of small-scale farmers, who do not use highly mechanised equipment which has the potential to heavily impact on soil structure (Chazdon, 2003), therefore extending the recovery period.

4.4 Implications of Miombo recovery for REDD+

The recovery of Miombo C stocks means fallows of slash and burn agriculture and charcoal production have the potential to be managed sustainably under REDD+ to ensure degraded forests recover their lost carbon stocks and biodiversity and restore the flow of various ecosystem services. This has the potential to generate income for local communities through the sale of carbon credits, subsequently diversifying their livelihood strategies beyond their use of traditional non-timber forest products. In the past, little attention has been paid to reversing forest degradation through restoration (Sasaki et al., 2011). The Kyoto Protocol's narrow focus on afforestation (establishing forests on land that has not previously been forested) and reforestation (planting trees on land that was previously a forest) excluded natural restoration. The post-Kyoto negotiations according to the Copenhagen accord of 2009 adopted at the 15th Conference of the Parties (COP 15) and subsequent meetings (Cancun and Durban COP 16 and 17 respectively) have opened a window of opportunity for forest restoration under improved forest management to enhance carbon sinks, conserving biodiversity and improving livelihoods (see UNFCCC, 2009, UNFCCC, 2010, UNFCCC, 2011). Forest restoration has a significant role to play in global climate change mitigation and supporting livelihoods (Sasaki et al., 2011). In the management of Miombo under the REDD+ initiative (for which Zambia is a pilot country), it is important that rather than only focusing on avoided deforestation, forest restoration management must be considered. In Miombo woodlands, promotion of mosaic restoration is ideal for small-scale farmers and charcoal producers since patches of forests are subjected to different uses. Mosaic restoration is suitable for areas with considerable differences in land use (such as agriculture, charcoal, human settlements, grazing) (IUCN, 2011), and populations that are between 10-100 persons/Km^2 (WRI, 2011), which are common in rural areas of Miombo. This will help degraded forests to recover their lost carbon stocks, biodiversity and provide an array of benefits to people, both as goods or other ecosystem services (Sasaki et al., 2011). Regrowth vegetation is important in offsetting GHG emissions from agriculture and other industries, and conserving biodiversity of native flora (Dwyer et al., 2009). Natural regeneration offers a suitable way to restore biodiversity habitats (Kim, 2004). Despite the observed uncertainties on the time required for Miombo biodiversity to recuperate after disturbances, regrowth under natural regeneration produces species that are adapted to local conditions and provides suitable habitat for local fauna (Bowen et al., 2007). Further, local people have realised the use of these species and so they are capable of providing more benefits to local people than forest plantations. Management of fallows for extended periods of time will allow local people to generate carbon credits through managing fallows, and further provides an opportunity to restore forest biodiversity which underpins many rural livelihood strategies. There is need for investment into Miombo recovery through local communities' participation, long-term political commitment and provision of long-term financial incentives for fallow management under any Post-Kyoto agreement. Lack of investment funds hampers restoration efforts (IUCN, 2011). To support forest restoration, appropriate national policies, institutional arrangements and local participation are needed (Sasaki et al., 2011). Once adopted under REDD+, managing fallows will be cost effective when compared to conventional planting, but it comes with the challenge of monitoring that fallows are managed.

5 Conclusion

The objective of this study was to determine aboveground C storage, structural and floristic composition and biodiversity of the Miombo and changes once charcoal production and slash and burn agriculture are abandoned. We have shown that the Miombo is a substantial AG C store. Once mature Miombo woodland are cleared in the study area, aboveground C stocks are reduced by 39.6 t C ha^{-1}, and after abandonment and subsequent recovery through natural succession, vegetation C accumulates at rates of 0.98 and 1.42 t C ha^{-1} for agriculture and charcoal land uses respectively, with accumulation increasing rapidly in the first 15 post-abandonment years. After 20 years, the C storage in regrowth sites shows no significant difference compared with mature woodlands. The study shows that Miombo is able to achieve mature vegetation structure (DBH, basal area) after 20 years of abandonment. Our empirical findings suggest charcoal production and slash and burn agriculture have the potential to be considered in emerging C markets, where incentives are given to local people to manage fallow areas to increase carbon storage and restore other ecosystem services. These land uses hold an enormous management potential which remains neglected in current forest management strategies.

