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Original research article

Effects of 34-year-old *Pinus taeda* and *Eucalyptus grandis* plantations on soil carbon and nutrient status in former miombo forest soils

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ABSTRACT

There is a strong need in Mozambique to counteract decades of deforestation and forest degradation by planting new forests. Plantations of *Pinus/Eucalyptus* species and maintenance of mature miombo forests are activities supported by the REDD+ mechanism (Reducing Emissions from Deforestation and forest Degradation) in climate negotiations. This study examined the effects of first-rotation *P. taeda* L. (Loblolly pine) and *E. grandis* Hill ex Maiden plantations (ca. 34 years old) on soil carbon status compared with adjacent dry miombo forest. At three study sites located in the Western Highlands of Manica Province, Mozambique, study plots with *Pinus taeda, Eucalyptus grandis* and mixed-deciduous miombo species were delineated. The selection criteria were (i) forest stand of first-rotation plantation of *Pinus/Eucalyptus*, located adjacent to miombo forest, (ii) plantations established on soils similar to miombo forest soils, and (iii) former land use similar to that at current miombo sites. Stocks of soil organic carbon (SOC), total nitrogen (N) and extractable phosphorus (P) were quantified. Soil pH (H_2O), cation exchange capacity (CEC) and base saturation (BS) were measured in soil extracted with ammonium acetate.

Plantations of *P. taeda* and *E. grandis* increased total SOC stocks (0–50 cm) and N stocks in the top 10 cm. Assuming steady state in the miombo stands, the estimated net stock change in soil carbon was 1.41 Mg ha^{-1} yr^{-1} in *P. taeda* and 1.53 Mg ha^{-1} yr^{-1} in *E. grandis* stands. Estimated N accumulation rate was 32 kg ha^{-1} yr^{-1} in *P. taeda* and *E. grandis* stands. *P. taeda* had no significant effect on extractable P, soil pH and BS, but had significantly higher. CEC compared with miombo forest soil. *E. grandis* decreased P stocks, but increased soil pH and BS.

Overall, *P. taeda* and *E. grandis* plantation had a large impact on SOC in dry miombo forest and also affected soil acidity and soil nutrient status, except for total soil N stocks. These effects of tree plantation on soil reflected differences in management practices between miombo forest and plantations, with the latter being subjected to better protection against fires and illegal cutting.

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1. Introduction

Pinus and *Eucalyptus* plantations can have a marked impact on carbon (C) pools and nutrient status in tropical soils. However, the magnitude and nature of these effects may vary depending on the features of particular sites (Bahn et al., 2009; Chang et al., 2014; Jandl et al., 2007; Paul et al., 2002). A meta-analysis of afforestation by Li et al. (2012) concluded that soil C stocks increased after plantation of hardwood species such as *Eucalyptus*, but did not change with softwood species such as *Pinus*. The increase in soil C stocks by *Eucalyptus* spp. occurred in both the organic horizon and the mineral soil, whereas *Pinus* spp. increased soil C stocks in the organic horizon, but generally depleted C stocks in the mineral soil, resulting in insignificant overall change (Li et al., 2012). Lower soil C stocks in the mineral horizon under *Pinus* spp. plantations could be a consequence of lower below-ground litter production compared with *Eucalyptus* spp.

Li et al. (2012) also found that changes in stocks of soil C and nitrogen (N) are positively correlated and have a similar temporal pattern, but that changes in soil N are detected at later age stages than changes in soil C. In general, changes in soil C stocks are detected 30–40 years after plantation, whereas significant changes in soil N stocks appear from around 50 years after plantation. One main cause of the strong positive correlation between changes in soil C and soil N is the fairly fixed stoichiometric relationship between C and N in plant litter, a relationship that also includes phosphorus (P). As with N, the relationship between soil C and soil P suggests that an increase in soil organic C is associated with increased stocks of soil P, dominated by P in organic form. This is partly because as soil C stock increases, P fixation in minerals decreases (El-Baruni and Olsen, 1979). However, the correlation between C and P is generally weak ($r = 0.40$–0.57; Hou et al., 2014).

In highly weathered, leached and acidic soils, such as those in Mozambique, available soil P is typically low and is often the major growth-limiting factor, whereas N is not a limiting nutrient (Aggangan et al., 1996; Binkley, 1997; Högberg, 1986). In highly weathered soils, the availability of P also depends on soil pH level. Plant P availability is highest in the soil pH range 6–7, while P fixation is at its lowest (Devau et al., 2009). Uptake of P by tree species may cause a reduction in plant-available P, particularly if forest growth also results in lower soil pH. Lower soil pH and base saturation (BS) can be expected, as a result of higher accumulation of base cations by plants (Nilsson et al., 1982). In addition, increased stocks of soil C can be expected to increase the organic acidity, manifested in lower BS, lower pH and higher cation exchange capacity (CEC) of the soil (Berthrong et al., 2009).

Knowledge of how tree species affect ecosystem C and nutrient status is of particular importance in Mozambique, where there is a strong need to counteract decades of deforestation and forest degradation by planting new forest. New plantations and maintenance of mature woodlands are activities supported by the REDD+ mechanism in the United Nations Framework Convention on Climate Change (UNFCCC) and to sustain livelihoods. In Mozambique, forest plantations are desired by society and the national aim is to establish 1 million ha of forest plantations by 2022, i.e. a 10-fold increase in area compared with the current situation (see Coetzee and Alves, 2005; MINAG, 2009). Forest plantations in Mozambique consist mainly of *Pinus* and *Eucalyptus* spp., which are also common in neighbouring countries (e.g. South Africa and Zimbabwe; FAO, 2001). The plantations are mostly established on degraded miombo woodlands, thickets and degraded agricultural land. On average, degraded miombo sites, or high-utilisation sites, store approximately 15 Mg C ha^{-1} in above-ground biomass, which is significantly lower average C storage than at medium and low utilisation sites (e.g. 31 Mg C ha^{-1}) (Jew et al., 2016). However, expansion of forest plantations in Mozambique can also be achieved through direct conversion of non-degraded forests (Indufor, 2012; Nhantumbo and Salomão, 2010). Despite this large-scale land use change, lack of knowledge on how forest plantations affect soil organic carbon (SOC) and nutrient status in the long run limit the possibility to develop local guidelines on sustainable forestry.

