Available online at www.sciencedirect.com
ScienceDirect
ELSEVIER
Forest Ecology and Management xxx (2007) xxx–xxx
Forest Ecology and Management
www.elsevier.com/locate/foreco

Carbon sequestration and biodiversity of re-growing miombo woodlands in Mozambique

M. Williams [a,*], C.M. Ryan [a], R.M. Rees [b], E. Sambane [a], J. Fernando [a], J. Grace [a]

[a] *School of GeoSciences, IAES, University of Edinburgh, Edinburgh EH9 3JN, UK*
[b] *SAC, West Mains Road, Edinburgh EH9 3JG, UK*

Received 20 March 2007; received in revised form 24 July 2007; accepted 31 July 2007

Abstract

Land management in tropical woodlands is being used to sequester carbon (C), alleviate poverty and protect biodiversity, among other benefits. Our objective was to determine how slash-and-burn agriculture affected vegetation and soil C stocks and biodiversity on an area of miombo woodland in Mozambique, and how C stocks and biodiversity responded once agriculture was abandoned. We sampled twenty-eight 0.125 ha plots that had previously been cleared for subsistence agriculture and had been left to re-grow for 2 to ∼25 years, and fourteen 0.25 ha plots of protected woodlands, recording stem diameter distributions and species, collecting wood for density determination, and soil from 0 to 0.3 m for determination of %C and bulk density. Clearance for agriculture reduced stem wood C stocks by 19.0 t C ha^{-1}. There were significant relationships between period of re-growth and basal area, stem numbers and stem biomass. During re-growth, wood C stocks accumulated at 0.7 t C ha^{-1} year^{-1}. There was no significant difference in stem C stocks on woodlands and on abandoned farmland 20–30 years old. Soil C stocks in the top 0.3 m on abandoned land had a narrower range (21–74 t C ha^{-1}) than stocks in woodland soils (18–140 t C ha^{-1}). There was no discernible increase in soil C stocks with period of re-growth, suggesting that the rate of accumulation of organic matter in these soils was very slow. The re-growing plots did not contain the defining miombo species, and total stem numbers were significantly greater than in woodland plots, but species richness and diversity were similar in older abandonments and miombo woodlands. Wood C stocks on abandoned farmland were capable of recovery within 2–3 decades, but soil C stocks did not change on this time-scale. Woodland soils were capable of storing >100 t C ha^{-1}, whereas no soil on a re-growing area exceeded 74 t C ha^{-1}, so there is a potential for C sequestration in soils on abandoned farmland. Management should focus on identifying C-rich soils, conserving remaining woodlands to protect soil C and preserve defining miombo species, and on investigating whether fire control on recovering woodland can stimulate accumulation of soil C and greater tree biomass, and restore defining miombo species.

Keywords: Savanna; Tropical deciduous forest; Basal area; Allometric equations; Wood density; Soil bulk density; Soil C stock; Biodiversity; Shannon index; Chronosequence

1. Introduction

The C cycle of tropical open woodlands is relatively understudied compared to other biomes. These woodlands are subject to frequent disturbance via fires and land clearance. Such woodland degradation threatens terrestrial carbon stocks (Chidumayo, 2002; Frost, 1996) but is little monitored or modelled. Climate change mitigation initiatives resulting from the United Nations Framework Convention on Climate Change are now managing tropical woodlands to sequester carbon (Silver et al., 2004), including 19 current projects in sub-Saharan Africa (Jindal, 2006). Alongside the potential to generate income through sales of C offsets, woodland management is likely to benefit other ecosystem services and biodiversity. However, the optimal management approaches are not yet clear, due to a lack of data on vegetation and soil C stocks, as well as biodiversity indices, on the dominant land use types. Here we quantify changing C stocks and biodiversity along a chronosequence of abandoned farmland in Mozambique, and in nearby protected miombo woodlands.

Miombo is the vernacular term for the seasonally dry deciduous woodlands that are widespread across southern Africa, dominated primarily by genera *Brachystegia*, *Julber-*

* Corresponding author.
E-mail address: mat.williams@ed.ac.uk (M. Williams).

0378-1127/$ – see front matter © 2007 Elsevier B.V. All rights reserved.
doi:10.1016/j.foreco.2007.07.033

nadia and/or *Isoberlinia* (Campbell, 1996). These open woodlands extend across 2.7 million km^2 of some of the world's poorest countries (Campbell, 1996; Frost, 1996). Low per capita income and high population growth rates in southern Africa mean that subsistence slash-and-burn farming within the miombo zone is the predominant way of life. Growing populations are increasing pressure from slash-and-burn. Loss of miombo woodland is also driven by increasing demand for fuel-wood (Abbot and Homewood, 1999).

Our objective was to determine how slash-and-burn agriculture affected soil and vegetation C stocks on an area of miombo woodland, and how C stocks recovered once agriculture was abandoned. We hypothesised that C stocks in both soils and vegetation of abandoned slash-and-burn plots would be lower than in woodland plots. Farmland is abandoned after a few years as soil fertility declines, and we expected this fertility to recover slowly following abandonment. Thus, we hypothesised that C stocks would recover more rapidly in vegetation than in soils. We hypothesised that the species dynamics in recovering plots would indicate a return towards the dominant and defining species of local miombo woodland. We also hypothesised that this successional change in species would result in an increase in mean wood density as pioneer species were replaced with slower growing, denser miombo dominants. This study is unique in collecting and analysing data on soil and stem C (the largest stocks of C in miombo), wood density and biodiversity on a range of different land use types within a small region of miombo woodland.

2. Study site

This case study is located in the small community of N'hambita in Sofala Province, Mozambique, located around the operational centre of an EU-funded C sequestration pilot project at 18°58′44″S, 34°10′37″E. This is an area with little or no infrastructure, and a community still recovering from decades of war. Recent population growth means that more land is coming under the traditional slash-and-burn agricultural system. Local households clear machambas (slash-and-burn plots covering 1–3 ha) by felling trees and then burning them, and then raise crops, which are planted with a stick, or primitive entrenching tool. After a few years of tillage, yields fall and the machambas are abandoned to woodland re-growth.

The N'hambita community is located on the western escarpment of the southern limit of the Rift Valley, near the Gorongosa National Park, and its ecology is comprehensively described by Tinley (1977). The area receives 690 mm mean annual precipitation (ranging from 407 to 1219 mm), 96% of which falls between October and April, based on data from 1999 to 2005 (Mozambique Central Water Board, ARA-Centro, 2005). Soils are highly weathered and generally freely drained sandy loams or sandy silt loams. Fire is a frequent, generally annual, disturbance agent; during June–October 2006 most natural miombo vegetation in the area was burned (personal observation).

The N'hambita community lands, which covers an area of 348 km^2, can be divided into three zones. In the east, the community lands lie within the Gorongosa National Park, and there is no settlement, agriculture or land clearance. In the centre, the community lands lie within the Park buffer zone. There are settlements, and slash-and-burn agriculture, but land clearance is restricted legally. In the west, the community lands are outside the buffer zone, population density is greater, and land clearance is ongoing.

3. Methods

We surveyed tree biodiversity, above-ground woody biomass, wood density and soil C across an area of largely undisturbed primary woodlands and a series of abandoned machambas of differing ages, during 2004–2005. All survey plots were located in central N'hambita, within the buffer zone of Gorongosa National Park, to avoid areas with heavy disturbance.

