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Forest Biomass and Carbon

Edited by Gopal Shukla and Sumit Chakravarty

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and **Sumit Chakravarty**

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Meet the editors

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Preface

Forests grow and their biomass increases, absorbing carbon from the atmosphere and storing it in plant tissue. Moreover, soil is one of the main sinks of carbon on earth, especially forest soil because it normally has higher soil organic matter. Active absorption of CO_2 from the atmosphere through photosynthesis, and its subsequent storage in the biomass of growing trees or plants, is known as carbon storage. Understanding the biomass of forest vegetation is essential for determining the storage of carbon in the dominant tree component and computing carbon cycling at a regional as well as global level. This book consisting of five chapters will give a comprehensive understanding of biomass production vis-à-vis carbon storage in relation to litter and nutrient dynamics of the forest by analyzing the mode and magnitude of biomass production and carbon storage as a function of various silvicultural factors.

Biomass has been gaining increased interest due to its importance in sustainable forest management and in carbon sequestration. The first chapter highlights the differences between pure even-aged, pure multi-age, mixed even-aged, and mixed multi-age stands. The authors measured and indexed the above-ground biomass for evaluation of biomass and carbon stocks. Details of structural analysis for the modeling of forest stands as well as for their management are noted. The results of this work indicate significant differences between the measures and indices calculated with the number of trees and basal areas, when compared with those calculated with above-ground biomass. The authors are of the view that the latter can be of importance when there is the need to discriminate between stands with a similar number of trees but with different proportions.

Forests in close proximity to urbanized locations in the tropics suffer high human pressure because of unregulated harvests that supply domestic energy and equipment. The second chapter assesses the stock, structure, and use of common woody species in a reserve forest close to an urban location and quantifies its carbon accumulation potential amid pressures from surrounding communities. The structure of the forest was altered due to unregulated tree harvesting, which would likely jeopardize the forest's ecosystem functioning and carbon accumulation. It was recommended that reserve sustainability should be managed, local communities should be made aware of forest biodiversity conservation, and the reserve for ecotourism potentials should be explored.

The third chapter is based on a study carried out on a cocoa-based agroforestry system, which documented its plant species diversity, ecological services, and carbon sequestration potential. The study found high flora diversity and carbon storage capacity of the system along with a potential for ecological services of remunerating carbon credits in terms of monetary gain to the growers. The chapter concludes that cocoa-based agroforestry systems are playing an important role in biodiversity conservation.

Forestry is a viable and feasible strategy for carbon cycle management since it promotes carbon accumulation in soils and vegetation. The fourth chapter is an attempt based on this principle. The chapter concludes that particular care must be taken when choosing forest management practices in tropical-weathered soils because they can oxidize a significant part of the SOC pool with negative consequences to soil fertility and aggregate stability. Forest management practices may have strong implications for soil organic carbon (SOC) pools that may offset the carbon biomass accumulation potential of plantations of fast-growing species, thus limiting their role in C sequestration and climate change mitigation. These implications are particularly critical in the case of the substitution of native forests by artificial plantations considering the possible negative consequences for biodiversity conservation.

Forest biomass, the most ancient of fuels, is again at the center of renewable energy production. The fifth chapter gives an insight into the drivers that influence the ecological sustainability of this energy source. The chapter describes the basic concepts of ecological sustainability, ecological rotation, and ecological thresholds with state-of-the-art approaches to assess sustainability of forest biomass production along with the available tools for decision-makers to evaluate the sustainability of forest biomass production and management. Furthermore, there is an elaboration on the advantages and drawbacks of forest certification, growth and yield tables, and ecological models in relationship to their use in sustainable forest management for biomass and energy production. The authors strongly advocate that the sustainability of forest biomass management depends on the ecological variables being assessed and the time frame for which the assessment is carried out.

This book is an attempt to elaborate information on a number of new emerging issues of forest biomass and carbon. We believe that this volume will be an interesting and helpful resource to all those who work for a sustainable greener future.

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Effects of Forest Stand Structure in Biomass and Carbon

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Additional information is available at the end of the chapter

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Abstract

Biomass has been gaining an increased interest due to its importance in sustainable forest management and in carbon sequestration. Biomass in each forest stand varies according to its structure and influences not only the biomass per area unit but also its distribution in space and time. The structure analysis with absolute stand density measures and structure and diversity measures and indices for the number of trees and basal area does not always reflect the above-ground biomass distribution and variability. The use of above-ground biomass as an absolute density measure and the development of diversity measures and indices derived from it enable further details in the stand structure characterisation. The results of this study highlighted the differences between pure even-aged, pure multiaged, mixed even-aged and mixed multiaged structures. The measures and indices of above-ground biomass are considered primordial as they integrate the horizontal and the vertical distribution, thus enabling a more detailed evaluation of biomass and carbon stocks.

Keywords: stand structure, biomass, density measures, structure and diversity indices

1. Introduction

Forest stands provide a wide range of products and services, from timber and other woody and nonwoody products to services [1, 2]. Traditionally, forest inventories evaluated forest area, crown cover, tree species, number of trees, diameter at breast height and total height [3–6]. National Forest Inventories started to evaluate biomass from the late twentieth century onwards in order to assess wood for timber and bioenergy, carbon stocks and carbon sequestration and losses [7–12]. Biomass is frequently estimated with indirect methods using tree-level species-specific and site-specific allometric functions [13–28].

The analysis of structure of any forest stand is described in almost all silviculture text books [29–36]. It is a useful tool for stand management, whether they are managed for products or services, as well as for describing the stand or the ecosystem conditions in a long-term monitoring, and for the silvicultural and management practices [33, 36–40]. In any forest system, stand structure is a primordial notion that refers to suite of patterns and interactions between the individuals in a stand. The stand structure can be a result of a planned design or a self-organisation process. In a stand the ongoing processes determine the spatial pattern of the system and thus their structure, but the system properties are also determined by its structure. For example, the spatial arrangement and the tree dimensions in a stand determine their structure in a point in time, but a disturbance either natural (e.g. windstorm or fire) or artificial (e.g. silvicultural practices) determines the future processes and thus the future structure. The variability of stand structure is wide and influences growth, mortality, silvicultural practices, harvests and regeneration, which in turn determine their structure [33–36]. Stand structure is also linked to some heterogeneity, which is associated to diversity, related not only to the number and proportion of species but also to the variability of the tree dimensions and their spatial arrangements [41]. The different stand structures bring to light the variability of the interactions between the trees in a stand. The interactions that occur between a reference tree and its neighbours will define its available growing space and therefore the competition which is also reflected in its growth and consequently biomass and carbon sequestration [33, 41]. Above-ground biomass is frequently used as a proxy to evaluate the carbon stocks [42]. Therefore, to compare stands or to highlight their dynamics, stand structure analysis makes the bridge between the individual tree interactions and the stand.

Stand structure can be described as the spatial and temporal distribution of the trees and other species [33] and encompasses both the horizontal and vertical distributions [9, 33, 35, 43]. It is classified in most silviculture text books [29–36] in two classes: even-aged and multiaged or uneven-aged. Between two extremes, stands with all the trees with the same age and stands with trees of all ages, a wide range of combinations can be found, hence originating many different stand structures. The variability increases from pure even-aged to mixed multiaged stands, thus enhancing the importance of characterising and analysing structure to evaluate the stands in a point in time, to study their dynamics, to model them or to implement management practices [29–36, 43, 44]. Several methods have been developed and used to describe stand structure, which include tree size distributions, density measures, structure indices and diversity indices. They serve as guides for forest management [29–36, 43].

Structure analysis is frequently done with the number of trees, diameter at breast height, basal area, tree height and number of species. The use of above-ground biomass is not frequent. This study will use absolute density and structure and diversity measures and indices, defined as the function of the former dendrometric variables and those defined as the function of above-ground biomass to evaluate different stand structures. The specific objectives include the analysis of the variability of stand structure between pure and mixed and even-aged and multiaged stands.

2. Materials and methods

A suite of plots that cover a wide range of forest areas and species in Portugal (from north to south) were selected. The plot locations are Mora (central coordinate 8°4′53.98″W and 38°51′16.12″N) of pure *Quercus rotundifolia*, pure *Quercus suber* and mixed *Quercus rotundifolia* and *Quercus suber*; Alcácer do Sal (central coordinate 8°40′28.20″W and 38°27′45.71″N) of pure *Pinus pinea*, pure *Quercus suber* and mixed *Quercus suber* and *Pinus pinea*; Pinheiro da Cruz (central coordinate 38°16′56″ N and 8°45′19″ W) of pure *Pinus pinaster*; Lousã (central coordinate 40°04′57″ N and 8°14′57″ W) of pure *Pinus pinaster* and mixed *Pinus pinaster*, *Castanea sativa* and *Quercus robur*; Arcos de Valdevez (central coordinate 41°49′52″ N and 8°29′38″ W) of mixed *Quercus robur*, *Quercus rubra* and *Betula celtiberica*; Montargil (central coordinate 39°07′08″ N and 8°08′49″ W) of pure *Quercus suber*; Extremoz (central coordinate 38°54′25″ N and 7°37′48″ W) of pure *Quercus suber*; Chamusca (central coordinate 39°21′19″ N and 8°26′05″ W) of pure *Quercus suber*; Coruche (central coordinate 39°06′27″ N and 8°21′48″ W) of mixed *Quercus suber* and *Pinus pinea* and mixed *Quercus suber*, *Pinus pinea* and *Pinus pinaster*. In the plots of Lousã, several other species were present in a very small number of individuals. The analysis of these plots will be focused in three main species, and all the other species in the plot were grouped in one class, as suggested by [45] since class bias could arise in the results, especially in the diversity indices. The plots used are pure even-aged (53), pure multiaged (129), mixed even-aged (20) and mixed multiaged (53) in a total of 255 plots. These set of plots was selected to enable the characterisation of different aspects of stand structure, in particular above-ground biomass. The diameter at breast height, total height and four crown radii (north, south, east and west) as well as recorded the species, for all with diameter at breast height ≥5 cm. The classification of the plots as pure or mixed was done using the four criteria classification [46] and as even-aged or multiaged using the diameter distributions with 2.5 cm classes [29, 31, 32, 34–36]. Above-ground biomass was calculated per species and per tree with the allometric functions at the tree level (**Table 1**).

It is not possible to describe stand structure with only one criterion; inversely, the combination of several criteria is needed. The stand structure has to be described with the characterisation of the spatial distribution, both horizontal and vertical, of the trees with their dendrometric parameters. From the latter, the use of the number of trees, the diameter at breast height and the total height are frequent. Less commonly used are the height of the beginning of the crown, the crown radii and the tree locations is frequent [29–36, 41, 43]. Seldom biomass is used [9].

The measures to analyse stand structure can be divided in three groups: (i) density measures, (ii) structure indices and (iii) diversity measures and indices. In even-aged stand structure, characterisation can be done using the first two groups, while for mixed stands, diversity measures and indices should be included as the former are not able to fully characterise it.

Density measures are stand-level parameters which are a proxy for competition between individual trees and growing space allocated to each tree. The *absolute stand density measures* are a unique measure for a stand usually obtained as the count, sum or average of a dendrometric parameter, frequently calculated for a standard area, typically the hectare.

Quercus rotundifolia and *Quercus suber* [20]

$$ww = 0.164185 \times d^{2.011002}$$

$$\mathrm{wb} = 0.600169 \times \mathrm{d}^{1.355957}$$

$$\mathrm{wc} = 1.909152 \times \mathrm{d}^{1.200354}$$

$$W = ww + wb + wc$$

Pinus pinea [22]

$$ww = 18.85 \times c^{1.68} \times h^{0.95}$$

$$wbr = 184.94 \times c^{3.03}$$

$$wl = 22.27 \times c^{1.76} \times \left(\frac{h}{d}\right)^{-0.5}$$

$$wb = 8.08 \times c^{1.55} \times h^{0.47}$$

$$W = ww + wbr + wl + wb$$

Quercus robur, *Quercus rubra* and *Betula celtiberica* [48]

$$ww = e^{(-3.323+(0.950\times\ln(d^2\times h)))}$$

$$wc = e^{(-14.246+(2.248\times\ln(d^2\times h))-0.01972\times(lc\times h))}$$

$$W = ww + wc$$

Castanea sativa [47]

$$ww = 0.02044 \times d^{1.76603} \times h^{1.16402}$$

$$wbr = 0.00440 \times d^2 \times h$$

$$wb = 0.06574 \times d^{1.84096}$$

$$W = ww + wb + wbr$$

Pinus pinaster [47]

$$ww = 0.0146 \times d^{1.94687} \times h^{1.106577}$$

$$wbr = 0.0308 \times d^{2.75761} \times \left(\frac{h}{d}\right)^{-0.39381}$$

$$wl = 0.09980 \times d^{1.39252} \times \left(\frac{h}{d}\right)^{0.71962}$$

$$W = ww + wbr + wl$$

where d is the diameter at breast height (in cm), h is the total height (in m), c is the circumference at breast height $c = (\pi \times d)/100$ (in m), lc is the crown length (in m), W is the total above-ground biomass (in kg), ww is the wood biomass (in kg), wbr is the branch biomass (in kg), wl is the leaf biomass (in kg) and wc s the crown biomass (in kg).

Table 1. Above-ground biomass allometric functions.

The most frequently used density measures are the number of trees per hectare, the basal area per hectare, the volume per hectare, crown cover and mean quadratic diameter [29–36, 43]. Of interest is also above-ground biomass, per hectare, and their mean, which are not usually used. In pure stands each absolute density measure results frequently in one value per stand. Conversely, in mixed and multiaged stands, evaluation should also be done per species and per height layer, respectively [46].

When tending a stand, there is the need to select the trees that will be maintained and removed in silvicultural practices [29, 31, 32, 34, 35] and thus to have a suite of tools for their selection. The *structure indices*, which are derived from dendrometric variables, are able to evaluate potential photosynthetic ability, potential tree growth, vigour and stability. From the indices described in literature, the most frequently used index to evaluate tree or stand stability and vigour is hd ratio [6, 36, 43, 49–52]; for the ability to withstand disturbances such as the windstorms is the linear crown ratio [4, 6, 31] and for the potential photosynthetic ability and growth rate are the crown length and the crown ratio [6, 33, 34, 36]. The tree stability and growth are usually evaluated as the function of threshold values. It should be noticed that variability is expected between tree species and between even-aged and multiaged stands. For even-aged stands, ageing and crown closure might increase instability and reduce growth and vigour. The multiaged systems tend to have higher stability and vigour, due to their wider variability of diameter, total height and crown diameters [35, 38, 51].

The hd ratio (*hd*) is the relation between the tree total height and its diameter at breast height, with both variables in the same metric units [6, 29, 36, 43]. It is usually calculated per tree and at the stand level with their mean value. It allows an evaluation of the tree stability but gives also some insights to the competition pressure that the tree was subjected in the past. The higher the competition the higher *hd* and the lower is stability [36, 43]. This can be explained by the distribution of photoassimilates, which are allocated first to height growth and only then to diameter [33], and consequently trees under stronger competition grow more in height and less in diameter and thus will have higher slenderness [43]. There is a straight link between *hd* and windthrow, the higher the first the higher the probability of wind damages [33, 36, 43]. For silviculture, the definition of thresholds for tree stability is of importance. Several authors [36, 49, 51, 53, 54] report that $hd \leq 85$ indicates stable trees and stands, $hd > 85$ unstable and $hd > 100$ very unstable. Trees with $hd < 45$ correspond to trees in free growth [49]. Nonetheless, the tree stability should not be analysed separately, as it also depends on the dimension of the tree crowns and stand density. Trees with large crowns have potentially higher stability than those with smaller crowns. Dense stands of trees with small crowns are potentially more affected by disturbances due to the higher *hd* and smaller crowns [36]. In even-aged stands, this can originate windthrow of large forest areas [38–40]. Multiaged stands tend to have smaller *hd* [38, 55] due to the variability of tree height, diameter and crowns, originating potentially greater stability [43].

The linear crown diameter (*lcr*) is defined by the ratio between the crown diameter and the diameter at breast height, with both variables in the same metric units [4, 6, 31]. In general, *lcr* has the tendency to increase in time in young stands, especially before crown closure. Also, *lcr* depends mainly on the aerial growing space, increasing from dominated to dominant trees, and tends to diminish with crown closure, as trees continue to increase in stem diameter, but not in crown diameter due to the lateral confinement of the crowns by their neighbours [31, 33]. [31] for even-aged stands reports a *lcr* of about 22 for broadleaved species and 12–18 for conifers.

The crown length (*lc*), defined as the difference between the total height and the height of the beginning of the live crown (in m), is a proxy for the evaluation of the trees' past competition pressures. Strong competition increases the death and fall of the inferior branches, phenomena known as crown shyness [6, 34, 36]. The threshold interval of *lc* for trees of good growth and vigour is between ⅓ and ½ of the total height of the tree [34, 36].

The crown ratio (*cr*) is the percent of the crown length in relation to the total height. This index is used as a surrogate for the photosynthetic rate and is strictly related with the stand management, for example, with stem height free of branches and *hd* [6, 33, 34]. The threshold for vigorous growth is $cr > 30\%$ and for stability should be $cr > 50\%$ [34]. Well-balanced crowns have *cr* between 30 and 50% [33].

The stand structure and its complexity can be evaluated by *diversity measures* and *indices*, enabling to predict the growth and growth patterns' dynamics [33]. Frequently, one measure or index is not able to quantify the different stand characteristics; thus, it is common to use more than one. The most frequently used measures are related to the number of species, the species proportions and their distribution, and are based on the number of trees, basal area and tree height [41, 52, 56–63]. They are selected so that the horizontal and vertical distributions are characterised. These indices are frequently applied to mixed and/or multiaged stands. The commonly used indices are species richness, relative density, relative basal area, Simpson index, Shannon and Weaver index for the horizontal distribution and A index for the vertical distribution [41, 52, 56–63]. Some can also be used in even-aged stands to evaluate the tree horizontal and vertical distribution.

Species richness (*SR*) refers to the number of species in a stand. The higher the number of species the richer is the stand [56, 59]. It gives insights regarding diversity, but it lacks information in what concerns the frequency of each species and/or their dimensions. Thus, stands with the same number of species are included in the same class, though they can have different proportions of species [45, 56, 59, 64, 65].

Other indices enable the heterogeneity quantification [56] through the importance of each species in the mixture; the relative density (R_N) quantifies the number of individuals of a species in relation to the total number of individuals in a stand, relative basal area (R_G) the proportion of the basal area of a species in relation to the total basal area in a stand [45, 56, 59, 64, 65] and the relative biomass (R_{AGB}) the proportion of above-ground biomass of each species in relation to the total above-ground biomass in a stand [9]. These measures can also be applied to tree dimensions, such as height layers or diameter at breast height or above-ground biomass classes.

Another two indices that characterise the heterogeneity are the Simpson index and Shannon and Weaver index. The Simpson index (*D*) measures the probability of two individuals belonging to the same species, assuming they were chosen randomly [56, 59]. This index varies between 0 and 1. It is 1 for stands with only one species and decreases with the increase of both the number of species and the similarity between their frequencies. Shannon and Weaver index (*H*) measures the probability of one individual chosen randomly to belong to a certain species. The inclusion of the Napierian logarithm in its formula results in a disproportional variation of their values, enabling larger increases for the rare species than for the abundant ones [61, 66]. This index increases with the increase of both the number of species and the equality of their frequencies [45, 61, 65–67].

The formula of Shannon and Weaver index enables its division in additive components [60, 61, 66, 68, 69]. While some authors considered diameter at breast height classes [61, 68, 69], others used basal area [60] and height [66] classes. The A index, based in Shannon and Weaver index, enables the characterisation of the vertical profile of the stand in a number of individuals. It considers the vertical profile divided in three height zones, defined as proportions of the maximum height of the stand, namely, inferior zone 0–50% of the maximum height, intermediate zone between 50 and 80% and superior zone >80%. It is 0 for pure one-layer stands and increases with the increase of the number of species and their equality per layer, reaching the higher values in mixed multiaged stands [66, 67].

Structure indices		**Diversity measures and indices**			
Name	**Formula**	**Name**	**Formula**	**Name**	**Formula**
hd ratio	$hd = \frac{h}{d}$	Species richness	$SR = \sum_{i=1}^{k} Sp_i$	Simpson index	$D = \sum_{i=1}^{N'} \left(\frac{N_i^i(N_i' - 1)}{N'(N' - 1)}\right)$
Linear crown ratio	$lcr = \frac{d_c}{d} \times 100$	Relative density	$R_N = \frac{N_i'}{N} \times 100$	Shannon and Weaver index	$H = -\sum_{i=1}^{k} p_i \times \ln p_i$
Crown length	$lc = h - h_c$	Relative basal area	$R_G = \frac{G_i'}{G} \times 100$	A index	$A = -\sum_{i=1}^{k} \sum_{J=1}^{z} p_{ij} \times \ln p_{ij}$
Crown ratio	$cr = \frac{lc}{h} \times 100$	Relative above ground biomass	$R_{ABG} = \frac{ABG_i'}{ABG} \times 100$		

where *d* is the diameter at breast height, *h* the total height, *lc* the crown length, *dc* the crown diameter, h_c the height of the beginning of the life crown, Sp_i the species i, N_i the number of individuals of species i, *N* the total number of individuals, G_i the basal area of specie i, *G* the total basal area, AGB_i the above-ground biomass of species *i*, A*GB* the total above-ground biomass, p_i the probability of an individual belonging to the *i*th species, p_{ij} the probability of an individual belonging to the *i*th species and the *j*th height zone.

Table 2. Structure and diversity measures and indices.

For each plot the following absolute density measures were calculated: number of trees per hectare (*N*), basal area per hectare (*G*), basal area of the average tree (*gm*), mean quadratic diameter (dg), above-ground biomass per hectare (*AGB*) and its arithmetic mean (*AGBm*). Four structure and seven diversity measures and indices were used (**Table 2**). In order to better characterise above-ground biomass in the horizontal and vertical planes, Simpson, Shannon and Weaver and A indices were adapted to *G* and *AGB*, and the former two were also adapted to classes of 500 kg. The plots were grouped in four structure classes: pure even-aged (PE), pure multiaged (PM), mixed even-aged (ME) and mixed multiaged (MM). To understand better the variability between the different stand compositions, plots were grouped in the following classes: pure *Quercus rotundifolia* (QR); pure *Quercus suber* (QS); pure *Pinus pinea* (PP); pure *Pinus pinaster* (PPi); mixed *Quercus rotundifolia* and *Quercus suber* (QRS); mixed *Quercus suber* and *Pinus pinea* (SP); mixed *Quercus suber*, *Pinus pinea* and *Pinus pinaster* (SPP); mixed *Pinus pinaster*, *Castanea sativa* and *Quercus robur* (PCR); and mixed *Quercus robur*, *Quercus rubra* and *Betula celtiberica* (RRB). The comparison between the different measures and indices and between pure, mixed, even-aged and multiaged plots was carried out with non-parametric Wilcoxon test for paired and independent samples, respectively [70]. The statistical analysis was implemented in R [71].

