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Determinants of aboveground carbon offset additionality in plantation forests in a moist tropical forest in western Kenya

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ABSTRACT

Few studies have attempted to identify factors that contribute to aboveground carbon offset additionality in forest restoration planting in the tropics. Moreover, those that have compared aboveground carbon offset potential of naturally regenerating secondary forests and plantation forests have yielded conflicting results regarding the ability of the latter to attain carbon offset additionality, thus limiting broad adoption of carbon-driven forest restoration interventions. We assessed woody species diversity, stem density, stem diameter and wood specific gravity of secondary and plantation forests in Kakamega Forest in western Kenya to identify determinants of aboveground carbon offset additionality in plantation forests. Secondary forests comprised old-growth, middle-aged and young vegetation stands. Plantation forests consisted of mixed indigenous, *Maesopsis eminii* indigenous monoculture and *Cupressus lusitanica*, *Pinus patula* and *Bischofia javanica* exotic monoculture stands. Assessment was carried in 135 sample plots in three forest blocks using stratified systematic sampling in nested plots. Analysis of variance indicated that there was no significant difference in woody species diversity between secondary and plantation forests due to natural forest succession in both forest types. Mixed indigenous plantation had more aboveground carbon stock than secondary forest stands of comparable stand age due to its greater proportion of tree species with high wood specific gravity and large tree diameter. Old-growth secondary forest had more aboveground carbon stock than monoculture forest plantations due to its relatively higher wood specific gravity. Middle-aged secondary forest had relatively lower aboveground carbon stock than plantation forests of comparable stand age because of its smaller tree diameter. The results suggest that stem diameter and wood specific gravity are the most important determinants of aboveground carbon offset additionality. Thus, forest managers and investors in carbon offset projects can achieve aboveground carbon offset additionality in forest restoration interventions by planting tree species with relatively higher wood specific gravity and manipulating them to attain large stem diameter through silvicultural management.

1. Introduction

Tropical forest ecosystems play a significant role in the global carbon balance. They account for only 37% of the estimated 1150 Gt of carbon that resides in forest ecosystems, but have the greatest impact on the terrestrial carbon balance (Malhi et al., 1999; Lewis et al., 2004; Lewis, 2006; Chaturvedi et al., 2011; Martin et al., 2013). This is because most of the carbon in tropical forests is stored in standing vegetation unlike the case of boreal and temperate forests where a greater proportion of the carbon is stored in the soil and in peat bogs (Malhi et al., 2002). Global wood utilization trends also indicate that wood supply in developing countries, most of which occupy the tropical zone, is driven primarily by the demand for fuel wood (Siry et al., 2005). Thus, most of the 5.6–8.6 billion tons of carbon, which is emitted annually into the atmosphere through deforestation and forest degradation (approximately 18% of present greenhouse gas emissions), originates from tropical forests (Glenday, 2006; Brickell, 2009; Keenan, 2009; Verburg et al., 2009; van der Werf et al., 2009; Orihuela-Belmonte et al., 2013). Despite such a huge contribution of deforestation and forest degradation to global greenhouse gas emissions, forest restoration efforts hold a great potential to significantly offset atmospheric carbon emissions (Brown et al., 2000; Laurance, 2007). However, large

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scale adoption of carbon-driven forest restoration strategies has been hampered by poor understanding of the aboveground carbon offset potential of passive and active forest restoration interventions (Lewis, 2006; Gibbs et al., 2007; Holl and Zahawi, 2014; Kinyanjui et al., 2014).

Comparisons of aboveground carbon offset potential of naturally regenerating secondary forests and plantation forests in tropical forest ecosystems have yielded conflicting findings regarding which of the two forest types has superior offset potential (Chaturvedi et al., 2011; Omeja et al., 2012; Bonner et al., 2013; Marin-Spiotta and Sharma, 2013). A number of studies have indicated that natural forest regeneration accumulates significantly more aboveground carbon stock than restoration planting (Glenday, 2006; Ch'ng et al., 2011), while other studies have reported that the latter sequesters significantly more aboveground carbon stock (Zheng et al., 2008; Baishya et al., 2009; Chaturvedi et al., 2011). Other studies have indicated that plantation forests are only marginally superior to secondary forests in aboveground carbon accumulation (Omeja et al., 2012; Bonner et al., 2013). However, ITTO and FAO (2009) and Kanowski and Catterall (2010) state that plantation forests also differ in aboveground carbon accumulation potential due to variation in tree species mix and stand structural attributes and are therefore not expected to give the same results. Since the findings of each of these studies are considered accurate, it is likely that there exist specific variables that determine the aboveground carbon accumulation potential in both naturally regenerating secondary forests and planted forests. For instance, Baishya et al. (2009) attributed superior aboveground carbon accumulation potential in plantation forests over secondary forests to good silvicultural management. It was unclear, however, whether the aboveground carbon accumulation potential of naturally regenerating secondary forest stands would also increase if they were subjected to similar silvicultural treatment. Given that carbon offset benefits can only be derived from forest restoration interventions with the capacity to achieve carbon offset additionality over the reference scenario (natural forest regeneration in this case) (Valatin, 2011; Omeja et al., 2011; Gillenwater, 2012), failure to clearly identify the determinants of aboveground carbon offset additionality has slowed the broad adoption of carbon-driven forest restoration planting. The situation is attributed to the fact that many actors in carbon offset schemes are apprehensive of engaging in restoration efforts whose aboveground carbon sequestration potential is inferior to natural forest regeneration. It is prudent, therefore, to provide a clear picture on the determinants of aboveground carbon offset additionality in order to inform forest managers and investors in carbon offset schemes of the best forest restoration approaches to employ.