Our results show that although 20 years is sufficient for the forest structure of re-growing Miombo to resemble mature woodlands, this time is not sufficient for the floristic composition to recuperate. High species diversity in regrowth stands does not always correlate with species similarities with primary forest. Caution therefore must be taken in the interpretation of diversity indices in developing management strategies. It must be ensured that attention is paid to actual species composition and the presence of Miombo dominant species.

In view of the unclear time required for the floristic composition of regrowth to recuperate to mature woodland there is need to conserve the existing mature Miombo for various ecological and socio-economic benefits.

The results provided in this study are important in providing insights into the scope and nature of REDD+ initiatives in Zambia and more broadly in global drylands, providing empirical evidence on C storage and how C and biodiversity changes after disturbances from the main drivers of forest loss in tropical drylands. The results presented in this paper are also important to policy makers for understanding carbon changes in forests and biodiversity and can guide them in developing policy interventions on which the emerging initiatives of C payments are to be based.

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7 References

BAKER, T. R., JONES, J. P. G., THOMPSON, O. R. R., CUESTA, R. M. R., DEL CASTILLO, D., AGUILAR, I. C., TORRES, J. & HEALEY, J. R. 2010. How can ecologists help realise the potential of payments for carbon in tropical forest countries? *Journal of Applied Ecology,* 47**,** 1159-1165.

BARBOUR, M. G., BURK, J. H. & PITTS, W. D. 1987. *Terrestrial plant ecology*, Benjamin/Cummings Pub. Co.

BOALER, S. B. & SCIWALE, K. C. 1966. ECOLOGY OF A MIOMBO SITE LUPA NORTH FOREST RESERVE TANZANIA .3. EFFECTS ON VEGETATION OF LOCAL CULTIVATION PRACTICES. *Journal of Ecology,* 54**,** 577-&.

BOMBELLI, A., HENRY, M., CASTALDI, S., ADU-BREDU, S., ARNETH, A., GRANDCOURT, A. D., GRIECO, E., KUTSCH, W. L., LEHSTEN, V., RASILE, A., REICHSTEIN, M., TANSEY, K., WEBER, U. & VALENTINI, R. 2009. The Sub-Saharan Africa carbon balance, an overview. *Biogeosciences Discuss,* 6**,** 2085–2123.

BOWEN, M. E., MCALPINE, C. A., HOUSE, A. P. N. & SMITH, G. C. 2007. Regrowth forests on abandoned agricultural land: A review of their habitat values for recovering forest fauna. *Biological Conservation,* 140**,** 273-296.

BREARLEY, F. Q., PRAJADINATA, S., KIDD, P. S., PROCTOR, J. & SURIANTATA 2004. Structure and floristics of an old secondary rain forest in Central Kalimantan, Indonesia, and a comparison with adjacent primary forest. *Forest Ecology and Management,* 195**,** 385-397.

BROWN, S. 2002. Measuring carbon in forests: current status and future challenges. *Environmental Pollution,* 116**,** 363-372.

BROWN, S., GILLESPIE, A. J. R. & LUGO, A. E. 1989. Biomass estimation methods for tropical forests with applications to forest inventory data *Forest Science,* 35**,** 881–902.

BROWN, S. & LUGO, A. E. 1982. The storage and production of organic matterin tropical forests and their role in global carbon cycle *Biotropica* 14**,** 161-167.

BRYAN, J., SHEARMAN, P., ASH, J. & KIRKPATRICK, J. B. 2010. Estimating rainforest biomass stocks and carbon loss from deforestation and degradation in Papua New Guinea 1972-2002: Best estimates, uncertainties and research needs. *Journal of Environmental Management,* 91**,** 995-1001.

CHAPIN, F. S., ZAVALETA, E. S., EVINER, V. T., NAYLOR, R. L., VITOUSEK, P. M., REYNOLDS, H. L., HOOPER, D. U., LAVOREL, S., SALA, O. E., HOBBIE, S. E., MACK, M. C. & DIAZ, S. 2000. Consequences of changing biodiversity. *Nature,* 405**,** 234-242.

CHAVE, J., CONDIT, R., AGUILAR, S., HERNANDEZ, A., LAO, S. & PEREZ, R. 2004. Error propagation and scaling for tropical forest biomass estimates. *Philosophical Transactions of the Royal Society of London Series B-Biological Sciences,* 359**,** 409-420.