3. Results and discussion

The absolute density measures show a wide variation, larger for *N* and *AGB* than for *G* for all the plots. Variability is larger in the MM, as reported by several authors [29–37]. In general it increases from the pure to the mixed and from the even-aged to the multiaged plots (**Figure 1**). Similarly, *gm*, *dg* and *AGBm* are larger for multiaged than for even-aged plots though with less variability (**Figure 1**). Nonetheless, there are significant differences between the absolute density measures for the structure classes, except for *N* between ME and MM ($W = 514$, $p = 0.85$)

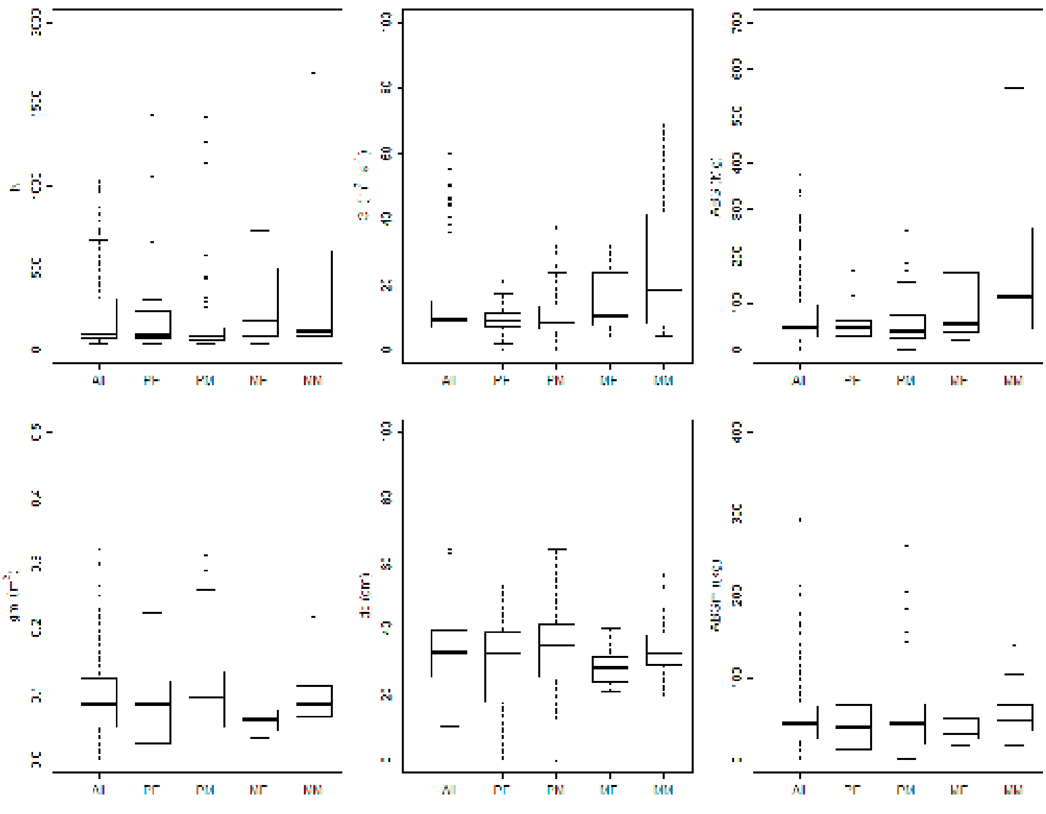

Figure 1. Boxplots of *N*, *G*, *AGB*, *gm*, *dg* and *AGBm* for all plots and per structure classes.

and for *G* and *AGB* between PE and PM (W = 3776, $p = 0.27$; W = 3828, $p = 0.21$, respectively). In spite of the similar tends, there are significant differences between *N* and *G*, *G* and *AGB* and *N* and *AGB*, for all structure classes (*all*, $p < 0.001$). In fact the absolute density measures encompass different aspects of stand structure that are complementary. While *N* reports only to the number of individuals, *G* relates to their stem diameter and *AGB* to the relation between diameter at breast height and total height, incorporating both the horizontal and the vertical dimensions in the absolute density measure.

The lowest variability is found in the plots managed as agroforestry systems with lower *N*, *G* and *AGB* but larger *gm*, *dg* and *AGBm*. This is characteristic of management systems focused on stem and crown diameter growth [44] and is especially visible for PP (**Figure 2**). Inversely, the plots where management is directed towards timber production have higher *N*, *G* and *AGB*, but the individual trees have lower diameter at breast height and smaller *AGBm*, though considerably larger *AGB* (**Figure 2**). Noteworthy are QS plots that have a rather high variability. This is due to the development stage of the stands, while some are young with higher *N*, *G* and *AGB* and smaller *gm*, *dg* and *AGBm*; in the adult plots, the opposite is observed (**Figure 3**).

The values of *hd* are indicative of good stability (**Figure 4**) with most values < 80 [36, 49, 51, 53, 54]. Many plots have *hd* ≤ 45 indicative of trees in free growth, which is the case of most

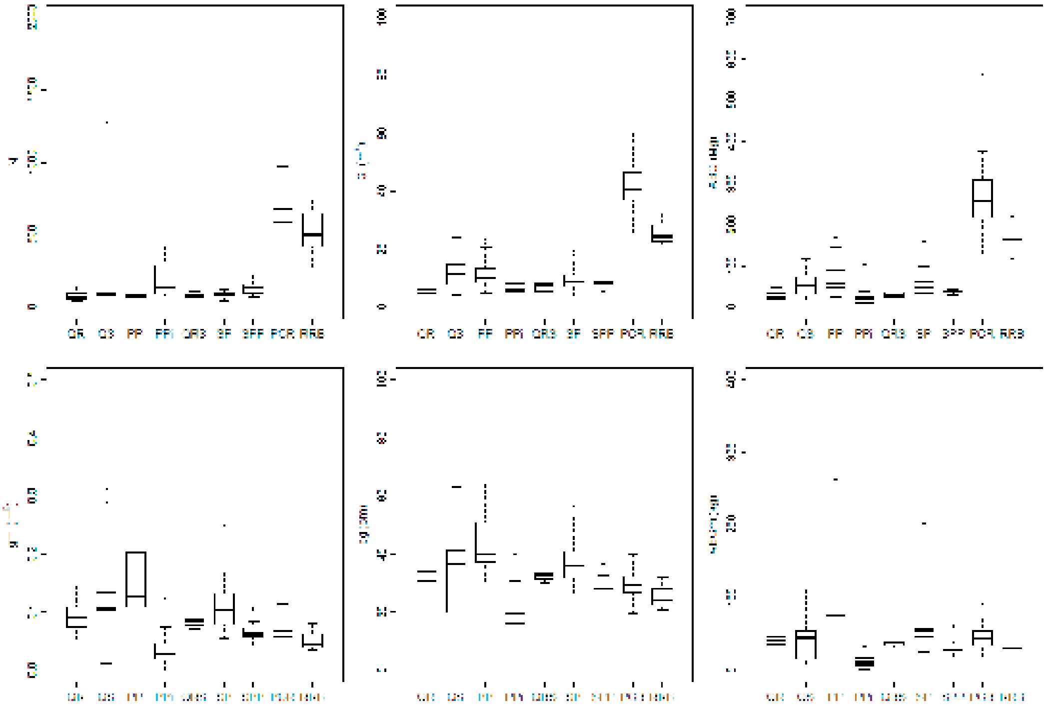

Figure 2. Boxplots of *N*, *G*, *AGB*, *gm*, *dg* and *AGBm* per composition classes.

QR, QS and PP plots that are managed in agroforestry systems. The stands managed for timber PPi, PCR and RRB have higher *hd* (**Figure 5**). There is a decrease of *hd* from even-aged to multiaged plots [38, 43, 49, 55] and an increase from pure to mixed plots, though the latter show a wider variability (**Figure 4**), denoted by the significant differences between PE and ME and PM and MM (*all*, $p < 0.001$). The analysis per composition classes outlines the differences between the plots managed in agroforestry systems (QR, QS, PP, QRS, SP and QSP) and those managed for timber (PPi, PCR and RRB), with the former with *hd* < 45 indicative of many trees in free growth [49] and the latter for most trees *hd* < 85, thus indicating good tree and stand stability [36, 49, 51, 53, 54]. Another source of variability is the tree and/or stand development stage; young individuals and stands have higher *hd* as a result of the high growth rates in height and low in stem diameter [54].

For all plots and per structure classes, *lcr* is larger than the inferior threshold [31], though with larger variability for MM (**Figure 4**), denoted by the significant differences between PE and ME and PM and MM (*all*, $p < 0.001$). Conversely, no significant differences were found between PE and PM ($W = 3402$, $p = 0.96$) and ME and MM ($W = 659$, $p = 0.11$). In the multiaged structures, as trees develop their crowns in different height layers, crown horizontal confinement is not as strong as in the even-aged ones; thus, the crown lateral growth still continues in the former opposite to what happens when crown closure occurs in the latter [33, 35]. The analysis per composition classes (**Figure 5**) shows larger variability for the plots managed for

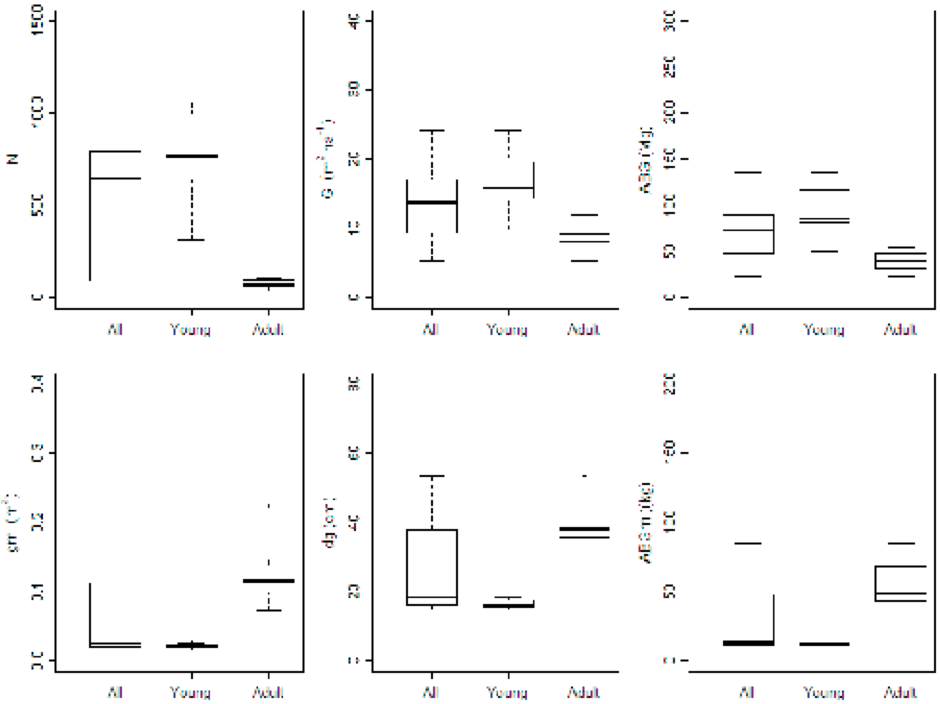

Figure 3. Boxplots of *N*, *G*, *AGB*, *gm*, *dg* and *AGBm* for all young and adult QS plots.

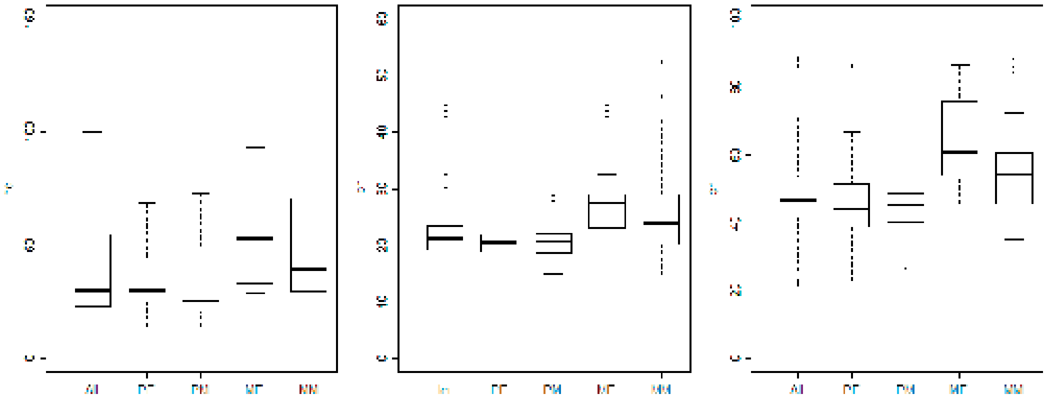

Figure 4. Boxplots of *hd*, *lcr* and *cr* for all plots and per structure classes.

timber both pure and mixed (PPi, PCR and RRB) and for QS. The former is due to the presence of different cohorts (PCR) and to the high density (RRB), and the latter as aforementioned is due to the difference in development stage (young *vs.* adult). The lower variability of QR, QS and PP is a reflection of low density where stem and crown diameter growths are promoted and as the trees are frequently in free growth, denoted by the *hd*.

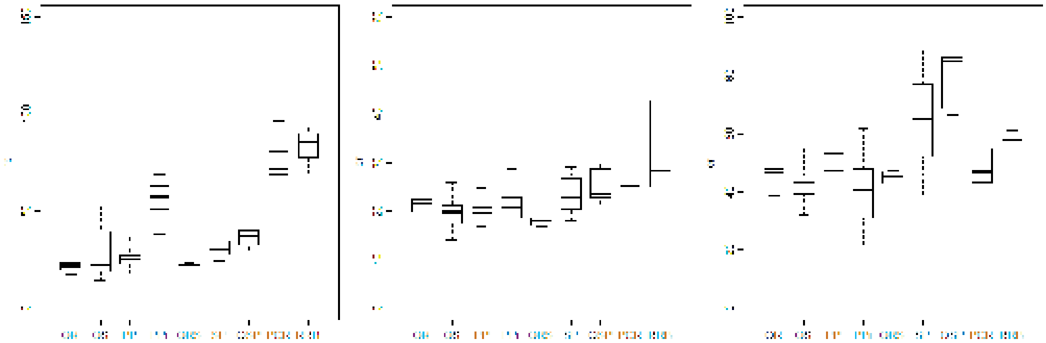

Figure 5. Boxplots of *hd*, *lcr* and *cr* per composition classes.

In 95% of the plots, $lc > \frac{1}{3}$ of the total height and the remaining 5% have $lc > 22\%$, indicating good tree and stand, growth and vigour [33–35]. Likewise, $cr > 30\%$ for most individuals and plots, increasing from pure to mixed plots and from multiaged to even-aged plots, though with larger variability in the mixed plots (**Figure 4**), indicates vigorous growth and well-balanced trees [33, 34], and $cr > 50\%$ good stability [34]. The variability between structure classes is denoted by the significant differences between PE and ME, PM and MM and ME and MM (all, $p < 0.05$). The smaller cr and variability are found for QR, QS and QRS plots, which are a reflection of management, where trees are periodically pruned to promote fruit production, especially adult stands [44]. In contrast, the mixtures of broadleaved and conifer species (SP, QSP, PCR) or of broadleaved species (RRB) have higher cr and larger variability (**Figure 5**). A possible explanation can be the effect of competition between trees, as these stands have higher densities, shade and branch abrasion phenomena can happen and the trees with competition advantages tend to expand their growing space, thus reducing the cr of those trees with fewer advantages and hence increasing the variability [33–35].

Species richness is lower in pure than in mixed plots [46, 56, 59]. Pure plots have one (55%), two (37%) or three or more species (8%). The mixed plots have two or three (40% each) or four or more species (20%). PE and PM with more than one species correspond to 43% and 45% of the total number of plots. Though N, G and AGB of the secondary species are much smaller than that of the main species, their presence is reflected in the relative density measures (**Table 3**). Consequently, as referred by several authors [45, 56, 59, 64, 65], other indices should be used to evaluate diversity. D and H variability in the pure plots is derived from the presence of more than one species. Thus, diversity can increase with only a few individuals (**Figures 6** and **7**), though in mixed plots is higher, denoting both the number of species and the equality of their proportions [45, 64, 65]. For D_N, D_G and D_{AGB}, significant differences were found between PE and PM and between PM and MM and for D_{AGB} also between ME and MM (all, $p < 0.001$). When comparing the different formulations of this index, significant differences are found between D_N and D_G for MM, between D_N and D_{AGB} for PM and MM and between D_G and D_{AGB} for the four structure classes (all, $p < 0.05$). A possible explanation is the number of individuals of the secondary species and the dimensions of their individuals. D_G does not account for the tree

Species composition	R_N		R_G		R_{AGB}	
	min	**max**	**min**	**max**	**min**	**max**
Pure even-aged						
QR	96.6	100.0	96.9	100.0	96.8	100.0
QS	83.0	100.0	87.2	100.0	90.5	100.0
PP	80.0	100.0	84.2	100.0	87.4	100.0
PPi	100.0	100.0	100.0	100.0	100.0	100.0
Pure multiaged						
QR	87.5	100.0	85.4	100.0	85.7	100.0
PP	83.3	100.0	85.4	100.0	85.7	100.0
PPi	93.8	100.0	95.1	100.0	99.9	100.0
PCR	85.3	100.0	97.3	100.0	98.6	100.0
Mixed even-aged						
Cr	32.7	80.0	42.1	80.0	45.8	91.3
Mixed multiaged						
PPi	39.1	80.0	77.7	80.0	80.9	99.7

Table 3. Minimum and maximum proportion of R_N, R_G and R_{AGB} per species for pure plots and per main species for mixed plots.

height contrary to what happens with D_{AGB} (cf. **Table 1**). Therefore, it can be said that D_{AGB} enables the incorporation of two dimensions, thus discriminating stands where the secondary species have the similar diameter but different height distributions. The Shannon and Weaver index shows the same trends as Simpson's [60, 61, 66, 68, 69], as denoted by the significant differences between PE and ME and PM and MM for H_N, H_G and H_{AGB} and between ME and MM for H_G and H_{AGB} (all, $p < 0.001$). The comparison of the three formulations of H shows significant differences for PM, ME and MM between H_N and H_G, for PM and MM between H_N and H_{AGB} and for PE and MM between H_G and H_{AGB} (all, $p < 0.01$).

Simpson and Shannon and Weaver indices formulated for 500 kg *AGB* classes enable further details of the differences between structure classes. A general decreasing trend for the former and increasing for the latter are observed from PE to MM, though with wider variability (**Figure 8**), which is clearer than with the formulation per plot. Hence, it enables to differentiate further the structure according to the proportions of *AGB* per structure classes (cf. **Figure 6**). A similar trend was found by [69].

Most plots have trees in all height zones (**Table 4**). The even-aged plots have more than 67, 86 and 86% of *N*, *G* and *AGB* in the superior and intermediate layer. Inversely, the multiaged plots have between 46 and 49% of *N* in the inferior layer, corresponding to 8–15% of *G* and 5–16% of *AGB*. This distribution is also reflected in the A index, though differently for A_N, A_G and A_{AGB} (**Figure 6**). In general, it increases from PE to MM, in accordance to several authors

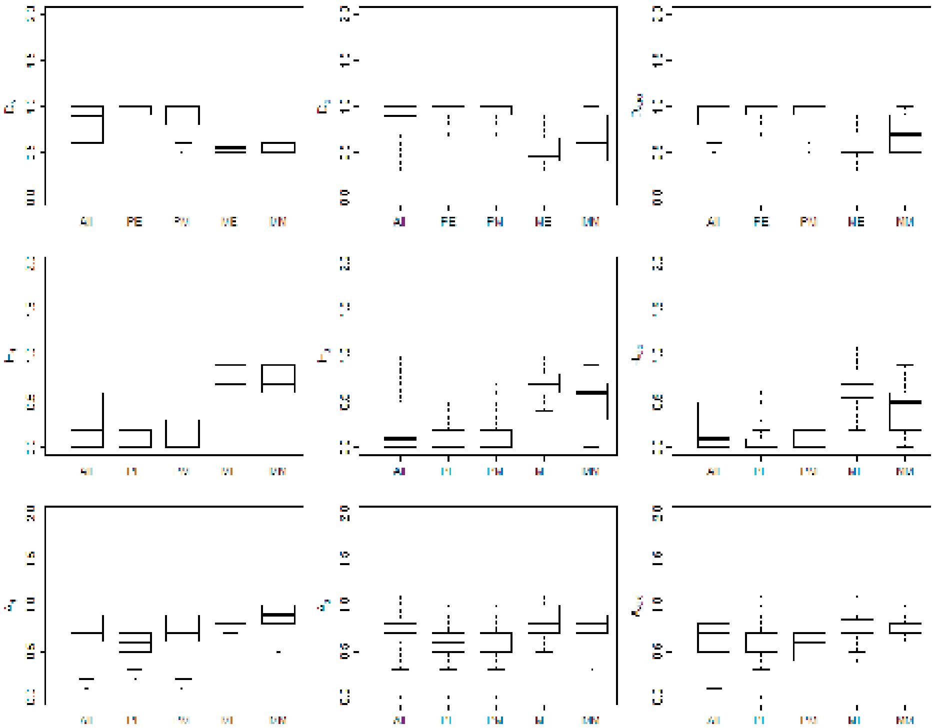

Figure 6. Boxplots of D_N, D_G, D_{AGB}, H_N, H_G, H_{AGB}, A_N, A_G and A_{AGB} for all plots and per structure classes.

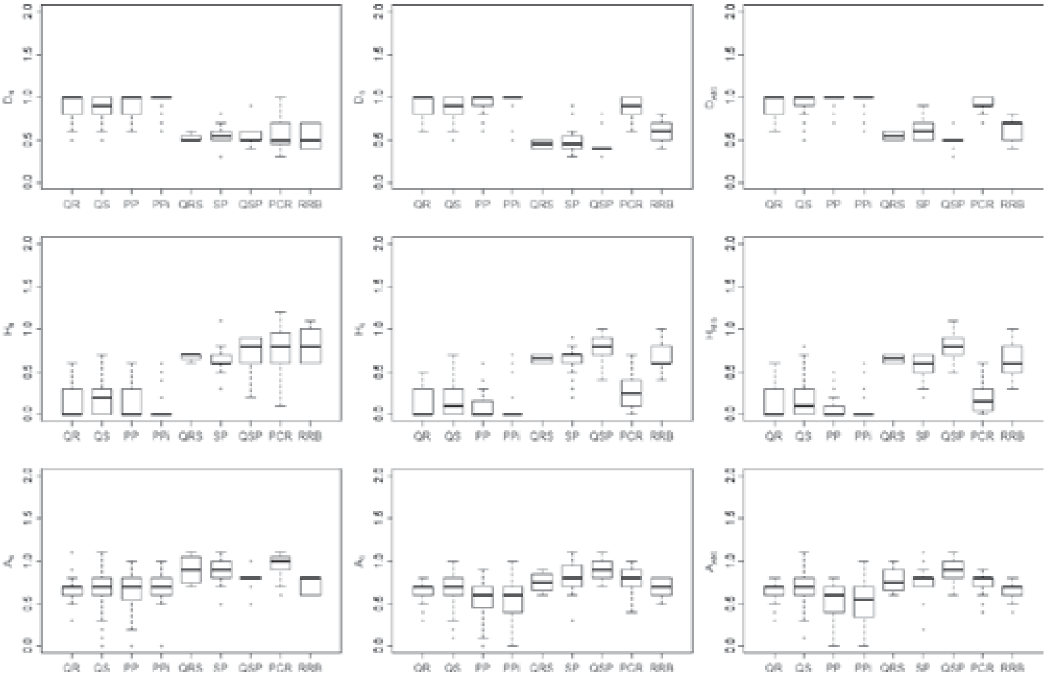

Figure 7. Boxplots of D_N, D_G, D_{AGB}, H_N, H_G, H_{AGB}, A_N, A_G and A_{AGB} per composition classes.

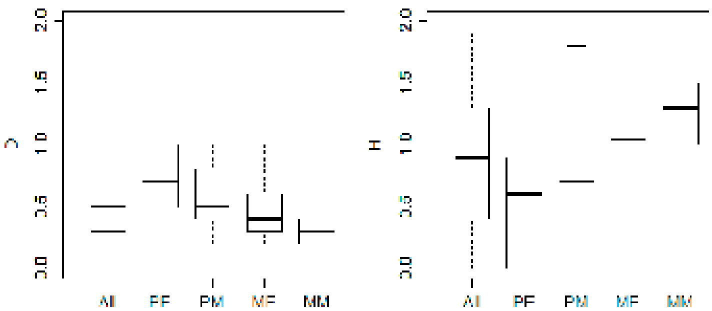

Figure 8. Boxplots of D and H per 500 kg *AGB* classes for all plots and per structure classes.