In this paper, we assess the aboveground carbon offset potential of different secondary and plantation forest types using a chronosequence study in Kakamega Forest in western Kenya. The forest is an eastern relic of the African equatorial rainforest and one of the forest ecosystems that have been subjected to a great deal of degradation for close to a century (Lung and Schaab, 2006; Schaab et al., 2010). It has lost about 75% of its primary forest cover as a result of a series of disturbance events that occurred between 1930s and 1990s (Wass, 1995). Whereas most of the degraded forest sites regenerated naturally and ended up as secondary forests, some of the sites were placed under plantation forests of both indigenous and exotic tree species (Glenday, 2006; Otuoma et al., 2014). Other sites were subjected to repeat incidences of disturbance and degenerated into open fields that are presently used for grazing. The study analysed possible variation in woody species diversity, stand structural attributes and wood specific gravity in relation to aboveground carbon stock in order to identify variables that determine superiority in carbon offset potential between secondary and plantation forests in moist tropical forests. Secondary forest types comprised old-growth, middle-aged and young vegetation stands, while plantation forests consisted of mixed and monoculture indigenous and exotic stands. Findings of this study are expected to inform policy makers, forest managers and prospective investors in carbon credit schemes about tree species attributes and managerial aspects that forest restoration interventions should focus on in their endeavour to secure aboveground carbon offset additionality for future carbon offset projects.

2. Materials and methods

2.1. Study site

The study was carried out in Kakamega Forest between February 2013 and January 2015. The forest is located in western Kenya between latitudes 0°10′N & 0°21′N and longitudes 34°47′E & 34°58′E at an elevation of 1600 m above sea level (Fashing and Gathua, 2004; Farwig et al., 2008). The area has a hot and wet climate characterised by a mean temperature of 25 °C and an annual precipitation of 1500–2000 mm with a dry season between December and March (Glenday, 2006; Mitchell and Schaab, 2008). The forest has over 400 plant species (of which about 112 are tree species), over 300 bird species and about seven endemic primate species (Kokwaro, 1988; Otuoma et al., 2014).

The forest's vegetation comprises a disturbed primary forest, secondary forests in different stages of succession, mixed indigenous plantation forests, indigenous and exotic monoculture plantation forests, and both natural and man-made glades (Tsingalia and Kassily, 2009). Closed canopy old-growth natural forest stands are dominated by evergreen tree species such as *Funtumia africana* (Benth.) Stapf, *Strombosia scheffleri* Engl., *Trilepisium madagascariense* DC., *Antiaris toxicaria* Lesch., *Ficus exasperata* Vahl, *Croton megalocarpus* L. and *Celtis gomphophylla* Baker (Glenday, 2006; Lung, 2009). The forest supports an adjoining human population of about 280,000 people who are distributed in surrounding farmlands and urban centres (Otuoma et al., 2014). Some of the resources that they obtain from the forest include fuel wood, timber, construction poles, herbal medicine, fibre, pasture for livestock, indigenous fruits and traditional vegetable (Musila et al., 2010).

2.2. Study design

The study employed a nested experimental design (Kuehl, 2000; Onwuegbuzie and Leech, 2007). Assessment was carried out in three forest blocks, namely: Yala, Kibiri and Isecheno. Each of the forest blocks had nine different forest types, which were the treatments in the study. The nine forest types were disturbed primary forest (which was the control), old-growth secondary forest, middle-aged secondary forest, young secondary forest, mixed indigenous plantation, *Maesopsis eminii* Engl. indigenous monoculture plantation, and *Bischofia javanica* Blume, *Cupressus lusitanica* Mill. and *Pinus patula* Schlechtend. & Cham. exotic monoculture plantations. The treatments were treated as sub-blocks, which were nested within each of the three forest blocks.