3.1. Primary woodlands

During December 2004, surveys were undertaken of the woody vegetation in the part of the N'hambita community that lies within the buffer zone of the Gorongosa National Park. Plots were not surveyed if they showed signs of previous cultivation or charcoal burning, or if local informants knew them to have been utilised. Fourteen 0.25 ha plots were established in pairs at seven randomly selected locations, equally spaced along the road and track network within the community buffer zone. At the seven locations, two 50 m × 50 m plots were set out at 200, 450, 700 or 950 m from the road, along a transect line perpendicular to the road. The distances for the paired plots were selected randomly from these four options.

All living woody specimens >0.05 m diameter at breast height (DBH) were measured, recording species local name (provided by a local informant knowledgeable in botany), species botanical name (Coates Palgrave et al., 2002; de Koning, 1993; Van Wyk and Van Wyk, 1997; Van Wyk, 1993), and DBH (m) using diameter callipers or tape. On trees forking below 1.3 m from the ground level, each stem was measured and recorded separately. Trees forking above 1.3 m were measured at breast height. In one plot there was a single very large (2.17 m DBH) baobab tree (*Adansonia digitata*), a species with an unusual shape that does not conform to generic allometric relationships. No other baobabs were found anywhere else in the study and this single specimen was excluded from plot calculations as it heavily skewed the analyses for its plot compared to others.

3.2. Abandoned machambas

In June 2005, 28 abandoned agricultural fields were surveyed, each identified through talking with the farmers of the N'hambita community. Farmers were asked to locate machambas abandoned over past years and decades. The shortest period of re-growth was 2 years, and the longest exceeded 20 years. It was easier to obtain precise age estimates for more recently abandoned sites (up to 19 years). Site ages were confirmed by the secretary of the community association. Robertson (1984) used this method to obtain the fallow age of plots in Malawi and found it agreed well with estimates from aerial photographs. Several sites had been abandoned >20 years ago, but since independence in 1975, and so the ages for these sites were between 20 and 30 years. Abandoned machambas were divided into four classes, depending on fallow age: class 1, 1–5 years; class 2, 6–10 years; class 3, 11–19 years; and class 4, 20–30 years. Individual plots on age classes 1–3 have been aged precisely, but those on age class 4 have not, and may be aged between 20 and 30 years.

To sample the abandoned machambas, which usually extended over an area of 1–3 ha, each was divided into four approximately equal parts, and each of these parts was sampled by one 10 m radius subplot. Each subplot was randomly located, but locations were discounted that caused overlap between quadrants, or that placed subplots over the agreed boundary of the abandoned machamba. The total area of the four subplots was thus 1256 m^2, ~1/8 ha. Within each subplot, species and DBH for all stems >0.05 m DBH were recorded, a total of 1955 stems. The abandoned machambas were not randomly distributed in the area with respect to their age, but were clustered in groups of similar age.

3.3. Stem wood density

Dry bulk stem density was measured for the 21 most populous species found on the woodland and abandoned machamba plots. These species accounted for 980 of the 3166 stems enumerated, and 72% of the basal area. For the remaining, un-sampled species, each was assigned the average density, weighted by basal area, of species sampled on plots of the same age class.

To determine dry bulk density, during November 2005 one or more branches between 0.01 and 0.02 m diameter were taken from three or more trees and cut to ~0.15 m length, and the bark removed. Fresh volume was determined by displacement of water in a measuring cylinder (precision 1000 mm^3) and also by calculation from three measurements of diameter and one of length (precision 0.1 mm). The samples were returned to the lab and dried at 100 °C to constant weight. Density was calculated from the average of the fresh volume derived from the two methods, divided by the dry weight.

This method only determines the dry bulk density of the sapwood of the tree, which may differ from the density of the heartwood, which is present in most miombo species. However, sampling heartwood would have required the destruction of a large numbers of trees, so a sapwood–heartwood comparison was limited to five species, which had been felled fortuitously. The heartwood samples were much larger than the sapwood samples, either circular cross sections of the bole at different heights (0.1–0.2 m thick) or cubes sawn from the centre of a cross-section (~0.15 m × 0.15 m × 0.15 m). Volume was calculated both by measurement of diameter and thickness and also by measuring the weight of water displaced when placed into a full bucket. The wood samples were dried by episodic heating in a microwave to constant weight.

3.4. Stem C stocks

Stem biomass was calculated using allometric relationships (Table 1) developed in the miombo woodlands of nearby countries (Abbot et al., 1997; Chidumayo, 1997; Frost, 1996) and also a generic equation for the tropics (Brown et al., 1989). Two of the equations relate DBH or basal area to estimate volume, and then our wood density data were used to estimate biomass. The other two equations estimated biomass directly from DBH data. Multiple approaches to biomass estimation allowed realistic uncertainties to be generated. Wood biomass was assumed to be 50% C (Nabuurs et al., 2003).

3.5. Soil properties and C stocks

At all 28 abandoned machambas soil samples were collected from four subplots. In each subplot ten soil samples were collected from the 0–0.03 m horizon and mixed to form a single composite sample for analysis. In one of the subplots soil samples from four depths were collected (0–0.03 m, 0.03–0.1 m, 0.1–0.2 m and 0.2–0.3 m). Soils data were similarly collected from nine subplots in an area of 1 ha within 500 m of the woodland plots. At five of the subplots a depth profile was

Table 1
Allometric equations used in this paper

Reference	Equation(s)	Source country	Notes
Abbot et al. (1997)	$V = 10^{(-4.22+2.76 \log D)}$ $V = (0.057 + 0.000918D^2)^2$	Malawi	For canopy trees, assumed to be >4 m in height for this study. Phuyu site Understorey species, assumed to be trees less than 4 m high for our study. Phuyu site
Frost (1996)	$V = 6.18A^{0.86}$	Zaire, Malawi, Zambia and Zimbabwe	Equation applied on a stand basis
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
Brown et al. (1989)	$B = 34.47 - 8.067D + 0.659D^2$	Dry tropics	Not miombo specific, developed in "the dry tropics"

B, biomass (t); *V*, Volume (m^3); A, basal area ($m^2\ ha^{-1}$); *D*, diameter (cm) at 1.3 m (DBH). Note that for Abbot and Frost equations, wood density values determined at the site were used to calculate biomass from volume (Table 3).

sampled (0, 0.05, 0.15, 0.25 m depth) and at the remaining four, only a surface sample was taken. In the field, all soil textures were determined by hand texturing (Rural Development Service, 2006).

Soil samples were dried, sieved (2 mm sieve) and ball-milled to produce a fine flour. Percentage soil C was determined on dried samples using a Carbo-Erba/400 automated CN analyser. At each soil plot, soil bulk density measurements were determined using steel rings of known volume. Before weighing, the soils samples were dried in an oven at 40 °C for 24 h. Bulk density values were calculated by dividing the mass (t) of the soil sample by volume of the cylinder (m^3). Total soil C stock was determined by stepwise integration of the profile data of soil C content from 0 to 0.3 m.

Textural analysis on soils of abandoned machambas and woodland sites indicated that all were dominated by sand loams and sand loam silts. Single factor ANOVA revealed that there was no significant relationship between site age and bulk density (BD) on machambas, and so the data were pooled to provide a single bulk density estimate (1.26 t m^{-3}, identical to the mean woodland BD) which was applied at all sites. The pooled BD data were combined with soil %C data, and multiplied by the mass fraction of soil remaining after sieving, to generate estimates of total soil C stock for each soil layer.

3.6. Biodiversity

The Shannon index (H') is a measure of biodiversity calculated from the relative abundance of species in a community:

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

where $p_i = n_i/N$, n_i is the number of individuals present of species i, N the total number of individuals, and S is the total number of species. The Shannon index was determined for woody species >0.05 m DHB for each abandoned machamba and woodland plot.