Structure class	Height zone								
	N			*G*			*AGB*		
	1	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**
All	41.9	36.8	21.2	11.4	47.1	41.5	8.2	46.9	44.9
Pure even-aged	32.6	47.1	20.3	13.8	50.0	36.2	13.9	52.9	33.3
Pure multiaged	49.4	31.2	19.3	15.2	40.4	44.4	15.7	40.3	44.0
Mixed even-aged	13.0	49.3	37.7	7.8	45.9	46.3	4.5	43.4	52.1
Mixed multiaged	46.5	34.2	19.2	8.2	51.4	40.4	4.5	49.3	46.1

where 1 is the inferior zone, 2 the intermediate and 3 the superior.

Table 4. Mean proportion of ***N***, ***G*** and ***AGB*** per height zone, for all plots and per structure class.

[45, 64–67]. Also, it is denoted by the significant differences between the four structure classes for A_N and between PE and ME and PM and MM for A_G and A_{AGB} (*all*, $p < 0.001$). The analysis per composition classes shows the increase of diversity from pure to mixed plots, though the variability within each group is rather wide, consequence of the number of individuals per height zone as well as their dimensions. Similarly to the former, higher values are attained for A_N than for A_G and A_{AGB} (**Figure 7**). This variability can be explained by the dimension of the individuals of the inferior layer. Though they can be in a rather high number, their diameter at breast height and total height are much smaller than those of the individuals of the two upper layers; thus, both the basal area and the above-ground biomass are also much smaller. This variability is also denoted by the significant differences for all formulations between PM and MM (*all*, $p < 0.001$). As expected PE and ME present no significant differences for A_N and A_G ($V = 259$, $p = 0.34$; $V = 44$, $p = 0.90$, respectively), for A_N and A_{AGB} ($V = 397$, $p = 0.37$; $V = 104$, $p = 0.43$, respectively) and for A_G and A_{AGB} ($V = 52$, $p = 0.38$; $V = 43$, $p = 0.13$, respectively). This can be, at least partially, explained by the small proportions of individuals in the inferior layer (cf. **Table 4**).

4. Conclusions

Structure analysis is of primordial importance for the study and modelling of forest stands as well as for their management. The most frequently used measures and indices characterise the stands with the number of individuals, stem diameter and total height. Above-ground biomass, as by their formulation incorporates both diameter and total height, is able to incorporate in the measures and indices the horizontal and the vertical dimensions. Also, forest biomass can give further insights to the carbon sequestration.

The results revealed that there are significant differences between the measures and indices calculated with the number of trees and basal area, when compared with those calculated with above-ground biomass. The latter can be of importance when there is the need to discriminate stands with similar number of trees but with different dimension proportions.

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Tree Stock, Structure and Use of Common Woody Species of a Town Neighboring Forest Reserve in Tanzania: Implication for Managing Carbon Accumulation

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Abstract

Town neighboring forests in the tropics suffer high human pressure owing to unregulated harvest to supply domestic energy and equipment. Although this causes considerable source of income among communities, it poses significant deforestation, thus, jeopardizing carbon accumulation potentials of most of the forests. This study therefore assessed the stock, structure and use of common woody species in a town neighboring forest reserve to elucidate the reserve's carbon accumulation potential amid pressures from surrounding communities. It was found out that the structure of the forest had been altered following unregulated tree harvest. There were selective harvesting and removal of valued timber trees, and key species that otherwise are responsible in regulating the ecosystem functioning of the reserve. It was apparent that the unregulated harvest is likely to jeopardize the ecosystem functioning and carbon accumulation potential of the reserve. Thus, to manage the reserve sustainably, awareness education on forest biodiversity conservation among surrounding local communities is recommended. We also recommend exploration of the reserve for ecotourism potentials. This might stimulate ecotourism activities in the area and provide an alternative source of income among the local population. This would add value and sense of ownership and stimulate self-mobilization for protection of the reserve.

Keywords: miombo, savanna, harvest, exploitation, community, tropics, Africa

1. Introduction

The ecosystems of tropical natural forests are undergoing rapid destruction due to increasing of many factors including anthropogenic activities, natural catastrophes and climate change. Understanding in detail the drivers of such destruction and implied effects on carbon accumulation is of vital importance for conservation [1]. The majority of the forest ecosystems especially in the tropic are never been stable following anthropogenic pressures, climatic, and geo-morphological influences [2]. Overall, the threats facing the tropical forests due to anthropogenic activities are habitat modification and destruction [3] of which burning and harvesting are the most pressing [3–9]. The reasons behind burning are many including attainment of higher visibility to evade wild carnivores, to find prey among hunters and initiation of off-season re-growth of perennial fodders [5, 6]. Harvesting, on the other hand, comes in many forms to fulfill needs such as firewood, building materials, domestic items (e.g., mortar, bow, arrow handle, spear handle), fencing materials, among others. While these forms of use of forest products cause significant source of income for many small scale farmers in developing countries, they pose significant deforestation [10]. For example, in the 1990s, there were approximately 40 million people living in Africa's miombo woodlands—all with an additional 15 million urban residents depending on the woodlands for domestic fuel—especially charcoal [11]. Moreover, the building materials and fuel energy are needed in high supply particularly for forests neighboring residential areas [1, 10]. In Kenya, for instance, some tree species such as *Brachylaena huillensis* and *Dalbergia melanoxylon* are highly sought and have been depleted in forests near towns [12]. In Tanzania, tree species such as *Pterocarpus angolensis, Milicia excelsa, Afzelia quanzensis, Khaya anthotheca* and *D. melanoxylon* are categorized as protected trees due to overexploitation and harvesting them even on private land requires the government authorization [13, 14].

Some studies indicate that valued timber trees such as *P. angolensis* suffers high human pressure and are highly sought throughout south-central Africa [15, 16], such that the stock in most places has been depleted to vulnerable numbers [17, 18], and the structure has been damaged beyond repair in the near future in some reserves [19]. In Msagania Forest Reserve for example, Schwartz et al. [18] reported that loggers have reduced the population density of *P. angolensis* from 11.4 to 3.7 trees/ha, whereby, trees are left standing only if they are not of harvestable size yet. This kind of selective use has a profound impact on tree stock and structure and can jeopardizes the forests carbon accumulation potential [20] as well as the general biodiversity [21, 22]. The fact that most important sites and habitats in Tanzania are not included in a reserved areas system, and most of those under the reserved areas system are not well protected [1, 23], the forests are likely to continue experiencing unregulated exploitation from surrounding communities [1, 23, 24], thus jeopardizing their carbon accumulation potential.

This study therefore aimed at assessing the stock, structure and use of common woody species in a town neighboring forest reserve to elucidate its carbon accumulation potential and put forward management recommendations. The forest, hereby referred to as Ndege Forest Reserve (NFR) is a small reserve located about 10 km north of Morogoro Municipality, Tanzania (**Figure 1**). The surrounding local communities have depleted its biological and ecological resources to unprecedented level. There is widespread tree felling, encroachment and bushfires [14]. The northwest slope of the reserve is already completely deforested and its

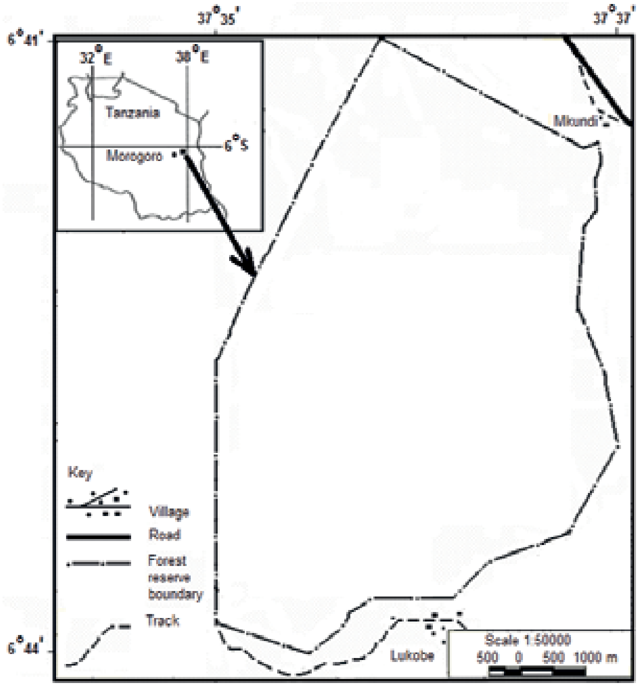

Figure 1. Map showing location of the study area: modified after national tree seed programme in Tanzania, NTSP [30].

entire valued tree has been wiped out [25]. However, there is lack of information on natural stands of different plant species in this reserve. The fact that information on threats facing trees is usually scanty and anecdote, assessment of tree stock, structure and human use for the NFR may shade light on our knowledge on carbon accumulation potential of the reserves neighboring towns in tropical Africa.

2. Materials and methods

This study was conducted in Ndege Forest Reserve (NFR). This reserve is found between 6°41′ to 6°44′ S and 37° 35′ to 37°37′ E [25, 26]. It is a small reserve covering an area of about 36.14 km^2 and is located about 10 km north of Morogoro Municipality, Tanzania (**Figure 1**). The rainfall is estimated at 850 mm/year and the temperatures range from 20–25°C [25]. The vegetation of the area is mainly woodlands [25, 26]. The dominant tree species of the reserve are *Brachystegia microphylla, Brachystegia boehmii* and *Julbernardia globiflora*. Some other trees include *Albizia harveyi, Brachystegia spiciformis, Sterculia africana, Sterculia quinqueloba* and *Xeroderris stuhlmannii*. Important timber species in the reserve are *Brachystegia spp., P. angolensis* and *Afzelia quanzensis* [26].

During data gathering, we sampled 50 sites in which we took vegetation measurements and indices of tree use. The detailed study design and vegetation sampling procedure is explained in Modest et al. [14]. In brief, we measured diameter at breast height (dbh), crown diameter and height of trees and counted saplings (dbh 1 ≤ 10 cm) and trees (dbh > 10 cm). The stump diameters were as well measured. This data was collected within 50 quadrats measuring 20 by 20 m spread randomly in the study area. Furthermore, additional 50 quadrats measuring 5 by 5 m were established within the 20 by 20 m quadrats for sampling seedlings which were regarded as small trees with dbh < 1 cm [27, 28]. We also recorded the global positioning system (GPS) data for each quadrat for use in determining the distribution of valued timber tree species in the reserve.

We analyzed the data, first by determining the floristic composition of all woody species through preparing a checklist; then, using previously published literature, we assigned the tree species in their respective families and defined their habit. Second, from the checklist we determined the common woody species using the Important Value Index (IVI) criteria following Kent and Coker [29], as follows:

$$\text{IVI} = \text{relative dominance} + \text{relative frequency} + \text{relative density}$$

where

$$\text{Relative dominance} = (\text{Sum of basal area of the ith species/total basal area of all species}) \times 100.$$

$$\text{Relative density} = (\text{Number of stems/ha of the ith species/total number of stems/ha of all species}) \times 100.$$

$$\text{Relative frequency} = (\text{Frequency of the ith species/total frequency of all species}) \times 100.$$

Here, the first 10 species with the highest IVIs were picked as common species and the rest of the discussion in this chapter refers to them.

We proceeded in assessing the stock, structure and use of common woody species by determining their population structure, conservation priority, level of exploitation and spatial distribution. The population structure was assed based on tree population density, tree dbh, density distribution of seedlings, saplings and mature trees and canopy diameter as well as canopy area. The population density on the other hand was calculated as number of individual plants/ha [29, 31, 32]. The tree dbh were categorized into 10 classes from class 1 dbh 0–4.9 cm, class 2 dbh 5–9.9 cm, class 3 dbh 10–14.9 cm, class 4 dbh 15–19.9 cm, class 5 dbh 20–24.9 cm, class 6 dbh 25–29.9 cm, class 7 dbh 30–34.9 cm, class 8 dbh 35–39.9 cm, class 9 dbh 40–44.9 cm and class 10 dbh above 45 cm [33, 34]. The dbh were also grouped into three categories following Jones [35] to determine the density of seedlings, saplings and mature trees as follows:

seedlings, individuals with dbh < 1 cm

saplings, individual with dbh 1 ≤ 10 cm and

trees, individual with dbh > 10 cm

Moreover, the canopy diameters were standardized and presented as mean canopy diameter ± SE, while the canopy area (CA) was calculated following Cunningham [33] as shown below:

$$\mathrm{CA} = \mathrm{W1}/2 \times \mathrm{W2}/2$$

where W1 = widest canopy diameter and W2 = perpendicular diameter to the widest diameter.

The conservation priority for each of the 10 common woody species was determined using the IVI classes and the regeneration status based on density of seedlings and saplings following Shibru and Balcha [36]. For this case, the IVIs were categorized into five classes:

Class 5 = IVI (<1)

Class 4 = IVI (1–10)

Class 3 = IVI (10.1–20)

Class 2 = IVI (20.1–30) and

Class 1 = IVI (>30)

The seedlings and saplings on the other hand were grouped into three classes:

Class 1: species with 0 individual seedling and saplings ha–1

Class 2: species with 0 > 50 individual seedlings and saplings ha-1

Class 3: species with <50 individual seedlings and saplings ha–1

To assess the level of exploitation of the trees, we determined the basal area of the stumps. First, the stump diameters were converted into dbh using this linear regression equation presented in Luoga et al. [37]; $\mathrm{dbh} = -3.17 + 0.961\mathrm{SD}$, where SD = stump diameter. Then, the resulting dbh were used to calculate the basal area following Martin [38] as follows; $\mathrm{BA} = \pi(\mathrm{dbh}/2)^2$, where dbh is diameter at breast height. Finally, we assessed the distribution of the highly valued timber species *P. angolensis* using the ArcView computer software version 3.2.

3. Results and discussion

3.1. Floristic composition and population structure of the common woody species

A total of 102 woody species were recorded in the reserve and were represented by 30 families as presented in **Table 1**. The most dominant family was Fabaceae with 31 species, whereas several other families were represented by only one species. Of the 102 woody species recorded, 78 were trees while 24 were shrubs. The first 10 common woody species determined using the IVI classification procedure were *Acacia nilotica, Brachystegia boehmii, Brachystegia microphylla Combretum molle, Dalbergia boehmii, Diplorhynchus condylocarpon, Julbernardia globiflora, Lannea welwitschii, Pteleopsis myrtifolia* and *Pterocarpus angolensis.* The

SN	Species name	Family	Habit	
			Shrub	Tree
1	*Ozoroa insignis* Del.	Anacardiaceae	*	
2	*Ozoroa reticulata* (Baker F.) R. Fern. & A. Fern.	*	*	
3	*Lannea schimperi* (A. Rich.) Engl.	*		*
4	*Lannea welwitschii* (Hiern) Engl.	*		*
5	*Sclerocarya birrea* (A. Rich.) Hochst.	*		*
6	*Sorindeia madagascariensis* Baill	*		*
7	*Annona senegalensis* Pers	Annonaceae		*
8	*Steganotaenia araliacea* Hostchst.	Apiaceae	*	
9	*Diplorhynchus condylocarpon* (Muell-Arg.) Pichon	Apocynaceae		*
10	*Kigelia africana* (Lam) Benth.	Bignoniaceae		*
11	*Markhamia obtusifolia* (Baker) Sparague	*		*
12	*Stereospermum kunthianum* Cham.	*		*
13	*Bombax rhodognaphalon* K. Schum.	Bombacaceae		*
14	*Ehretia amoena* Klotzsch	Boraginaceae		*
15	*Commiphora africana* (A. Rich.) Engl.	Burseraceae	*	
16	*Boscia salicifolia* Oliv.	Capparidaceae	*	
17	*Maerua triphylla* A.Rich	*		*
18	*Maytenus senegalensis* (Lam.) Excell	Celastraceae		*
19	*Parinari excelsa* Sabine	Chrysobalanaceae		*
20	*Garcinia huillensis* Welw	Clusiaceae		*
21	*Combretum adenogonium* Steud. ex A. Rich	Combretaceae		*
22	*Combretum collinum* Fresen.	*		*
23	*Combretum molle* R.Br. ex G. Don	*		*
24	*Combretum padoides* Engl. & Diels	*		*
25	*Combretum zeyheri* Sond.	*		*
26	*Pteleopsis myrtifolia* Engl. & Diels	*		*
27	*Terminalia sericea* Burch. ex DC	*		*
28	*Diospyros consolatae* Chiov.	Ebenaceae	*	
29	*Diospyros zombensis* (B.L. Burt) F. White	*	*	
30	*Diospyros kirkii* Hiern	*		*
31	*Diospyros mespiliformis* Hochst. ex A. DC	*		*
32	*Drypetes gerrardii* Hutch	Euphorbiaceae	*	
33	*Bridelia cathartica* Bertol.F.	*		*
34	*Croton macrostachyus* Hochst. ex Delile	*		*

SN	Species name	Family	Habit	
35	*Croton megalocarpus* Hutch	*		*
36	*Margaritaria discoidea* (Baill.) Webster	*		*
37	*Pseudolachnostylis maprouneifolia* Pax	*		*
38	*Spirostachys africana* Sond	*		*
39	*Suregada zanzibariensis* Baill.	*		*
40	*Acacia pentagona* (Schumach.) Hook. F.	Fabaceae	*	
41	*Bauhinia petersiana* C. Bolle	*	*	
42	*Cassia abbreviata* Oliver	*	*	
43	*Dichrostachys cinerea* (L.) Wight & Arn	*	*	
44	*Ormocarpum kirkii* S. Moore	*	*	
45	*Acacia goetzei* Harms.	*		*
46	*Acacia goetzei subsp. microphylla* Brenan	*		*
47	*Acacia nigrescens* Oliver	*		*
48	*Acacia nilotica* (L.) Willd.ex Del	*		*
49	*Acacia polyacantha subsp.*	*		*
50	*Afzelia quanzensis* Welw.	*		*
51	*Albizia harveyi* Fourn	*		*
52	*Albizia petersiana* Oliver	*		*
53	*Brachystegia boehmii* Taub.	*		*
54	*Brachystegia microphylla* Harms	*		*
55	*Brachystegia spiciformis* Benth	*		*
56	*Dalbergia boehmii* Taub	*		*
57	*Dalbergia melanoxylon* Guill.& Perr	*		*
58	*Dalbergia nitidula* Baker	*		*
59	*Dalbergia obovata* E. Mey.	*		*
60	*Erythrina abyssinica* Lam.ex DC	*		*
61	*Erythrophleum africanum* (Benth.) Harms	*		*
62	*Julbernardia globiflora* (Benth.) Troupin	*		*
63	*Lonchocarpus bussei* Harms	*		*
64	*Millettia usaramensis* Taub	*		*
65	*Pterocarpus angolensis* DC.	*		*
66	*Pterocarpus tinctorius* Welw.	*		*
67	*Scorodophloeus fischeri* (Taub.) J. Leon	*		*
68	*Swartzia madagascariensis* Desv.	*		*
69	*Xeroderris stuhlmannii* (Taub.) Mendoca & Sousa	*		*

SN	Species name	Family	Habit	
70	*Pterocarpus rotundifolius* (Sond.) Druce	*		*
71	*Strychnos spinosa* Lam.	Loganiaceae		*
72	*Ficus bussei* Mildbr.& Burret	Moraceae		*
73	*Ficus exasperata* Vahl	*		*
74	*Ficus sycomorus* L.	*		*
75	*Treculia africana* Decne. ex Trecul	*		*
76	*Ochna leptoclada* Oliver	Ochnaceae		*
77	*Ximenia americana* L.	Olacaceae		*
78	*Ximenia caffra* Sond.	*		*
79	*Ziziphus mucronata* Willd	Rhamnaceae		*
80	*Crossopteryx febrifuga* (G.Don) Benth.	Rubiaceae	*	
81	*Gardenia ternifolia* Schumach. & Thonn.	*	*	
82	*Pavetta crassipes* K.Schum	*	*	
83	*Vangueria infausta* Burch.	*	*	
84	*Catunaregam spinosa* (Thumb) Tirveng.	*		*
85	*Grumilea riparia* K. Schum & K. Krause	*		*
86	*Toddalia asiatica* Baill.	Rutaceae	*	
87	*Zanthoxylum chalybeum* Engl.	*	*	
88	*Allophyllus africanus* P. Beauv.	Sapindaceae	*	
89	*Deinbollia borbonica* Scheff.	*	*	
90	*Haplocoelum inopleum* Radlk	*		*
91	*Lecaniodiscus fraxinifolius* Bak.	*		*
92	*Zahna africana* (Radlk.) Exell	*		*
93	*Synsepalum brevipes* (Baker) T.D. Penn	Sapotaceae		*
94	*Harissonia abyssinica* Oliv.	Simaroubaceae	*	
95	*Dombeya cincinnata* K. Schum. ex Engl.	Sterculiaceae		*
96	*Dombeya rotundifolia* (Hochst.) Planch.	*		*
97	*Sterculia africana* (Lour.) Fiori.	*		*
98	*Sterculia quinqueloba* (Garcke) K. Schum	*		*
99	*Grewia ectasicarpa* S. Moore	Tiliaceae	*	
100	*Grewia bicolor* Juss.	*		*
101	*Heteromorpha trifoliata* (Wendl.) Eckl. & Zeyh.	Umbelliferae	*	
102	*Vitex payos* (Lour.) Merr.	Verbenaceae		*

Table 1. Checklist for the tree species recorded during field survey. Under the "Family" column, the star "*" denote that the species belongs to the family mentioned above it. The same star under the "Habit" column denote that the species is either a Shrub if the star is under the "Shrub" column, otherwise the species is a Tree.

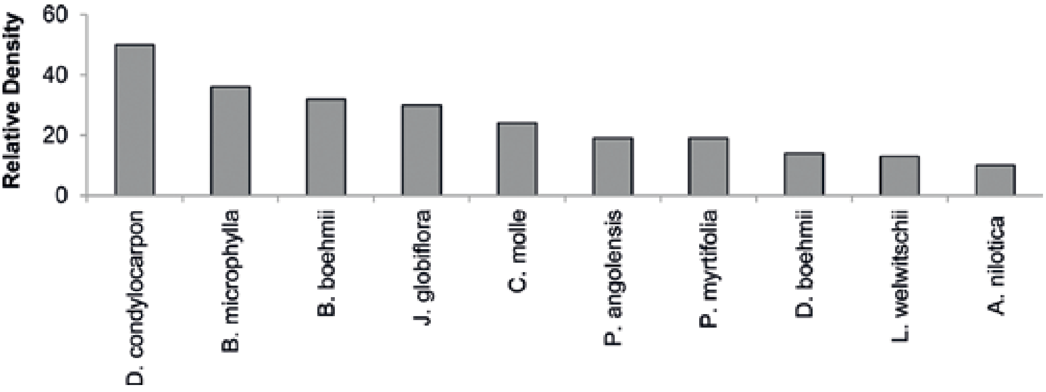

Figure 2. Relative density of the common woody species in NFR.

relative density of these common woody species for stems with dbh > 4 cm is presented in **Figure 2**, whereby, *D. condylocarpon* had the highest relative density of 50 stem/ha, while *L. welwitschii* had the least density that is,13 stem/ha.