Assessment was carried out in the nine sub-blocks using a variable area technique, which ensured that woody species of different stem sizes were assessed in sampling plots of different sizes to enhance the probability of obtaining tree data in equal proportions (NAFORMA, 2010; Nath et al., 2010). The sampling unit comprised a concentric sample plot of 30 m radius with stratified sub-plots of 15 m, 10 m, 5 m and 2 m radius from the center of the sample plot. The sub-plots were nested within the sample plot. There were five sample plots in each sub-block, which gave a total of 135 sample plots in three forest blocks. The first concentric sample plot was located in the middle of a sub-block. Subsequent sample plots were located away from the centre in a Cartesian approach in a clockwise fashion (NAFORMA, 2010). The distance between any two sample plots was at least 100 m.

2.3. Data collection

Data were collected on woody species types, stem diameter at breast height (DBH) and wood cores for estimating wood specific gravity (Gregoire and Valentine, 2007; Coe, 2008). The 30 m radius plot was used to measure the DBH of trees >50 cm in diameter. The 15 m radius sub-plot was used to measure the DBH of trees of 20.1–50 cm in diameter. The 10 m radius sub-plot was used to measure the DBH of trees of 10.1–20 cm in diameter. The 5 m radius sub-plot was used to measure the DBH of trees of 5.1–10 cm in diameter; while the 2 m radius sub-plot was used to measure the DBH of saplings $\geqslant$1.5 m in height, but less than 5 cm in DBH (NAFORMA, 2010).

Woody species were identified by their botanic names with the assistance of a plant taxonomist. Stem DBH was measured in centimetres at 1.3 m above the ground using a diameter tape for all stems $\geqslant$1 cm in DBH. The DBH of trees with a buttress was measured above the buttress. Tree stems were cored at 1.3 m height starting from the bark to the pith using an increment borer of 12 mm in diameter (Chave et al., 2006; Condit, 2008; Williamson and Wiemann, 2010). For every tree species that was represented in a sample plot or sub-plot, a single tree stem was randomly selected and cored. Only tree stems $\geqslant$10 cm DBH were cored in order to avoid destroying younger tree stems. Core samples were stored in zip-lock bags and labelled by block, sub-block, sample plot number, sub-plot, tree species and DBH (Condit, 2008). The volume of fresh wood cores was estimated using the water displacement method within three days of data collection (Condit, 2008). The wood cores were thereafter oven-dried until they attained constant weight (Chave et al., 2006; Williamson and Wiemann, 2010). The dry weight was divided by the volume of fresh weight to estimate the wood specific gravity (Chave et al., 2006; Condit, 2008). Information was obtained on the stand age of each forest type from forest compartment registers at the Kenya Forest Service office in Kakamega Forest (KFS, 2010). The information was corroborated by elderly members of forest adjoining communities who worked as casual labourers during commercial logging and forest plantation establishment.

2.4. Data analysis

The data that were obtained from the field were used to derive woody species richness, species diversity, stem density, basal area and aboveground biomass. Tree species diversity was calculated using Shannon diversity index (Pena-Claros, 2003; Magurran, 2004; Newton, 2007). Aboveground biomass was calculated using the allometric equation for moist tropical forests (Chave et al., 2005). The wood specific gravity of most of the tree species was not similar to those of the regional wood density database for tropical forests (Chave et al., 2006), but the differences were fairly small in most cases. Given that the carbon content of aboveground wood biomass is globally considered to constitute about 50% of wood biomass (Brown, 1997; Chave et al., 2005; Gibbs et al., 2007), our estimates of aboveground biomass were divided by two to obtain aboveground carbon stock. Analysis of variance was used to test for possible variation in woody species diversity, stem density, stand basal area and aboveground carbon stock among different forest types at 5% significance level in Genstat statistical software version 17 (Buysse et al., 2004; Sokal and Rohlf, 2012; VSN International, 2014). Post hoc tests were carried out to detect significant differences among means using the Ryan–Einot–Gabriel–Welsch Multiple Range Test at 5% significance level (Krull and Craft, 2009; Sokal and Rohlf, 2012; Holt et al., 2013). Analysis of covariance was used to assess variation in aboveground carbon stock under uniform stand age and stem density (Sokal and Rohlf, 2012; VSN International, 2014).