The Jaccard similarity coefficient (J) is a statistic used for comparing the similarity and diversity of sample sets. We used J to determine the degree of similarity of species composition of different age classes. The Jaccard coefficient is defined as the size of the intersection divided by the size of the union of the sample sets:

$$J(A,B) = \frac{|A \cap B|}{|A \cup B|}$$

where A and B are the binary descriptions of species presence/absence in given age classes. A value of 1 indicates complete similarity, while 0 indicates complete dissimilarity. We used a list of 161 species, comprising all those recorded in surveys within the N'hambita community on woodlands and abandoned machambas, during 2003–2005.

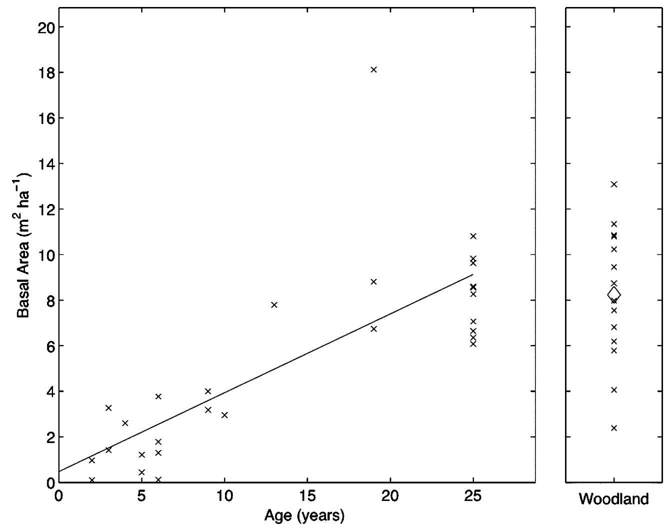

Fig. 1. Measured basal area plotted against age for all abandoned machambas (left panel), and for woodland plots (right panel, diamond indicates mean), with a linear regression assuming oldest abandoned machambas, age class 4, are 25 years. The age of the oldest abandoned machambas is given as 25 years, but lies between 20 and 30 years. Regression parameters are: $y = 0.27x + 1.13$; $P < 0.001$; $r^2 = 0.55$ (age class 4 = 30 years), $y = 0.35x + 0.48$; $P < 0.001$; $r^2 = 0.61$ (age class 4 = 25 years), $y = 0.47x - 0.41$; $P < 0.001$; $r^2 = 0.68$ (age class 4 = 20 years).

4. Results

4.1. Vegetation structure and C stocks

There were clear changes in vegetation characteristics along the chronosequence of machamba abandonments. Basal area was significantly correlated ($P < 0.001$) with time since abandonment (Fig. 1). A linear regression was able to explain 55–68% of the observed variability (uncertainty in ages of age class 4 abandonments, which might be 20–30 years old, accounts for the range of R^2 values reported here). The slopes of the regressions indicated that basal area increment was 0.25–0.47 m^2 ha^{-1} $year^{-1}$ (again the uncertainty here reflects uncertainty in age of the oldest abandonments). Basal area recorded in the woodland plots varied from 2.4 to 13.1 m^2 ha^{-1}, reflecting the highly heterogeneous forest cover in this area. The mean value ± standard deviation, S.D., for the woodland plots was 8.2 ± 3.0 m^2 ha^{-1}. However, there was no significant difference between basal area recorded in the oldest abandonments (>20 years, mean = 8.2 m^2 ha^{-1}) and the woodland plots (t-test, two sample assuming equal variances, $P > 0.05$). For the woodland plots there was no significant relationship between basal area and distance from the road (ANOVA, $P = 0.59$).

Stocking density (number of tree stems >0.05 m DBH ha^{-1}) also varied along the chronosequence, initially increasing with age, peaking after 10–20 years of abandonment, and then declining (Fig. 2). A third-order polynomial fit was able to explain 57% of observed variability, and revealed a highly significant relationship between stocking density and age ($P < 0.001$). There was a significant difference between the stocking density of machambas abandoned for >20 years and the

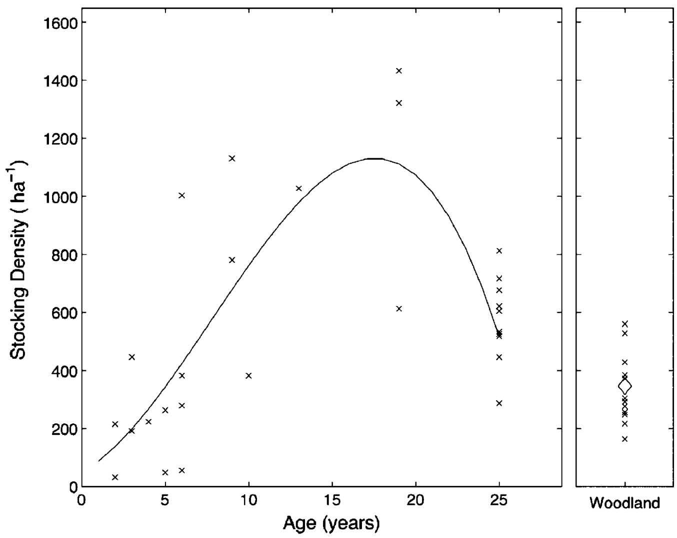

Fig. 2. Measured stem stocking density (stems per ha) plotted against age (years since abandonment) for all abandoned machambas (left panel), and for woodland plots (right panel, diamond indicates mean). The age of the oldest abandoned machambas is given as 25 years, but lies between 20 and 30 years. A third-order polynomial is fitted to the data, S.D. = $49.3 + 31.2t + 7.0t^2 - 0.30t^3$, where t is the time in years.

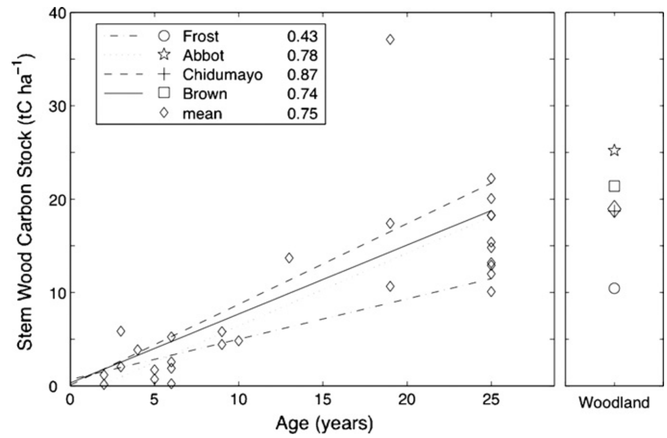

Fig. 3. Estimate wood C stock plotted against age for all abandoned machambas (left panel), and for woodland plots (right panel). The age of the oldest abandoned machambas is given as 25 years, but lies between 20 and 30 years. C stock is calculated from basal area, wood density and four different allometric relationships. The data here are the mean estimates from the four different allometric equations. The linear regressions determined from each individual relationship are plotted to indicate the uncertainty in biomass estimates. The legend indicates the author of the allometric relationship and the slope of the C stock vs. age relationship (i.e. annual wood C productivity, t C ha^{-1} year^{-1}).

woodland plots (means are 574 and 373 trees ha^{-1}, respectively, t-test, two sample assuming equal variances, $P < 0.01$).