The population structure of the common woody species in NFR was assessed based on dbh class size distribution, mean crown diameter and area, and density distribution of seedlings, saplings and mature trees. The dbhs of the common woody species recorded during the study are presented in **Figure 3**. All the 10 representative woody species had more individuals in their first classes with an abrupt drop beginning in the following class indicating the characteristic inversed J-shapes. However, except for *C. molle* of which its graph had a smooth inversed J-shape, the graphs for the rest of the species seemed interrupted. This is an indication of selective removal of individuals, especially those at pole size stages and the mature ones. The Morogoro region in which NFR is located has a history of human exploitation of tree species. For example, Wells and Wall [39], reported that, between 1970 and 1980 Morogoro Region was the source of much of the hardwood timber going to the Dar es Salaam market, but from early 1990s, the Morogoro region was exhausted of valued timber species such as *P. angolensis* and the production shifted to Tabora region, and even further west to Rukwa Region by late 1990s. Therefore, this denotes that, NFR being in Morogoro region, its quality timber trees has long been extracted, which is also evidenced by small dbhs in almost all the common tree species **Figure 3**. Moreover, the trend in dbh classification of the 10 common woody species (**Figure 3**) indicates that the graphs of some species such as *B. microphylla* are almost flat for individuals from the sapling stage. This is not a health state for the NFR since it is evident that trees are harvested in all stages from sapling to mature trees. Newton [40] pointed out that, when there is a very few trees recorded in lower dbh classes, extracting of large sized trees is detrimental to the future populations. Therefore, if unsustainable harvest continues in NFR, it is likely that the knock-off effect of the forest valuable trees will occur within a short span of time.

Figure 4 shows that of the common woody species *B. microphylla* had the biggest mean canopy diameter while that of *C. molle* was the smallest. Moreover, the mean canopy area was also the biggest for *B. microphylla* and the smallest for *C. molle* as shown in **Table 2**. The bigger crown diameter and area of *B. microphylla* as well as its higher density are indication that the species is potential especially in influencing the ecosystem functioning of the reserve. Thus, *B. microphylla* can be regarded as a foundation species for the NFR. By definition, a

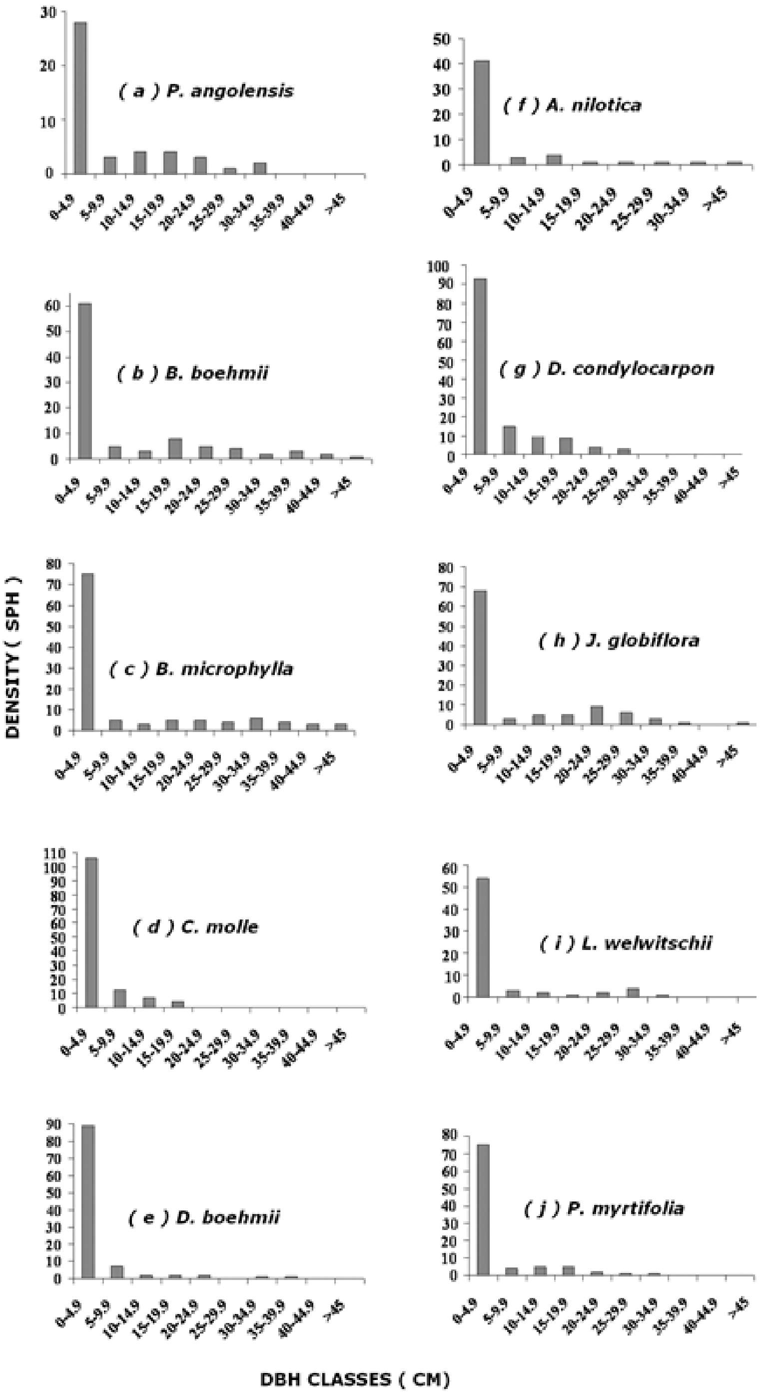

Figure 3. DBH class distribution of the common woody species recorded in NFR.

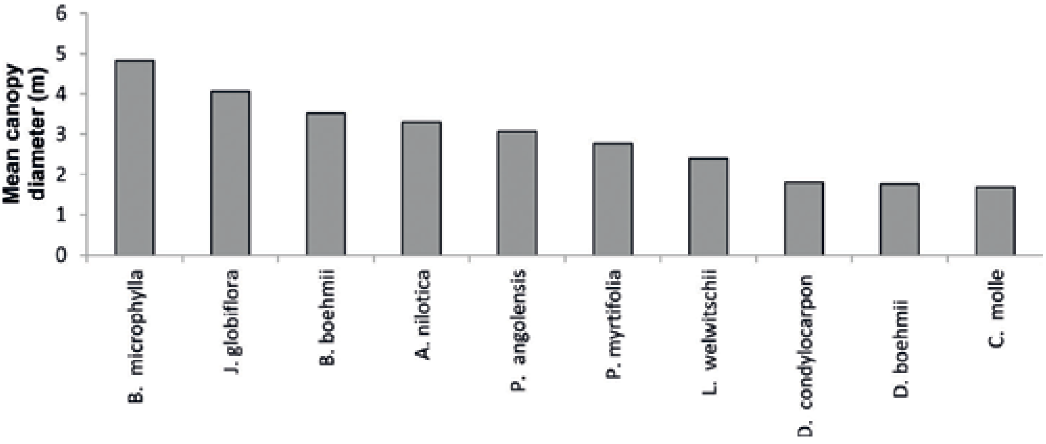

Figure 4. Mean canopy diameter of the common woody species.

Scientific name	Mean canopy area/ha (m^2)	Std. error of the mean (SEM)	Lower 95% conf. limit	Upper 95% conf. limit
B. microphylla	11.55	1.46	8.64	14.46
J. globiflora	8.19	0.94	6.31	10.08
B. boehmii	6.26	0.79	4.46	7.84
A. nilotica	5.44	0.94	0.40	13.74
P. angolensis	4.70	0.82	3.05	6.35
P. myrtifolia	3.42	0.04	2.42	4.37
L. welwitschii	3.40	0.89	01.58	5.21
D. boehmii	1.85	0.53	0.78	2.63
D. condylocarpon	1.51	0.14	1.23	1.79
C. molle	1.37	0.17	1.04	1.72

Table 2. Mean canopy area/ha (m^2) of the common woody species recorded in NFR.

foundation species is "a single species that defines much of the structure of a community by creating locally stable conditions for other species and by modulating and stabilizing fundamental ecosystem processes" [41, 42]. These kinds of species with greater influence on ecosystem usually possess greater repercussions on ecosystem functioning in case they disappear [43]. Therefore, for NFR *B. microphylla* can be regarded as a candidate foundation species, and in case it disappears there could be far reaching consequences [43]. Probably the species we see today in NFR especially the shade tolerant ones are a masterpiece of *B. microphylla*. Thus, if *B. microphylla* vanishes from NFR the species composition is likely to change with cascading effects on carbon accumulation potential of the reserve.

The classification of stems into seedlings, saplings and mature trees showed the least number of seedlings/ha for the highly valued timber species *P. angolensis* **Figure 5**. The 24 seedlings/ha of *P. angolensis* were far less compared to that of *C. mole* and *D. condylocarpon* (88/ha) respectively. This could be an indication that there are difficulties in seedling germination in NFR for this species. Probably fires, damages from tree felling activities and rolling logs, or all of these combined are hampering the regeneration of the species. Other common woody species namely *B. boehmii, B. microphylla, J. globiflora* and *P. myrtifolia* had a bit higher number of seedlings but the saplings seemed to suffer a high mortality. This could be an indication that the saplings are removed to cater for different purposes such as house construction or instrument making [10].

3.2. Tree conservation priority

The tree conservation priority was assessed based on IVI classes and regeneration status (**Table 3**). Categorization of the common woody species into conservation priority classes based on IVI criteria placed *L. welwitschii, D. boehmii, P. angolensis* and *A. nilotica* in class 4. This means that these species have insufficient stock and are recommended for conservation priority, for example, Zegeye et al. [44], Kacholi [45]. The IVI criterion is used in determining the conservation priority of species whereby those with low IVI values are considered of main concern for conservation [44]. Therefore, the management of NFR should place special consideration in protecting *A. nilotica, D. boehmii, L. welwitschii* and *P. angolensis* against anthropogenic pressures for their persistence. On the other hand, the remainder of the common woody species recorded in NFR, that is, *J. globiflora, B. boehmii, D. condylocarpon, C. molle, P. myrtifolia* and *B. microphylla* are of less conservation concern since they fall under a higher IVI class value. However, the management of NFR is not advised to relax as protection of every single species in the reserve is paramount. As pointed out earlier, *B. microphylla* has been categorized by this study as a foundation species, thus, despite the fact that this species appears common, it still deserves special consideration for protection in respect of its ecological implication in case it disappears.

Moreover, classification of the 10 common woody species into regeneration status placed the highly valued timber species *P. angolensis* in class 2 denoting that the species had insufficient number of seedlings and saplings (**Table 4**). This suggests that either the species recruitment is insufficient or the seedlings and perhaps the saplings suffer high mortality before making it to maturity, the phenomena which are likely to jeopardize its long-term persistence in the reserve. As pointed out earlier, *P. angolensis* had the least number of seedlings observed among the 10 common woody species and this is attributed to a collective number of factors including disturbances brought about by tree felling activities, or fires. Omeja et al. [34] claims that human disturbance influence seed dispersal mechanisms, fruiting, germination and regeneration of tree species. Thus, disturbance by activities involved during charcoal burning (see **Plate 1 a** and **b**) perhaps contributes in killing young seedlings as logging was observed almost everywhere in the reserve especially on lower altitudes. Similarly, van Daalen [46], pointed out that very high intense fire reduces the viability of seeds of *P. angolensis* limiting their germination potential. Thus, the intense fires that the NFR faces (Personal Communication) could be killing more seeds of *P. angolensis*, allowing only a few to germinate.

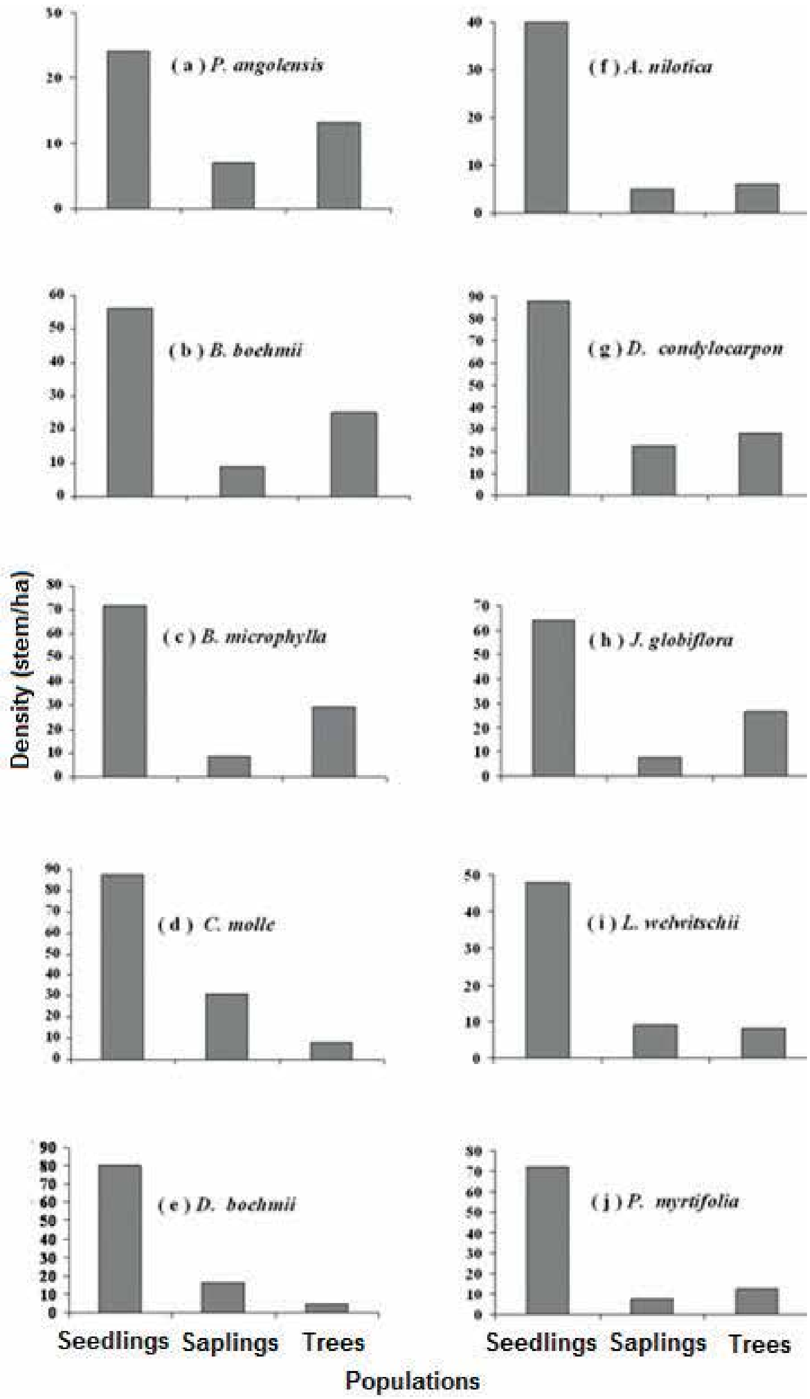

Figure 5. The population of seedlings, saplings and mature trees of the common woody species recorded in NFR.

IVI priority classes	IVI value class	Species
5	<1	—
4	1–10	*L. welwitschii* *D. boehmii* *P. angolensis* *A. nilotica*
3	10.1–20	*J. globiflora* *B. boehmii* *D. condylocarpon* *C. molle* *P. myrtifolia*
2	20.1–30	*B. microphylla*
1	>30	—

Table 3. IVI priority classes for the 10 common species recorded in NFR.

Class 1	Class 2	Class 3
0 individuals ha^{-1}	0 > 50 Individuals ha^{-1}	<50 Individuals ha^{-1}
	P. angolensis	*B. microphylla* *D. condylocarpon* *J. globiflora* *B. boehmii* *D. boehmii* *A. nilotica* *P. myrtifolia* *C. molle* *L. welwitschii*

Table 4. Regeneration status of the 10 common woody species based on individual seedlings and saplings/ha.

3.3. Exploitation and spatial distribution of the common woody species

Harvesting rate of the common woody species was evaluated based on basal area/ha of the cut trees. During this study, some stumps could not be identified simply because most of them had already lost identification evidences such as barks and had no sprouts. Of the 10 common woody species, stumps were recorded for *B. microphylla, C. molle, J. globiflora, P. angolensis* and *P. myrtifolia* (**Table 5**). From this perspective, the total basal area for the harvested species seemed to be small (0.32 m/ha). Of the common woody species found harvested, *J. globiflora* seemed to be the highly preferred species accounting for 0.14 m^2/ha followed by *P. angolensis* with 0.10m^2/ha, while *B. microphylla* was the least harvested. The fact that this study has

Plate 1. (a) Pieces of wood found cut for charcoal burning in NFR, (b) the kiln previously used for charcoal burning.

Scientific name	Basal area/ha (m^2)	Percentage	Rank
J. globiflora	0.14	43.34	1
P. angolensis	0.10	32.00	2
C. molle	0.03	10.70	3
P. myrtifolia	0.03	8.41	4
B. microphylla	0.01	4.54	5

Table 5. Basal area/ha of the common woody species found harvested in NFR.

considered *B. microphylla* as a foundation species following its bigger canopy area, the removal of this species though at a smaller scale triggers alarm. As pointed out earlier, this species has a high chance of influencing other species especially in suppressing shade intolerant species. Therefore, if harvest continues for *B. microphylla,* there is a high possibility of seeds for other species that were being suppressed to start emerging and this could change the species composition and ultimately the ecosystem functioning and carbon accumulation potential of the area. Some previous studies pointed out that increasing ground illumination through felling bigger canopy trees has many consequences including allowing colonization of new plant species and development of new communities such as grasses [9, 20], thus impairing the ecological succession and carbon accumulation potential of the affected area [20]. Therefore, exploitation of trees in NFR especially among key species needs to be monitored by the responsible authorities for stability of ecosystem processes in the reserve.

One of the important questions under the current study concerned the distribution of valued timber species in NFR. This was so in order to elucidate the probable influence of the surrounding community on the reserve's valued biological resources. To understand this, we plotted the spatial distribution of the highly valued timber species *P. angolensis* on the reserve's map as shown in **Figure 6**. From the map, it was apparent that quadrats into which *P. angolensis* were recorded concentrated more in the higher altitudes of the reserve to the interior. We did not

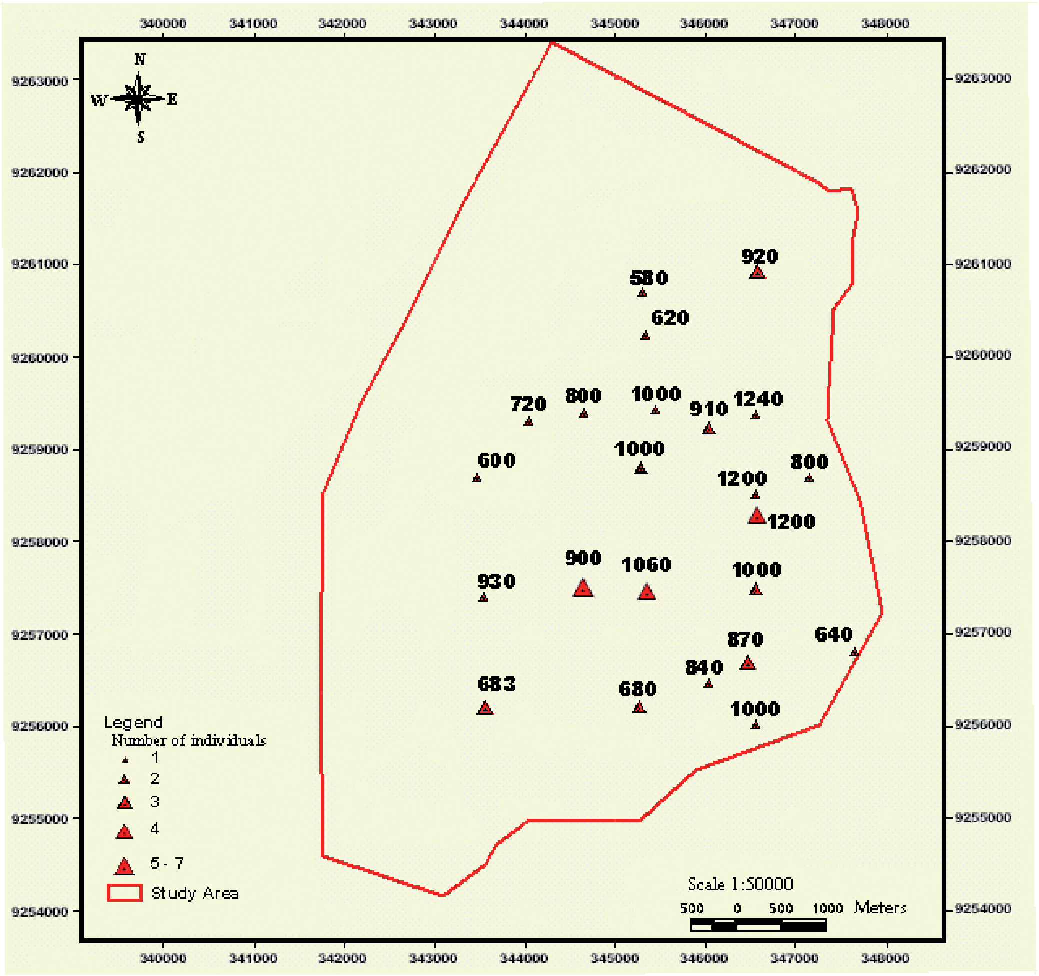

Figure 6. Map of the study area showing the distribution of *P. angolensis* as displayed by the ArcView version 3.2 computer software. Numbers inside the map boundaries show the altitudes while triangles represent plots where *P. angolensis* was recorded.

record *P. angolensis* from the northern, western or southern parts of the reserve. Moreover, the range at which *P. angolensis* occurred in NFR was between 600 to 1240 m a.s.l. The absence of *P. angolensis* especially in the northern, western and southern parts of the reserve perhaps was a result of selective logging, fires or edaphic factors. The eastern part of NFR consists of steep slopes and difficult terrain. This could be a reason why mature *P. angolensis* were recorded just close to the boundary of the eastern part of the reserve—perhaps tree loggers might be experiencing difficulties in accessing this area. According to Lovett et al. [26], the edges of NFR face fire incidences, deforestation for charcoal making and tree cutting for timber sawing. The villagers at Mkundi and Lukobe also disclosed that people prefer poaching trees for charcoal making and timber just a few hundreds of meters from the edges of the reserve in fear of the forest officers who usually conduct ambush patrols (Personal Communication). Therefore, tree cutting and frequent fires in the edge of NFR are probably the root of mortality and poor re-establishment of *P. angolensis* in the northern, western and southern parts of the reserve. Although *P. angolensis* is considered to be resistant to fires, heavy and/or frequent fires damage seedlings and prevent them passing to sapling stages [47].

4. Implications of the results on carbon management

The information we have provided in this chapter is expected to guide the authority responsible in overseeing NFR on managing carbon in the forest. Specifically, implementation of the recommendations on fire control and halt of tree cutting highlighted later is necessary to allow for tree restoration, hence increased capacity of the forest in capturing carbon. Awareness raising campaign on biodiversity conservation also insisted later under the recommendations section is essential on deviating the interest of the surrounding communities from overdependence on the forest's biodiversity—to allow the forest to regenerate and augment its capacity in seizing carbon.

5. Conclusion and recommendations

This study concludes that the population structure of the common woody species in NFR did not show a natural stand. Fires and selective logging seemed to influence the distribution of timber trees as our model timber species that is, *P. angolensis* was not found in fire prone part of the reserve and the species was as well recorded mostly in hard to reach places. Of the 10 common woody species on the other hand, *B. microphylla* can be regarded as a foundation species in NFR and its removal should be regulated as when it disappears for example, the ecosystem functioning and the carbon accumulation potential of the reserve might be jeopardized. To manage the tree species sustainably in NFR, this study recommends awareness education on forest biodiversity conservation among surrounding local communities. Regular patrol in the reserve is also needed to stop illegal logging. Fire control is needed to allow for regeneration of the vegetation in the reserve and to minimize seed mortality in order to attain maximum recruitment of the tree species. A study to compare dry and wet season status is required in order to explore the species in its full life cycle as this will take care of the individuals that might be overlooked during dry seasons while in their underground shoot-die back stages. A study to investigate if edaphic factors apart from fires and human disturbances do influence the distribution of the preferred timber species that is, *P. angolensis* is also recommended. Finally, we recommend an exploration of the ecotourism potential of NFR to be undertaken. This might stimulate ecotourism activities in the reserve to provide an alternative source of income and generate jobs among the local population. This would add value and sense of ownership and stimulate self-mobilization among the surrounding communities for protection of the reserve.