Pinus/*Eucalyptus* plantations and native miombo forest differ in several aspects. The non-native species are easily managed, show rapid adaptability, high above-ground carbon sequestration rate and high production and productivity of wood (Dohrenbusch, 2011; FAO, 2001; Lugo and Brown, 1993). Most species in miombo forest have low growth rates (Ciais et al., 2011; Lupala et al., 2014; Sitoe, 1999) and natural regeneration can be relatively slow, due in part to lack of forest management practices (Jew et al., 2016; Kalaba et al., 2013; Williams et al., 2008). However, high production in plantations is generally associated with a high demand for soil nutrients, which may adversely affect soil properties (Berthrong et al., 2009; Jien et al., 2011; Knoepp et al., 2000; Liao et al., 2012; Zhang et al., 2012).

Dry miombo woodland (hereafter referred to as 'miombo forest') is defined as a sub-type of dry forest located in tropical climates, with summer rains bringing 500–1500 mm annually and a dry period of 5–8 months (Blackie et al., 2014; FAO, 2012). Miombo forest covers a large part (around 67%) of the inland of Mozambique (Sitoe et al., 2012), and is in fact the most extensive tropical seasonal woodland and dry forest formation in Africa (Frost, 1996). It is dominated by the genera *Brachystegia, Julbernardia* and *Isoberlina* (Fabaceae, subfamily Caesalpinioideae).

Miombo forest provides a wide range of goods and ecosystems services, thus playing an important role for the food security and livelihood of households, as well as climate change mitigation and adaptation. Despite its great importance, miombo forest is severely under threat (Bruschi et al., 2014; Cuambe and Marzoli, 2006; Jansen et al., 2008; Sawe et al., 2014; Sitoe et al., 2012). Increased population density is an important driver of deforestation and miombo degradation through increased wood extraction for energy (e.g. charcoal), clearance for subsistence agriculture and uncontrolled fires (see Backéus et al., 2006; Bruschi et al., 2014; Nhantumbo et al., 2001; Sawe et al., 2014; Sitoe et al., 2012).

The process of miombo degradation is generally taking place by selective logging, fires and clearance of small areas within forests (subsistence agriculture), resulting in a mosaic pattern of agriculture within the forest (Sitoe et al., 2012). Although dominant tree species in miombo forest are fire-resistant (Frost, 1996), fires cause serious damage to the ecosystem, including reducing litter carbon (C). Fires not only volatilise soil C, but also soil nitrogen (N). Here we define degraded miombo forest, or high-utilisation sites as designated by Jew et al. (2016), as forest with reduced capacity to store C as a result of anthropogenic changes (adapted from Thompson et al., 2013) and which has lost much of its original biomass, structure and productivity (Lamb and Gilmour, 2003). This definition also denotes reduced forest cover, as long as the site, by definition, remains a "forest" (Sitoe et al., 2012; Tavani et al., 2009).

The aim of this study was to examine the effects of first-rotation *Pinus taeda* L. (Loblolly Pine) and *Eucalyptus grandis* Hill ex Maiden plantations (ca. 34 years old) on soil C status compared with that of adjacent dry miombo forest. The pH (H_2O) and stocks of SOC, total N, extractable P and exchangeable base cations in the soil were determined. The hypothesis was that higher biomass production in *P. taeda* and *E. grandis* plantations compared with miombo forest leads to (i) higher soil C and N stocks than in degraded dry miombo forest, (ii) with a stronger effect of *E. grandis* than that of *P. taeda*; (iii) decreased stocks of extractable P as a consequence of increased nutrient demand and uptake in trees; and (iv) increased soil acidification as a consequence of increased stocks of soil C and nutrient accumulation in the biomass.

2. Material and methods

2.1. Study area

The study area is located in the Western Highlands (1100–1700 m.a.s.l.) in Manica Province, Central Mozambique, and comprises three study sites: the Penhalonga forest unit (18°47′53″ S, 32°47′24″ E) and Rotanda (19°34′50″ S, 32°54′07″ E), both managed by the IFLOMA company, and the Inhamacari forest unit (18°56′40″ S, 32°42′43″ E), managed by the Machipanda Research Centre of Eduardo Mondlane University. The climate in the study area is temperate, with dry winters and warm summers (Cwb), according to the Köppen–Geiger climate classification (Peel et al., 2007). Mean annual temperature is 21.2 °C and mean annual precipitation is 1300 mm, with a dry season during May–October (IIAM, 2007). Ferralsols and Cambisols dominate the study area (IIAM, 2007). The vegetation in the region is dominated by dry miombo forest at high altitude and non-native commercial forest plantations. The area covered by these forest types is estimated to be 31 000 ha and 21 000 ha, respectively, based on a forest inventory of Manica Province (Cuambe and Marzoli, 2006).

Miombo forest is characterised by a canopy providing 10%–40% cover, average stand height 12 m and an understorey dominated by different grass species (Cuambe and Marzoli, 2006). As in other areas, local miombo forest in Mozambique is subjected to uncontrolled fires (often yearly), wood collection for energy and clearing for agriculture. Most mature plantations of pines and eucalyptus were established between 1977 and 1980 (Adam et al., 1991) and represent the first generation of large-scale commercial forest plantations in the country. The plantations consist of *P. taeda, P. patula, P. elliottii, E. grandis, E. camaldulensis* and *E. cloeziana* and receive no fertiliser. Between 1977 and 1992, these plantations became almost inaccessible due to the civil war in Mozambique, but they were still better protected against fire and illegal cutting than miombo forest (A. Esequias, personal communication, Feb. 2012).

2.2. Study design and data collection

The three sites were considered blocks in a randomised block design. On each site, forest stands that fulfilled the following criteria were identified: (i) Stand of first-rotation plantation of *P. taeda*/*E. grandis* (mean age 34 years ± 6%), located adjacent to miombo forest, (ii) forest plantation established on soil similar to miombo forest soil, and (iii) a former land use similar to that in the current miombo forest. The study plots were identified in the field with the help of senior staff from IFLOMA (M. Mariano, personal communication, Oct. 2012) and Machipanda Research Centre (A. Esequias, personal communication, Sept. 2012).