Combining all four allometric equations, the mean $\pm$ S.D. estimated stem C stock estimates in the woodland plots was 19.0 $\pm$ 8.0 t C ha^{-1}. Nine of the 14 woodland plots had mean biomass estimates from 12 to 24 t C ha^{-1}, but plot values ranged from 4.3 to 33.4 t C ha^{-1}. The mean estimated stem C stock for the oldest abandoned machambas (>20 years) was 15.7 $\pm$ 3.9 t C ha^{-1}, ranging from 10.1 to 22.2 t C ha^{-1}. There was no significant difference ($t = -1.21$, $P = 0.11$) in the stem C stock estimates between abandoned machambas >20 years old and woodlands (t-test, two sample assuming equal variances).

Stem C stock estimates in the abandoned machambas were significantly and positively correlated with time since abandonment ($P < 0.001$, Fig. 3). Using linear regressions for the four allometric equations, and assuming that the oldest abandonment were 25 years old, resulted in estimates of biomass accumulation rates varying from 0.43 to 0.87 t C ha^{-1} year^{-1}, and a mean $\pm$ S.D. of 0.70 $\pm$ 0.19. The uncertainty in productivity estimates introduced by uncertainty in ages of oldest abandonments is important. Using the Brown equation (Table 1) and assuming that trees in age class 4 were 30 years old resulted in productivity estimates of 0.57 t C ha^{-1} year^{-1}, but 0.74 t C ha^{-1} year^{-1} if age class 4 trees were 25 years old, and 1.00 t C ha^{-1} year^{-1} for 20 year old trees.

4.2. Wood density

Wood density for individual species ranged from 0.40 to 0.71 t m^{-3} (Table 2). The mean density $\pm$ S.D. was 0.56 $\pm$ 0.08 t m^{-3}. The lower bulk densities were associated with fruiting species such as marula (*Sclerocarya birrea*) and mango (*Mangifera indica*). The three defining miombo species (*Brachystegia boehmii*, *Brachystegia spiciformis* and *Julbernadia globiflora*) in our sites had density values of 0.52, 0.63 and 0.63 t m^{-3}, slightly below or above the average.

The average wood density for each plot was determined from the specific measurements (Table 2), species composition and weighted by the cube of the each stem's DBH. The lowest values were found in the most recently abandoned machambas, while values for all abandoned machambas >7 years old were ~0.55 t m^{-3} (Fig. 4), very close to the mean value from the tree samples. A non-linear curve fit using a saturation equation ($D = \chi t/(\varepsilon + t)$, where t is the time since abandonment and χ and ε are the parameters) was able to explain 30% of observed variation with a root-mean-square error (RMSE) of 0.006.

For the five species with both sapwood and heartwood samples, it was possible to make comparisons of wood density estimates from both types of wood (Table 3). The mean wood density was 0.57 t m^{-3} for heartwood and 0.55 t m^{-3} for sapwood. A t-test was used to determine if the heartwood and sapwood densities were significantly different. Only one species of the five had significantly different heartwood and sapwood density ($P < 0.05$, t-test, two sample assuming equal variances).

4.3. Soil C stocks

There were no clear trends in soil C stocks in the top 0.3 m along the abandoned machamba chronosequence (Fig. 6). The frequency distribution of soil C stocks differed between abandoned machambas and woodlands. The Shapiro–Wilks test (IMSL Stats Library) indicated that the abandoned machamba soil C stocks were normally distributed ($n = 28$, $P > 0.05$) while the woodland plot soil C stocks were not ($n = 25$, $P < 0.001$). The mean soil C stock ($\pm$S.D.) for abandoned

Please cite this article in press as: Williams, M., et al., Carbon sequestration and biodiversity of re-growing miombo woodlands in Mozambique, Forest Ecol. Manage. (2007), doi:10.1016/j.foreco.2007.07.033

Table 2
Stem sapwood dry bulk density data (mean and standard deviation) from the 21 most populous species found on the combined woodland and abandoned machamba plots

Species	*n*	Mean dry bulk density (t m^{-3})	Standard deviation (t m^{-3})
Mangifera indica	4	0.40	0.07
Khaya anthoteca	3	0.45	0.02
Acacia nigrescens	5	0.46	0.03
Commiphora mossambicensis	3	0.47	0.06
Sclerocarya birrea	12	0.47	0.12
Entada abyssinica	18	0.51	0.11
Brachystegia boehmii	6	0.52	0.09
Xeroderris stuhlmannii	3	0.52	0.02
Piliostigma thonningii	4	0.53	0.05
Combretum apiculatum	10	0.55	0.03
Philenoptera violacea	24	0.55	0.10
Albizia amara	10	0.57	0.11
Diplorhynchus condylocarpon	9	0.60	0.07
Julbernardia globiflora	8	0.63	0.07
Brachystegia spiciformis	8	0.63	0.04
Albizia lebbeck	4	0.63	0.15
Burkea africana	3	0.63	0.03
Erythrophleum africanum	3	0.64	0.15
Millettia stuhlmannii	12	0.68	0.07
Pterocarpus rotundifolius subsp. *rotundifolius*	8	0.65	0.04
Cleistochlamys kirkii	3	0.71	0.11

Samples were collected in N'hambita in 2006, and are sorted by increasing mean bulk wood density. *n* indicates number of sample collected per species.

machambas was 45.2 (±14.1) t C ha^{-1}. The woodland plots had a clear bimodal distribution of soil C (Fig. 7), with 9 of the 28 plots having values from 10 to 30 t C ha^{-1} and another 9 having values from 60 to 90 t C ha^{-1}. The median soil C stocks were 57.9 t C ha^{-1} for woodlands and 44.9 t C ha^{-1} for abandoned machambas. The highest (140 t C ha^{-1}) and lowest (18 t C ha^{-1}) soil C stocks found in all soil profiles were in woodland plots. A Wilcoxon rank sum test (IMSL Stats Library), the nonparametric equivalent of the two-sample *t*-test, indicated there was no significant difference between the woodland and abandoned machamba soil C stock estimates ($P > 0.05$).

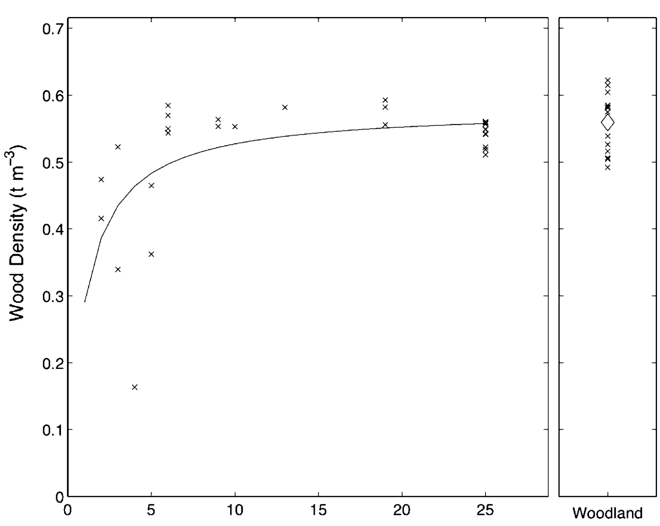

Fig. 4. Wood density (D, t m^{-3}) estimates for all abandoned machambas plotted against age, and for all woodland plots (right panel, diamond indicates mean). Wood density of 21 dominant species was determined and combined with species data on each plot. A non-linear curve fit for the chronosequence of abandoned machambas is shown, using a saturation equation, $D = \chi t/(\varepsilon + t)$, where t is the time since abandonment (years) and χ and ε are the parameters. For the best fit $\chi = 0.58$ and $\varepsilon = 1.00$.