CHAZDON, R. L. 2003. Tropical forest recovery: legacies of human impact and natural disturbances. *Perspectives in Plant Ecology Evolution and Systematics,* 6**,** 51-71.

CHIDUMAYO, E. N. 1987. Woodland structure, destruction and conservation in the Copperbelt area of Zambia. *Biological Conservation,* 40**,** 89-100.

CHIDUMAYO, E. N. 1988a. A RE-ASSESSMENT OF EFFECTS OF FIRE ON MIOMBO REGENERATION IN THE ZAMBIAN COPPERBELT. *Journal of Tropical Ecology,* 4**,** 361-372.

CHIDUMAYO, E. N. 1988b. REGENERATION OF BRACHYSTEGIA WOODLAND CANOPY FOLLOWING FELLING FOR TSETSE FLY CONTROL IN ZAMBIA. *Tropical Ecology,* 29**,** 24-32.

CHIDUMAYO, E. N. 1991. Woody biomass structure and utilisation for charcoal production in a Zambian Miombo woodland. *Bioresource Technology,* 37**,** 43-52.

CHIDUMAYO, E. N. 1997. *Miombo ecology and management: an introduction,* London, IT Publications in association with the Stockholm Environment Institute.

CHIDUMAYO, E. N. & FROST, P. 1996. *Population biology of Miombo trees,* Bogor, Indonesia, Center for International Forestry Research (CIFOR).

CHIDUMAYO, E. N. & GUMBO, D. J. 2010. *The dry forests and woodlands of Africa,* London, UK, Earthscan.

CHIRWA, P. W., SYAMPUNGANI, S. & GELDENHUYS, C. J. 2008. The ecology and management of the Miombo woodlands for sustainable livelihoods in southern Africa: the case for non-timber forest products. *Southern Forests,* 70**,** 237-245.

CHISHA-KASUMU, E., WOODWARD, S. & PRICE, A. 2007. Comparison of the effects of mechanical scarification and gibberellic acid treatments on seed germination in Pterocarpus angolensis. *Southern Hemisphere Forestry Journal,* 69**,** 63-70.

CONNELL, J. H. & SLATYER, R. O. 1977. Mechanisms of Succession in Natural Communities and Their Role in Community Stability and Organization. *The American Naturalist,* 111**,** 1119-1144.

CURTIS, J. T. & MCINTOSH, R. P. 1951. AN UPLAND FOREST CONTINUUM IN THE PRAIRIE-FOREST BORDER REGION OF WISCONSIN. *Ecology,* 32**,** 476-496.

DENSLOW, J. S. & GUZMAN, S. G. 2000. Variation in Stand Structure, Light and Seedling Abundance across a Tropical Moist Forest Chronosequence, Panama. *Journal of Vegetation Science,* 11**,** 201-212.

DEWEES, P. A., CAMPBELL, B. M., KATERERE, Y., SITOE, A., CUNNINGHAMS, A. B., ANGELSEN, A. & WUNDER, S. 2010. Managing the miombo woodlands of southern Africa: Policies, incentives and options for the rural poor. *Natural Resources Policy Research,* 2**,** 57-73.

DITT, E. H., MOURATO, S., GHAZOUL, J. & KNIGHT, J. 2010. FOREST CONVERSION AND PROVISION OF ECOSYSTEM SERVICES IN THE BRAZILIAN ATLANTIC FOREST. *Land Degradation & Development,* 21**,** 591-603.

DWYER, J. M., FENSHAM, R. J., BUTLER, D. W. & BUCKLEY, Y. M. 2009. Carbon for conservation: Assessing the potential for win-win investment in an extensive Australian regrowth ecosystem. *Agriculture, Ecosystems & Environment,* 134**,** 1-7.

FERREIRA, L. V. & PRANCE, G. T. 1999. Ecosystem recovery in terra firme forests after cutting and burning: a comparison on species richness, floristic composition and forest structure in the Jaú National Park, Amazonia. *Botanical Journal of the Linnean Society,* 130**,** 97-110.

FIGUEIREDO, J. & PEREIRA, H. 2011. Regime shifts in a socio-ecological model of farmland abandonment. *Landscape Ecology,* 26**,** 737-749.

