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Floristic composition, species diversity and carbon storage in charcoal and agriculture fallows and management implications in Miombo woodlands of Zambia

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Abstract

Globally, there are increasing demands for land use changes aimed at restoring Carbon (C) and biodiversity in degraded forest ecosystems. This study provides an integrated understanding of aboveground (AG) C storage, structural and floristic composition in charcoal and agriculture fallows in Miombo woodland systems of Zambia. We present the findings of 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 Mg C ha^{-1} AG, while after clearance, C stocks accumulated at 0.98 and 1.42 Mg 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 decreased with fallow age. Importance values (IVI) of tree species show low presence of less fire resistant tree species such as *Uapaca kirkiana* in the initial regrowth of post agriculture 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 systems recover relatively fast in terms of species diversity and C storage, species composition takes longer to recuperate. The findings show that agriculture and charcoal fallows hold enormous management potential for emerging C-based payments for ecosystem services such as through United Nations Reduction of Emissions from Deforestation and forest Degradation-

plus (REDD+) programme and Voluntary Carbon Market projects. 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; Species diversity; Fallow; Floristic composition; Miombo woodland; REDD+

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 (Nasi et al., 2002; Gibbs et al., 2007). There is growing interest in understanding the capacity of forest ecosystems to sequester and store C in developing countries (Walker et al., 2004), which 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). Miombo woodland is the most extensive dry forest formation in Africa, with an estimated area of 2.7million km^2 (White, 1983; Frost, 1996), and is rich in plant diversity, with about 8500 species of higher plants of which 54% are endemic (Chirwa et al., 2008), making them one of the world's high-biodiversity hotspots (Mittermeier et al., 2003).

The C cycle in Miombo and other tropical woodlands is comparatively understudied (Williams et al., 2008; Bombelli et al., 2009). 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 C stores in woodlands uncertain (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 et al., 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. Slash and burn agriculture and charcoal production are the major causes of forest loss in Miombo woodlands (Stromgaard, 1987; Chidumayo, 1991; Malambo et al., 2008), and have been linked to huge losses of C and biodiversity of forest systems (Kotto-Same et al., 1997). Vegetation structure and floristic compositional changes in forest recovery has been discussed mainly in post -slash and burn agriculture abandonment sites in tropical rainforests (Guariguata et al., 1997; Ferreira et al., 1999; Denslow et al., 2000), with a few studies in African woodlands (Williams et al., 2008; Syampungani et al., 2010), though floristic composition in regrowth sites remains contested. Some studies (e.g Stromgaard, 1985; Kappelle et al., 1996; 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). Furthermore, some studies have suggested it takes centuries for forest to return to primary forest species composition and argue that forests may not return to their original composition after severe disturbances (Jacobs et al., 1988; Meng et al., 2011).

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 et al., 2012). The aim of this study 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 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; Stringer et al., 2012).

2 Research design and methods

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).