Three square sampling plots (30 m × 30 m) were delineated at each site, with one forest type each (*P. taeda; E. grandis* and miombo forest). In each plot, diameter at breast height (DBH) was measured on all trees and shrubs with DBH $\geq$ 5 cm and each individual was identified at species level in the field with the help of a botanically skilled technician. Soil and litter mass were sampled using a grid sampling design involving 10 spots in each plot, in order to cover the expected variability in soil and litter mass.

The litter layer was sampled within a 15 cm × 15 cm frame. Soil samples were collected with a steel corer (28 mm diameter) and soil cores were divided into depth sections (0–10, 10–30 cm) and pooled together to one sample per soil layer and main plot. For the deeper layers (30–50 cm), soil samples were collected from one soil pit in each plot. In total, nine soil samples were collected per forest type for chemical analyses (three sites × three soil layers). A 50 mm diameter steel corer was used to collect nine additional soil samples for determination of soil texture. The forest and soil inventory was carried out in September–October 2012.

2.3. Laboratory analyses

Soil analyses were performed at the Faculty of Agronomy and Forest Engineering (FAEF), University Eduardo Mondlane (UEM) in Maputo, Mozambique. Sieved (<2 mm) and oven-dried soil samples (40 °C) were analysed for SOC content using the Walkley–Black (WB) method (Houba et al., 1989). A correction factor of 1.3 was used to convert the WB organic carbon value to total SOC. Total N content was determined with the Kjeldahl method (Westerhout and Bovee, 1985). Exchangeable cation concentrations were determined by extracting dry soil with 1 M ammonium acetate. Concentrations of exchangeable base cations (i.e. Ca^{+2}, Mg^{+2}, K^+ and Na^+) were determined by atomic absorption spectrophotometry according to Houba et al. (1989). Exchangeable acidity ($Al^{3+} + H^+$) was determined by complexometric titration (Houba et al., 1989). The Bray I method (Houba et al., 1989) was used for determination of extractable P. Soil pH was measured potentiometrically in a 1:2.5 soil: water solution (Houba et al., 1989). Soil texture was determined by mechanical analyses and the Robinson pipette method was used for determination of the clay content (Westerhout and Bovee, 1985).

2.4. Calculations

The dry mass of the litter layer was determined after oven drying (65 °C) to constant weight. The C content in the litter layer was then estimated assuming that C represented 37% of the dry mass, according to the protocol of the Intergovernmental Panel on Climate Change (IPCC, 2006). Cation exchange capacity (CEC) was estimated from the sum of equivalent charges of base cations (Ca^{2+}, Mg^{2+}, K^+, Na^+) and exchangeable acidity (Al^{3+}, H^+) (Bache, 1976). Base saturation was calculated from the sum of the base cations divided by CEC.

The amount of different elements per unit area and soil layer (Mg ha^{-1}) was calculated by multiplying the soil bulk density (BD), soil layer thickness and element concentration per unit dry mass. The total stocks of soil C and nutrients in the soil profile were calculated as the sum of stocks in all layers (0–10, 10–30 and 30–50 cm depth) for each study plot. Soil BD (g cm^{-3}) was estimated per soil layer using the following general pedotransfer function (Guo and Gifford, 2002; Paul et al., 2002; Post and Kwon, 2000):

$$\mathrm{BD} = 100/((\%\mathrm{OM}/\mathrm{BD}_{\mathrm{OM}}) + ((100 - \%\mathrm{OM})/\mathrm{BD}_{\mathrm{min\ soil}})) \quad (1)$$

where %OM is soil organic matter (i.e. SOM) as estimated in the field (in %), BD_{OM} is density of organic matter (0.244 g cm^{-3}, i.e. mean organic matter bulk density) and $BD_{\mathrm{min\ soil}}$ is the bulk density of soil mineral material (1.64 g cm^{-3}, i.e. a high degree of compaction). Annual net change in SOC stocks in plantations was estimated assuming a steady state in the miombo forest over 34 years. Mean age of the non-native plantations was 34 years (±6% uncertainty error) according to oral sources, which are supported by a SIDA evaluation report of the Mozambique-Nordic Agricultural Programme (MONAP) (Adam et al., 1991).

The stocks of N and P in above-ground tree biomass were computed in order to relate nutrient stock changes in the soil to accumulation in biomass. Published allometric equations, which were selected according to criteria recommended by Smith et al. (2014), were used for these calculations. Destructive sampling included 29–33 trees for each forest type. Tree species selection in the miombo stands was based on the Importance Value Index (IVI), which measures the dominance of species in a specific area by calculating the number of stems per hectare, basal area (m^2 ha^{-1}) and relative frequency of the tree species (Agarwal, 2008; Dash, 2001; Kacholi, 2014). Generalised allometric models that had the best fit to our data were selected from linear regression. Furthermore, relative mean predicted error was computed (Chidumayo, 2013; Gonzalez-Benecke et al., 2014; Mwakalukwa et al., 2014; Ngomanda et al., 2014).

The following allometric equations (Eqs. (2)–(4)) were selected to determine the total dry weight per tree (tDW, kg $tree^{-1}$) from the stem diameter at breast height (DBH, cm at height 1.3–13.7 m).

For *P. taeda* plantations (Brown, 1997):

$$\mathrm{tDW}\ (\mathrm{kg\ tree}^{-1}) = \mathrm{Exp}\,(-1.170 + 2.119 * \mathrm{LN}\,(\mathrm{DBH})) \quad (2)$$

where DBH range was 9–39 cm, relative mean predicted error = −4%, precision = 98%, $p < 0.0001$.

For *E. grandis* plantations (Eamus et al., 2000):

$$\mathrm{tDW}\ (\mathrm{kg\ tree}^{-1}) = 0.4506 * \mathrm{DBH}^{2.082} \quad (3)$$

where DBH range was 5–74 cm, relative mean predicted error = +11%, precision = 95%, $p < 0.0001$.