FISHER, B., LEWIS, S. L., BURGESS, N. D., MALIMBWI, R. E., MUNISHI, P. K., SWETNAM, R. D., KERRY TURNER, R., WILLCOCK, S. & BALMFORD, A. 2011. Implementation and opportunity costs of reducing deforestation and forest degradation in Tanzania. *Nature Clim. Change,* 1**,** 161-164.

FROST, P. 1996. *The ecology of miombo woodlands,* Bogor, Center for international forestry research (CIFOR)

GIBBS, H. K., BROWN, S., NILES, J. O. & FOLEY, J. A. 2007. Monitoring and estimating tropical forest carbon stocks: making REDD a reality. *Environmental Research Letters,* 2.

GRACE, J., JOSÉ, J. S., MEIR, P., MIRANDA, H. S. & MONTES, R. A. 2006. Productivity and carbon fluxes of tropical savannas. *Journal of Biogeography,* 33**,** 387-400.

GRUNDY, I. M. 1995. Wood biomass estimation in dry miombo woodland in Zimbabwe. *Forest Ecology and Management,* 72**,** 109-117.

GRZ 1998. An overview of Copperbelt Forestry Action Plan. Ndola: PFAP.

GUARIGUATA, M. R., CHAZDON, R. L., DENSLOW, J. S., DUPUY, J. M. & ANDERSON, L. 1997. Structure and floristics of secondary and old-growth forest stands in lowland Costa Rica. *Plant Ecology,* 132**,** 107-120.

GUARIGUATA, M. R. & OSTERTAG, R. 2001. Neotropical secondary forest succession: changes in structural and functional characteristics. *Forest Ecology and Management,* 148**,** 185-206.

GUTIÉRREZ, A. G. & HUTH, A. 2012. Successional stages of primary temperate rainforests of Chiloé Island, Chile. *Perspectives in Plant Ecology, Evolution and Systematics,* 14**,** 243-256.

GUY, P. R. 1981. CHANGES IN THE BIOMASS AND PRODUCTIVITY OF WOODLANDS IN THE SENGWA WILDLIFE RESEARCH AREA, ZIMBABWE. *Journal of Applied Ecology,* 18**,** 507-519.

HOGBERG, P. 1982. MYCORRHIZAL ASSOCIATIONS IN SOME WOODLAND AND FOREST TREES AND SHRUBS IN TANZANIA. *New Phytologist,* 92**,** 407-415.

IUCN. 2011. *The value of investing in locally-controlled forestry* [Online]. International Union for Conservation of Nature. Available: http://www.iucn.org/about/work/programmes/forest/?6853/valuelocally-controlledforestry [Accessed 10th August 2011].

JACOBS, M., KRUK, R. & OLDEMAN, R. A. A. 1988. *The tropical rain forest: a first encounter*, Springer.

JOHNSON, C. D. & SKOUSEN, J. G. 1990. *TREE SPECIES COMPOSITION, CANOPY COVERAGE, AND IMPORTANCE ON SEVERAL AML SITES IN NORTHERN WEST-VIRGINIA*.

KALABA, K. F., CHIRWA, P., SYAMPUNGANI, S. & AJAYI, C. O. 2010. Contribution of agroforestry to biodiversity and livelihoods improvement in rural communities of Southern African regions. *Tropical Rainforests and Agroforests under Global Change: Ecological and Socio-economic Valuations***,** 461-476.

KAPPELLE, M., GEUZE, T., LEAL, M. E. & CLEEF, A. M. 1996. Successional age and forest structure in a Costa Rican upper montane Quercus forest. *Journal of Tropical Ecology,* 12**,** 681-698.

KOTTO-SAME, J., WOOMER, P. L., APPOLINAIRE, M. & LOUIS, Z. 1997. Carbon dynamics in slash-and-burn agriculture and land use alternatives of the humid forest zone in Cameroon. *Agriculture, Ecosystems & Environment,* 65**,** 245-256.

LAWTON, R. M. 1978. STUDY OF DYNAMIC ECOLOGY OF ZAMBIAN VEGETATION. *Journal of Ecology,* 66**,** 175-&.

LEAKEY, R. R. B. & AKINNIFESI, F. K. 2008. Towards a domestication strategy for indigenous fruit trees in the tropics. *Indigenous Fruit Trees in the Tropics: Domestication, Utilization and Commercialization***,** 28-49.