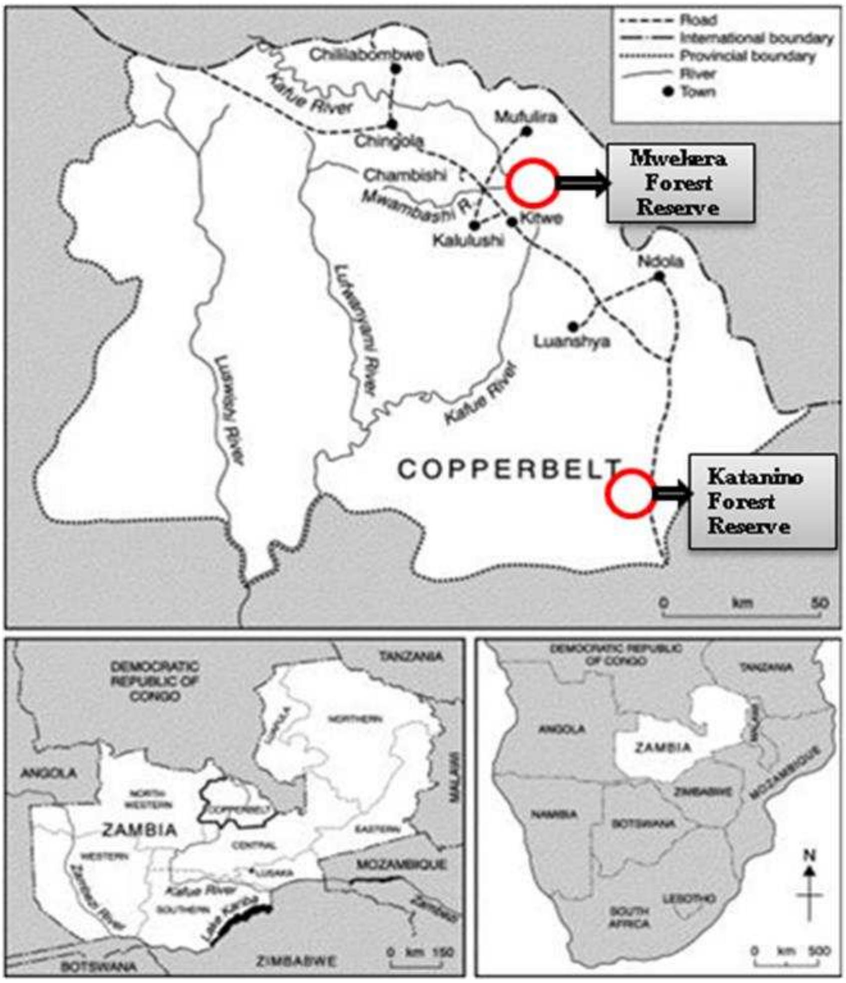

Fig 1: Location of study sites

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.

2.2 Site selection and data collection

The study sites were selected using stratified purposive sampling (Creswell, 1998). Three different land use categories (i.e. treatments) were identified for Miombo woodlands; (1) undisturbed Miombo, (2) Slash and burn fallows, and (3) Charcoal fallows.

We used analogous sites to provide insights on changes in floristic composition and carbon storage overtime. Investigating succession using analogous sites (spatial) rather than temporal chronosequence has a limitation of ensuring various stands of different ages along the identified chronosequence have similar soils, vegetation composition, climatic histories, and previously subjected to similar disturbances (Schoonmaker and McKee 1988). This challenge was addressed by conducting the study in the same sub-region agro-ecological zone and creating a criterion for sample selection in the different land-use categories (Table 1).

Table 1: Descriptions of main characteristics of land use categories

Category	**Main characteristics**
Undisturbed Miombo	• Not been cleared or cultivated. • No records of forest management treatments as supported by Forest Department records. • Not experienced any major human or natural disturbances.
Slash and burn fallows	• Abandoned after slash and burn agriculture. • Knowledge of age of fallows. • Non-mechanised tillage. • Rain fed. • No evidence of post-abandonment removal of some trees (e.g. cutting of trees for poles) • Free of agrochemicals. • Not experienced any major human or natural disturbances.
Charcoal fallows	• Abandoned after cutting trees for charcoal production. • Knowledge of age of fallow • No evidence of use of fallows for agriculture purposes. • No evidence of post-abandonment removal of some trees (e.g. cutting of trees for poles) • Not experienced any major human or natural disturbances.

Data collection was conducted from December 2011 to April 2012.

2.2.1 Sampling and plot establishment

Undisturbed Miombo

Ground inventories were done in the identified land use categories. Twenty-four 50 m x 50 m (0.25 ha) plots were established in undisturbed Miombo (i.e. 16 plots in Mwekera Forest Reserve and 8 in Katanino Forest Reserves). 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.