4.4. Biodiversity

There were 69 different woody species (DBH > 0.05 m) in the 14 woodland plots (total survey area 3.5 ha), which contained in total 1211 stems >0.05 m DBH. Five species contributed 54% of the total stem count, and five species contributed 46% of the total basal area (Tables 4 and 5). *Diplorhynchus condylocarpon* and *B. boehmii* were the dominant species (by stocking density and basal area, respectively). The dominant species of the woodland plots are typical of those for dry miombo (Kanschik and Becker, 2001).

We identified 67 woody species (DBH > 0.05 m) in the 28 abandoned machamba plots (total survey area ~3.5 ha), which contained in total 1955 stems >0.05 m DBH. The dominant species in abandoned machambas differed according to time since abandonment (Tables 4 and 5), and according to whether stocking density or basal area were used as measures of dominance. The youngest plots (age class 1) were dominated by fruit trees, some exotic, such as papaya (*Carica papaya*) marula (*S. birrea*) and mango (*M. indica*), or trees with other domestic uses, such as monkey bread (*Bauhinia thonningii*). In older abandonments (age-classes 2–4), the trees were dominated by small-to-medium sized native trees such as *Philenoptera violacea* and *Combretum apiculatum*. The dominant species in the abandoned machambas largely differed from those in the woodland plots. Comparing the oldest abandonments to the

Table 3
Heartwood and sapwood dry bulk densities for five dominant species

Species ID	*Sclerocarya birrea*		*Millettia stuhlmannii*		*Philenoptera violacea*		*Albizia amara*		*Entada abyssinica*	
	Heart[a]	Sap[a]	Heart[a]	Sap[a]	Heart[a]	Sap[a]	Heart[a]	Sap[a]	Heart[a]	Sap[a]
n	4	12	5	12	3	24	2	10	5	13
Mean (t m^{-3})	0.50	0.47	0.56	0.68	0.61	0.55	0.64	0.57	0.55	0.49
S.D.	0.03	0.12	0.05	0.07	0.04	0.10	n/a	0.11	0.03	0.12
t-Test *P* value	0.383		0.002*		0.116		n/a		0.131	

For each species are listed the number of branch samples measured (*n*), the mean wood density of the samples, and their standard deviation (S.D.). A *t*-test was used to determine if the heartwood and sapwood densities were significantly different; the *t*-test *P* values are given in the final row. There was only one significant difference (marked *) at the 5% level.
[a] Wood type.

Table 4
The five most dominant species, ranked by stocking density in each age class of abandoned machamba and in woodland, are listed in order

Order	1 (1–5 years)	2 (6–10 years)	3 (11–20 years)	4 (20–30 years)	Woodland
1	*Sclerocarya birrea*	*Philenoptera violacea*	*Entandrophragma caudatum*	*Combretum apiculatum*	*Diplorhynchus condylocarpon*
2	*Piliostigma thonningii*	*Entandrophragma caudatum*	*Philenoptera violacea*	*Philenoptera violacea*	*Pterocarpus rotundifolius rotundifolius*
3	*Mangifera indica*	*Piliostigma thonningii*	*Piliostigma thonningii*	*Dalbergia boehmii* Taub.	*Combretum apiculatum*
4	*Acacia nigrescens*	*Albizia lebbeck*	*Combretum apiculatum*	*Commiphora mossambicensis*	*Brachystegia boehmii*
5	*Philenoptera violacea*	*Sclerocarya birrea*	*Sterculia appendiculata*	*Vitex doniana* Sweet	*Cleistochlamys kirkii*
Percent of total	60%	67%	57%	51%	54%
n	7	7	4	10	14
Species richness	5.2	7.4	17.5	13.8	15.1
J	0.15	0.19	0.19	0.31	N/A

Also shown are the percentage of total stems made up by the five dominant species, the number of plots sampled in each age class (*n*), and mean species richness. The final row shows the Jaccard similarity coefficient (*J*) for species composition of abandoned machambas of different ages (classes 1–4) and that of woodland.

woodlands, the dominant five species by basal area had no species in common, while the dominant five by stocking density had just one in common (Tables 4 and 5). The Jaccard similarity coefficient for species composition of abandoned machambas compared to woodland ranged from a minimum of 0.15 between recent abandonments and woodland plots, to a maximum of 0.31 between the oldest abandonments and woodland plots (Table 4). The defining miombo species found in the woodland plots (*B. boehmii*, *B. spiciformis*, *J. globiflora*) were completely absent from abandoned machambas of all ages in stems >0.05 m DBH.

The lowest Shannon indices were found in the most recently abandoned machambas, and increased with time since abandonment, but then saturated at greater ages (Fig. 5). A non-linear curve fit using a saturation equation ($H' = \alpha t/(\beta + t)$, where t is the time since abandonment and α and β are the parameters) was able to explain 62% of observed variation with a root-mean-square error (RMSE) of 0.12. Species richness also increased with time since abandonment, saturating with a similar pattern to the Shannon Index (Table 4). There were no significant differences between abandoned machambas >20 years old and the woodland plots for either Shannon indices ($P = 0.15$) or mean species richness ($P = 0.22$, t-test, two sample assuming equal variances).

5. Discussion

Because of access issues and chronosequence measurement, the sampling strategy employed was not completely randomised, and so some caution is required in interpreting the results. There was no significant effect of distance from road on woodland structure, so locating plots within easy walking

Table 5
Dominant species type ranked by basal area in all abandoned machambas, and in woodland

Rank order	Age class 1	Age class 2	Age class 3	Age class 4	Woodland
1	*Carica papaya*	*Philenoptera violacea*	*Albizia lebbeck*	*Combretum apiculatum*	*Brachystegia boehmii*
2	*Trichilia emetica*	*Piliostigma thonningii*	*Philenoptera violacea*	*Philenoptera violacea*	*Acacia nigrescens*
3	*Mangifera indica*	*Albizia lebbeck*	*Piliostigma thonningii*	*Commiphora mossambicensis*	*Diplorhynchus condylocarpon*
4	*Acacia nigrescens*	*Entandrophragma caudatum*	*Combretum apiculatum*	*Faidherbia albida*	*Brachystegia spiciformis*
5	*Sclerocarya birrea*	*Sclerocarya birrea*	*Sclerocarya birrea*	*Albizia amara*	*Erythrophleum africanum*
Percent of total	69%	66%	67%	55%	46%

The five most abundant species are listed in order. The final row shows the percentage of basal area made up by the five dominant species.

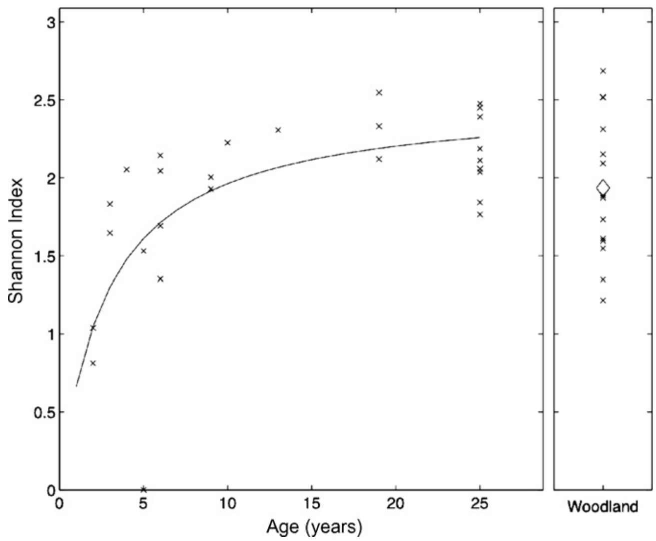

Fig. 5. Shannon index (H') of species diversity plotted against age for all abandoned machambas (left panel), and for woodland plots (right panel, diamond indicates mean). The age of the oldest abandoned machambas is given as 25 years, but lies between 20 and 30 years. A non-linear curve fit for the chronosequence of abandoned machambas is shown, using a saturation equation, $H' = \alpha t/(\beta + t)$, where t is the time since abandonment (years) and α and β are the parameters. For the best fit $\alpha = 2.51$ and $\beta = 2.79$.

distance of tracks is unlikely to have affected the results. Woodland plots were randomly located along the road/track network, but the abandoned machambas were selected on the basis of farmer interviews. The abandoned machambas were all located around the N'hambita village, and are clustered in age groups, according to the historical development of the community. Due to their proximity to human settlements, it is likely that the abandoned machambas were more disturbed than woodland plots, for example by fires, which are often started near settlements or roads, or fuel-wood harvesting. It is also possible that farmers selected woodland areas with richer soils for clearance.