For miombo forest (Chamshama et al., 2004):

$$\mathrm{tDW}\ (\mathrm{kg\ tree}^{-1}) = 0.0625 * \mathrm{DBH}^{2.553} \quad (4)$$

where DBH range was 4–36 cm, relative mean predicted error = −5%, precision = 99%, $p < 0.0001$. Biomass per forest type corresponded to the mean total dry weight of the three plots (30 m × 30 m) at the different sites, as described previously, and was expressed on a per hectare basis.

Stocks (kg ha^{-1}) of N and P in tree biomass were calculated from the above-ground biomass in plots and the N and P concentrations in the biomass, using values from the following sources. Data on the concentrations of N and P in the stem, bark and branches were taken from Turner and Lambert (1983) for *Eucalyptus* and from Dovey (2009) for *Pinus*. Foliar nutrient concentrations in native forest trees were set to 16.5 mg N g^{-1} and 1.65 mg P g^{-1}, based on Chidumayo (1994). The concentration of N in biomass of miombo trees was assumed to be 2 mg g^{-1} in stems and 3 mg g^{-1} in branches, assuming similarity in concentrations with *Pinus* and *Eucalyptus*. The P concentration in woody tissues was assumed to be 10% of N levels, based on general stoichiometric relations in plants under N- and P-limited conditions (Ågren, 2008). The concentrations of N and P in foliage of *Eucalyptus* and *Pinus* were assumed to be the same as in the native species.

Table 1

Concentrations of selected soil parameters: SOC = soil organic carbon, N = total soil nitrogen, P = extractable phosphorus; C/N ratio = ratio of organic carbon and total soil nitrogen. Values represent the mean for individual soil layers (± Standard Deviation). Different letters indicate significant difference between stand types ($p < 0.05$).

Parameter	Soil layer	Miombo forest	*P. taeda*	*E. grandis*
SOC (g kg^{-1} soil)	0–10 cm	18.6^a (3.8)	38.3^b (14.6)	37.3^b (14.1)
	10–30 cm	15.2 (4.1)	22.5 (10.1)	20.4 (11.2)
	30–50 cm	7.1^a (1.7)	10.8^a (7.2)	11.6^b (4.7)
N (g kg^{-1} soil)	0–10 cm	0.75^a (0.26)	1.73^b (0.65)	1.50^b (0.26)
	10–30 cm	1.12 (0.51)	1.07 (0.2)	1.18 (0.63)
	30–50 cm	0.60 (0.30)	0.63 (0.40)	0.63 (0.06)
P (mg kg^{-1} soil)	0–10 cm	9.6 (0.57)	10.5 (0.49)	6.7 (0.42)
	10–30 cm	4.8^a (0.20)	6.4^a (0.35)	3.3^b (0.21)
	30–50 cm	2.1 (0.07)	2.5 (0.09)	1.9 (0.09)
C/N ratio	0–10 cm	26 (5)	23 (6)	24 (5)
	10–30 cm	14^a (3)	20^b (6)	17^a (0)
	30–50 cm	14 (7)	17 (1)	19 (8)

2.5. Statistical analyses

The Linear Mixed-Effects Model in the R package "lme4" was used to analyse the data (Bates et al., 2015). The model was run with study sites treated as random effect and tree species and soil properties as fixed effects. A two-way analysis of variance technique was used to evaluate the changes in soil C and nutrient status driven by the plantations. Pair-wise comparisons between the vegetation types were based on Tukey's post hoc test, with P-value (p) adjusted using the single-step method and a significance level of $p < 0.05$ (Hothorn et al., 2008). The p values for the model were determined using the package "multcomp" (Hothorn et al., 2015). When the F-test (F) value is presented in the Results section, the three studied forest types did not differ significantly, while when Z-test (Z) value is presented this indicates that pair-wise comparisons were performed and significant differences were found between some forest types, but not all.

Correlation analyses were performed using Pearson product-moment correlation coefficient (r). All statistical analyses and tests were carried out using R software, version 3.1.2 (RCoreTeam, 2014).

3. Results

3.1. Soil and litter carbon

The concentration of SOC was slightly higher in the top 10 cm of soil in *P. taeda* and *E. grandis* compared with miombo forest ($Z = 2.29$; $p = 0.06$). The total stocks of SOC for the whole soil profile (0–50 cm) in *E. grandis* (139 Mg ha^{-1}; $Z = 2.46$; $p < 0.05$) and *P. taeda* (135 Mg ha^{-1}; $Z = 2.25$; $p = 0.05$) was significantly higher than in miombo forest (87 Mg ha^{-1}) (Fig. 1(a)). Assuming steady state stocks of soil C in miombo forest over 34 years, net accumulation of C was 1.41 Mg ha^{-1} yr^{-1} in *P. taeda* and 1.53 Mg ha^{-1} yr^{-1} in *E. grandis* stands. Litter layer C (Fig. 1(a)) in miombo forest was 60% and 79% less than in plantations of *E. grandis* ($Z = 2.93$; $p < 0.01$) and *P. taeda* ($Z = 4.76$; $p < 0.001$), respectively. Litter C content ranged from 2.6 Mg ha^{-1} in miombo forest to 12.1 Mg ha^{-1} in *P. taeda*, with *E. grandis* intermediate (6.6 Mg ha^{-1}). The *Pinus* and *Eucalyptus* plantation did not differ significantly in this regard ($Z = 1.84$; $p > 0.1$).

3.2. Total soil nitrogen and carbon-to-nitrogen ratio

The three forest types did not differ significantly in terms of total N stocks in the 0–50 cm layer ($F = 0.74$; $p > 0.1$). The stock of N ranged from 5.9 Mg ha^{-1} in miombo forest to 7.0 Mg ha^{-1} in *E. grandis*, with *P. taeda* intermediate (6.7 Mg ha^{-1}) (Fig. 1(b)). However, in the top 10 cm soil layer (Fig. 1(b)), the pine and eucalyptus stands had on average 116% (2.03 Mg ha^{-1}) higher total soil N stocks than the miombo forest (0.94 Mg ha^{-1} of N; $Z = 2.45$; $p < 0.05$). Assuming steady state stocks of total N in the topsoil of miombo forest over 34 years, the net accumulation of N by plantations was 3.2 g m^{-2} yr^{-1}. The concentration of total N per unit dry mass was also significantly higher in the top 10 cm soil layer of *P. taeda* stands ($Z = 3.04$; $p < 0.01$) and *E. grandis* ($Z = 2.66$; $p < 0.05$) compared with miombo forest (Table 1). The C/N ratio in the 10–30 cm layer in *P. taeda* soils was slightly higher than in miombo forest ($Z = 2.22$; $p = 0.06$), but there were no significant differences for individual soil layers (Table 1). The mean value of the C/N ratio (0–50 cm depth) was 15, 21 and 22 for miombo forest, *E. grandis* and *P. taeda* soils, respectively.