LEWIS, S. L., LOPEZ-GONZALEZ, G., SONKE, B., AFFUM-BAFFOE, K., BAKER, T. R., OJO, L. O., PHILLIPS, O. L., REITSMA, J. M., WHITE, L., COMISKEY, J. A., DJUIKOUO K, M.-N., EWANGO, C. E. N., FELDPAUSCH, T. R., HAMILTON, A. C., GLOOR, M., HART, T., HLADIK, A., LLOYD, J., LOVETT, J. C., MAKANA, J.-R., MALHI, Y., MBAGO, F. M., NDANGALASI, H. J., PEACOCK, J., PEH, K. S. H., SHEIL, D., SUNDERLAND, T., SWAINE, M. D., TAPLIN, J., TAYLOR, D., THOMAS, S. C., VOTERE, R. & WOELL, H. 2009. Increasing carbon storage in intact African tropical forests. *Nature,* 457**,** 1003-U3.

MAASS, J. M., BALVANERA, P., CASTILLO, A., DAILY, G. C., MOONEY, H. A., EHRLICH, P., QUESADA, M., MIRANDA, A., JARAMILLO, V. J., GARCIA-OLIVA, F., MARTINEZ-YRIZAR, A., COTLER, H., LOPEZ-BLANCO, J., PEREZ-JIMENEZ, A., BURQUEZ, A., TINOCO, C., CEBALLOS, G., BARRAZA, L., AYALA, R. & SARUKHAN, J. 2005. Ecosystem services of tropical dry forests: Insights from long-term ecological and social research on the Pacific Coast of Mexico. *Ecology and Society,* 10**,** Article No.: 17.

MAGURRAN, A. E. 2004. *Measuring Biological Diversity,* Oxford, Blackwell.

MALAMBO, F. M. & SYAMPUNGANI, S. 2008. Opportunities and challenges for sustainable management of miombo woodlands: the Zambian perspective. *Working Papers of the Finnish Forest Research Institute* 98**,** 125-130.

MALHI, Y. & GRACE, J. 2000. Tropical forests and atmospheric carbon dioxide. *Trends in Ecology & Evolution,* 15**,** 332-337.

MALIMBWI, R. E., SOLBERG, B. & LUOGA, E. 1994. Estimation of biomass and volume in miombo woodland at Kitulangalo Forest Reserve, Tanzania. *Journal of Tropical Forest Science,* 7**,** 230-242.

MARGALEF, R. 1972. HOMAGE TO EVELYN HUTCHINSON OR WHY THERE IS AN UPPER LIMIT TO DIVERSITY. *Transactions Connecticut Academy of Arts and Sciences,* 44**,** 211-235.

MBATA, K. J., CHIDUMAYO, E. N. & LWATULA, C. M. 2002. Traditional regulation of edible caterpillar exploitation in the Kopa area of Mpika district in northern Zambia. *Journal of Insect Conservation,* 6**,** 115-130.

MCGREGOR, J. 1995. GATHERED PRODUCE IN ZIMBABWE COMMUNAL AREAS CHANGING RESOURCE AVAILABILITY AND USE. *Ecology of Food and Nutrition,* 33**,** 163-193.

MENG, J., LU, Y., LEI, X. & LIU, G. 2011. Structure and floristics of tropical forests and their implications for restoration of degraded forests of China's Hainan Island. *Tropical Ecology,* 52**,** 177-191.

MITTERMEIER, R. A., MITTERMEIER, C. G., BROOKS, T. M., PILGRIM, J. D., KONSTANT, W. R., DA FONSECA, G. A. B. & KORMOS, C. 2003. Wilderness and biodiversity conservation. *Proceedings of the National Academy of Sciences of the United States of America,* 100**,** 10309-10313.

MUNISHI, P. K. T., MRINGI, S., SHIRIMA, D. D. & LINDA, S. K. 2010. The role of the Miombo Woodlands of the Southern Highlands of Tanzania as carbon sinks. *Journal of Ecology and the Natural Environment,* 2 261-269.

MUNISHI, P. K. T., PHILIPINA, F., TEMU, R. P. C. & PIMA, N. E. 2008. Tree species composition and local use in agricultural landscapes of west Usambaras Tanzania. *African Journal of Ecology,* 46**,** 66-73.