Space-for-time studies have been criticised for producing artefacts because of non-random site selection (Frost, 1996). If the cleared and abandoned land was never originally miombo, then the chronosequence approach is compromised. The balance of evidence suggests that these lands were most likely dry miombo. Firstly, the nearest woodland plots (three were within 2.5 km to the NW, N and SE of the cleared areas) were all dominated by the defining dry miombo species. Local seed sources were thus available, and the local climate was suitable. Secondly, there were no clear topographical or soil textural differences between these three woodland plots and the abandoned machambas lying between them. However, the lack of information concerning the natural vegetation of the machamba areas is a major, but unavoidable, cause of uncertainty.

We hypothesised that C stocks in vegetation and soils of abandoned machambas would be lower than in woodland plots, and that C stocks would accumulate more rapidly after abandonment in vegetation than in soils. The data for vegetation largely supported these hypotheses, but the results for soils are less clear.

5.1. C stocks in vegetation

For the abandoned machambas, there was a clear relationship between time since abandonment and increased wood C stocks (Figs. 1 and 3). While the oldest abandoned machambas (>20 years) and woodland plots had similar basal area and biomass, suggesting a large degree of recovery, there were still significant differences in stocking density (Fig. 2), indicating that structural differences remained. The woodland plots had a larger variation in biomass than the abandoned machambas, which we attribute to either variable natural disturbance generating a mosaic of differently structured woodland stands (Fig. 7) or perhaps local variations in hydrology. However, further work is required to determine whether differences in soils or hydrology can explain these variations, and whether wood C stocks of low biomass woodlands are aggrading at the same rate as on abandoned machambas.

After abandonment, wood C stocks increased by 0.7 t C ha^{-1} year^{-1}. The mean annual increment (MAI) of above-ground woody C was similar to other figures for dry miombo, 0.9 t C ha^{-1} year^{-1} over 35 years in Zambia (Chidumayo, 1997), 0.5 t C ha^{-1} year^{-1} in 16 years old coppiced miombo woodland in northern Zambia (Stromgaard, 1985) and 0.75 t C ha^{-1} year^{-1} over 50 years, calculated from data in Frost (1996). There is uncertainty in the productivity estimate ($\pm$0.2 t C ha^{-1} year^{-1}), due to uncertainty in aging the oldest abandoned machambas, and in the allometric relationships used to generate biomass from stem diameter and wood density measurements. These two uncertainties are roughly equal—the range in productivity estimates is 0.44 t C ha^{-1} year^{-1} due to uncertainty in allometric equations, and 0.43 t C ha^{-1} year^{-1} due to uncertainty in time since abandonment. Thus, improvements to the productivity estimates at this site will require both locally determined allometric equations and a more thorough investigation of site histories and constraints on date of abandonment.

5.2. C stocks in soils

Disturbance of soils associated with cultivation generally leads to a rapid decline in soil organic C content as a consequence of enhanced microbial respiration (King and Campbell, 1994; Schlesinger, 1986). In a global survey Guo and Gifford (2002) found C losses of 42% due to land use conversion to agriculture from native forest. Walker and Desanker (2004) observed C reductions of 40% after conversion to agriculture from Malawian miombo woodland. In the present study, abandoned agricultural land had a median C stock 23% lower than the surrounding woodlands. There was no significant difference between the pooled abandoned machamba soil C stocks and those of woodlands, so there was no clear support for our expectation of a widespread drop in soil C stocks after slash-and-burn. However, it is possible that farmers selected areas with richer soils for clearance, in which case a real loss of C stocks may be obscured by the comparison with randomly sampled woodland. The broad range of soil C stocks in woodland

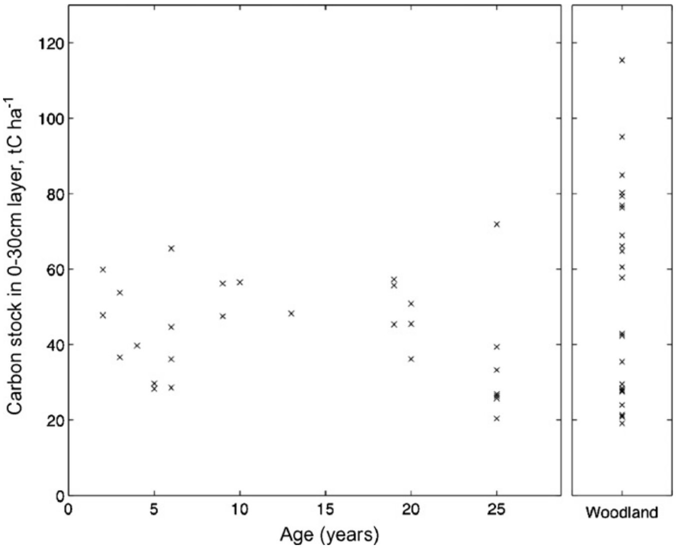

Fig. 6. Total soil C content in surface 0.3 m for all sites plotted against age for all abandoned machambas (left panel) and for woodlands (right panel). The age of the oldest abandoned machambas is given as 25 years, but lies between 20 and 30 years.

suggests major variations in organic matter and probably fertility (Figs. 6 and 7).

Subsequent reforestation often leads to a steady but slower accumulation of organic C as plant C inputs accumulate in the surface soil horizons (Guo and Gifford, 2002; Jarecki and Lal, 2003). In a survey of re-growing tropical forests, mostly from the moist and wet tropics, Silver et al. (2000) found that soil C increased at 1.3 t ha^{-1} $year^{-1}$ in the first 20 years after abandonment. However, in a review of rates of C sequestration following land use change, Post and Kwon (2000) reported annual rates of C accumulation of as little as 0.03 t C ha^{-1} $year^{-1}$ in arid locations. These low rates of recovery are consistent with the lack of any identifiable change in soil C on older abandoned machambas at our dry tropical sites, and suggest very slow additions of organic matter to the soil. In a study of miombo woodlands in Malawi, Walker and Desanker (2004) also found that soil C stocks did not increase after abandonment of agriculture. This lack of accumulation is likely a result of frequent fire disturbance (Bird et al., 2000; Chidumayo and Kwibisa, 2003), or perhaps termite activity (Konate et al., 2003).

The soil C stocks of the woodland plots are notable for their bimodal distribution (Fig. 7). The significant departure from normality suggests a complex spatial pattern of soil C stocks, which is not easily explained given the lack of textural or bulk density differences among soil samples. Site variables such as drainage are known to influence soil organic C concentrations, but were not characterised in this study. It is possible that disturbance history, particularly frequency of fires, is a critical factor in development of soil C stocks through vegetation-fire-fuel-load feedbacks (Frost, 1996; Stromgaard, 1985, 1986). Further sampling of soils and other potentially linked environmental controls are required to test whether the bimodal distribution is widespread and to understand its cause.