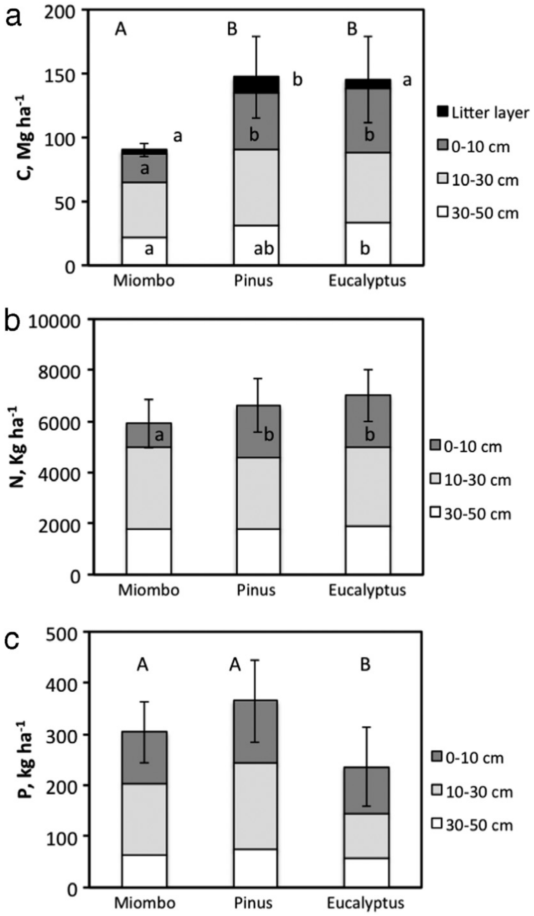

Fig. 1. Mean values of stocks of (a) soil carbon (C), (b) total soil nitrogen (N) and (c) extractable phosphorus (P) in the three forest types studied and across soil layers. The error bars indicate standard deviation of the mean for the whole soil profile. Different lower-case and upper-case letters indicate significant difference for individual soil layers or for the whole soil profile, respectively, between forest types ($p < 0.05$). N and P in the litter layer was not determined.

3.3. Extractable phosphorus

The *E. grandis* soils (0–50 cm depth) had 22% and 35% lower extractable P stocks than miombo forest ($Z = -2.65$; $p < 0.01$) and *P. taeda* ($Z = 4.31$; $p < 0.001$), respectively (Fig. 1(c)). This lower availability of extractable P in *E. grandis* plantations was also evident as significantly lower P concentration in the 10–30 cm layer (Table 1), compared with the corresponding layer in miombo forest ($Z = -2.87$; $p < 0.05$) and *P. taeda* ($Z = 4.35$; $p < 0.001$).

The estimated mean amount of P in above-ground biomass of *E. grandis* (67 kg ha^{-1}) was of a similar magnitude to the apparent decline in P content in the soil compared with miombo forest (38 kg ha^{-1}) (Fig. 2). In contrast, there were no differences in soil P stocks between miombo forest and *P. taeda* plantations, while the amounts of P accumulated in the biomass of pines was substantial (Fig. 2). In both the *E. grandis* and *P. taeda* plantations, extractable P and SOC were positively ($r = 0.97$) and strongly correlated to each other ($p < 0.001$) (Fig. 3(a)).

3.4. Soil acidity, base saturation and cation exchange capacity

The soils in *E. grandis* stands were significantly less acid than those in the other two stand types, which were characterised as only weakly or moderately acid (Fig. 4). In *E. grandis* stands, soil pH in the 10–30 cm layer ($Z = 2.29$; $p = 0.05$) and 30–50 cm ($Z = 2.51$; $p < 0.05$) was 0.6 and 0.8 pH units higher than in miombo forest. Furthermore, BS in the 10–30 cm fsoil layer of *E. grandis* was around 34% higher than in other two stand types ($Z = -2.39$; $p < 0.05$) (Table 2). A similar trend was observed for the whole soil profile (0–50 cm), which also had significantly higher BS ($Z = -2.95$; $p < 0.01$), of which calcium (Ca^{+2}) accounted for 50% or more (Table 2).

Cation exchange capacity in the top 10 cm soil layer of *P. taeda* plantation was 45% higher than in miombo forest ($Z = 2.94$; $p < 0.01$), but in the lower soil layers no significant difference was detected (Table 2). Mean CEC in the 0–50 cm

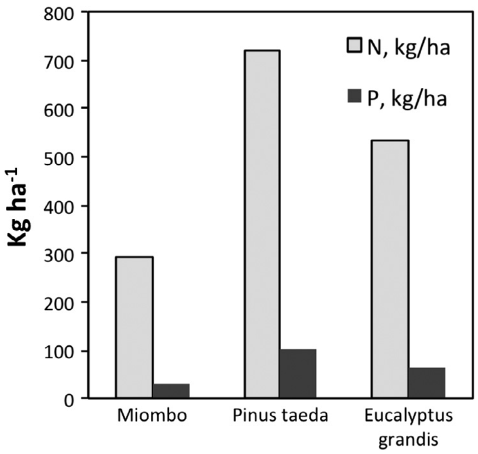

Fig. 2. Total phosphorus (P) and total nitrogen (N) stocks in above-ground biomass in the three forest stands studied.