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Conflict of interest

The authors declare no conflict of interest for publication of this study.

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Chapter 3

Plant Diversity, Ecological Services, and Carbon Stock Assessment in Cocoa Agroforestry Plantations of Forest and Savannah Transitions in Cameroon

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Additional information is available at the end of the chapter

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Abstract

This study was carried out in cocoa-based agroforestry systems in Mbam and Inoubou department, Center Region, Cameroon. The study aimed at assessing the plant species diversity, ecological services, and carbon sequestration potentials of diverse trees associated in cocoa agroforestry systems. Twenty-seven sampling plots of 100 × 20 m were established in cocoa agroforestry systems in three villages. Our results registered the occurrence of 238 plant species grouped into 16 families in the sampled area. Sterculiaceae, Burseraceae, and Moraceae were the three dominant families. The species richness and diversity that were assessed using the Shannon index were 0.62, 0.66, and 0.68, respectively, while using the Simpson index, they were 1.421, 1.409, and 0.349, respectively, for Mouko, Rionong, and Nyamsong 3. Carbon stock sink was also estimated at 92.03, 55.18, and 46.83 tC/ha. Our results indicate a high flora diversity in cocoa-based agroforests especially with respect to fruit trees where *Tetracarpidium conophorum* is introduced. The total amount of CO_2/ha per village plots is estimated at 337.46, 202.32, and 171.71 tCO_2/ha. Then, the ecological services that should be paid according to 10 US$ per ton of carbon are evaluated at 3374.6, 2023.2, and 1717.1 US$ to the owner of the selected farms.

Keywords: carbon sequestration, REDD+, agroforestry systems, climate change, mitigation

1. Introduction

The concept of REDD+ is always limited to emissions due to deforestation and the increase of carbon sink in land use systems. In order to contribute to the fight against climate change issues in Cameroon, we should develop suitable proposals which target the Clean Development Mechanism (CDM) [1]. The particularity of these projects is to reduce the emission of CO_2.

Agroforestry can be defined as a collective name for land use systems and technologies where woody perennials (trees, shrubs, palms, bamboo, etc.) are deliberately used on the same land management units as agricultural crops [2]. Agroforestry can provide benefits as (i) linking poor households to markets for high-value fruits, (ii) balancing improved productivity with the sustainable management of natural resources, and (iii) maintaining or enhancing the supply of environmental services in agriculture and landscapes for water, soil health, carbon sequestration, and biodiversity conservation [3–4]. In our land use systems, there are non-timber forest products (NTFPs). They are considered with higher potentialities as medicinal and economic values which contribute to break the chain of poverty in rural areas [5–6]. *T. conophorum* (**Photograph 1**) is a local vine from the Euphorbiaceae family which is also called African walnut, cashew nut, conophor seeds, or conophor nuts [7–9]. This species has several properties such as cholesterol-lowering and triglyceride-lowering properties which have been reported [10]. The nutritional properties of seeds have also been fully demonstrated [11–13].

During some inventories, it was estimated that tropical forests can store more than 200 t C/ha in trees [14]. Carbon management in agroforestry systems is a new global concern to mitigate the increased concentration of greenhouse gases in the atmosphere in Congo Basin

Photograph 1. Fruits of *Tetracarpidium conophorum* assessed in Rionong village.

countries. Reforestation cover and finding low-cost methods to sequester carbon in land use systems are emerging tools. As trees grow and their biomass increases, they absorb carbon from the atmosphere and store in the plant tissues and roots. We need to contribute to the Reduction of Emissions derived from Deforestation and Degradation (REDD+). Carbon stock sink varies following the type of trees, and it has been demonstrated that diameter at breast height (DBH) and height are important factors in carbon stock sink variation [15]. Several studies were carried out on the biomass and carbon stock assessment in all ecosystems all over the world [16–20]. These aspects have been partially studied, and there is only a limited amount of work which upholds the notion of the potential of diversity and associated species with *Tetracarpidium conophorum* in agroforestry systems which can contribute to mitigate effects of climate change and improve livelihood options of local farmers. At this time when natural ecosystems are disappearing at an alarming rate, it is clearly necessary today to outline the carbon sequestration potential of agroforestry systems, so that their compensatory role in the mitigation process of climate change be made known in Cameroon.

2. Materials and methods

2.1. Study area

The study was carried out in the Mbam and Inoubou department in Cameroon (**Figure 1**) which appears to be a transitional area covered by forest and savannah. The main cultivation of species there is cocoa plantation. This area is located between 4′39 and 4′49 north and then 11′4 and 11′19 east. Altitude varies from 600 to 900 m [21]. We choose this site because it is located near Yaounde, which includes several markets around where the conophor nuts can be found. Secondly, it is classified among the main area in terms of productivity of *T. conophorum* fruit in Cameroon [13]. There are many vegetal formations that belong to Sterculiaceae (*Pterygota macrocarpa, Sterculia tragacantha, Cola gigantea, Cola altissima, Cola cordifolia, Triplochiton scleroxylon, Mansonia altissima*, etc.) and Ulmaceae (*Celtis zenkeri, Celtis tessmannii*) [22].

2.2. Data collection

2.2.1. Floristic inventories

Floristic inventories were performed in cocoa-based agroforestry plantations where the liana (*T. conophorum*) is introduced for several purposes. Data were collected in nine sample plots of 2000 m^2 in each sample village. Trees with a DBH > 5 cm were also assessed with the aim of evaluating the typology of cocoa agroforests associated with liana cultivation. In each plot, species were identified with the use of identification tool keys using various books [23–25] and various volumes of flora of Cameroon. **Figure 2** illustrates the experimental design of inventories carried out in the various cocoa agroforestry sample plots where we record the presence of *T. conophorum*.

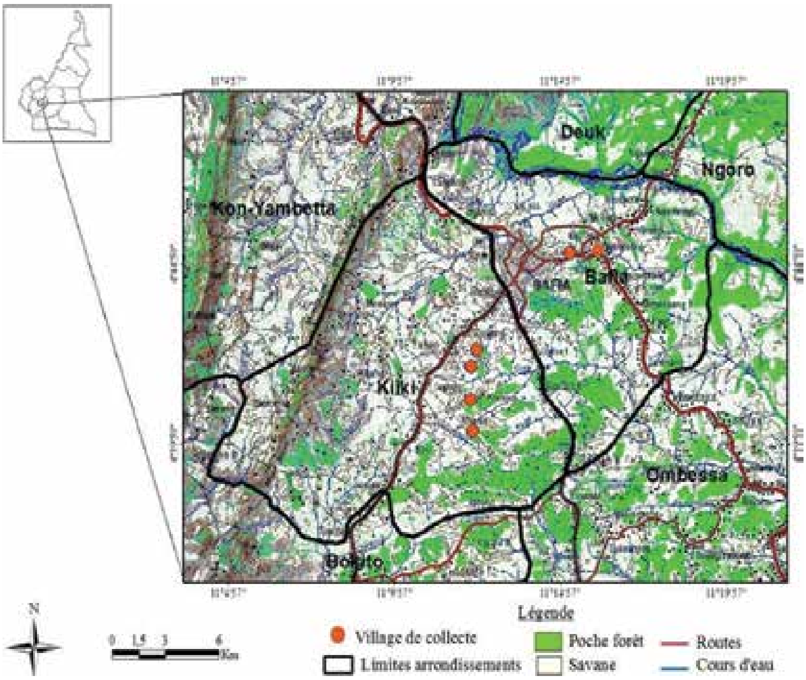

Figure 1. Localization of study site.

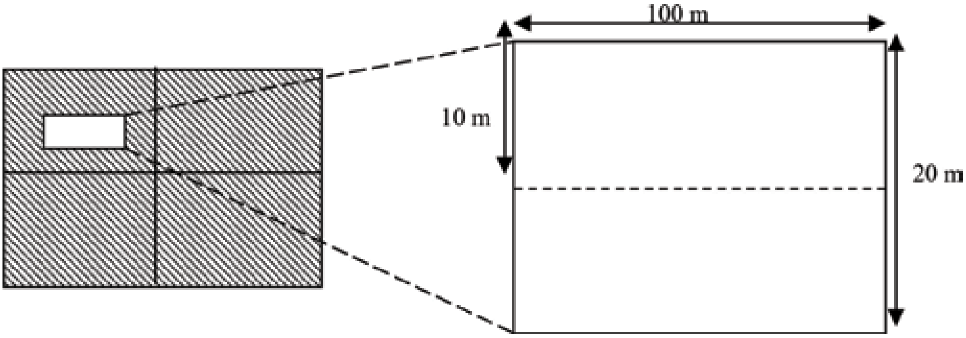

Figure 2. Experimental design of inventories carried out in cocoa based agroforests where *T. conophorum* is introduced.

2.2.2. Aboveground biomass (AGB)

AGB was performed using the methodology described by measuring carbon stock manual [26]. This method illustrated by **Figure 3** consists in delimiting a main plot (2000 m^2) where all trees with a dbh ≥ 30 cm have been recorded. Another subplot of 40 × 5 m was designed in order to assess trees (30 < dbh < 5 cm). In the subplot 40 × 5 m, other plots of 1 × 1 m were designed in order to collect litter and herbs of understory.

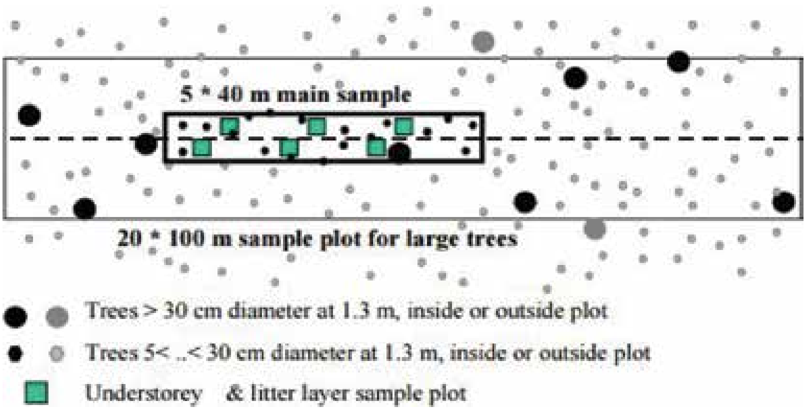

Figure 3. Details of sampling methods according to Hairiah et al. [26].

2.3. Data analysis

2.3.1. Floristic inventories

According to the following formula, we calculated structural parameters and determined the dominant species importance value index (IVI).

- Frequency was estimated for each species with the formula F = (number of plot containing × species/total number of plot) × 100.
- Abundance was estimated for each species using the formula A = (ni/N) × 100, where "ni" is the number of individuals of species i and "N" is the total of the flora.
- The diversity was calculated using the Shannon index to compare the data of various sample sites in terms of diversity of plant species. ISH = −Σ pi $\log^2$ (pi), where "pi" is the frequency of species i (ni/N), "ni" is the number of individuals of species i, and "N" is the number of individuals of all species.
- IVI = frequency + abundance + dominance [27].

Diversity parameters were also calculated according to the following formula:

- Shannon index (H′) measures uncertainty about species belonging to a randomly selected individual in the sample. It is expressed according to the proportions of each species: The formula is H′ = −Σ = pi log2 (pi) with "pi" proportion of the species "i".
- Simpson index (D′) is a measure of dominance and expresses the probability that two individuals drawn at random from an infinite population belong to the same species. It is expressed from the "pi" frequencies of species where $D' = \Sigma pi^2$. The value 0 of this index indicated a maximum diversity, while value 1 represents the minimum diversity.

2.3.2. Aboveground biomass (AGB)

The methodology used in this study was described in the document [1, 28] in cocoa agroforestry systems. Biomass was estimated using allometric model. We used allometric model of Chave [29] to evaluate carbon stock sink sequestration by each tree species. The formula is $AGB = Exp (-2.977 + LN (DBH^2H\varrho))$ where "ϱ" is the wood density and "DBH" is the diameter at breast height. We, respectively, use the density of all those trees which correspond to each plant species according to wood density database of the Food and Agricultural Organization (FAO). The total carbon stock sink in the selected trees was estimated by summing the values at the level of timbers, herbs, and litters.

2.3.3. Ecological services

To determine ecosystem services derived from the use of species in agroforests, we used ratio CO_2/C (44/12) molecular weight to convert carbon stocks (tC/ha) into tCO_2/ha and, thus, the total CO_2 sequestrated in the farmer's agroforestry demonstration plot [1, 28]. The transaction price for conservation was estimated at 10 US$/$tCO_2$; we used this ratio to estimate the ecological service value derived from the utilization of those land use systems [30].

3. Results

3.1. Floristic composition of cocoa agroforestry systems with *T. conophorum* as an additional plant species for several purposes

The results revealed during our inventories that in different cocoa-based agroforests in the Mbam and Inoubou a total of 230 plant species were recorded belonging to 16 families. These species was distributed as follow: Mouko (86 individuals representing 36% of the total number of individuals), Rionong (53 individuals representing 22% of the total number of individuals), and Nyamsong 3 (99 individuals representing 42% of the total number of individuals). **Figure 4** represents that among the plant species families inventoried in the cocoa-based agroforestry plantations, the six main dominant families are Sterculiaceae, Burseraceae, Moraceae, Anacardiaceae, Bombacaceae, and Araliaceae.

Inventoried cocoa-based agroforestry sampling plots allow to represent relative abundance, relative dominance, relative frequency, and importance value index of species which enable us to characterize farmer's cocoa agroforestry systems in the Mbam and Inoubou as illustrated by **Table 1**. As part of the results, we obtain a Shannon index estimated at 0.62, 0.66, and 0.68, respectively, while Simpson index was 1.421, 1.409, and 0.349, respectively, for Mouko, Rionong, and Nyamsong 3.

3.2. Species plant diversification in cocoa agroforestry systems

During our inventories, we recorded a total number of 230 plant species in sampling plots where *T. conophorum* was present. With these results, it appears that 53 fruiting species

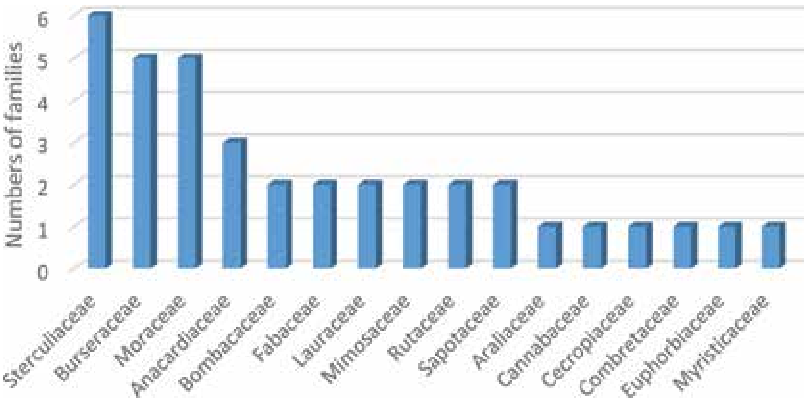

Figure 4. Numbers of family plants species inventoried in the sampled villages in the Mbam and Inoubou.

Scientific names	RA	DR/Ha	RF	IVI
Albizia zygia	4.80	4.80	8.33	17.94
Canarium schweinfurthii	3.77	3.77	5.88	13.42
Ceiba pentandra	2.32	2.32	3.63	8.28
Celtis tessmannii	1.16	1.16	1.81	4.14
Citrus sp	3.77	3.77	5.88	13.42
Cola acuminata	1.92	1.92	3.33	7.17
Dacryodes edulis	5.81	5.81	9.09	20.71
Erythrophleum suaveolens	2.32	2.32	3.63	8.28
Ficus exasperata	3.84	3.84	6.66	14.35
Gambeya lacourtiana	2.32	2.32	3.63	8.28
Mangifera indica	12.79	12.79	20,00	45.58
Milicia excelsa	2.32	2.32	3.63	8.28
Persea americana	3.84	3.84	6.66	14.35

Notes: RA, relative abundance; RD= relative dominance/hectare; RF, relative frequency; IVI, importance value index.

Table 1. values of abundance, dominance, relative frequency and IVI of various plants species characterizing cocoa agroforestry systems in the 3 villages.

represent 23% of the total number of individuals, while 149 cocoa trees represent 65% of the total number of individuals. And the rest of 28 other trees such as medicinal plants, highly marketable trees represent 12% of the total number of individuals as detailed in **Table 2**. From that table, it can be concluded that Nyamsong 3 village has the highest fruit tree rate (40%) compared to Mouko (36%) and Rionong (24%). Floristic inventories reveal that *Mangifera indica* is the most important species according to the dominant species importance value index (IVI). Classes of diameters obtained allow us to appreciate the behavior

Villages	Mouko	Rionong	Nyamsong 3	Total
Fruits trees	19	13	21	53
	36%	24%	40%	
Cocoa trees	55	34	60	149
	37%	23%	40%	
Others trees	4	6	18	28
	14%	22%	64%	
Total	78	53	99	230

Table 2. Diversity of plants species in cocoa based agroforestry plantations in order to diversify farmer's income.

of the vegetation and the most dominant species. But we record that trees with a diameter (9–24 cm) are highly represented in the sampled area. Producers deliberately introduced fruiting species (*Dacryodes edulis*, *Persea americana*, etc.) in order to diversify their cocoa agroforests where *T. conophorum* stems are also introduced. In addition, 69–84 cm class represents trees with higher diameters, and they are present in the three various villages. This class of trees is made up of species such as *Mangifera indica* and *Dacryodes edulis* which appears as to be fruiting species used by producers in order to support the heavy weight of the liana.

3.3. Aboveground biomass of sequestrated agroforestry farmer's plantations with *T. conophorum* as an associate plant species

Results indicated that cocoa agroforestry plantations found in selected villages during sampling permit to assess aboveground biomass through various plant species. A diversity of plants were found, and they can help to estimate the quantity of carbon sink that each tree can store to contribute to fight against the effects of climate change in Congo Basin countries especially in Cameroon. The total carbon sequestered by those sampling plots in recorded villages was estimated at 92.03, 55.18, and 46.83 tC/ha, respectively, at Mouko, Rionong, and Nyamsong 3.

3.4. Ecological services derived from the utilization of trees in cocoa agroforestry systems by farmers

Ecological services are those derived from the use and utilization of plant species in a land use system. From our results, those services can be estimated as follows. The total amount of CO_2/ha per village plots was estimated at 337.46, 202.32, and 171.71 tCO_2/ha, respectively, for Mouko, Rionong, and Nyamsong 3. These results could be a significant importance for targeting the reduction of effects of climate modifications in Cameroon. Then, the ecological services which should be paid according to 10 US$ per ton of carbon was evaluated at 3374.6 $, 2023.2 $, and 1717.1 $ to the owner of the farms in the Mbam and Inoubou department.

Several ecological services can be derived from the utilization of trees in cocoa-based agroforestry systems. By including trees in agricultural systems, agroforestry can increase the amount of carbon stored in lands devoted to agriculture. We can have several services notably: provision services (diverse products as food, timber, and welfare of the household), regulation services (climatic variation moderation, carbon stock assessment), and support services (biodiversity conservation, soil fertility). Agroforestry can also have an indirect effect on carbon sequestration when it helps decrease pressure on natural forests, which are the largest sink of terrestrial carbon. Another indirect avenue of carbon sequestration is through the use of agroforestry technologies for soil conservation, which could enhance carbon storage in trees and soils.

4. Discussion

4.1. Plant species diversity importance

Floristic inventories revealed the presence of several multipurpose fruiting tree species associated with the cultivation of *T. conophorum* in the Mbam and Inoubou cocoa-based agroforestry plantations. Our floristic inventories reveal the presence of 230 plant species belonging to 16 botanic families. We equally assess an important number of fruiting species which are associated at the cultivation of African walnut. However, these data are different by the ones [31–32]. The results from the previous authors assess a diversity of 116 and 206 species, respectively, in cocoa agroforestry systems in Cameroon. Our results are different from the previous authors because producers intensively introduce fruiting species in their plantations for several ecosystem services. Carbon stock, soil fertilization, and reconstitution are the services provided by cocoa agroforestry systems that local producers can benefit from the integration of local fruiting species. From our results, we can say that cocoa agroforestry plantations have an important diversity in terms of fruiting and associated species. Nevertheless, our findings revealed the presence of the most three important families such as Sterculiaceae, Burseraceae, and Moraceae which are different from the other results [32–34] in the same agroecological area and in the center region in cocoa agroforestry plantations. This observation can be justified by the choice of species which are introduced by the producer at the moment of selection.

Concerning plant species composition, diversity indices calculated revealed that cocoa-based agroforests are less diversified. We obtain a Shannon index of 0.62, 0.66, and 0.68, while the Simpson index as 1.421, 1.409, and 0.349, respectively, for Mouko, Rionong, and Nyamsong 3. These results are different by the ones [35–37] who obtained values such as 4.39 and 4.63 for Shannon index. The Simpson index reveals the way species are distributed or dispatched within the different sites/cocoa agroforest sample sites. And, the results indicated that these agroforests are quite little bit diversified.

4.2. Trees associated with *T. conophorum* in cocoa agroforests

According to inventories and frequency of associated species, we noted that *Mangifera indica* was the most used fruiting species in cocoa-based agroforestry plantations in association with *T. conophorum* in the Mbam and Inoubou. Fruiting species are added to cocoa plantations according to the needs and preferences of local producers and also his desire to diversify his home garden or cocoa farm for more productivity. Another reason for introducing species is to diversify the source of income in order to improve livelihood options. On the other hand, *Mangifera indica* tree species are specially used by local producers as tutors for the liana (**Photograph 2**) because when growing it has a big stem and needs to be supported by a big tree in order to grow well and provide shade for cocoa plantations.

The high percentage of fruiting tree species and useful species in cocoa agroforests and their increased abundance in the more intensely used landscape in the world reflect the fact that farmers intentionally introduce useful tree species in their environment. In Cameroon, one household lives with less than 1US$ per day, and that is why the presence of fruiting species in cocoa agroforests helps farmers achieve their basic needs of food, health, energy, and housing. Results published [32, 38] demonstrated that trees with edible products were the main common tree species found in cocoa agroforests. All are considered as agroforestry tree products (AFTPs) because they are derived from agroforestry trees in the same piece of land. We can have *Dacryodes edulis*, *Mangifera indica*, *Citrus spp.*, *Theobroma cocoa*, *Allanblackia floribunda*, etc.

Photograph 2. One of the biggest stems of *Tetracarpidium conophorum* found in Rionong village in cocoa agroforests.

4.3. Environmental services and carbon sinks

For this work, the estimated percentage of carbon stock sink sequestrated per locality varies as 92.03, 55.18, and 46.83 tC/ha, respectively, to Mouko, Rionong, and Nyamsong 3. Those values represent less than one-fourths of the biomass estimated in some selected agroforestry systems around natural forest stands in the Dja biosphere reserve in the East province of Cameroon. We can say that density and number of species can greatly impact the quantity of carbon sink in some land uses systems (LUS). Diversification is an important tool which contributes to mitigate effects of climate change and climate modifications [39, 34]. Considering the fact that each plant/tree specie can store a specific quantity of carbon through leaves and roots, it is therefore recommended to plant more species in our LUS in order to gather more carbon. Our results obtained on the carbon sink potential are important like the others conducted in cocoa agroforestry systems around the world. Results obtained in our study are superior in Mouko village (92.03 tC/ha), while different from the results found in [40] where the authors revealed an amount of 68.12 tC/ha at Kédia and 76.99 tC/ha at Ediolomo. These results demonstrated that carbon stock sink in cocoa-based agroforestry plantations is higher when there is more fruiting species and other useful tree species for the local producer. It can be explained by the fact that carbon sink depends on several parameters such as the quality and quantity of species which are introduced for diversification or shade purposes. Following this, a study was conducted on carbon stocks [34], and it reveals a higher amount of carbon sink (197.5 tC/ha) in the cocoa agroforests of Bokito in the same geographical area with the sampled localities. This amount of carbon is higher because during the creation of cocoa farms and local producers/farmers introduce several species (fruiting species and medicinal, etc.) in order to provide many services in the farm and then conserve more land for land restoration and protection. The presence of several species in our LUS increases the total quantity of CO_2, which appears to be very high, thus explaining an important ecological and ecosystem service provided by those cocoa agroforests to local farmers and land owners.