3. Results

3.1. Woody species diversity

A total of 7625 woody stems from 85 tree species were recorded. The distribution of these woody species among the nine forest types ranged between 5.0 ± 1.0 and 34.0 ± 2.3 species per ha, which resulted in a significant variation in woody species richness among the forest types ($F_{(1,8)} = 12.30$; $p < 0.001$). The multiple range test indicated that the variation was caused by significantly lower woody species richness in young secondary forest, *B. javanica* monoculture plantation and *C. lusitanica* monoculture plantation (Table 1). There was, however, no significant variation in woody species richness among the rest of the forest types (disturbed primary forest, old-growth secondary forest, middle-aged secondary forest, mixed indigenous plantation and *M. eminii* and *P. patula* monoculture plantations) ($F_{(1,5)} = 0.96$; $p = 0.463$). Similarly, there was a significant variation in Shannon diversity index ($F_{(1,8)} = 5.46$; $p = 0.002$), which was attributed to significantly lower woody species diversity indices in young secondary forest and *B. javanica* monoculture plantation than the other forest types (Table 1). Old-growth and middle-aged secondary forests had relatively higher woody species diversity indices than mixed indigenous plantation and *C. lusitanica, M. eminii* and *P. patula* monoculture plantations, but the difference was not statistically significant (Table 1).

3.2. Stem density and basal area

There was a significant variation in stem density among the nine forest types ($F_{(1,8)} = 2.98$; $p = 0.029$). The variation was caused

Table 1

Stand age, woody species richness and Shannon diversity indices of secondary and plantation forest types in Kakamega Forest in western Kenya. Forest types with dissimilar letters in superscripts had significantly different woody species richness and Shannon diversity indices.

Forest type	Mean stand age (years)	Woody species richness (species per ha)	Shannon diversity index
Old secondary forest	63.3 ± 7.9	34.0 ± 3.5^{d}	14.3 ± 2.8^{c}
Middle-aged secondary forest	33.3 ± 4.9	25.7 ± 5.4^{cd}	12.3 ± 3.0^{bc}
Young secondary forest	10.0 ± 0.0	5.0 ± 1.0^{a}	2.7 ± 0.0^{a}
Bischofia javanica plantation	40.1 ± 0.7	10.7 ± 0.3^{ab}	2.5 ± 0.5^{a}
Cupressus lusitanica plantation	34.3 ± 2.2	10.0 ± 1.5^{ab}	4.8 ± 0.7^{ab}
Maesopsis eminii plantation	51.0 ± 9.3	20.3 ± 3.8^{bc}	8.3 ± 1.6^{abc}
Mixed indigenous plantation	70.3 ± 2.9	26.7 ± 2.7^{cd}	9.0 ± 0.9^{abc}
Pinus patula plantation	42.9 ± 2.0	19.3 ± 2.4^{bc}	9.5 ± 1.5^{abc}
Disturbed primary forest (control)	150.0 ± 0.0[*]	32.7 ± 2.0^{cd}	10.9 ± 2.2^{abc}
p value		<0.001	0.002
l.s.d.		8.34	5.34

[*] The mean stand age of the disturbed primary forest was estimated at 150 years as suggested by Holl and Zahawi (2014). The presented values are means with standard error of mean.

by a significantly higher stem density in young secondary forest stands (6580 ± 256.3 stems per ha) and significantly lower stem densities in middle-aged secondary forest stands (3311 ± 624.9 stems per ha) and *C. lusitanica* exotic monoculture plantations (3800 ± 426.1 stems per ha) (Table 2). The stem densities of the rest of the forest types were not significantly different from each other.

Basal area varied significantly among the nine forest types ($F_{(1,8)} = 13.02$; $p < 0.001$) because of significantly lower stand basal area in young secondary forest (18.0 ± 1.0 m^2 ha^{-1}) and a significantly higher stand basal area in the disturbed primary forest (53.5 ± 7.0 m^2 ha^{-1}) (Table 2). A comparison of stand basal area among plantation and secondary forests of comparable stand age indicated that the former tended to have relatively larger stand basal area than the latter, but the difference was not statistically significant (Table 2). For instance, mixed indigenous plantation forests had relatively larger basal area (39.0 ± 3.2 m^2 ha^{-1}) than old-growth secondary forest stands (34.9 ± 4.2 m^2 ha^{-1}). Similarly, all monoculture plantations had relatively larger basal area than middle-aged secondary forest (Table 2).

3.3. Aboveground carbon stock

There was a significant variation in aboveground carbon stock among the forest types ($F_{(1,8)} = 15.4$; $p < 0.001$) (Table 3). With 345.0 ± 54.4 tons per ha, the disturbed primary forest had significantly more aboveground carbon stock than all other forest types. Mixed indigenous plantation forest had significantly more aboveground carbon stock (240.3 ± 23.3 tons per ha) than all secondary and plantation forest types ($F_{(1,7)} = 10.6$; $p < 0.001$). Old-growth secondary forest stands had significantly more aboveground carbon stock (195.8 ± 36.8 tons per ha) than middle-aged and young secondary forests, and all monoculture plantation forests ($F_{(1,6)} = 9.51$; $p < 0.001$). The aboveground carbon stock of *B. javanica* and *M. eminii* monoculture plantations was comparable, but significantly more than that of middle-aged and young secondary forests, and *C. lusitanica* and *P. patula* monoculture plantations (Table 3).