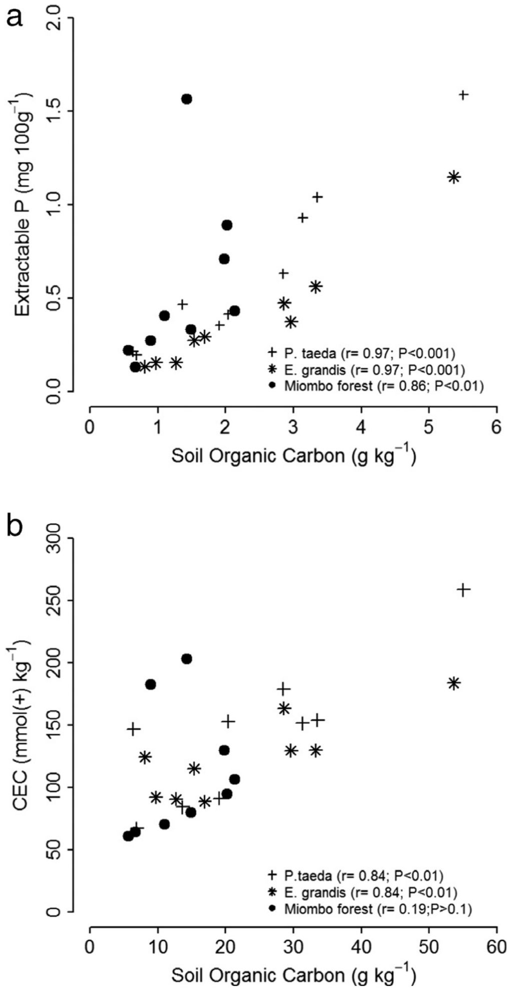

Fig. 3. (a) Correlation between soil organic carbon (SOC) and soil extractable phosphorus (P) and (b) SOC and cation exchange capacity (CEC) in the three forest types studied. Pearson's correlation coefficient (r) values for paired relationships are indicated in the plots.

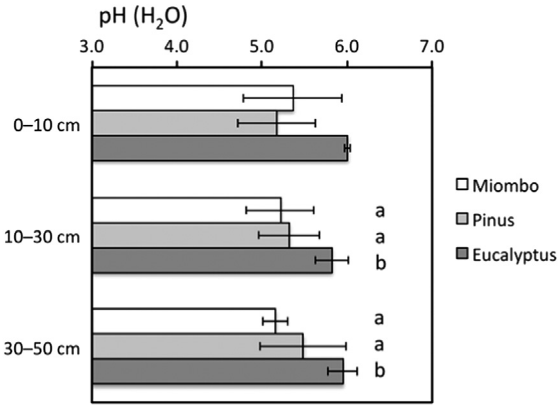

Fig. 4. Mean values (± Standard Deviation) of soil pH (H_2O) depending on stand type and soil layer. Different letters indicate significant difference between forest types ($p < 0.05$).

Table 2
Mean values (±Standard Deviation) of cation exchange capacity (CEC), base saturation (BS) and exchangeable calcium (Ca^{2+}, % of CEC) in the stand soil types studied. Different lower-case letters indicate significant difference between stand types ($p < 0.05$).

Parameter	Soil layer	Miombo forest	*P. taeda*	*E. grandis*
CEC (mmol (+) kg^{-1})	0–10 cm	135^a (60)	196^b (56)	159^a (27)
	10–30 cm	94 (32)	130 (40)	112 (20)
	30–50 cm	103 (69)	102 (41)	102 (20)
BS (%)	0–10 cm	44.2 (23.1)	37.1 (33.5)	57.4 (23.4)
	10–30 cm	33.7^a (22.7)	34.9^a (31.0)	45.9^b (26.9)
	30–50 cm	30.6 (21.9)	34.4 (28.6)	40.5 (30.2)
Ca^{2+} (%)	0–10 cm	22.8 (15.2)	22.8 (22.6)	38.0 (19.0)
	10–30 cm	18.2 (13.2)	21.2 (23.2)	24.9 (18.3)
	30–50 cm	15.2 (12.2)	17.1 (14.1)	19.2 (13.9)

layer of *P. taeda* stands was slightly higher ($Z = 2.13$; $p = 0.06$) than in miombo forest (Table 2). In contrast, *E. grandis* and miombo forest did not differ significantly, either in the top 10 cm ($Z = 1.53$; $p > 0.1$) or in the whole soil profile ($Z = 1.39$; $p > 0.1$). An increase in CEC due to the increase of SOC content was evident in both *E. grandis* and *P. taeda* plantations, as demonstrated by a significant ($p < 0.01$) positive correlation ($r = 0.84$) (Fig. 3(b)).

3.5. Characterisation of soil physical properties and the forest stands

Soil texture in the 0–30 cm layer did not differ significantly between the three stand types studied ($F = 0.57$; $p \geq 0.1$). Clay content in the 30–50 layer of *P. taeda* stands was 67% ($Z = 3.10$; $p < 0.01$) higher than in miombo forest (Table 3). Soil bulk density (Table 3) did not differ significantly between miombo forest and planted forest ($F = 0.35$; $p > 0.1$). The three stand types were similar in terms of number of stems per hectare ($F = 1.18$; $p > 0.1$), but basal area in miombo forest was on average 35.3 m^2 ha^{-1} smaller (Table 3) than in either *P. taeda* ($Z = 3.73$; $p < 0.001$) or *E. grandis* ($Z = 3.95$; $p < 0.001$) stands.

4. Discussion

4.1. Soil and litter carbon

An important finding was that planted stands of *P. taeda* and *E. grandis* had higher total SOC stocks than miombo forest, confirming hypothesis (i), but there was no marked difference between the pine and eucalyptus stands in terms of SOC stocks. This partially contradicts hypothesis (ii), that the effect of *E. grandis* is stronger than that of *P. taeda* (see also Li et al., 2012). This similarity between plantations can be explained by the fact that the *Pinus* and *Eucalyptus* stands studied here did not differ in litter layer C stocks (Fig. 1(a)).

The SOC stocks in the top 60 cm layer of African savannah and woodland generally vary between 30 and 140 Mg ha^{-1} (Ciais et al., 2011; Ryan et al., 2011; Shelukindo et al., 2014; Williams et al., 2008). The estimated SOC stocks in miombo forest in the present study (Fig. 1(a)) were within the range reported by those authors and were similar to findings by Mujuru et al.

Table 3
Soil physical properties and selected features of the forest types studied. MDH = mean dominant height of the five tallest trees per sampled plot. The values represent the mean for forest type or individual soil layer depending on forest type (±Standard Deviation). Different lower-case letters indicate significant difference between forest types ($p < 0.05$).