Recovering Miombo

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, after the criteria summarized in table 1), and then age of fallows, after which plots were established at random locations within the identified age categories. Land-use history and fallow age were obtained through informal interviews with local farmers, charcoal producers and traditional councillors (Ba filolo). 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 consistent with other studies in Miombo woodlands (Robertson, 1984; Walker et al., 2004; Syampungani, 2009; Mwampamba et al., 2011).

Slash and burn recovering fallows ranged between 5-58 years. 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 et al., 2004). 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). At least 4 plots were surveyed in recovering Miombo for each identified fallow age. These fall within the plot numbers used in similar studies (Williams et al., 2008; Syampungani, 2009).

2.2.2 Field measurements

In the established plots, the tree diameters were measured using a diameter tape at breast height (i.e. 1.3 m above ground) (Lawton, 1978; Malimbwi et al., 1994; Ditt et al., 2010) for all trees (trees defined as woody plants more than 2 m (Frost, 1996)). 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 lead authors' knowledge were also used in identifying tree species. For trees that were difficult to identify, voucher specimens were taken to the Kitwe Forest Research Herbarium for identification. We recorded a total of 8031 stems in the sampled plots.

2.3 Data analysis

2.3.1 Floristic indices and biodiversity

To describe the tree species composition and vegetation structure of the plots, this study used the Importance Values Index (IVI), which is a summation of the relative density, dominance and frequency of species, i.e.

IVI= (Relative frequency + relative basal area + relative density) / 3 (Curtis et al., 1951).

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.

J was calculated using the formula: J= A /(A+B+C) (Chidumayo, 1997)

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.

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

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

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 (Shannon, 1948; Chidumayo, 1997). The current study complemented the Shannon index with the Simpson index (D) which is a useful index for relatively small samples (Magurran, 2004). This was important in getting a better informed evidence of the biological diversity of trees, measured using two different diversity indices.

2.3.2 Quantifying aboveground C

We used allometric equations to estimate tree biomass (Table 2). 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. 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 (Malimbwi et al., 1994; Gibbs et al., 2007). 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 et al., 1982; Williams et al., 2008; Bryan et al., 2010).

Table 2: 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)	$B= D^{2.516} / e^{2.462}$	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 reverse 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.

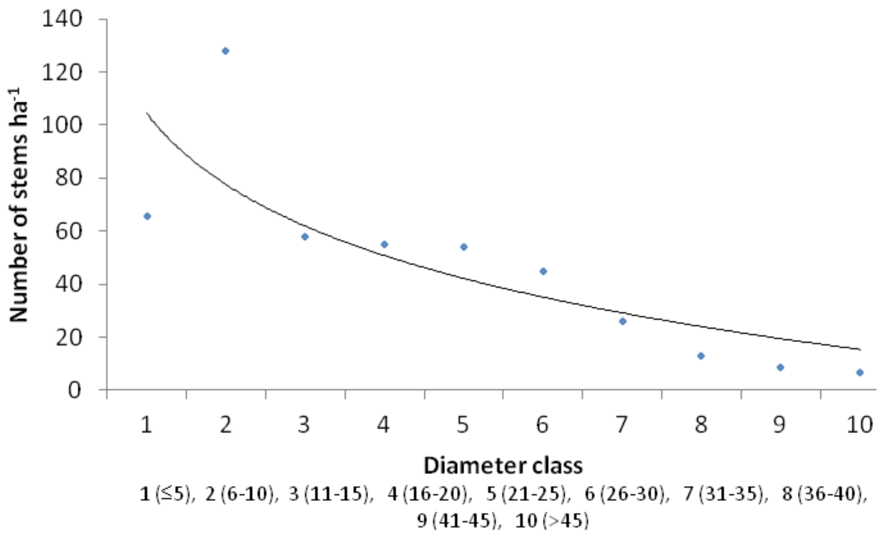

Fig 2: Diameter distribution showing reverse-J- shaped size classes

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).