The management implications are clear for sequestering C in wood. Natural regeneration will restore stem C stocks in 2–3 decades, but rates of accumulation are low and total stem C stocks in the natural vegetation are relatively small compared to soil C stocks. The potential to sequester C in soils is less clear. There is no increase in soil C stocks along the chronosequence, which indicates that inputs of organic matter to soils in re-growing miombo are very small. Woodland soil C stocks vary across almost an order of magnitude (Fig. 7). If boosting the C storage of miombo soils were possible (Jarecki and Lal, 2003; Lal, 2003), this would provide a valuable means to sequester C, with an equal or greater potential for C sequestration than restoring abandoned machambas to miombo woodland under current disturbance conditions. Whether a reduction in

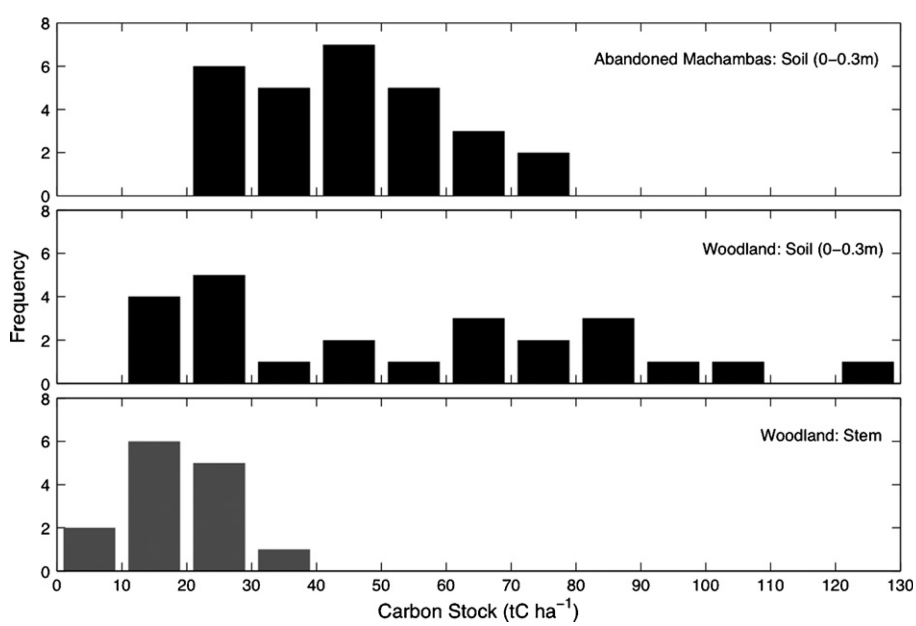

Fig. 7. Frequency distributions of C stocks (t C ha^{-1}) in abandoned machamba soils (top panel), woodland soils (middle) and woodland stem C (bottom panel).

disturbance (i.e. fire intensity and frequency) would increase long-term wood C stocks is also a critical management issue (Bond et al., 2005; Trapnell, 1959).

5.3. Changes in stem wood density

We hypothesised that the successional change in species would result in an increase in mean wood density as pioneer species were replaced with slower growing miombo dominants. The evidence supports this hypothesis, but only for the very early part of the succession (<10 years) where fruit trees are dominant (Fig. 4). These trees have the lowest wood density values (Table 2), but the data suggest that fruit trees are rapidly lost from re-growing woodlands. There was little difference in wood density calculated for sapwood and heartwood from a selection of common species (Table 3) suggesting that the biomass estimates derived from sapwood density data alone were reasonable. However, we were not able to sample the heartwood of larger trees, including the defining miombo species, due to logistical difficulties. It is possible that using sapwood estimates of wood density for these species has resulted in an under-estimate of wood biomass and C stocks.

5.4. Species dynamics

We hypothesised that the species dynamics in recovering plots would indicate a return towards the dominant species of local miombo woodland. We reject this hypothesis. None of the defining miombo species are present in any of the abandoned machambas (Tables 4 and 5), as was also found by Stromgaard (1986). However, the secondary dominant species of miombo are found on abandoned machambas, and there is greater similarity in species composition between older abandonments and woodlands (Table 4). Even though most species differed between abandoned machambas and woodlands (as indicated by low Jaccard similarity coefficients), the biodiversity of woody species (i.e. Shannon index and species richness) of abandoned machambas >10 years old was similar to that found in woodlands (Fig. 5). Overall tree biodiversity has not been degraded by the slash-and-burn disturbance, but the time-scale of recovery of defining miombo species is unclear.

Miombo species are known to be able to survive the destruction of their above-ground parts (Chidumayo, 1997; Frost, 1996; Nyerges, 1989; Robertson, 1984). They are generally good at re-sprouting and can reproduce from root suckers; 15 years of mattocking were required to kill *Brachystegia* spp. (Robertson, 1984), and re-sprouting is a common response to destruction by fire. The defining miombo species are, however, thought to be relatively fire-tender (Cauldwell and Zieger, 2000; Trapnell et al., 1976). We have observed that fires are commonplace in the N'hambita community, recurring every year or two. It is possible that in the open, early successional areas, the frequent fires, high grass biomass, and thus high fuels loads, mean that only fire-tolerant species can re-establish (e.g. *P. violacea*, *D. condylocarpon*, *Combretum* spp.). The data show that fruit trees are replaced by fire-tolerant miombo species, but the defining miombo species do not establish over 2–3 decades.

Further work is required to study the woody biomass belowground in these sites, which is potentially significant given the prodigious sprouting behaviour from root stock of many miombo species. The bimodal distribution of soil C stocks in woodlands needs to be tested by further detailed surveys including analyses of nutrient dynamics and soil hydrology. Finally, the role of fire in miombo systems is significant and the C dynamics of these woodlands can only be fully understood and predicted within the context of fire disturbance.

6. Conclusions

Our objective was to determine how slash-and-burn agriculture affected soil and vegetation C stocks, and biodiversity on an area of miombo woodland, and how C stocks and species changed once agriculture was abandoned. We have shown that clearance for agriculture reduces stem wood C stocks by 19.0 t C ha^{-1} and the years following abandonment wood C stocks accumulated at 0.7 t C ha^{-1} $year^{-1}$. However, the re-growing areas do not contain the defining miombo species, and stem numbers are significantly greater than in woodland plots. Whether typical miombo species regain dominance is less clear, so conserving existing miombo woodlands is vital for maintaining the defining species, and their rich associated fauna. If fire disturbance on the abandoned machambas is heightened by proximity to human settlements (as is the case here) then more fire resistant species may dominate instead. Because woodland soil C stocks vary so much, it is not clear whether slash-and-burn reduces soil C stocks. Analysis of the abandonment chronosequence shows that there is no accumulation of soil C after 20–30 years of abandonment. This suggests that the rate of accumulation of organic matter in these soils has been very slow.

Our study emphasises the importance of measuring and monitoring *ecosystem* C stocks when assessing the potential for C sequestration. While the woodland stem C stocks are capable of recovery within a few decades, the soil stocks do not accumulate following abandonment over a few decades. Woodland soils were capable of storing >100 t C ha^{-1}, whereas no abandoned machamba soil exceeded 74 t C ha^{-1}, and no stem wood stock exceeded 33 t C ha^{-1}, so there is a potential for C sequestration in woodland soils that should be investigated as a management option, probably through experiments with fire exclusion during woodland re-growth (Chidumayo, 1997). Management for C sequestration should also focus on identifying C-rich soils and conserving remaining woodlands to protect soil C. To ensure that local communities can meet agricultural needs without permanent loss of woodlands, land management must provide approaches to increase crop output at low cost, e.g. agro-forestry and intercropping. The major challenge for C management is to understand the observed variability in vegetation and soil C stocks in woodlands, and use this understanding to manage existing forests and re-growing areas for greater C storage.