Parameter	Soil layer	Miombo forest	*P. taeda*	*E. grandis*
Density (stems ha^{-1})		971 (216)	853 (291)	629 (312)
Basal area (m^2 ha^{-1})		17.4^{a} (15.7)	51.7^{b} (17.7)	53.7^{b} (30.3)
MDH (m)		12	27	39
Clay (%)	0–10 cm	32.3 (9.3)	32.1 (8.9)	22.2 (18.9)
	10–30 cm	33.5 (10.1)	32.6 (3.6)	35.5 (10.7)
	30–50 cm	35.1^{a} (9.8)	58.7^{b} (20.1)	46.4^{a} (13.9)
Silt (%)	0–10 cm	32.4 (12.4)	36.2 (9.0)	47.3 (21.6)
	10–30 cm	34.7 (10.9)	40.3 (6.8)	38.7 (13.2)
	30–50 cm	32.6^{a} (17.7)	13.1^{b} (14.7)	28.7^{a} (28.6)
Sand (%)	0–10 cm	35.4 (14.3)	31.7 (16.9)	30.5 (16.9)
	10–30 cm	31.8 (9.4)	27.1 (6.2)	25.9 (18.8)
	30–50 cm	32.3 (20.2)	28.3 (5.7)	24.9 (24.1)
Bulk density (g cm^{-3})	0–10 cm	1.2 (0.3)	1.2 (0.1)	1.3 (0.1)
	10–30 cm	1.4 (0.1)	1.3 (0.1)	1.4 (0.1)
	30–50 cm	1.5 (0.0)	1.5 (0.1)	1.5 (0.1)

(2014) (85 Mg C ha^{-1}). The increase in SOC observed in *P. taeda* compared with miombo forest was of the same magnitude as reported by Mujuru et al. (2014) in Zimbabwe, although their values referred to younger plantations of *P. patula* (137 and 121 Mg C ha^{-1} in 10- and 25-year-old *P. patula* stands, respectively).

Forest plantations generally affect C pools in the organic horizon more strongly than in mineral soil (Jandl et al., 2007; Li et al., 2012). This effect was also observed in the present study, particularly in the *P. taeda* stands. The annual net increment observed in SOC stocks for *P. taeda* and *E. grandis* was within the range reported by Post and Kwon (2000) (0-3-.3 Mg ha^{-1} yr^{-1}), but higher than that reported by Li et al. (2012) (0.15–0.75 Mg ha^{-1} yr^{-1}).

The effect of *P. taeda* in this study was higher than could be expected from other studies (e.g. (Li et al., 2012)), but the effect of *E. grandis* was in line with effects reported for *E. grandis* by Li et al. (2012). Significant changes in soil organic C stocks are generally detected around 30–40 years following plantation (Li et al., 2012), but the findings by Mujuru et al. (2014) for plantations of *P. patula* in Zimbabwe indicate that changes can also occur over the shorter term (e.g. 10–25 years after plantation). The incorporation of C from both above-ground and root litter in the mineral soil C pools is a gradual process (Jandl et al., 2007). In the *E. grandis* stands in the present study, SOC increments were more evenly distributed across soil layers than in the pine stands (Fig. 1(a)), suggesting a stronger relative impact of root litter on SOC accumulation than in *P. taeda* stands.

The plantations are generally more protected against fires and illegal cutting than native miombo woodlands. Partly due to this, biomass production is typically larger in forest plantations. The increased total SOC stocks observed in both *P. taeda* and *E. grandis* sites were probably due to three factors. First, higher above-ground litter production (Fig. 1(a)). The SOC stock difference, including litter and soil layer, is largely determined by species differences in above-ground and below-ground litter production (Lemma et al., 2007; Li et al., 2012). Second, the simultaneous increase in SOC and soil N (as described below) could have been an effect of stronger protection against fires. Third, in the tropics, increase in SOC and soil N can result of termite activity (e.g. (Ackerman et al., 2007)), although this could not be determined from our data.

Anthropogenic fires occur frequently in miombo forest, often once a year, and are started by local people in connection with hunting practices, shifting cultivation or with the purpose to produce a new flush of grass for grazing animals. However, human fires rarely affect the soil C and fine root mass in the mineral soil (Rodeghiero et al., 2009), as also indicated by the finding that mycorrhizal associations to trees in miombo forest generally occur in the soil depth range 10–30 cm (Högberg and Piearce, 1986). In addition to fires, illegal cutting and logging can severely damage the stand structure in miombo forest (Jew et al., 2016), by decreasing above-ground C stocks, reducing above-ground litter production and consequently reducing accumulation of SOC, at least in the topsoil of miombo forest.

4.2. *Total soil nitrogen and carbon-to-nitrogen ratio*

In contrast to SOC, there were non-significant effects of stand type on total soil N stocks (0–50 cm layer), despite statistically significant accumulation of N in the top 10 cm soil of the plantations (Fig. 1(a) and (b)). This finding largely supports hypothesis (i). However, it suggests a less marked effect of a change in tree species on N than SOC. This agrees with the general observation that a longer time (∼50 years) is needed to detect increases in soil N than increases in SOC following plantation (Li et al., 2012). A possible explanation is that the higher variability in N determination makes it more difficult to detect differences.

The estimates of total N stocks (0–50 cm) for miombo forest and *P. taeda* plantations were similar to those reported by Mujuru et al. (2014), i.e. 5 Mg ha^{-1} of nitrogen in miombo forest and 7.5 Mg ha^{-1} of nitrogen in *P. patula* plantation. Our estimate of the annual net increment in N stocks for *Pinus and Eucalyptus* plantations represented a substantial accumulation rate of total soil N stocks in relation to the expected magnitude of N fluxes. The source of this N is unknown as the forest plantations studied here were not fertilised and extra N inputs from N fixation can be expected to be low, although some species common in miombo forest are N-fixers (Brown, 1997; Kuyah et al., 2014). Thus, the increase in N stocks in the top 10 cm of soil in the plantations (Fig. 1(b)) was probably due to a combination of other factors. Better protection from fires in plantation forests would prevent N losses from volatilisation of N in litter. Assuming no difference in the C/N ratio of fresh litter, increased litter production should result in increased stocks of N in the litter layer and also in soil horizons with a high fine root turnover rate.