MUNISHI, P. K. T. & SHEAR, T. H. 2004. Carbon storage in afromontane rain forests of the Eastern Arc Mountains of Tanzania: Their net contribution to atmospheric carbon. *Journal of Tropical Forest Science,* 16**,** 78-93.

MUNYANZIZA, E. 1996. *Domestication of mushrooms from the miombo woodlands: current status and crucial issues for agroforestry* [Online]. FAO. Available: http://www.fao.org/docrep/w3735e/w3735e26.htm [Accessed 30 th August 2012].

MWAMPAMBA, T. H. & SCHWARTZ, M. W. 2011. The effects of cultivation history on forest recovery in fallows in the Eastern Arc Mountain, Tanzania. *Forest Ecology and Management,* 261**,** 1042-1052.

NASI, R., WUNDER, S. & CAMPOS, J. J. 2002. Forestry ecosystem services: can they pay our way out of deforestation? . *Discussion paper prepared for the* *forestry roundtable.* New York: Global Environment Facility (GEF), United Nations Forum on Forestry (UNFF) II.

ORWA, C., MUTUA, A., KINDT, R., JAMNADASS, R. & SIMONS, A. 2009. *Agroforestree Database:a tree reference and selection guideversion 4.0* [Online]. World Agroforestry Centre. Available: http://www.worldagroforestry.org/treedb2/AFTPDFS/Uapaca_kirkiana.pdf [Accessed 31 st August 2012].

PATTON, M. Q. 1990. *Qualitative evaluation and research methods,* London, SAGE Publications.

PETERS, C. M. 1994. *Sustainable Harvest of Non-timber Plant Resources in Tropical Moist Forest: An Ecological Primer* [Online]. Washington DC. Available: http://www.harmlesscoconut.com/resources/14.pdf [Accessed 6th Agust 2012].

PRSP 2002. Poverty Reduction Strategic Paper. Lusaka: Government of the Republic of Zambia.

ROBERTSON, E. F. 1984. *Regrowth of Two African Woodland Types After Shifting Cultivation.* PhD, Aberdeen University.

RODGERS, A., SALEHE, J. & HOWARD, G. 1996. The biodiversity of miombo woodlands. *In:* CAMPBELL, B. (ed.) *The miombo in transition: woodlands and welfare in Africa.* Bogor: Center for international forestry research (CIFOR).

SALDARRIAGA, J. G., WEST, D. C., THARP, M. L. & UHL, C. 1988. Long-Term Chronosequence of Forest Succession in the Upper Rio Negro of Colombia and Venezuela. *Journal of Ecology,* 76**,** 938-958.

SASAKI, N., ASNER, G., KNORR, W., DURST, P., PRIYADI, H. & PUTZ, F. 2011. Approaches to classifying and restoring degraded tropical forests for the anticipated REDD+ climate change mitigation mechanism. *iForest - Biogeosciences and Forestry,* 4**,** 1-6.

SCHONGART, J., ARIEIRA, J., C. FELFILI FORTES, ARRUDA, E. C. D. & CUNHA, C. N. D. 2008. Carbon dynamics in aboveground coarse wood biomass of wetland forests in the northern Pantanal, Brazil

Biogeosciences Discuss, 5**,** 2103–2130.

SHACKLETON, S. & GUMBO, D. 2010. Contribution of non-wood forest products to livelihoods and poverty alleviation. *In:* CHIDUMAYO, E. N. & GUMBO, D. J. (eds.) *The dry forests and woodlands of Africa: managing for products and services.* London: Earthscan.

SHANNON, C. E. 1948. A MATHEMATICAL THEORY OF COMMUNICATION. *Bell System Technical Journal,* 27**,** 379-423.

SHIRIMA, D. D., MUNISHI, P. K. T., LEWIS, S. L., BURGESS, N. D., MARSHALL, A. R., BALMFORD, A., SWETNAM, R. D. & ZAHABU, E. M. 2011. Carbon storage, structure and composition of miombo woodlands in Tanzania's Eastern Arc Mountains. *African Journal of Ecology,* 49**,** 332-342.

STRANG, R. M. 1974. Some Man-Made Changes in Successional Trends on the Rhodesian Highveld. *Journal of Applied Ecology,* 11**,** 249-263.