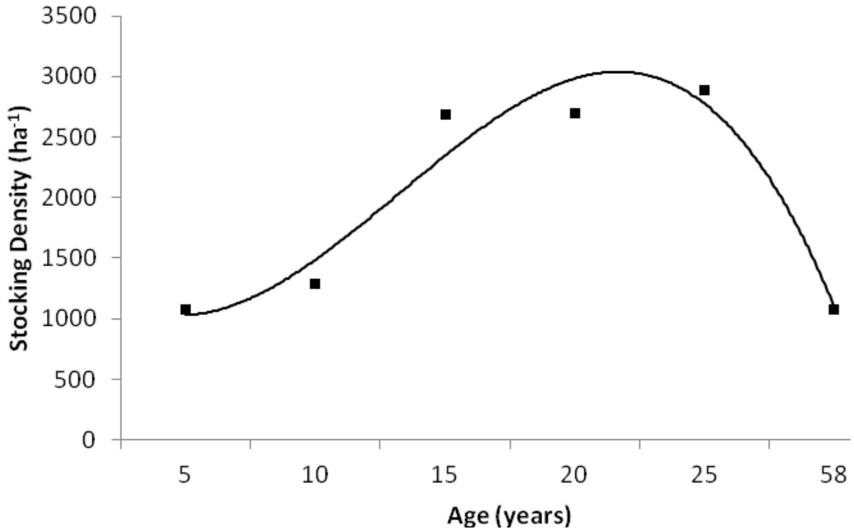

Fig 3: Stocking density (stems per ha) of slash and burn fallows plotted against age of plots

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.

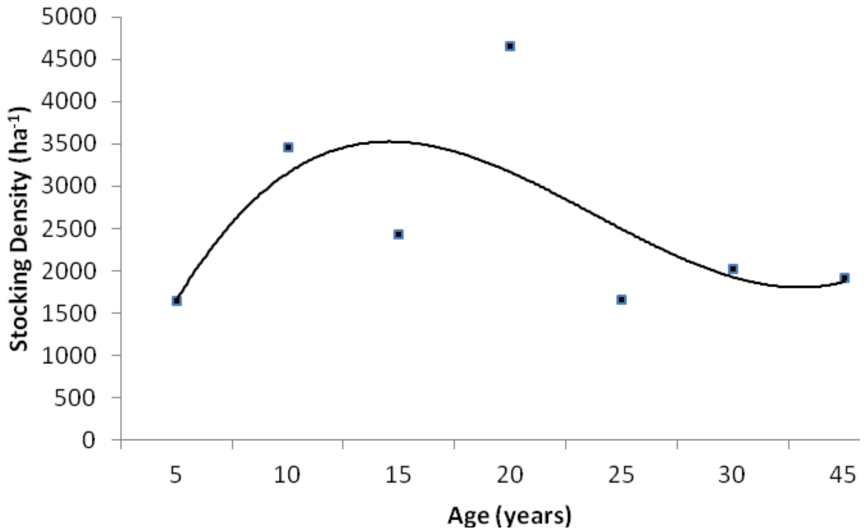

Fig 4: Stocking density of charcoal fallows plotted against age of plots

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 fallows at 10 – 15 years. Tree density then later decreased 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 reverse-J shape as observed in mature woodlands.

Basal area for both slash and burn and charcoal regrowth sites were significantly positively correlated with time since abandonment ($r^2 = 0.93$, $P < 0.001$ and $r^2 = 0.92$, $P < 0.001$ respectively) 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 $m^2 ha^{-1}$ along chronosequence of recovery), increasing at an average rate of 0.73 m^2ha^{-1}