Forest stands with relatively larger basal area tended to have significantly more aboveground carbon stock. This was the case with old-growth forest stands, such as the disturbed primary forest, mixed indigenous plantation and old-growth secondary forest (Table 3). Since, the variation was partly attributable to differences in stand age, analysis of covariance indicated that only the mixed indigenous plantation would register significantly more aboveground carbon stock ($F_{(1,8)} = 2.35$; $p = 0.037$) if all the forest types were of the same stand age (Table 3).

Table 3
Aboveground carbon stock of different secondary and plantation forest types in Kakamega Forest in western Kenya. Aboveground carbon stocks were adjusted through analysis of covariance in order to eliminate variation attributable to difference in mean stand age. Forest types with distinct letters in superscripts had significantly different aboveground carbon stocks. The presented values are means with standard error of mean.

Forest type	Mean stand age (years)	Aboveground carbon stock (tons ha^{-1})	
		Actual carbon stock	Adjusted carbon stock
Old secondary forest	63.3 ± 7.9	195.8 ± 36.8^{bc}	156.4 ± 19.9^{a}
Middle-aged secondary forest	33.3 ± 4.9	100.7 ± 15.2^{ab}	133.8 ± 19.6^{a}
Young secondary forest	10.0 ± 0.0	59.3 ± 4.8^{a}	150 ± 22.8^{a}
Bischofia javanica plantation	40.1 ± 0.7	149.2 ± 3.2^{abc}	164.1 ± 11.2^{a}
Cupressus lusitanica plantation	34.3 ± 2.2	96.4 ± 0.2^{ab}	133.6 ± 10.7^{a}
Maesopsis eminii plantation	51.0 ± 9.3	157.6 ± 21.0^{abc}	148.0 ± 19.2^{a}
Mixed indigenous plantation	70.3 ± 2.9	240.3 ± 23.3^{c}	183.5 ± 20.7^{ab}
Pinus patula plantation	42.9 ± 2.0	106.3 ± 7.5^{ab}	121.4 ± 10.7^{a}
Disturbed primary forest (control)	150.0 ± 0.0	345.0 ± 54.4^{d}	93.9 ± 39.5^{a}
p value		<0.001	0.037
l.s.d.		70.2	58.34

3.4. Wood specific gravity

Analysis of wood specific gravity among tree species indicated that *Manilkara butugi* Chiov., *Craibia brownii* Dunn, *Cussonia arborea* Hochst. ex Delile, *Olea capensis* L., *Acacia abyssinica* Hochst. ex Benth., *Celtis gomphophylla* Baker, *Margaritaria discoidea*, *Celtis africana* Burm.f., *Drypetes gerrardii* Radcl.-Sm. and *Rawsonia lucida* had the highest wood specific gravity in descending order. Young secondary forest was not represented in this analysis because it did not have any of these tree species. The analysis recorded a significant variation in the abundance of these tree species among different secondary and plantation forest types ($F_{(1,7)} = 2.96$; $p = 0.012$) (Table 4). However, higher abundance of these tree species did not necessarily result in more aboveground carbon stock. For instance, *P. patula* and mixed indigenous plantations had significantly higher abundance of these tree species than other forest types, but the aboveground carbon stock of *P. patula* was significantly lower while that of mixed indigenous plantation was significantly higher (Table 4). The results indicated that *P. patula* plantation had a significantly lower aboveground carbon stock

Table 2
Variation in stem density, stem DBH, stand basal area and wood specific gravity among different secondary and plantation forest types in Kakamega Forest in western Kenya. Forest types with dissimilar letters in superscripts had significantly different stem densities, mean DBH, stand basal area and mean wood specific gravity. The presented values are means with standard error of mean.