STROMGAARD, P. 1984. The immediate effect of burning and ash-fertilization. *Plant and Soil,* 80**,** 307-320.

STROMGAARD, P. 1985. BIOMASS, GROWTH, AND BURNING OF WOODLAND IN A SHIFTING CULTIVATION AREA OF SOUTH CENTRAL-AFRICA. *Forest Ecology and Management,* 12**,** 163-178.

STROMGAARD, P. 1987. EARLY SECONDARY SUCCESSION ON ABANDONED SHIFTING CULTIVATOR'S PLOTS IN THE MIOMBO OF SOUTH CENTRAL AFRICA. *Biotropica,* 18**,** 97-106.

SYAMPUNGANI, S. 2009. *Vegetation change analysis and ecological recovery of the Copperbelt Miombo woodland of Zambia.* PhD, University of Stellenbosch.

SYAMPUNGANI, S., CHIRWA, P. W., AKINNIFESI, F. K., SILESHI, G. & AJAYI, O. C. 2009. The miombo woodlands at the cross roads: Potential threats, sustainable livelihoods, policy gaps and challenges. *Natural Resources Forum,* 33**,** 150-159.

SYAMPUNGANI, S., GELDENHUYS, C. J. & CHIRWA, P. W. 2010. The use of species-stem curves in sampling the development of the Zambian miombo woodland species in charcoal production and slash-and-burn regrowth stands. *Southern Forests,* 72**,** 83-89.

TRAPNELL, C. G. 1959. Ecological Results of Woodland and Burning Experiments in Northern Rhodisia. *Journal of Ecology,* 47**,** 129-168.

TRAPNELL, C. G., FRIEND, M. T., CHAMBERLAIN, G. T. & BIRCH, H. F. 1976. The Effects of Fire and Termites on a Zambian Woodland Soil. *Journal of Ecology,* 64**,** 577-588.

UNFCCC. 2009. *Decision -/ CP15. "Copenhagen Accord"*

[Online]. Available: http://unfccc.int/files/meetings/cop_15/application/pdf/cop15_cph_auv.pdf [Accessed 20th September 2012].

UNFCCC. 2010. *Decision -/ CP.16* [Online]. Available: http://unfccc.int/files/meetings/cop_16/application/pdf/cop16_lca.pdf [Accessed 20th September 2012].

UNFCCC. 2011. *Draft decision -/ CP.17* [Online]. Available: http://unfccc.int/files/meetings/durban_nov_2011/decisions/application/pdf/cop17_lcaoutcome.pdf [Accessed 20th September 2012].

VAN ANDEL, T. 2001. Floristic composition and diversity of mixed primary and secondary forests in northwest Guyana. *Biodiversity and Conservation,* 10**,** 1645-1682.

VINYA, R., MALHI, Y., BROWN, N. & FISHER, J. 2012. Functional coordination between branch hydraulic properties and leaf functional traits in miombo woodlands: implications for water stress management and species habitat preference. *Acta Physiologiae Plantarum***,** 1-10.

VON DER HEYDEN, C. J. & NEW, M. G. 2004. Groundwater pollution on the Zambian Copperbelt: deciphering the source and the risk. *Science of the Total Environment,* 327**,** 17-30.

WALKER, S. M. & DESANKER, P. V. 2004. The impact of land use on soil carbon in Miombo Woodlands of Malawi. *Forest Ecology and Management,* 203**,** 345-360.

WHITE, F. 1983. The Vegetation of Africa. *UNESCO, Natural Resources Research,* 20**,** 86-101.

WILLIAMS, M., RYAN, C. M., REES, R. M., SARNBANE, E., FEMANDO, J. & GRACE, J. 2008. Carbon sequestration and biodiversity of re-growing miombo woodlands in Mozambique. *Forest Ecology and Management,* 254**,** 145-155.

WILSON, K. B. 1989. TREES IN FIELDS IN SOUTHERN ZIMBABWE. *Journal of Southern African Studies,* 15**,** 369-383.

WRI. 2011. *Opportunities for forest and landscape restoration in Africa* [Online]. World Resources Institute. Available: http://www.wri.org/map/opportunities-forest-and-landscape-restoration-africa [Accessed 10th August 2011].