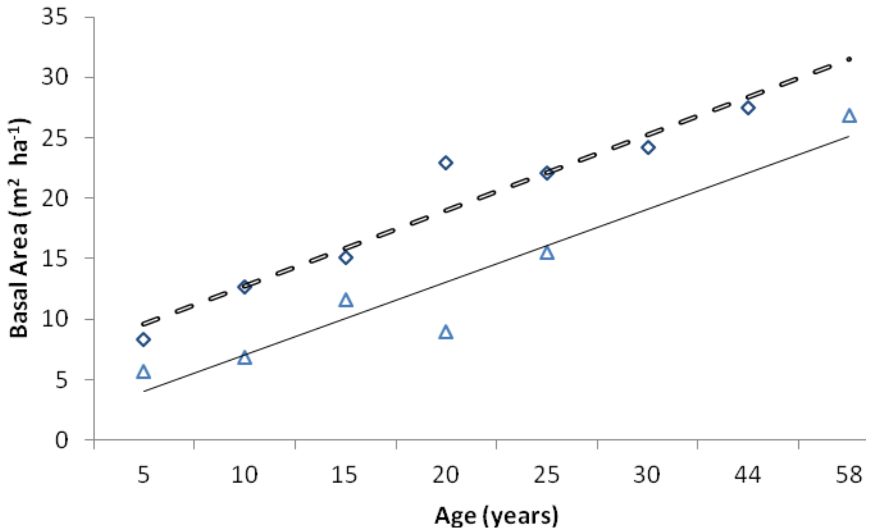

Fig 5: Basal area plotted against age of abandonment for Slash and burn (diamonds, and dash line) and charcoal (triangles, and solid line)

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 y and t represent the basal area and time after abandonment respectively.

3.1.2 Floristic composition

The total number of species identified in the mature woodlands was 83 belonging to 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, Marquesia macroura, and Diplorhynchus condylocarpon. The 20 most frequently occurring tree species in descending order are summarized in Table 3. These species are typical of the wet Miombo systems of this eco-region (Stromgaard, 1985; Vinya et al., 2012).

Table 3: 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	2.4	83.3	3.0	29.5
5	Pericopsis angolensis	2.1	79.2	1.2	27.5
6	Isoberlinia angolensis	5.5	66.7	8.4	26.9
7	Brachystegia speciformis	3.5	70.8	6.2	26.8
8	Pseudolachnostylis maprouneifolia	2.8	75	1.0	26.3
9	Monotes africanus	2.5	66.7	1.1	23.4
10	Brachystegia longifolia	3.3	62.5	3.7	23.1
11	Albizia antunesiana	3	62.5	1.5	22.3
12	Syzygium guineense	1.7	62.5	0.3	21.5
13	Ochna pulchra Hook	1.6	58.3	0.2	20.1
14	Phyllocosmus lemaireanus	4.3	54.2	0.6	19.7
15	Brachystegia boehmii	3	50	6.0	19.7
16	Uapaca kirkiana	2.4	54.2	1.0	19.2
17	Anisophyllea boehmii	2.9	54.2	0.6	19.2
18	Pterocarpus angolensis	1.0	54.2	0.3	18.5
19	Baphia bequaertii	1.9	50	0.5	17.5
20	Brachystegia floribunda	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 4 and 5).

It was observed that in the early recovering plots (5-10 years), D. condylocarpon dominated slash and burn followed by I. angolensis, Securidaca longepedunculata, Bridelia micrantha 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 4: The ten most dominant species, ranked by IVI (in parenthesis) in each age class of abandoned slash and burn fallow, species richness, Jaccard similarity coefficient and diversity indices

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 (36.3)	B. bequaertii (39.2)	B. speciformis (38.4)	O. pulchra (38.2)	Swartzia madagascariensis (37.3)	B. floribunda (37.9)
4	B. micrantha (36.1)	D. condylocarpon (39.0)	Uapaca nitida (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 . (23.9)	Dichrostachys cinerea. (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 (33.9)	Vitex doniana (12.6)	B. floribunda (30.3)	Lannea discolour (35.3)
8	B. floribunda (28.2)	B. speciformis (27.0)	Strychnos pungens (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	S. guineense (12.6)	Hymenocardia acida	O. pulchra (27.6)

			(20.6)		(27.3)	
10	Ekebergia benguelensis (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
H	2.1	2.4	2.5	2.1	2.6	2.8
D	0.80	0.87	0.89	0.83	0.90	0.92

Where; J = Jaccard similarity coefficient, H = Shannon index and D = Simpson diversity index.