Forest type	Stem density (stems ha^{-1})	Stem DBH (cm)	Stand basal area (m^2 ha^{-1})	Mean wood specific gravity
Old secondary forest	4570 ± 435.9^{ab}	61.56 ± 10.95^{bc}	46.02 ± 9.75^{c}	0.516 ± 0.013^{cd}
Middle-aged secondary forest	3311 ± 624.9^{a}	34.95 ± 3.93^{a}	24.99 ± 2.15^{ab}	0.473 ± 0.018^{ab}
Young secondary forest	6580 ± 256.3^{b}	31.0 ± 0.37^{a}	21.63 ± 1.16^{a}	0.546 ± 0.001^{d}
Bischofia javanica plantation	5071 ± 113.9^{ab}	30.55 ± 0.16^{a}	47.05 ± 1.32^{c}	0.511 ± 0.002^{bcd}
Cupressus lusitanica plantation	3800 ± 426.1^{a}	30.72 ± 0.26^{a}	32.2 ± 0.53^{abc}	0.449 ± 0.002 ^{a}
Maesopsis eminii plantation	5339 ± 1079.8^{ab}	34.57 ± 5.52^{a}	42.13 ± 4.24^{bc}	0.480 ± 0.004^{abc}
Mixed indigenous plantation	3938 ± 194.8^{ab}	51.71 ± 4.39^{ab}	43.72 ± 0.54^{c}	0.541 ± 0.005^{d}
Pinus patula plantation	4352 ± 268.8^{ab}	31.14 ± 0.13^{a}	35.17 ± 0.32^{abc}	0.455 ± 0.002^{a}
Disturbed primary forest (control)	4907 ± 685.8^{ab}	75.49 ± 6.71^{c}	70.06 ± 5.59^{d}	0.542 ± 0.011^{d}
p value	0.029	<0.001	<0.001	<0.001
l.s.d.	1682.3	14.49	11.17	0.027

Table 4
Relating the abundance of ten tree species with the highest wood specific gravity to mean stem DBH and aboveground carbon stock in different secondary and plantation forest types in Kakamega Forest in western Kenya. Forest types with distinct letters in superscripts had significantly different tree species abundance, mean stem DBH and aboveground carbon stock. The presented values are means with standard error of mean.

Forest type	Tree species abundance (%)	Mean wood specific gravity	Mean stem DBH (cm)	Aboveground carbon stock (tons ha^{-1})
Old secondary forest	16.37 ± 2.95^{a}	0.098 ± 0.046^{a}	34.93 ± 16.76^{ab}	38.82 ± 24.95^{ab}
Middle-aged secondary forest	8.58 ± 1.21^{a}	0.036 ± 0.066^{a}	13.39 ± 7.64^{a}	10.23 ± 9.14^{a}
Bischofia javanica plantation	6.68 ± 0.01^{a}	0.023 ± 0.123^{a}	17.9 ± 2.26^{a}	3.11 ± 0.52^{a}
Cupressus lusitanica plantation	10.02 ± 1.93^{a}	0.048 ± 0.087^{a}	1.59 ± 0.21^{a}	0.27 ± 0.03^{a}
Maesopsis eminii plantation	17.16 ± 2.97^{a}	0.062 ± 0.058^{a}	16.37 ± 4.63^{a}	6.87 ± 1.36^{a}
Mixed indigenous plantation	27.55 ± 5.32^{ab}	0.290 ± 0.059^{a}	61.44 ± 1.17^{b}	108.73 ± 33.93^{c}
Pinus patula plantation	33.52 ± 2.1^{ab}	0.209 ± 0.174^{a}	8.4 ± 1.59^{a}	2.72 ± 0.26^{a}
Disturbed primary forest	19.17 ± 3.571^{a}	0.179 ± 0.055^{a}	66.57 ± 23.84^{bc}	108.73 ± 33.93^{c}
p value	0.012	0.076	0.002	<0.001
l.s.d.	15.33	0.2493	29.87	49.77

than mixed indigenous plantation because its trees had a significantly smaller mean stem DBH than those of mixed indigenous plantation (Table 4).

Forest stands which had both larger stem DBH and higher wood specific gravity, such as disturbed primary forest and mixed indigenous plantation, had higher aboveground carbon stock (Table 4). Forest stands whose stem DBH was large, but had a lower mean wood specific gravity, such as *B. javanica* and *M. eminii* monoculture plantations, had relatively lower aboveground carbon stock than those with larger stem DBH and higher wood specific gravity. Those that had significantly smaller stem DBH and comparable or lower wood specific gravity, such as young and middle-aged secondary forest stands; had significantly lower aboveground carbon stock (Table 4).

4. Discussion

4.1. Aboveground carbon stock

The range of aboveground carbon stock in this tropical forest, as recorded in this study, is within what has been reported in other studies in tropical forests (Glenday, 2006; Keith et al., 2009; Stegen et al., 2011). Moreover, the study shows that aboveground carbon stocks vary across forest stands. The results suggest that stand age is one of the most important factors explaining the variation in aboveground carbon stock among forest stands. For instance, the disturbed primary forest had significantly more aboveground carbon than all other forest types. Similarly, mixed indigenous plantation forest and old-growth secondary forest stands had more aboveground carbon stock than relatively younger secondary and plantation forest stands. The role of stand age in explaining the variation in aboveground carbon stock among forest stands is attributable to the fact that older tree stands often comprise relatively larger trees, which, as illustrated by Omeja et al. (2011) and Ifo et al. (2014), tend to hold more aboveground carbon than younger forest stands. The results reaffirm the role of old-growth forest stands, such as the disturbed primary forest, old-growth secondary forest and mixed indigenous plantation, as major terrestrial carbon sinks (Stegen et al., 2011) whose disturbance may have a significant impact on the global carbon balance.