Please cite this article in press as: Williams, M., et al., Carbon sequestration and biodiversity of re-growing miombo woodlands in Mozambique, Forest Ecol. Manage. (2007), doi:10.1016/j.foreco.2007.07.033

Acknowledgements

We dedicate this paper to the late Patrick Mushove, of Harare, whose assistance with the sampling and analysis of the woodland plots was vital. We also thank Piet van Zyl and his colleagues at Envirotrade for their assistance with field operations. We thank the N'hambita community for their hospitality and assistance. Funding for this work was provided by the European Development Fund, EuropeAid, and NERC. We thank Paul Stoy for his comments. M.W. wrote the paper and analysed data. M.W., J.G. and B.R. devised the experiments and observations. E.S. and J.F. collected part of the data as a component of their MSc theses. C.M.R. collected further data, and analysed the complete data set, and contributed to the writing. J.G. and B.R. provided input to the writing.

References

Abbot, J.I.O., Homewood, K., 1999. A history of change: causes of miombo woodland decline in a protected area in Malawi. J. Appl. Ecol. 36, 422–433.
Abbot, P., Lowore, J., Werren, M., 1997. Models for the estimation of single tree volume in four Miombo woodland types. For. Ecol. Manage. 97, 25–37.
Bird, M.I., Veenendaal, E.M., Moyo, C., Lloyd, J., Frost, P., 2000. Effect of fire and soil texture on soil carbon in a subhumid savanna (Matopos, Zimbabwe). Geoderma 94, 71–90.
Bond, W.J., Woodward, F.I., Midgley, G.F., 2005. The global distribution of ecosystems in a world without fire. New Phytol. 165, 525–538.
Brown, Gillespie, R., Lugo, E., 1989. Biomass estimation methods for tropical forests with applications to forest inventory data. For. Sci. 35, 881–902.
Campbell, B.M., 1996. The Miombo in Transition: Woodlands and Welfare in Africa. Center for International Forestry Research, Bogor, Indonesia.
Cauldwell, A.E., Zieger, U., 2000. A reassessment of the fire-tolerance of some miombo woody species in the Central Province, Zambia. Afr. J. Ecol. 38 138–146>.
Chidumayo, E., 2002. Changes in miombo woodland structure under different land tenure and use systems in central Zambia. J. Biogeogr. 29, 1619–1626.
Chidumayo, E.N., 1997. Miombo Ecology and Management: An Introduction. IT Publications in association with the Stockholm Environment Institute, London.
Chidumayo, E.N., Kwibisa, L., 2003. Effects of deforestation on grass biomass and soil nutrient status in miombo woodland, Zambia. Agric. Ecosyst. Environ. 96, 97–105.
Coates Palgrave, K., Drummond, R.B., Moll, E.J., Coates Palgrave, M., 2002. Trees of southern Africa, 3rd ed./New ed. revised and updated by Meg Coates Palgrave (Ed.). Struik Publishers, Cape Town, Great Britain.
de Koning, J., 1993. Checklist of Vernacular Plant Names in Mozambique. Wageningen Agricultral University, Wageningen.
Frost, P., 1996. The ecology of miombo woodlands. In: Campbell, B. (Ed.), The Miombo in Transition: Woodlands and Welfare in Africa. CIFOR, Bogor.
Guo, L.B., Gifford, R.M., 2002. Soil carbon stocks and land use change: a meta analysis. Global Change Biol. 8, 345–360.
Jarecki, M.K., Lal, R., 2003. Crop management for soil carbon sequestration. Crit. Rev. Plant Sci. 22, 471–502.
Jindal, R., 2006. Carbon Sequestration Projects in Africa: Potential Benefits and Challenges to Scaling Up. EarthTrends, World Resources Institute.
Kanschik, W., Becker, B., 2001. Dry miombo—ecology of its major plant species and their potential use as bio-indicators. Plant Ecol. 155, 139–146.
King, J.A., Campbell, B.M., 1994. Soil organic matter relations in five land cover types in the Miombo Region (Zimbabwe). For. Ecol. Manage. 67, 225–239.
Konate, S., Le Roux, X., Verdier, B., Lepage, M., 2003. Effect of underground fungus-growing termites on carbon dioxide emission at the point- and landscape-scales in an African savanna. Funct. Ecol. 17, 305–314.
Lal, R., 2003. Offsetting global CO_2 emissions by restoration of degraded soils and intensification of world agriculture and forestry. Land Degrad. Dev. 14, 309–322.
Nabuurs, G.-J., Ravindranath, N.H., Paustian, K., Freibauer, A., Hohenstein, W., Makundi, W., 2003. Forest land. In: Penman, J., Gytarsky, M., Hiraishi, T., Krug, T., Kruger, D., Pipatti, R., Buendia, L., Miwa, K., Ngara, T., Tanabe, K., Wagner, F. (Eds.), Good Practice Guidance for Land Use, Land-Use Change and Forestry, Intergovernmental Panel on Climate Change. UNEP, Kanagawa.
Nyerges, A., 1989. Coppice swidden fallows in tropical deciduous forest: biological, technological, and sociocultural determinants of secondary forest successions. Hum. Ecol. 17, 379–400.
Post, W.M., Kwon, K.C., 2000. Soil carbon sequestration and land-use change: processes and potential. Global Change Biol. 6, 317–327.
Robertson, E.F., 1984. Regrowth of two African woodland types after shifting cultivation. Ph.D. Thesis. University of Aberdeen, Aberdeen.
Rural Development Service, 2006. Soil Texture, Technical Advice Note 52. DEFRA, London.
Schlesinger, W.H., 1986. Changes in soil carbon storage and associated properties with disturbance and recovery. In: Trabalka, J.R., Reichle, D.E. (Eds.), The Changing Carbon Cycle. A Global Analysis. Springer Verlag, New York, pp. 194–220.
Silver, W.L., Kueppers, L.M., Lugo, A.E., Ostertag, R., Matzek, V., 2004. Carbon sequestration and plant community dynamics following reforestation of tropical pasture. Ecol. Appl. 14, 1115–1127.
Silver, W.L., Ostertag, R., Lugo, A.E., 2000. The potential for carbon sequestration through reforestation of abandoned tropical agricultural and pasture lands. Restorat. Ecol. 8, 394–407.
Stromgaard, P., 1985. Biomass, growth, and burning of woodland in a shifting cultivation area of South Central Africa. For. Ecol. Manage. 12, 163–178.
Stromgaard, P., 1986. Early secondary succession on abandoned shifting cultivator's plots in the miombo of South Central Africa. Biotropica 18, 97–106.
Tinley, K., 1977. Framework of the Gorongosa Ecosystem. Univeristy of Pretoria, Pretoria.
Trapnell, C.G., 1959. Ecological results of woodland burning experiments in Northern Rhodesia. J. Ecol. 47, 129.
Trapnell, C.G., Friend, M.T., Chamberlain, G.T., Birch, H.F., 1976. Effects of fire and termites on a Zambian woodland soil. J. Ecol. 64, 577–588.
Van Wyk, B., Van Wyk, P., 1997. Field Guide to Trees of Southern Africa. Struik Publishers, Cape Town.
Van Wyk, P., 1993. Southern African Trees. New Holland, London.
Walker, S., Desanker, P., 2004. The impact of land use on soil carbon in Miombo Woodlands of Malawi. For. Ecol. Manage. 203, 354–360.