In the present study, the higher N concentration in the 10–30 cm layer of the miombo forest than in the 0–10 cm layer indicates that N losses were a significant element in the N balance of that stand type. In addition, deep root systems in all stands, mining N sources in deep soil layers, would result in transport of N contributing to biomass production and litter accumulation. For example, Laclau et al. (2013) demonstrated that young *E. grandis* plantations in Brazil had rooting depths down to 10 m in a sandy soil. Dominating species in miombo have both lateral roots and deep trunk roots that can reach well below 2 m depth (Chidumayo, 1994; Ryan et al., 2011).

Our estimates of C/N ratio (0–10 and 30–50 cm depth; Table 1) were high compared with e.g. those reported for Ferralsols and other tropical soils (e.g. Mujuru et al., 2014), but were within the range reported for other similar sites (e.g. King and Campbell, 1994). In a study in Zimbabwe, Mujuru et al. (2014) reported a mean C/N ratio of 13 and 17 in the 0–60 cm soil layer of miombo forest and 30-year-old *P. patula*, respectively. However, King and Campbell (1994) reported higher values for stands in Zimbabwe: a C/N ratio of 21 in miombo forest, 22 in *P. patula* and 26 in *E. grandis* stands.

4.3. Extractable phosphorus

Hypothesis (iii) stated that the higher biomass production in plantations would decrease plant extractable (P) stocks in the soil. The results indicated that this was only supported in the *E. grandis* stands, where extractable P stocks and concentrations were significantly lower than in the other stand types. The allocation pattern of P stocks in the above-ground biomass of the three forest stands studied differed (Fig. 2). The difference between stands of *Pinus* and *Eucalyptus* may derive from species differences in e.g. nutrient uptake capacity and mycorrhizal associations to roots. The results suggest that *P. taeda* has a greater capacity than *E. grandis* to utilise other P sources in the soil than measured by the extraction method used here. Another explanation is that extractable P depends on soil pH (Devau et al., 2009), and organic P is an important source of plant P uptake in weathered soils. The significant positive correlation between SOC and P found in the planted stands (Fig. 3(a)), but not in miombo forest, was not surprising, since an increase in soil organic C should be associated with increased availability of P (Rashidi and Seilsepour, 2008).

4.4. Soil acidity, base saturation and cation exchange capacity

The direction of change in soil acidity found in this study was unexpected and did not support hypothesis (iv). Soil pH (Fig. 4) and BS (Table 2) in the top 10 cm of *P. taeda* plantations were lower than in miombo forest, although the difference was not significant. The effect of *E. grandis* plantation on soil acid–base status was the opposite to that of *P. taeda*. A clear and consistent trend of lower soil acidity was observed in *E. grandis* soils, except in the top 10 cm soil layer (Fig. 4). The findings with respect to *P. taeda* plantations were in line with observations by Mujuru et al. (2014) in Eastern Zimbabwe, where the soil pH in 30-year-old *P. patula* plantations did not differ from that in miombo forest. The effect of *E. grandis* plantations on soil pH in the present study can be considered high, since changes of 0.2–0.3 pH units due to tree species effects have been reported in other studies (Binkley and Giardina, 1998).

However, FAO (1985) reported a high soil pH change in a 14-year-old *Eucalyptus* plantation in India compared with an adjacent natural forest. It is more likely that aggrading forest results in soil acidification rather than soil alkalinisation, as a result of excess uptake of nutrient cations compared to nutrient anions, which is acidifying (Nilsson et al., 1982). The role of N uptake (ammonium versus nitrate) for soil acidity would have no effect if the N source is mineralisation of organic matter. Another possible mechanism for soil alkalinisation is nutrient uptake from deeper soil layers, and this may have occurred in the present case.

There was also a tree species effect on CEC, as *P. taeda* stands had higher CEC in the top 10 cm soil layer than miombo forest and *E. grandis* stands (Table 2). This difference in CEC was closely associated with the change in SOC, as indicated by the strong correlation between CEC and SOC in plantation stands, but not in miombo forest (Fig. 3(b)). Accumulation of SOC and CEC in *P. taeda* stands should have generated higher soil acidity in terms of pH and BS, unless the increase in CEC was balance by increased stocks of exchangeable base cations.

4.5. Characterisation of soil physical properties and the forest stands

It is possible that the higher clay and lower silt content in *P. taeda* stands at 50 cm depth (Table 3) was established prior to plantation, because clay formation is a very slow process. However, comparable studies (King and Campbell, 1994; Michelsen et al., 1996) also report higher clay content in lower soil layers of a *Pinus* plantation in Zimbabwe. Conditions such as pH, decreased CEC and leaching of base cations in soil (e.g. Al^{3+}; Ca^{2+}) generally precede translocation of clay minerals.

The estimates of bulk density (BD), which were computed from pedotransfer functions that included SOC concentrations in the soil, indicated no significant differences between the forest types studied (Table 3). The BD estimates were also similar to findings for comparable sites in the region, e.g. King and Campbell (1994) reported BD values of 1.29–1.60 g cm^{-3} and Mujuru et al. (2014) values of 1.43–1.93 g cm^{-3}.

5. Conclusions

Old (34-year) first-rotation plantations of *P. taeda* and *E. grandis* in Mozambique had significantly larger SOC stocks in the soil profile and significantly larger N stocks in the top 10 cm soil layer than adjacent miombo stands. The net stock change in soil C in the pine and eucalyptus stands studied was 1.41 Mg ha^{-1} yr^{-1} and 1.53 Mg ha^{-1} yr^{-1}, respectively, assuming no SOC change in miombo stands. Thus, if management aims to increase soil carbon stocks, degraded miombo woodlands can be converted into fire protected plantations of *Eucalyptus* and *Pinus.* The results also showed that plantations of *P. taeda* and *E. grandis* affected soil acidity and soil nutrient status differently, except for total soil N stocks. Stands of *P. taeda* had no effect on soil pH and extractable P, but increased CEC, whereas plantations of *E. grandis* increased BS and soil pH, but decreased stocks of extractable P.

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