5. Conclusion

This study was carried out in order to assess plant diversity and carbon stock assessment by some selected cocoa-based agroforests in the Mbam and Inoubou department of Cameroon. Then, the interest in this work was to show that some tree species introduced in cocoa-based agroforests can store an important quantity of carbon which will contribute to fight, reduce climate change modifications, and diversify farmer's income during the year in order to improve livelihood options. We also estimate ecological services which can be derived from the utilization of those land use systems depending on the trees which have been integrated or associated according to farmer's preferences and needs. We have shown that percentage of fruiting species within the villages was very high, and this demonstrates that cocoa-based agroforestry plantations sometimes need associated trees in order to provide shade for cocoa development. Moreover, we can recommend that cocoa agroforestry systems can also play an important key role in biodiversity conservation. Also, different species were found to be more or less as reported by other studies using the same methodologies, and we can note that characterization allows us to demonstrate the various family species of trees associated with cocoa agroforestry systems in the Mbam and Inoubou department.

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Chapter 4

Effects of *Eucalyptus* and *Pinus* Forest Management on Soil Organic Carbon in Brazilian Wooded-Savanna

Leda Lorenzo Montero and Welington Delitti

Additional information is available at the end of the chapter

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Abstract

Forestry has been recommended for carbon cycle management since it promotes carbon accumulation in soils and vegetation. Soil organic carbon (SOC) is fundamental to fertility and crop production in tropical soils and its conservation is critical to sustainable land management of neotropical savannas. Thirty to forty years of *Eucalyptus* and *Pinus* forestry in original Brazilian wooded-savanna affected forest floor layers, SOC and organic matter (OM) quality. *Eucalyptus* and *Pinus* showed higher forest floor carbon stocks than natural forest plots. On the surface soil layer, plantation effects on SOC were mediated by site-dependent factors. Below 10 cm, both plantations showed lower SOC than the native forest. The relationship between carbon and clay contents was significant in subsurface soil layers, suggesting that the particulate OM pool had been depleted by plantation activities. Plantations lead to soil OM replacement to a depth of 5 cm within 30 years. The new litter and OM in the plantations had lower quality (higher C:N ratios) than in natural forests. Our results indicate that particular care must be taken when choosing forest management practices in tropical-weathered soils because they can oxidize a significant part of the SOC pool with negative consequences to soil fertility and aggregate stability.

Keywords: carbon stocks, organic matter, litter, forest floor, plantation, afforestation, forestry, silviculture, subtropical forest, *Cerrado*

1. Introduction

Forest management is an available option for climatic change mitigation through carbon cycle management [1–3]. In that context, soil carbon is decisive in the long term [2] because soils contain two to three times the amount of carbon in vegetation [4] in chemical forms that are much more stable than in biomass, with residence times extending from decades to millions of years [5] (**Figure 1**). Soil organic carbon (SOC) is defined by climate, soil type, plant cover, decomposer activity, perturbations and management [6]. Vegetation affects SOC because, through root and leaf production, it determines soil organic input quantity and quality, which are major decomposition control factors [7]. Soil type affects SOC decomposition and stabilization through drainage, structure, texture [8], the presence and type of clays, sesquioxides and other stabilizers [9]. Management practices affect disruption and aggregation of organic matter (OM), and thus, their influence on SOC is determinant [10] (**Figure 1**).

SOC is fundamental to fertility and crop production in tropical soils. In tropical savannas, most soils have a predominance of highly weathered clays; thus, they are acidic, low-fertility soils characterized by a low cation exchange capacity, low base saturation and high Al toxicity. Because the mineral fraction is dominated by low-activity clays, SOC is especially important in these soils, where OM is the main nutrient source for plants, soil fauna and microorganisms [11, 12]. SOC also plays a fundamental role in soil aggregation, and thus, it is essential for water supply and soil structure maintenance [13]. Hence, the conservation of SOC is critical to sustainable land management of neotropical savannas [14].

The Brazilian wooded-savanna's ecological complex (*Cerrado* biome) is important to national agricultural production, and it represents the national forestry core. Brazilian forestry generates approximately 4% of the total gross national product (www.ibge.gov.br). However, there are international concerns about the environmental impacts of exotic, fast-growing tree species on tropical soils [15, 16]. A significant part of the SOC pool is extremely sensitive to soil management or disturbance, and it can be easily lost with negative consequences to soil fertility and aggregate stability [14]. SOC losses can occur under intensive forest management, driven by enhanced oxidation due to soil preparation activities [17].

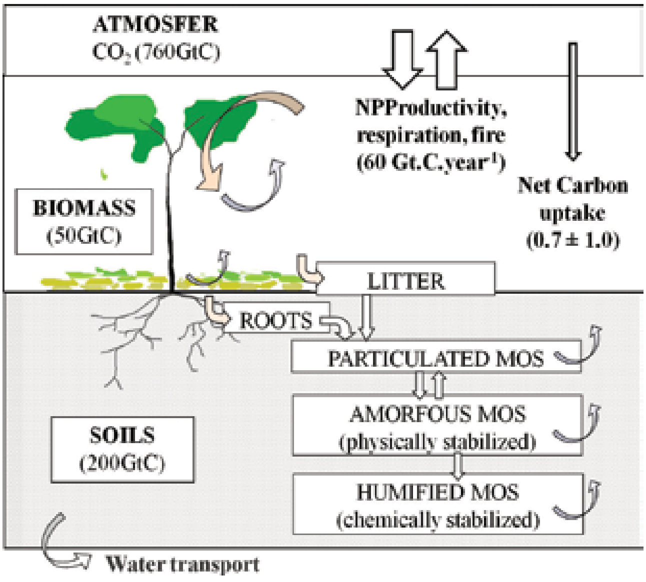

Figure 1. A general model of carbon in terrestrial ecosystems. Stocks are reported in Gt (10^{15}g) and fluxes in Gt year^{-1}. Note that arrows are not proportional to values. Values from [1].

Despite the importance of SOC to forest sustainable management in neotropical savannas, literature on the subject is scarce, and it mostly remains in thesis and regional papers where it is difficult to be consulted. Furthermore, available research papers conclude either that there is no effect or that contradictory effects are shown [18], as it has been recently appointed in a review of intensive logging effects on SOC [19]. The effects of *Eucalyptus* and *Pinus* forest management in SOC are not completely understood because the literature often reports contradictory results and conclusions. Although some authors have found significant SOC loss under *Eucalyptus* plantations compared to *Cerrado* [17], others found no significant change [20] or even increases in SOC [21, 22]. The same discrepancy is reported for *Pinus* stands, with depletion [14, 17, 21, 23], no change [20, 24] and increases [24], all being reported. These contradictory results can be explained by different experimental site conditions, such as soil type, stand age and tillage [17], as well as methodological differences, such as soil sampling depth.

Given the wide occurrence of forest plantations in tropical soils and the importance of SOC conservation to soil fertility, crop production and sustainable land management of neotropical savannas, we aimed to assess alterations in SOC content produced after 30 years of intensive management of *Eucalyptus* and *Pinus* on originally Brazilian wooded-savanna–covered areas. For this purpose, we have compared litter and soil C and N levels in planted and natural forests. To clarify previously reported contradictory results, we replicated our experimental work in four distant locations with the same experimental conditions.

2. Material and methods

2.1. Site description

We collected samples at four locations in the State of Sao Paulo, SE Brazil. They are approximately 100 km apart, located in the Luiz Antônio (21°61′S;47°75′W) (LZ), Mogi Guaçú (22°24′S;47°15′W) (MG), Pederneiras (22°34′S;48°89′W) (PD) and Itirapina (22°19′S;47°94′W) (IT) districts. At each location, we studied *Eucalyptus* and *Pinus* stands of similar ages as well as the nearest natural forest patches, which were considered to be the control areas. Thus, we defined 12 plots based on location and forest type (**Figure 2**).

Climate is characterized as tropical type II [25]. Climatic conditions are homogeneous among sites, with temperatures ranging between 19 and 22°C (mean annual T = 20°C) and with an annual mean precipitation of approximately 1200 mm year^{-1}. Rainfall is concentrated from October to March, and thus, there is a dry winter season between June and September when water demand exceeds water availability and soil moisture limits plant growth, biomass production, SOC decomposition and other ecosystem process.

Sites belong to the *Cerrado* biome, which is characterized by flat to gently undulating topography, with laterization as the main soil formation process. The studied soils were characterized as oxisols (*Humic Rhodic Haplustox*) at LZ, Inceptisols (*Oxic Dystrudepts*) at MG and PD and entisols (*Ustic Quartzipsamments*) at IT [26]. They represent a texture gradient, with average clay contents of 59% in LZ, 25% in MG, 15% in PD and 11% in IT. Altitudes range between 516 and 740 m. All sites are flat or have gentle slopes <7% and no signs of erosion were present.

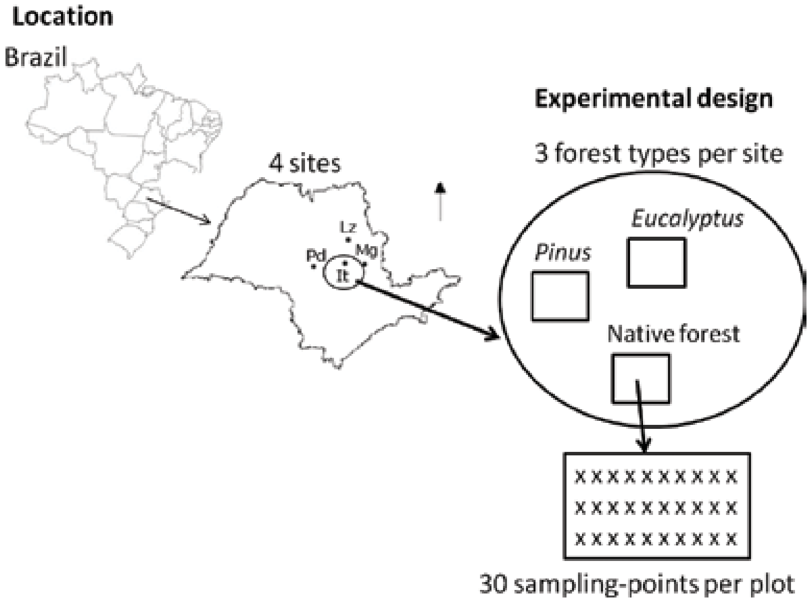

Figure 2. Location of the study area and experimental design.

The original *Cerrado* landscape consisted of grassland, savanna and wooded-savannas and dry seasonal forest patches. Control areas were located in remnants of Brazilian wooded-savanna (*cerradão*). They are characterized by closed canopies approximately 20 m tall. Leguminosae, Myrtaceae, Melastomataceae and Rubiaceae are the most represented families. There are no clear species dominance patterns, but some common species well-represented in the four sites are as follows: *Anadenanthera peregrina* var. *falcata* (Benth.) Altschul, *Qualea grandiflora* Mart., *Aspidosperma tomentosum* Mart., *Qualea multiflora* Mart. and *Roupala montana* Aubl.

All study stands are older than 30 years. They were planted in 1962, 1965, 1966, 1969 and 1972. Natural vegetation was first cleared, and after slash and burn of the original forest, a heavy disk plow was used to open seedling lines (~20 cm depth). Trees were planted manually. The used species were *Pinus elliottii* and *Eucalyptus citriodora* with the exception of the Itirapina site where *E. saligna* was the planted *Eucalyptus* species. No fertilizer or lime had ever been used at the sites, as they were planted for experimental purposes and for wood, not pulp, production. After 7–8 years of growth, 35% of trees are normally clear cut, which permits tree diameter to increase. The studied stands were clear cut about five times before the experimental work was performed. The length of rotation varied between 5 and 8 years according to market oscillations in the wood price, rather than planted species; therefore, there are slight differences on rotation times between sites, but they are not dependent on the planted species. At sampling time, the stands had closed canopies approximately 30 m tall.

One hundred trees per plot were measured for characterization of the vegetation. Mean basal area values ranged from 21 to 44 m^2 ha^{-1} in the planted stands and from 23 to 43 m^{-2} ha^{-1} in the native forests. Vegetation densities (299–786 trees ha^{-1} in plantations and 1200–1600 trees ha^{-1} in native forests) and diameter distributions (70% of trees were smaller than 10-cm diameter breast height in native forest) showed structural differences between natural and planted areas. Neither planted nor natural sites had been burnt in the last three decades.

2.2. Experimental design

Thirty sampling points were randomly selected at each plot (forest type × site) for soil and litter collection. The procedure was repeated four times at different locations in order to guarantee real independence of the observations (**Figure 2**). Our experimental work was then carried out following a randomized block design with sampling replication within the blocks [27], as we collected 30 samples inside each of the three forest type treatments (*Cerrado, Eucalyptus* and *Pinus*), and we replicated this complete design in four study locations (LZ, MG, PD and IT) corresponding to the four blocks. Sampling replication within the blocks permits us to test the block-treatment interaction. It is especially important when differences between blocks may be strong [27], as could be the present case.

2.3. Sampling and laboratory analysis

Collection was performed in 2004 at the end of the dry season during the maximum litter accumulation period. Forest floor samples were collected using a 25-cm^2 metal frame. All materials were collected, including not only litter but also fibric and humic horizons when present. Samples were oven dried to constant weight and ground for chemical analysis. C and N were determined by wet combustion [28] on 360 samples.

Soil samples were collected at three depths: 0–5, 10–25 and 35–50 cm. Pits were open to profile description and undisturbed soil sample collection. Undisturbed soil samples were collected using 5-cm diameter metal cores at four random replicates per plot. Then, soil bulk density was calculated as the oven-dry sample mass divided by the sampled volume for a total of 144 samples. Disturbed samples were collected with an auger at 30 sampling points per plot. After collection, soil samples were air dried and individually sieved through 2-mm mesh. SOC was determined by the Walkley & Black wet combustion method [29] following a tropical soil-adapted protocol [28] for a total of 600 samples. Since the studied soils are free of stones and gravel, corrections for those fractions were unnecessary. Texture was determined by the pipette method [28]. Sand (<2 mm to 64 μm), silt (<64 μm to 2 μm) and clay (<2 μm) fractions were determined for 216 samples.

Soil carbon stocks (Mg ha^{-1}) were calculated using bulk density (g cm^{-3}) and carbon content data (g kg^{-1}). Because we lack continuous sampling data, we used pedotransfer functions to estimate soil carbon stocks into the soil profile. Those functions, which related carbon content with soil depth or texture, can precisely calculate soil carbon stocks [30]. SOC exponentially decays with depth; therefore, most pedotransfer functions are based on the exponential model equation [22, 31, 32]. We used its more general form, which is:

$$y = \mathrm{a}\exp(-\mathrm{b}x), \tag{1}$$

where y is carbon (kg m^{-2}) and x is depth (cm). First, the exponential model parameters (a and b) were calculated using field data. Then, the model was integrated between the desired depths (x_1 and x_2, in cm), resulting in the function:

$$SOC_{est} = -a/b\left(\exp(-b\,x_1) - \exp(-b\,x_2)\right), \quad (2)$$

which estimates the SOC stock. We estimated the 0–30-cm depth carbon stocks following IPCC protocols [1–3].

Because the Kjeldahl acid digestion method loses accuracy when analyzing acidic, N-poor soils, we used a CN analyzer (Leco CN-2000) to determine soil C and N by dry combustion and gas chromatographic separation of 36 soil surface samples to obtain the A horizon C:N values. C:N ratios were then calculated for 720 samples; 36 samples from the soil surface (A horizon, 0–5 cm depth) and 360 from the forest floor.

2.4. Statistical analysis

We used general linear models to test the SOC content (g kg^{-1}) and C:N ratio responses to the three forest treatments (*Cerrado* considered as control, *Eucalyptus* and *Pinus*), which were considered as fixed factors, on four blocks (LZ, MG, PD and IT), taken as random factors. We used the AIC criteria to evaluate the interaction term significance [33]. Linear regression analysis was used to test the clay-SOC relationship. Data analysis and graphs were performed with R language [34].

3. Results and discussion

3.1. Forest floor

Forest floor layers under the native *Cerrado* forest consisted of fresh litter (Oi horizon) with discontinuous points of humified material at waterlogged spots. Similar conditions were found in *Eucalyptus citriodora* stands. We observed a thick organic layer in the *Eucalyptus saligna* stand, which was formed from fresh litter (Oi), fragmented debris (Oe) and a dark color–layer containing decomposed organic materials (Oa). Despite the sharpness of the transition between the organic and the A mineral horizon, this forest floor type can be considered a moder horizon because of its structure and morphological characteristics [35]. We also found clear forest floor layering under *Pinus elliottii*. Recognizable horizons were formed by freshly fallen (Oi horizon) and fragmented needles (Oe horizon). Humified material was scarce, and the transition between organic and A mineral horizons sharp; thus, these horizons are considered mor type according to Ponge's criteria [35]. Forest floor layer morphology differences between planted stands and natural forests could be explained on the basis of litter quality and soil fauna activities. High litter quality and activity of soil fauna incorporating SOC into the mineral soil lead to less organic material accumulation and most likely to faster nutrient cycling in the native forest, whereas in planted stands, lixiviation may be the principal cause of SOC incorporation into the surface A horizon, without almost any soil fauna intervention.

Forest floor organic carbon stocks were one to two times larger under the *Pinus* and *Eucalyptus* stands than under the native *Cerrado* forest (**Table 1**). Litter collected at the planted stands had higher carbon concentration and C:N ratios, leading to higher organic material accumulation

Treatment		**Litter**[a]				**Soil**[b]
		Mass (Mg ha^{-1})	**OC (g kg^{-1})**	**OC (Mg ha^{-1})**	**C:N**	**C:N**
Ce	LZ	8.194 (2.405)	416.1 (28.7)	3.523 (1.04)	33 (3.91)	15 (0.27)
Eu	LZ	10.014 (3.129)	440.2 (11.4)	4.340 (1.39)	84 (12.93)	21 (1.13)
Pi	LZ	10.322 (4.023)	453.0 (14.2)	4.697 (1.84)	66 11.96)	21 (0.63)
Ce	MG	8.025 (2.699)	429.8 (15.6)	3.366 (1.20)	43 (6.23)	15 (0.60)
Eu	MG	5.411 (1.664)	438.5 (13.7)	2.347 (0.75)	61 4.40)	18 (0.70)
Pi	MG	8.666 (3.855)	439.2 (25.2)	3.948 (1.78)	65 (9.20)	23 (1.07)
Ce	PD	8.146 (2.143)	399.8 (26.7)	3.158 (0.94)	33 (3.81)	17 (1.50)
Eu	PD	11.583 (3.183)	428.7 (30.9)	5.010 (1.48)	71 (12.08)	18 (0.35)
Pi	PD	11.862 (4.027)	450.9 (11.1)	5.318 (1.84)	94 (12.57)	21 (3.20)
Ce	IT	10.706 (2.981)	439.2 (15.3)	4.731 (1.32)	36 (4.93)	18 (1.61)
Eu*	IT	19.160 (3.334)	474.9 (2.8)	9.049 (1.58)	68 (4.78)	26 (0.83)
Pi	IT	16.328 (4.298)	420.0 (39.6)	6.339 (1.86)	64 (12.64)	21 (1.18)

Average litter dry weight mass (Mg ha^{-1}), litter organic carbon (OC) concentration (g kg^{-1}) and stocks (Mg ha^{-1}) and litter and soil A horizon (0–5 cm) C:N.

[a] Mean values of 30 samples; standard deviations are in brackets.

[b] Mean values of 3 samples; standard deviations are in brackets.

Table 1. Forest floor parameters under the *Cerrado* native forest (Ce), *Eucalyptus citriodora* (Eu) (**E. saligna*) and *Pinus elliottii* (Pi) stands in four locations in SE Brazil (LZ, MG, PD and IT).

than in the native forest (**Table 1**). Similar results are common in the literature: several authors reported higher forest floor accumulation [17, 36–38] and higher litter C stocks [21, 38] in *Pinus* stands than in the native *Cerrado* forest, as well as moder horizon formation [21, 37]. Higher forest floor accumulation and litter C stocks have been also reported in *Eucalyptus* stands compared to native *Cerrado* forest, although results are not as consistent as for *Pinus* [21, 22]. Less litter accumulation under *Eucalyptus* than under the *Cerrado* forest was reported; however, results were dependent on soil type because the differences were significant on loamy oxisol but not on sandy entisol [17]. We hypothesize than the observed *E. saligna* thick organic layer is due to the planted species, which is considered to be a strong forest floor accumulator in the Brazilian *Cerrado* [22] and in other areas with similar conditions in Congo [39] and South Africa [40]. However, we cannot confirm this hypothesis with our data because of the lack of replication of the *E. saligna* stand.

Forest floor accumulation depends on the input/output balance, which is controlled by litter production and decomposition [41]. It is known that *Eucalyptus* and *Pinus* stands had higher litter production than the *Cerrado* [14, 36, 37] and that this litter has lower nutritional content in our study sites [42]. These two features can explain the organic material accumulation and organic layer formation. We suggest that decomposition is strongly inhibited by *Pinus* plantations. Other authors' results support this idea; slow decomposition rates and longer residence times were reported when comparing *Pinus* with the adjacent native *Cerrado* near the study areas [36, 37].

3.2. Soil organic carbon

Our results show that soil carbon is related to forest type, soil depth and texture.

SOC distribution is heterogeneous in the soil profile. SOC fitted a lognormal distribution, as expected from the literature [43, 44]. SOC heterogeneity decreased from surface to subsurface soil layers. We found variation coefficients from 18 to 58% at the 0–5-cm soil layer and from 5 to 26% in layers below 10-cm soil depth. Standard deviations increased between 10 and 25 cm and 35–50 cm layers in four sites (*Eucalyptus* and *Pinus* stands at Luiz Antônio and *Cerrado* and *Eucalyptus* stands at Mogi Guaçú) (**Table 2**). Literature reports SOC variation coefficients from 5 to 59% in semiarid croplands with different land use intensities [45]. Other authors also report higher organic carbon variability in the surface than in deep layers in fast growing species plantations and natural *Cerrado* soils [22, 30]. High variability at the surface layer was expected and it can be explained by high spatial variation in surface organic carbon determinants such as soil fauna activities and litterfall. We found some samples with an extremely high C concentration that probably corresponds to local characteristics, such as charcoal, pieces of a termite mound or anthill and localized OM accumulation. Such samples were found especially in the Itirapina *Eucalyptus saligna* stand and the same situation has been reported under similar soil and vegetation conditions near our study site [22]. Therefore,

Treatment		**Soil organic carbon (g kg^{-1})**			**Total SOC (Mg ha^{-1})**	
		0–5 cm^{a}	**10–25 cm^{b}**	**35–50 cm^{b}**	**0–30 cm^{c}**	**R^2**
Ce	LZ	28.41 (06.12)	14.59 (1.27)	12.73 (1.04)	55.8	0.64
Eu	LZ	34.61 (06.33)	13.31 (1.27)	11.40 (1.56)	68.3	0.83
Pi	LZ	43.14 (17.01)	11.8 (0.55)	10.64 (0.78)	56.3	0.50
Ce	MG	24.67 (06.18)	13.2 (2.66)	10.93 (2.83)	56.9	0.51
Eu	MG	11.65 (02.79)	9.65 (2.14)	9.13 (2.21)	40.8	0.16
Pi	MG	35.45 (14.73)	8.90 (1.20)	7.79 (0.56)	54.3	0.66
Ce	PD	30.48 (15.31)	8.08 (1.01)	6.34 (0.70)	59.7	0.64
Eu	PD	10.83 (01.91)	6.92 (0.97)	6.10 (0.79)	39.6	0.53
Pi	PD	12.26 (03.88)	5.64 (0.73)	5.58 (0.56)	36.0	0.47
Ce	IT	36.86 (19.66)	8.43 (1.20)	6.63 (0.62)	54.4	0.53
Eu*	IT	65.99 (38.28)	7.56 (1.81)	6.40 (1.16)	85.7	0.76
Pi	IT	15.58 (06.18)	5.41 (1.16)	5.06 (0.87)	33.7	0.40

The uncertainty of the stocks estimations is expressed by the pedotransfer functions adjusted coefficients (R^2).