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 throughout the chronosequence (Table 5).

Table 5: The ten most dominant species, ranked by IVI (in parenthesis) in each age class of abandoned charcoal fallow, species richness, Jaccard similarity coefficient and diversity indices

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 (34.1)	B. boehmii (36.5)	I. angolensis (28.4)	P. curatellifolia (19.5)	Uapaca benguelensis (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. speciformis (26.7)	M. africana (35.5)	Brachystegia manga (24.3)	P. lemaireanus (34.9)	A. boehmii (26.1)	Dalbergia nitudula (18.4)	U. kirkiana (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
H	2.0	2.4	2.3	2.5	2.6	2.6	2.7
D	0.78	0.88	0.87	0.89	0.90	0.89	0.91

Where; J = Jaccard similarity coefficient, H = Shannon index and D = Simpson diversity index.

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 4). In charcoal regrowth sites, the Jaccard coefficient ranged from a minimum of 0.26 to a maximum of 0.44 (Table 5). 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.

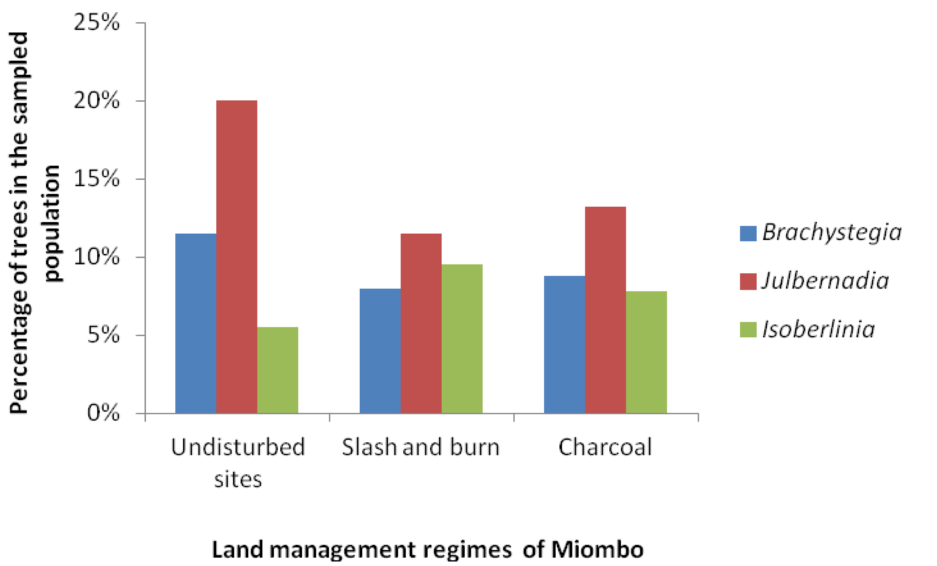

Fig 6: Distribution of dominant leguminous genera in undisturbed site and oldest regrowth sites

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. There was however 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). There was a significant positive correlation between species richness and 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 ($F = 1.48$, $P = 0.24$).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 (Table 4). In charcoal regrowth plots, *H'* values ranged from 2 to 2.7 with diversity increasing with age (Table 5). In mature woodlands, the mean *H'* was 2.8 ± 0.1, while the Simpson index (D) was 0.92. 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 estimated C was 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 regeneration 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 differences in carbon storage were also observed at 15-16 years, while there were no significant differences in C storage between the two management regimes after 20 years (Table 6).