4.2. Aboveground carbon offset additionality

A key result of this study was the observation that mixed indigenous plantation forest had significantly more aboveground carbon stock than secondary forest stands of comparable stand age, while the other plantation forest types did not. The result suggests that mixed indigenous plantation forests have the potential to secure aboveground carbon offset additionality in moist tropical forest ecosystems. Analysis of variables that were expected to contribute to the aboveground carbon offset additionality of this indigenous plantation forest gave interesting results. One variable that was expected to contribute to aboveground carbon offset additionality, but did not, is woody species diversity. The results indicated that secondary forests were not significantly different from a majority of plantation forests in woody species diversity in this tropical forest. The fact that mixed indigenous plantation was one of these plantations, effectively eliminated woody species diversity as a possible determinant of aboveground carbon offset additionality. The acquisition of additional woody species by plantation forests, which placed them at par with secondary forest stands in woody species richness, was traced to earlier studies in this forest by Farwig et al. (2009) and Otuoma et al. (2014). These studies illustrated that natural forest succession is active in both secondary and plantation forests of this tropical forest. Moreover, Farwig et al. (2009) indicated that the recruitment of indigenous tree species in the plantations of this tropical forest has progressed to an extent that some monoculture plantations may be hard to distinguish from secondary forest stands in the future. This observation is consistent with those of other studies in tropical forest ecosystems that have reported natural forest regeneration within plantation forest stands leading to forest plantations that are extremely rich in tree species diversity (Lugo, 1997; Parrotta et al., 1997; Duncan and Chapman, 2003; Omeja et al., 2011; Holl and Zahawi, 2014).

Another variable that was expected to explain the likely aboveground carbon offset additionality of mixed indigenous plantation forest is stand structure. However, stem density, which is one of the attributes of stand structure, was ruled out. Mixed indigenous plantation forest did not have a significantly higher stem density than other forest types. Similarly, young secondary forest, which had a significantly higher stem density than other forest types had the least aboveground carbon stock. Nonetheless, stand basal area was found to have a strong influence on aboveground carbon stock. The results indicated that forest stands with larger basal area had relatively more aboveground carbon stock than those with smaller basal area. Since stand basal area is a function of both stem DBH and stem density, and the latter had been confirmed to have negligible effect on aboveground carbon stock, the results suggest that the contribution of basal area was attributable to stem DBH. This observation is consistent with those of Chaturvedi et al. (2011), Omeja et al. (2011) and Ifo et al. (2014), which reported that large trees, though less abundant, often store more aboveground carbon than smaller ones, which are normally significantly more abundant in tropical forest stands.

There were instances where plantation forest stands with larger basal area did not hold more aboveground carbon stock than secondary forests. For instance, *B. javanica* and *M. eminii* monoculture plantations were observed to have larger basal area than old-growth secondary forest, but had significantly lower aboveground carbon stock than the latter. The results indicated that this was caused by a higher mean wood specific gravity in old-growth secondary forest stands than in the two plantation forests. These observations suggest that wood specific gravity and stand basal area synergized to influence aboveground carbon stock. A comparison of our results with other studies indicated that some studies (e.g. Ifo et al., 2014) identified a strong correlation between stem diameter and aboveground biomass, but they did not consider wood specific gravity as a major predictor of aboveground carbon stock. Perhaps this arose as a result of the failure to look at the synergistic relationship of the two variables. Other studies (e.g. Chaturvedi et al., 2011) identified the role of the two variables in predicting the carbon offset potential of forest stands.