[a]Mean values of 30 samples; standard deviations are in brackets.

[b]Mean values of 10 samples; standard deviations are in brackets.

[c]Estimated through pedotransfer functions based on exponential model.

Table 2. Soil organic carbon (SOC) concentration (g kg^{-1}) and organic carbon stocks (Mg ha^{-1}) under the *Cerrado* native forest (Ce), *Eucalyptus citriodora* (Eu) (**Eucalyptus saligna*) and *Pinus elliottii* (Pi) stands in four study locations in SE Brazil (LZ, MG, PD and IT).

they probably represent the spatial variability of the reality. However, they must be retired to perform the tests of significance because they were three standard deviations larger than mean values and their inclusion on the data set will invalidate the analyses of variance [46].

SOC content decreased as the depth increased in the soil profile (**Table 2**); 81–78% of the SOC contained 50 cm depth was concentrated on the surface soil layer (data not shown). More SOC content on the surface than on deeper layers is due to soil organic inputs being mostly superficial. Furthermore, literature reports that SOC distribution in the soil profile is well explained by indirect exponential functions in temperate [31] and tropical forest soils [32]. In this study, pedotransfer functions based on the exponential model properly fitted the observed data, with coefficients varying between 0.40 and 0.83. Similar pedotransfer function coefficients, from 0.54 to 0.73, were reported in oxisols of the Western Brazilian Amazon [32]. These results reinforce the idea that exponential models are useful for carbon stock estimations [22, 30–32]. Even continuous sampling with volumetric cores could improve the models' fit, collection at three depths is considered appropriate to SOC stocks estimation [31] and it demands significantly less field effort. The estimated SOC stocks (**Table 2**) match the values reported for *Cerrado*, *Eucalyptus* and *Pinus* plantations growing on similar soil and climatic conditions [22, 24].

Our soil bulk density values (**Table 3**) also fit the ranges reported in other studies conducted at Brazil [17, 22, 24, 47]. We found soil bulk density increments in plantations compared with natural *Cerrado* forests at every studied depth in *Eucalyptus* stands. At *Pinus* stands,

Treatment		**Bulk density (g cm^{-3})**		
		0–5 cm[a]	**10–25 cm**[b]	**35–50 cm**[b]
Ce	LZ	0.93 (0.02)	0.85 (0.03)	0.94 (0.09)
Eu	LZ	1.20 (0.04)	1.00 (0.06)	0.96 (0.05)
Pi	LZ	0.89 (0.05)	1.01 (0.06)	0.95 (0.03)
Ce	MG	1.03 (0.09)	1.23 (0.06)	1.12 (0.09)
Eu	MG	1.27 (0.06)	1.32 (0.06)	1.30 (0.05)
Pi	MG	1.15 (0.05)	1.10 (0.13)	1.03 (0.12)
Ce	PD	1.22 (0.02)	1.40 (0.04)	1.41 (0.02)
Eu	PD	1.28 (0.03)	1.41 (0.04)	1.44 (0.03)
Pi	PD	1.28 (0.06)	1.49 (0.02)	1.40 (0.08)
Ce	IT	0.86 (0.07)	1.30 (0.11)	1.36 (0.01)
Eu*	IT	1.00 (0.16)	1.46 (0.07)	1.38 (0.05)
Pi	IT	1.15 (0.10)	1.42 (0.04)	1.42 (0.06)

Higher values in plantations suggest compaction.

[a] Mean values of 30 samples; standard deviations are in brackets.

[b] Mean values of 10 samples; standard deviations are in brackets.

Table 3. Soil bulk density (g cm^{-3}) under the *Cerrado* native forest (Ce), *Eucalyptus citriodora* (Eu) (**Eucalyptus saligna*) and *Pinus elliottii* (Pi) stands in four study locations in SE Brazil (LZ, MG, PD and IT).

we found four exceptions (the 0–5-cm depth soil layer at Luiz Antônio, the 10–25-cm and 35–50-cm depth soil layers at Mogi Guaçú and the 35–50-cm depth soil layer at Pederneiras stands) (**Table 3**). Increases in bulk density under *Eucalyptus* plantation in *Cerrado* soils are also reported in other studies [17, 22] suggesting soil compaction. Soil mass rather than soil volume is conserved [48]; thus, compaction could mislead SOC stock comparisons on a soil volume basis, because greater SOC stocks could be detected as a result of soil mass increments and not necessarily by carbon enrichment or gains into the solid phase of the soil. Significant differences on SOC accumulation between *Eucalyptus* and *Cerrado* soils working on a depth but not on a mass basis support this idea [22].

We found gains as well as losses of carbon stocks into the upper 0–30-cm depth mineral soil layer working on a soil volume basis. Under *Eucalyptus* plantation, we found carbon decreases at Mogi Guaçú and Pederneiras stands, but increases at Luiz Antônio and Itirapina stands. Under *Pinus* plantation, we found decreases of carbon on mineral soil, with a unique exception into the Luiz Antônio stand (**Table 2**). However, rather than providing clarity, traditional corrections for bulk density often obscure soil carbon comparison, because the corrected values reflect differences in both carbon content and soil mass [48]; indeed, proper corrections for bulk density require estimates of C stored in an equivalent soil mass [48], but we lack continuous sampling data to properly do those estimates. Detecting relatively small changes of large compartments may still prove difficult and it has been asserted that differences on carbon stocks under different land uses are usually small when compared with the uncertainties in their measurements [19, 49, 50]. Therefore, we decided to directly use the soil mass field data to a comparative approach between forest types, because the lack of information into the 5–10-cm and 25–35-cm soil layers is an error source that will mask slight changes in SOC.

Our SOC results fit the literature-reported values [17, 24]. As expected, forest type had significant effects on SOC concentration; however, responses to forest type cannot be described straightforwardly because they were dependent on soil depth and site (**Table 4**). There were strong SOC

Soil layer	**Factor**	**df**	**F**	**p**
0–5 cm	Site	3	26.286	***
	Forest type	2	9.642	***
	Site×forest type	6	30.395	***
10–25 cm	Site	3	125.318	***
	Forest type	2	44.318	***
	Site × forest type	6	1.968	ns
35–50 cm	Site	3	117.654	***
	Forest type	2	19.321	***
	Site × forest type	6	1.626	ns

Soil organic carbon (SOC) content (g kg^{-1}) is affected by site and forest type. The significant interaction term at 0–5-cm depth soil layer indicates that the effects of forest type on surface SOC vary between sites.

*** $p < 0.001$, ^{ns} nonsignificant at the 0.05 level.

Table 4. General linear model results.

differences according to forest type at the soil surface layer, but the net effect varied between sites, including losses, gains and even no significant changes compared to the reference native forest (**Figure 3**). These results suggest that there are site-dependent determinant factors affecting surface layer SOC. At deeper layers, we found the same pattern in all studied stands: SOC concentration decreased under plantations and SOC levels were lower in the *Pinus* than in the *Eucalyptus* stands (**Figure 3**). At the *Eucalyptus* plantations, mean SOC concentration had decreased about 15% at the 10–25-cm soil layer and 10% at the 35–50-cm layer, while, under the *Pinus* plantations, it had decreased about 28% at the 35–50-cm layer and 21% at the 35–50-cm layer (**Table 5**).

Meta-analysis results support our findings; soil carbon stocks decline about 13% after natural forests to plantation conversion [51]. Nevertheless, the net effect depends on the type of planted species; broad tree plantation placed onto prior native forests or pastures did not affect the SOC stock, whereas pine plantations reduced SOC stocks about 12–15% [51]. Moreover, most of the soil carbon was lost under softwoods plantations (particularly *Pinus radiata*), while SOC accumulation was greater under deciduous hardwood or N-fixers plantations, following afforestation [52]. Although a recent review has shown consistent long-term (>300 years) decreases in SOC when primary forests are logged and harvested, it has concluded that previous results indicating lack of depletion of SOC in mineral soils subjected to harvest may have been a function of their short time frames [19]. Indeed, the effects of *Eucalyptus* afforestation on SOC get stronger with plantation age [18]. Furthermore, *Eucalyptus* effects could be related to mean annual precipitation, since SOC gains in drier sites and SOC depletions in wetter ones had been reported [18].

Our results suggest that clay content regulates SOC responses to forest management. Surface SOC concentration changes in *Eucalyptus* stands were more pronounced in sandy than in clay soils. The higher the clay content, the weaker the SOC changes in *Eucalyptus* stands at the soil surface (**Table 5**). At deeper layers, SOC could be related to clay content because it accompanies the texture gradient below a 10-cm soil depth (**Figure 3**). Similar results were reported in the same biome [17]. In tropical sandy soils, there is almost no SOC-clay adsorption; thus, the adsorption of carbon compounds released during decomposition is very weak, and SOC is more susceptible to being lost by lixiviation than in clay soils. Usually, clay soils show higher SOC content than sandy soils, and they are more difficult to modify through forest management. It can be explained by management practices, which have a greater impact on the soil OM associated with the sand fraction than on the fractions bound to clay and silt [53]. Furthermore, vegetal residues usually decompose more rapidly in sandy than in clay soils [52].

Our contrasting surface SOC results could be explained by tillage and/or soil preparation differences between sites at the initial plantation time, as no evident differences exist in climate, main soil formation processes, stand age and management practices, and differences in clay content are not related to SOC in the surface layer (see Section 3.3). Small differences in soil tillage and management practices at soil preparation time can generate significant losses of SOC, and the patterns of loss and accumulation of SOC strongly vary according to location [54]. More than 30% of the forestland and 50% of the grassland surface SOC pool variation were attributed to site variables in Ohio in the Great Lakes region of the USA [55]. Different patterns of surface SOC dynamics at each study site in *Eucalyptus* and *Pinus* stands on *Cerrado* soils were also reported in a previous study [17]. It concluded that the dynamics of soil OM in these stands depends on a set of environmental conditions and management practices that

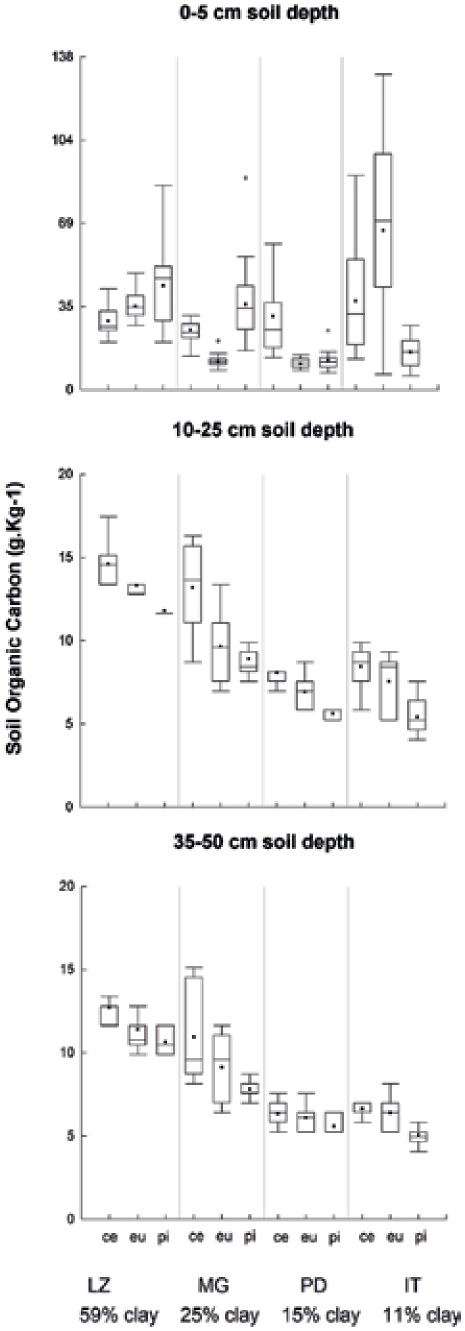

Figure 3. Soil organic carbon concentration (g kg^{-1}) as function of forest type in three soil layers at four studied locations in SE Brazil (LZ, MG, PD and IT). "Ce" depict data for *Cerrado* native forest; "eu" for *Eucalyptus* stands; "pi" for *Pinus* stands. Average soil clay content is reported for every location. Graph scale differs between surface and lower layers to facilitate interpretation. Effects of fast-growing species plantation on surface SOC depends on the site. Below 10-cm soil depth, plantations lead to SOC depletion. General linear model results are reported in **Table 4** and medium SOC changes in **Table 5**.

Treatment	**Soil layer**					
	0–5 cm				**10–25 cm**	**35–50 cm**
	LZ (59%)	**MG (25%)**	**PD (15%)**	**IT (11%)**	**All sites**	**All sites**
Eucalyptus	21.8% ^{ns}	−52.8%	−64.4%	79.0%	−15.5%	−9.8%
Pinus	51.8%	43.7%	−59.8%	−57.7%	−28.3%	−20.6%

Negative values indicate decreases and positive values increments. Mean site clay contents are reported in brackets.^{ns}Nonsignificant at the 0.05 level.

Table 5. Average soil organic carbon (SOC) concentration changes according to forest type in different soil layers in four study sites in SE Brazil (LZ, MG, PD and IT).

cannot be expected to follow an obvious and simple general tendency, particularly at surface layers where SOC is very dynamic [17]. Initial decreases in SOC after pasture to plantation conversion has been also observed and they are often attributed to site preparation [52]; however, the same study suggested that the lack of inputs into the soil (because lack of vegetation in the first years after conversion) rather than the soil disturbance during site preparation is responsible for the observed decreases.

Site preparation activities could be responsible for SOC losses at our sites. At plantation time, burning harvest residue for site preparation was the common practice and then, the SOC decreases founded in planted stands below 10-cm soil depth may be related to enhance SOC oxidation at plantation time. Activities carried out during site preparation, such as natural vegetation clearing and plowing (up to ~20-cm depth) probably lead to net SOC losses because they increase aggregates disruption and aeration as well as increase the availability of native labile organic carbon for decomposers. The particulate SOC pool, which is very sensitive to management [56], could be easily oxidized at plantation time due to soil preparation activities. The SOC pool associated with clays, which is more resistant to disruption, could be retained in the mineral soil, forming 60–70% of the actual carbon pool in the studied soils (see Section 3.3). Species richness itself could partially explain higher carbon contents in the subsoil of *Cerrado* forest than in plantations, since the carbon concentration per unit clay or fine silt in the subsoil has been found to be 30–35% higher in mixed than in monospecific stands in natural broad-leaved forests in Germany [57]. Other literature has reported SOC depletion in *Eucalyptus* and *Pinus* [14, 17, 21, 23] stands compared with the native *Cerrado* forest. Conservative practices are important at this point; for example, the adoption of minimum tillage in soil preparation in Brazil has implied significant reductions of SOC losses under plantations in the last decade [47]. Forest management is highly relevant in tropical soils where 20–40% of total SOC is related to particulate OM [8], which is the most sensitive pool to decomposition losses, and it is mainly controlled by management [56]. Not only coarse particulate SOC, but also the fine occluded fraction could have been oxidized, even in the B horizons where SOC may be less stable than is often thought [58]. Despite the fact that tropical forest plantations are often supposed to, and even designed to, sequester SOC, our study shows that significant losses still occur. However, these losses may be reduced through less intensive practices of soil preparation, such as reduced or no tillage [6, 10, 14, 17].

3.3. SOC-clay relationship

The SOC-clay equations fitted linear regression models; however, the relationship differed with soil depth. Clay content explained 62% of the SOC content at the 10–25 cm depth and 75% at the 35–50 cm depth, but clay and SOC levels were not related at the surface layer (0–5 cm depth) (**Table 6**). The SOC-clay direct relationship can be explained by SOC stabilization. Carbon is adsorbed on clay surface exchange sites, where it is protected from decomposition, lixiviation and water transport losses. Mineral fractions <20 μm are responsible for SOC physical protection because of its occlusion into microaggregates [56]. Thus, the more abundant the clay, the more protected the SOC.

In Brazilian oxisols, clay content is considered as a major controlling factor of slow SOC cycling [59] and SOC accumulation is often higher in clay than in sandy soils [17, 60]. Therefore, clay content has a strong influence on soil carbon dynamics and storage in this type of soils. Although experimental results do not always confirm the linear relationship between carbon and fine soil mineral particles [61, 62], several studies support significant relationships either with clay [8, 20] or with clay + silt [9, 30, 43, 57]. The linear relationship is related to the number of adsorption sites on the clay mineral surface per unit soil weight or volume. This linear relationship has been also reported in soils dominated by low activity clays [8]. Other authors found strong texture effects on SOC in shallow and deep soil layers but not at the surface layer [30, 57, 63]. The unclear textural effect at the surface layer found in the *Cerrado* biome has been related to the high particulate OM content in this layer, which would mask the interaction of humic carbon with the soil mineral matrix [30]. This explanation is plausible for the null relationship found in the surface layer in our study.

The SOC differences between the surface and the lower layers may be due to the surface SOC pool mostly containing labile forms that originated from particulate OM, which is composed of organic fragments up to 20–50 μm [8]. Below a 10-cm soil depth, the SOC pool may contain more stable forms of 20 μm or smaller in size that could be stabilized through clay association. However, the relationship between physical protection and chemical quality is not as simple. SOC recalcitrance decreases with depth have been found in Mediterranean forest soils [58]. The authors found that recognizable plant fragments constituting the free-light SOC fraction were not necessarily the youngest fraction and that the nondecomposed fraction of SOC presented intermediate degrees of recalcitrance [9].

Soil layer	**Linear regression analysis**		
	Adjusted model	**R^2**	**p**
0–5 cm			ns
10–25 cm	$SOC = 5.96 + 0.012\ Clay$	0.62	***
35–50 cm	$SOC = 4.74 + 0.011\ Clay$	0.75	***

The SOC-clay relationship varies with depth, being nonsignificant at the surface layer, while clay content explains 60–75% of SOC content below 10-cm soil depth.

$^{***}p < 0.001$, ^{ns} nonsignificant at the 0.05 level.

Table 6. Soil organic carbon (SOC) and clay contents (g kg^{-1}) linear regression analysis: adjusted models, regression coefficients (R^2) and significance level (p).

3.4. Organic matter quality: C:N ratios

We found higher C:N ratios in *Eucalyptus* and *Pinus* stands than in natural forests at the forest floor as well as into the surface A mineral horizon soil (**Figure 4**). Our data (**Table 1**) are in the range generally reported for tropical areas and forest [7, 64]. Commercial forest values are higher than those reported in the literature for the region [24, 37]. This discrepancy can be explained by the fertilization differences in stands managed for pulp or for timber production. *Eucalyptus* and *Pinus* stands increased mean C:N values about 1.9 times in the litter and 1.3 times in the A mineral soil (**Table 1**). OM C:N ratio increases are often related to the reduction in SOC quality, and they may have implications for nutrient cycling and fertility [64], especially in tropical soils where biotic factors have strong influences on decomposition dynamics [7].

Eucalyptus and *Pinus* are high nutrient-use efficient species [36, 37]; they produce high biomass growing in oligotrophic soils, but their leaves have low nitrogen concentrations. We know from a previous study that, at our study sites, litter nutrient content is significantly lower in planted stands than in the control native forests [42]. We suggest that this low-quality litter production leads to SOC quality degradation, which occurs gradually with litterfall increase during plantation development. New litter and the organic substances derived from decomposition are gradually incorporated into mineral soil where they will slowly replace the original SOC. The literature reports SOC replacement in *Cerrado* soils under *Eucalyptus* and *Pinus* culture [22, 38] and it reports higher soil C:N ratios in eight-year-old *Eucalyptus* and

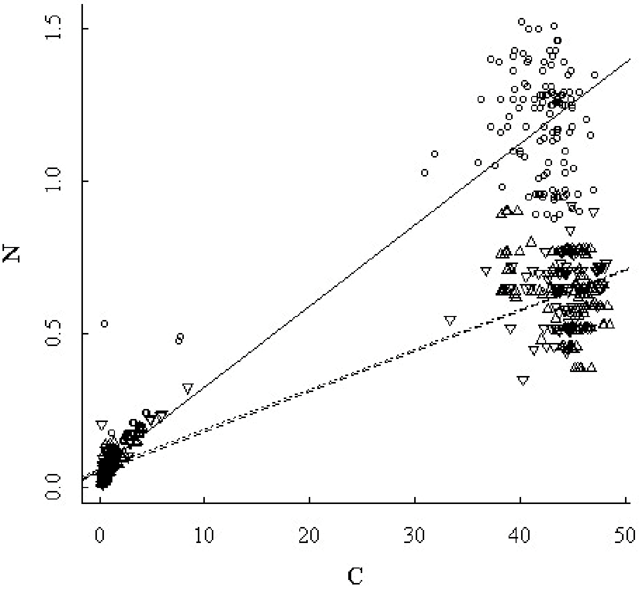

Figure 4. Organic matter C:N ratio as a function of forest type (circles and solid line depict data for *Cerrado* native Forest; triangles pointing down and dotted line for *Eucalyptus* stands; and triangles pointing up and dashed line for *Pinus* stands). Forest floor (higher C values on the right side of the plot) and upper A mineral horizons (lower C values on the left side of the plot) clearly differ in C and N contents. The increasing adjusted line slope indicates low C:N ratios. Plantations lead to significant C:N increases in the forest floor and mineral A horizons ($p < 0.001$).

Pinus stands than in the native forest in the same study area [20]. However, the reported differences were lower than ours, suggesting continuous longer term SOC replacement over time.

Our results indicate that SOC replacement occurred in 30 years in the A horizon to a depth of 5 cm in *Cerrado* soils transformed into *Eucalyptus* and *Pinus* plantations. Despite the fact that high litter C:N ratios could lead to the inhibition of decomposition [7], part of the decomposed OM is being incorporated into the mineral soil, as is shown by our topsoil C:N results. Changes in SOC C:N ratios may decrease the rates of nutrient cycling and affect the decomposer community, as indicated by diversity losses reported for various groups of soil fauna in SE Brazilian stands [65, 66].

4. Conclusions

Eucalyptus and *Pinus* plantations in the Brazilian wooded-savanna (*Cerrado*) lead to OM accumulation in the forest floor and higher litter carbon stocks, especially in *Pinus* stands, where mor humus-type was formed. *Eucalyptus* and *Pinus* litter showed high C:N ratios inducing OM accumulation compared to the native forest.

Eucalyptus and *Pinus* plantations significantly affected SOC concentration. In the surface soil layer, the effect on SOC was mediated by site-dependent factors, and a general pattern could not be identified. Gains, losses or no carbon changes with respect to the reference native *Cerrado* forest were observed at different sites under the same type of forestry. In lower layers (below a 10-cm soil depth), *Eucalyptus* and *Pinus* plantations lead to decreases in SOC concentration. In these subsoil soil layers, SOC was strongly related to clay content, suggesting that the particulate SOC pool was specifically oxidized by the soil preparation activities in plantations.