Table 6: 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. Carbon estimates in regrowth stands were positively and significantly correlated with 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 (Peters, 1994). When the forest canopy closes, some seedlings are stunted as some Miombo species require high light intensities for growth (Chidumayo et al., 1996). The diameter distribution obtained at 25 years old regrowth and older suggests that the Miombo has the capacity to achieve its mature vegetation structure after 25 years of abandonment. Our findings add to previous Miombo ecological assessments by demonstrating that Miombo systems return to primary forest characteristics within 2-3 decades of fallow after being degraded through either charcoal and agriculture production. This finding is 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 where trees are sometimes uprooted in land preparation, reducing future sources of propagules. Furthermore, 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 et al., 1999). The stem density per hectare declines with age of regrowth due to inter-shoot competition (Chidumayo, 1988b; Chidumayo, 1988a). 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 et al., 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 Strang, 1974; Lawton, 1978). These findings are consistent with the findings of Peters (1994) and Stromgaard (1984) who 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. High-intensity fires and subsequent high soil temperatures during slash and burn causes mortality of plant propagules of fire susceptible tree species (Beadle, 1940) which affects the rate of post-fire recolonisation. 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) which 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. 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 from the plots measured in their Mozambique study. The changes in species dominance along the chronosequence may be explained by the fact that tree species such as D. condylocarpon, B. Bequaertii dominate in initial Miombo recovery after slash and burn due to their rapid dispersal ability and fire tolerance, and occupy the 'empty area' (Strang, 1974; Lawton, 1978). In the middle stages of recovery, reduction in incidences of fire enhances growth conditions for less-fire resistant and light demanding species such as Uapaca spp and Albizia spp. These species are shade intolerant and cannot continue to grow under their own shade (Stromgaard, 1987). They start reducing with the age of the forest stand (Connell et al., 1977; Saldarriaga et al., 1988) as dominant Miombo species increase thus explaining the changes in species dominance. Initial stages of charcoal regrowth sites are dominated by a mixture of fire–tolerant and less tolerable species, while the presence of key Miombo woodland species in early recovery stages can be attributed to regeneration from stumps shoots and root suckers (Chidumayo, 1997; Stromgaard, 1985).

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 where 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) probably due to the comparable rainfall gradients. These results corroborate that within the Miombo region, our study region is biologically diverse at tree species level and could be important for various biogeochemical cycles since diversity often is indicative of better ecosystem functioning/productivity (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). Our findings show 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 et al., 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

Carbon 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), and Munishi et al., (2010), i.e. 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 drier Miombo than Copperbelt Zambia. Our estimated carbon storage is lower than estimates from tropical rainforests of Africa i.e. 202 Mg C ha^{-1} and over 350 Mg C ha^{-1} (Lewis et al., 2009, Munishi and Shear 2004). 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 C 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 et al., 1966; Guy, 1981; Chisha-Kasumu et al., 2007). Miombo species' main regeneration is through coppice regrowth and root suckers as opposed to seeds (Trapnell, 1959; Strang, 1974). 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 decrease with increasing fallow period and is not significantly different. 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). Recovery of forests is slow after disturbances that affect soil and aboveground vegetation (Chazdon, 2003). Cultivation using hand hoes has the potential to disturb the soil structure. This may partially explain why carbon accumulation was higher in charcoal regrowth plots than slash and burn agriculture. Furthermore, seedlings may have been left during charcoal production, therefore increasing C storage rapidly.

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 (Stromgaard, 1985; Kotto-Same et al., 1997; Williams et al., 2008) i.e. 0.7, 0.98 and 0.98 Mg 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 et al., 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 ($\geq$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. It should 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. 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 the management of the fallows.

5 Conclusions

Findings of this study have shown that the Miombo is a substantial AG C store. Once mature Miombo woodland are cleared, aboveground C stocks are reduced by 39.6 Mg C ha^{-1}, and after abandonment and subsequent recovery through natural succession, vegetation C accumulates at rates of 0.98 and 1.42 Mg 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. Miombo woodlands are able to achieve mature vegetation structure (DBH, basal area) after 20 years of abandonment. 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 fallows 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.

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. 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 can guide policy makers for understanding carbon changes in forests and biodiversity and in developing policy interventions on which the emerging initiatives of C payments are to be based.

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