Although up to this point our study has identified stem DBH and wood specific gravity as important variables in aboveground carbon accumulation, it has not explained how they contributed to the aboveground carbon offset additionality of mixed indigenous plantation over old-growth secondary forest given that the two forest types had largely similar tree species richness and stand age. The larger stem DBH of mixed indigenous plantation is likely to have occurred as a result of steady growth and persistence of planted trees in this plantation as opposed to the simultaneous recruitment and mortality of tree stems over several stages of successional change in the old-growth secondary forest. As illustrated by Montgomery and Chazdon (2001), McElhinny et al. (2005), Otuoma et al. (2014) on changes in structural complexity of forest stands, the difference in basal area between mixed indigenous plantation and old-growth secondary forest can be explained by the fact that it takes a secondary forest stand a relatively longer duration to attain structural complexity similar to that of a planted forest. According to Finegan (1996), Pena-Claros (2003), Norden et al. (2009) and Bonner et al. (2013), this phenomenon arises from the fact that early successional species, which occupy a secondary forest stand in the first one to three decades, disappear as intermediate successional species take over, while trees that are planted in plantation forests at the same time as early successional tree species persist to maturity thereby ending up with larger stems. Thus, the two forest types may have had fairly similar tree species in this tropical forest, but those in the old-growth secondary forest were relatively younger and hence had smaller stem DBH. The results indicated also that mixed indigenous plantation forest had a significantly larger proportion of individual trees with high wood specific gravity than old-growth secondary forest. This can be attributed to the fact that tree species in the mixed indigenous plantation were selected based on their superior wood attributes (KFS, 2010), while those in the old-growth secondary forest recruited naturally. Thus, the mixed plantation ended up with more individual trees with high wood specific gravity, which significantly increased its mean wood specific gravity compared to the old-growth secondary forest where the same tree species were represented but with a lower population.

4.3. Implications for carbon-driven forest restoration interventions

The finding of this study that mixed indigenous plantation forest had higher aboveground carbon stock than old-growth secondary forest, and yet they were of comparable stand age and woody species richness, provides useful insight for investment in carbon-driven forest restoration efforts. The result suggests that planting a large proportion of tree species with high wood specific gravity leads to significantly more aboveground carbon stock than passive restoration interventions, such as natural forest regeneration. This observation is consistent with findings of Oliver and Fried (2013) that passive forest restoration efforts may not produce the desired carbon offset benefits. The target restoration species should, however, comprise late successional tree species because of their likelihood to persist for a long period of time, unlike early and intermediate successional species which tend to disappear in the course of natural forest succession (Finegan, 1996; Bonner et al., 2013; Otuoma et al., 2014). The results suggest that planting tree species with lower wood specific gravity may not be a productive land use for forest restoration efforts that target carbon offset benefits, unless the plantations are intended for timber production in the long-term.

The results also indicate that it is not only planting late successional species with high wood specific gravity that leads to higher aboveground carbon stock, such trees must be managed to produce large stem diameter as well. Fortunately, stem diameter can be easily manipulated through silvicultural management operations, such as spacing, thinning or pruning (Baishya et al., 2009; Bonner et al., 2013). Although, this observation emphasizes the need for active silvicultural management of forest restoration stands in order to maximize on return on investment, it raises queries regarding what constitutes appropriate silvicultural operations for tree species planted primarily for aboveground carbon offset benefits. This is one of those areas that have not received adequate attention in tropical forest restoration efforts.

Overall, the results of this study suggest that investing in late successional tree species with high wood specific gravity and manipulating them to develop large stem diameter, without compromising on tree height, has huge potential to secure aboveground carbon offset additionality with significant monetary and environmental benefits. Besides having the capacity to offset significant amounts of greenhouse gas emissions, conservative estimates indicate that such forest restoration efforts can generate between US$400 and US$8000 per hectare of tropical forest depending on the prevailing market value of carbon (Laurance, 2007). However, the results suggest also that poor choice of tree species and passive silvicultural operations may result in failure to attain aboveground carbon offset additionality and lead to significant economic losses.

Despite illustrating the key determinants of aboveground carbon offset additionality and related monetary and environmental benefits, the study notes that forest managers and investors in carbon offset schemes are likely to encounter a few challenges and limitations in implementing these findings. One major limitation is lack of wood specific gravity values for tree species in some tropical regions (global database on wood specific gravity, Chave et al., 2006). Thus, some tropical forest restoration efforts may fail to secure aboveground carbon offset additionality due to poor choice of tree species. The situation calls for redoubling efforts to expand the global coverage of the wood specific gravity database with a focus on obtaining data from different tropical forests. The second challenge is the ability of forest managers to protect the gains made through aboveground carbon offset additionality by avoiding sources of leakage, such as unplanned logging and conversion of forests to other land use. Leakage remains one of the greatest challenges to sustaining gains made through aboveground carbon offset additionality, particularly under voluntary carbon offset schemes (Brown et al., 2000; Fahey et al., 2010).

5. Conclusion

Stem DBH and wood specific gravity are key determinants of additionality in aboveground carbon offset interventions in tropical forests. They function synergistically to give the desired aboveground carbon offset benefits. The two variables can be integrated by selecting late succession tree species with high wood specific gravity and employing sound silvicultural management operations that enhance tree stem development in forest restoration planting.

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