Eucalyptus and *Pinus* plantations lead to OM alterations in the forest floor and in the A mineral horizon. Plantations produced partial surface SOC replacement (down to a 5-cm soil depth) over 30 years. The new SOC has a higher C:N ratio that may cause SOC quality degradation.

According to our results, forest management practices may have strong implications for SOC pools that may offset the carbon biomass accumulation potential of plantations of fast-growing species, thus limiting their role in C sequestration and climate change mitigation. These implications are particularly critical in the case of the substitution of native forests by artificial plantations considering the possible negative consequences for biodiversity conservation.

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Chapter 5

Determinants and Tools to Evaluate the Ecological Sustainability of Using Forest Biomass as an Alternative Energy Source

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Additional information is available at the end of the chapter

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Abstract

Forest biomass, the most ancient of fuels, is again in the center of renewable energy production. This chapter provides an introductory view of the main factors that condition the ecological sustainability of this energy source. The basic concepts of ecological sustainability, ecological rotation, and ecological thresholds (among others) are presented. The state of the art on approaches to assess the sustainability of forest biomass production for heat and electricity is discussed, and tools available for decision-makers to evaluate the sustainability of forest biomass production and management are described. This chapter then describes the main advantages and drawbacks of forest certification, growth and yield tables, and ecological models in relationship to their use in sustainable forest management for biomass and energy production.

Keywords: ecological modeling, alternative energy, green energy, district heating, sustainable forest management

1. Introduction: Defining ecological sustainability

Forest biomass is a natural renewable resource with multiple uses, including energy production. In fact, forest biomass is still one of the main sources of energy for heating and cooking in many regions around the world, particularly in developing nations [1]. Indeed, even in many industrialized countries around the world, forest biomass has the potential to substitute an important share of fossil fuels [2–5]. However, biomass is just the result of ecological processes taking place in forests all around the world, in which the energy from the sun is combined with carbon dioxide from the atmosphere and water and nutrients from the soil to generate chemical energy bounded in the organic molecule product of the photosynthesis. Using forest biomass for sustainable energy production therefore requires understanding the functioning of forest ecosystems [6] and the environmental risks associated [7, 8].

Forests are complex ecological systems composed by many and very diverse elements. Topography, climate, and soil combined with animal, plant, fungi, and microorganism species to generate a unique forest in each site. The way such elements are combined is called ecosystem structure, and it encompasses both biological and nonbiological elements. As gearing parts, the ecological structure allows for the flows of sun energy (captured by plants through photosynthesis) and matter (water, nutrients, and carbon captured by plants through roots and leaves). In turn, such cycles allow animal, plant, fungi, and microorganism species to survive, reproduce, and evolve through time, colonizing new sites as they become available following natural and anthropogenic disturbances. Each of these ecological developments requires physiological processes (photosynthesis, water transport in plants, tissue growth, reproduction, etc.) or ecological processes (organic matter decomposition, seed and larvae dispersal, etc.) to run smoothly to keep the ecosystem working. In these processes, interactions among ecosystem elements (both biological and nonbiological) are determinant to produce the final amount of forest biomass that could be used to generate energy.

For example, when there is enough rain and medium to high temperatures (as in humid subtropical sites or in Mediterranean or dry subtropical areas during the rainy season), tree growth reaches its maximum rates. At the same time, soil microfauna is also in its most active moment, cutting up and processing litter and other soil organic matter in the forest soil. This facilitates the activity of fungi, bacteria, and other microorganisms that carry out the last steps of decomposing organic matter, releasing nutrients that can be used again by trees and plants to keep growing [9]. This complexity allows forest ecosystems to be resilient and recover from changes caused by disturbances.

Disturbances, defined by their intensity and frequency, cause important temporal changes in all ecosystems, causing significant fluctuations in some species populations but also providing new opportunities for species to establish and prosper. This process, called secondary succession, allows forest ecosystems to recover from disturbances by gradual changes in species composition and growing conditions that allow the recovery of the ecological community [10] and the capacity to produce woody biomass [11]. Based on this capacity to recover from change, the concept of sustainable forest management can be defined as the type of forest management that allows the recovery of the forest ecosystem to a situation similar to the one existing before the human intervention. The time needed for such recovery is called ecological rotation (**Figure 1**) [12], which is usually different from the economical or technological rotations, a fact that produces conflicting situations in forest management and planning. In fact, the term "forest biomass sustainability" is still up to discussion, and calls for international definitions and agreements are still common [13, 14].

The biomass obtained from forests should comply with the principles of sustainable forest management that are aimed at safeguarding economic, ecological, and social functions of forests and apply to all forest management activities. Sustainable forest management is the

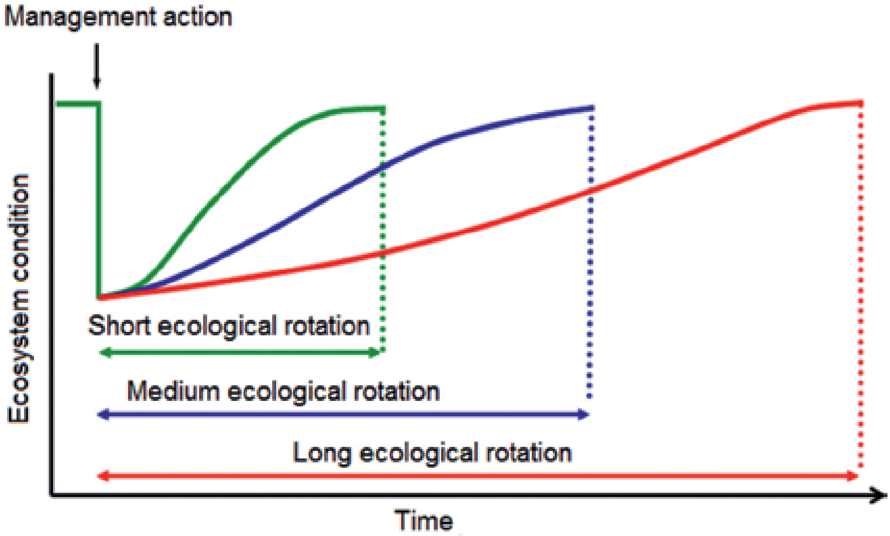


Figure 1. The concept of ecological rotation and its application to three ecosystems under the same management action but with different recovery rates.

practice of managing forests to meet the current needs and desires of society for forest resources, i.e., products, services, and values, without compromising the availability of them for future generations, as mentioned in the Brundtland Report [15]. More recently, new understanding of forests as ecosystems, along with societies' changing views of values derived from forests, has caused forest managers to adopt more comprehensive approaches to sustaining forests [16]. However, to become a useful tool, suitable indicators must be clearly defined and minimum/maximum thresholds for each indicator assigned [17].

Conceptually, forest management actions carried out by humans are not different from other natural disturbances. Both types of disturbances cause a temporal change in the ecosystem condition, which in turn can recover. Hence, from an ecological point of view, forest management actions can be defined by their intensity (how deep is the change in the ecosystem that they cause) and frequency (how often such change is imposed in the ecosystem). With just those two concepts, it is already possible to have a first glimpse of what sustainable forest management means from an ecological point of view (**Figure 2**).

Sustainability is an important concept currently at the forefront of many policy agendas. Sustainable forest management seeks to maintain or enhance ecosystem function and sustainability by emulating natural stand dynamics and disturbance regimes. Recently, interest in applying the ecological threshold concept has increased as a tool for approaching environmental problems, with the principles of sustainable and adaptive management. An ecological threshold is the point at which there is an abrupt change in an ecosystem quality, property, or phenomenon, or where small changes in an environmental driver produce large responses in the ecosystem [18]. The forest systems we deal with—ecological, economic, social, and integrated—are complex and operate by maintaining functional gradients away from equilibrium

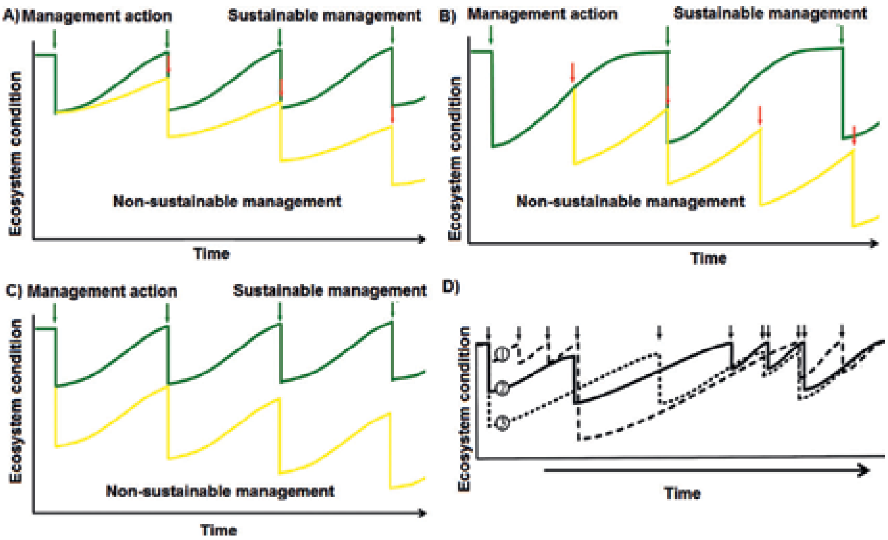


Figure 2. Three parameters can influence the sustainability of forest management: (A) different recovery rates can make a management plan sustainable in one ecosystem but not in another one; (B) human intervention frequency can be too high to allow for ecological recovery; (C) human intervention intensity can be too high to allow for ecological recovery; (D) sustainability should be framed for a given period of time and considered as the maintenance of a dynamic equilibrium of the target ecosystem condition over time.

[19]. Moreover, incorporating threshold concepts into environmental modeling, monitoring and management is a major advance in the ability to deal with ecological changes.

Ecological equilibrium should be then considered as a fluctuating condition that is bounded inside some limits. Therefore, the dynamic and long-term nature of changing ecological conditions in forest ecosystem brings a challenge when assessing if a given management plan to use forest biomass as a sustainable source of energy is sustainable. To support forest management planning and stakeholders' decision-making, several tools have been developed over the last decades, ranging from administrative processes (such as certification schemes) to high-level scientific simulators (empirical, theoretical, and hybrid ecological models).

2. Using forest biomass as an alternative energy source

2.1. State of the art

Forest biomass is a renewable, domestic fuel, which can be used for energy production. Sustainable use of forest biomass does not permanently increase the amount of carbon in the biospheric cycle, in contrast to fossil fuels, as the carbon in forest biomass comes from the atmosphere and not from the lithosphere. However, the difference between carbon sequestration rates by tree growth (commonly a slow process) and carbon releasing rates by biomass burning (a quick process) means that careful planning must be done to avoid the "carbon debt," i.e., increasing carbon in the atmosphere [6]. Current concerns regarding climate change and rising energy costs have noticeably increased the interest in the use of renewable and alternative energies. There is an increasing demand for biomass to be used for energy, and the use of forest biomass as an energy resource is growing as a result of increased energy costs and a desire to reduce the greenhouse emissions responsible for climate change. Forest biomass is currently used to generate electricity and heat, combined heat and power, liquid fuels and others.

Some industrialized countries derive a considerable amount of energy from biomass (i.e., Finland, Sweden, USA), but biomass resources are traditionally used in rural areas of developing countries where half the world's population lives and primary energy supply is provided by these forest resources [1]. Therefore, biomass already contributes a significant part of the world's energy. Currently, the major use of biomass is in the form of heat in rural and developing countries. Fuels such as firewood, dung, charcoal, peat, harvesting residues, methane, and alcohol are important sources of energy to many people in developing countries [20].

Wood is considered humankind's very first source of energy. It represents the only domestically available and affordable source of energy for cooking and heating in many households of developing countries [1, 5]. For example, the absolute number of people dependent on biomass fuels in sub-Saharan Africa will increase through 2030, suggesting that policy-makers should pay careful attention to the factors influencing supply, demand, and distribution of forest biomass fuels. This issue is particularly important in regions such as East Africa, where approximately 95% of the population use solid fuels for cooking and heating [1]. Thereby, bioenergy may represent opportunities for domestic industrial development and economic growth. In these countries, traditional biomass is often the dominant domestic fuel, especially in rural areas without access to electricity or other energy sources. There are multiple challenges and opportunities for bioenergy as a potential driver of sustainable development, given enough economic and technological support [21]. However, economic development, agricultural activity, and population pressure remain important drivers of deforestation at developing countries. Recent studies have highlighted trade of agricultural products as a potential driver of deforestation [22], pointing to the need for well-designed plans for reforestation and sustainable forest management.

Wood energy has entered into a phase of high importance and visibility with climate change and energy security concerns. There is a wide variety of biomass sources for bioenergy production: short-rotation woody crops of trees grown specifically for bioenergy, forest residues or woody materials remaining in the forest after harvest (e.g., tops, woody debris, stumps, and other logging residues), nonmerchantable biomass (e.g., small trees and noncommercial species), and waste from the creation or disposal of wood products [23]. The combination of wood biomass resources from forest and plantations will certainly play an important role in the development of energy alternatives [24]. In addition, the development of efficient cookware and modern kitchen installations could increase energetic efficiency in rural and developing areas. Otherwise, development leading to increased population and its associated agricultural activity and energy demands could cause rapid land use change, leading to large net losses in forest carbon stocks [1]. Such situation could have several unintended consequences. First, higher demand for forest biomass could increase the health burden on women and children due to the time needed to collect fuels and exposure to smoke. Also, goods and services provided by fully stocked tropical high forests could be lost by means of excessive biomass extraction [11]. Such issues could lead to increased climate impacts via carbon emissions from deforestation and forest degradation, and increased particulate and black carbon emissions [1].

On the other hand, use of biomass from residues of forest management aiming to reducing wildfire risks can support climate change mitigation efforts by helping to sequester and store more atmospheric carbon in forest standing biomass and soils [25]. Similar to any other forest management practice, ensuring the sustainability of biomass harvesting for energy requires attention to individual site conditions and consideration of multiple management objectives. Due to this increasing wildfire risks, there are economic as well as environmental reasons to increase the use of woody debris as a source of energy generated from the preventive forest management. Also, an environmental factor to be considered when using woody biomass as a source for heating fuel is the reduction of carbon dioxide emissions [25]. Integrated forest system analysis can provide insights on the overall climate implications of increasing use of bioenergy and biomaterials, which are often seen as a sustainable way to help mitigate climate change by substituting fossil fuels with forest biomass [26].

Global energy supply depends heavily on fossil fuels. Wood-based bioenergy is very competitive, when it is a product of the forest industry and used in highly efficient conversion plants. The competitiveness of forest biomass-based energy production can be further improved by better integration into industrial infrastructures of wood-processing industries, including their feedstock supply systems [27]. Integrated energy systems that supply a package of energy services including electricity, heat, and transport distance reduce the primary energy use and increase the climate benefits of woody biomass [28]. Development of energy conversion technologies and sectoral integration of energy systems could improve the primary energy efficiency of energy systems. Heat and electricity can be coproduced from woody biomass and used interchangeably in several end-use applications. However, the sustainability of all the management options associated with forest biomass should always be assessed.

2.2. Tools for monitoring ecological sustainability

Whether a forest biomass management plan is sustainable or not depends on multiple factors and determinants [29]. Due to this reason, multiple tools have been developed to estimate the sustainability of forest biomass management [30]. Some of them are forest management guidelines [31], administrative tools based on monitoring and indicators (e.g., certification schemes) [32], tools based on experience (empirical models) [33], and other tools based on the best available knowledge (process-based models) [6].

2.2.1. Forest certification: a powerful administrative tool

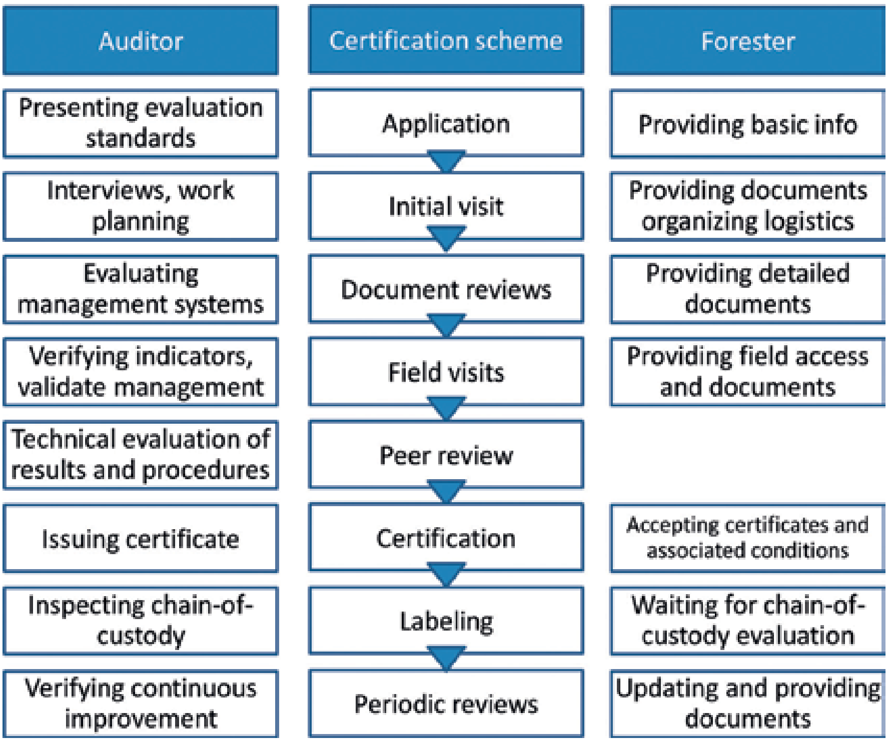
In forest biomass energy systems, both the management and the biomass product (i.e., chips, pellets, etc.) can be certified. Certification in forestry is a voluntary process in which an independent organization, recognized by the forest sector in the given country, certifies that forest operations during the management or the manipulation and transformation of the fuel have been done in a sustainable way. This is a process led by the market, in which the producer of forest products tries to get a "green label" that will raise him above the competition. The positive results obtained so far from this approach, together with the compulsory requirement for certified suppliers of many large distribution companies or public institutions, have produced a noticeable number of certification schemes in the last years (**Figure 3**).

The capacity of forest certification schemes to assess and ensure the sustainability of forest management is based on the use of criteria and indicators. A sustainability criterion is a rule, norm, target, or goal that if achieved indicates that management is sustainable. Given the inherent complexity of forest ecosystems, a set of numerous criteria are used, each of them focused on a specific part of the ecosystem [32]. For example, a sustainability criterion could be to keep a given number of dead standing trees (snags) per unit of surface in the forest that can be used as nesting ground for woodpeckers, beetles, or other fauna [34]. Sustainability criteria are usually based on empirical relationships between an ecosystem variable (i.e., number of snags per hectare) and a specific ecological process (i.e., the presence/absence or density of woodpecker birds by area unit, or other type of fauna). Such criteria are defined by the certification organizations and evaluated by the certification auditors during the certification process (**Figure 4**).

The most useful criteria to assess the sustainability of forest management for biomass production are those that can be measured or quantified. Such quantification allows for independent testing whether the sustainability criterion is being met. Such measurable variables are indicators, which can be compared against a threshold defined for a specific criterion [32, 35].

However, in certification processes, not only biological indicators are evaluated but also technical and administrative indicators (for example, creation of documents and files to follow forest management operations, handling of residues or chemicals in the forests, suitability of heavy machinery for different forest soils, etc.). The main advantage of forest biomass certification (in addition to facilitate meeting a given set of sustainability criteria) is that it generates and updates documentation on forest management that can be used for other purposes. In addition, it compares results at local scale (field data or filed documents) to international standards.

Figure 3. Forest certification labels issued by different international certification organizations.



Auditor	Certification scheme	Forester
Presenting evaluation standards	Application	Providing basic info
Interviews, work planning	Initial visit	Providing documents organizing logistics
Evaluating management systems	Document reviews	Providing detailed documents
Verifying indicators, validate management	Field visits	Providing field access and documents
Technical evaluation of results and procedures	Peer review	
Issuing certificate	Certification	Accepting certificates and associated conditions
Inspecting chain-of-custody	Labeling	Waiting for chain-of-custody evaluation
Verifying continuous improvement	Periodic reviews	Updating and providing documents

Figure 4. Roles of the three main actors involved in a forest certification process.

Last but not least, forest certification improves the marketability and image of forest producers in the eyes of sustainability-conscious consumers, which can be translated into improved sales in the long term or into being eligible for reaching given customers, such as public institutions that follow programs of buying goods only from sustainable providers. In fact, influencing sustainability of forest biomass production for energy is increasingly being seen as a policy tool that could steer intensive management away from sensitive forests [36]. To do so, technical guidelines for best practices are being developed, in which the precautionary principle is always suggested [31].

Given the increasing importance of the biomass for energy subsector inside the forest sector, certification schemes are being reviewed and adapted for the particular issues inherent to biomass production. Among them, water-related issues (both in terms of quality and availability) are being included. In addition, the overseeing and control of certification schemes must also been ensured, including mechanisms to ensure transparency and verification [37, 38].

However, forest biomass certification has some disadvantages. First, certification schemes assess the sustainability of forest biomass management practices and their ecological impact in a given point of time, but as seen before, sustainability depends on the time frame for which it is assessed. In other words, forest biomass certification studies a "picture" of the forest ecosystem, which may or may not be representative of the "movie" that in fact forest management is. For example, it could be possible to harvest all aboveground biomass in a forest stand (the so-called whole-tree harvesting) following sustainable indicators of forest management, but what could be sustainable if used once can become unsustainable if repeated over time if the increased nutrient exports in stems, branches, and leaves [39] surpass the inputs of nutrients in the system by deposition, fertilization, or other flows [40]. Other effects of forest biomass removal include changes in acid-base status and influence tree growth and survival [41]. If mineral deficiencies are generated by excessive biomass removal, reductions on forest productivity could last for decades, if not centuries [11, 42].

To address the issue of long-term sustainability evaluation, indicators can be evaluated against sustainability criteria thresholds over a period of time, and then a monitoring plan is created [37]. Controlling, measuring, and monitoring such sustainability indicators are therefore key actions in forest certification schemes. Monitoring plans can therefore detect temporal changes and define the timing when corrective actions should be carried out. For this reason, all certification schemes include periodic revisions of certified producers and owners. Although this is clearly an improvement over assessing sustainability at a unique point of time, it is possible that from an ecological point of view, monitoring plans used in certification are still using time frames too short to correctly assess sustainability of forest management, given the long timescales involved in the development of forest stands. Therefore, certification should ideally be accompanied with other tools to assess the sustainability of forest management in timescales meaningful for long-lived organisms like trees. Some of such tools are ecological models.

2.2.2. Ecological models

Historically, institutions related with forest management have generated large quantities of data and documents as reports, publications, inventories, books, etc. Many of such documents are based on field data from research plots or periodic inventories that those institutions have periodically carried out for many years, if not decades or in some cases even for more than a century [43]. An obvious application of such massive datasets is using their large coverage in time and space encompassing many different forest types (different species, tree ages, stand densities, climates, soils, etc.) to estimate how trees grow along a rotation. In other words: creating tree growth models.

A tree growth model is simply a set of rules, tables, or equations, which allows the estimation of the volume or biomass of a given forest stand at a given time. The most common growth models in forest management are growth and yield tables, which indicate for a site with a given growth potential (site quality) and a combination of tree species, stand density, and stand age, the expected volume to be harvested under different management regimes. These tables are the product of forest stakeholders' historical needs to estimate future forest productivity when planning forest management actions. Design criteria for such tools are simple: they must provide reliable predictions to avoid risks, must be based on data easy to measure in the field and must be able to use, summarize, and reproduce field data. However, as these models basically summarize observed data, their estimations will be valid only if future growing conditions are similar enough to the conditions that existed when field data were collected (**Figure 5**).

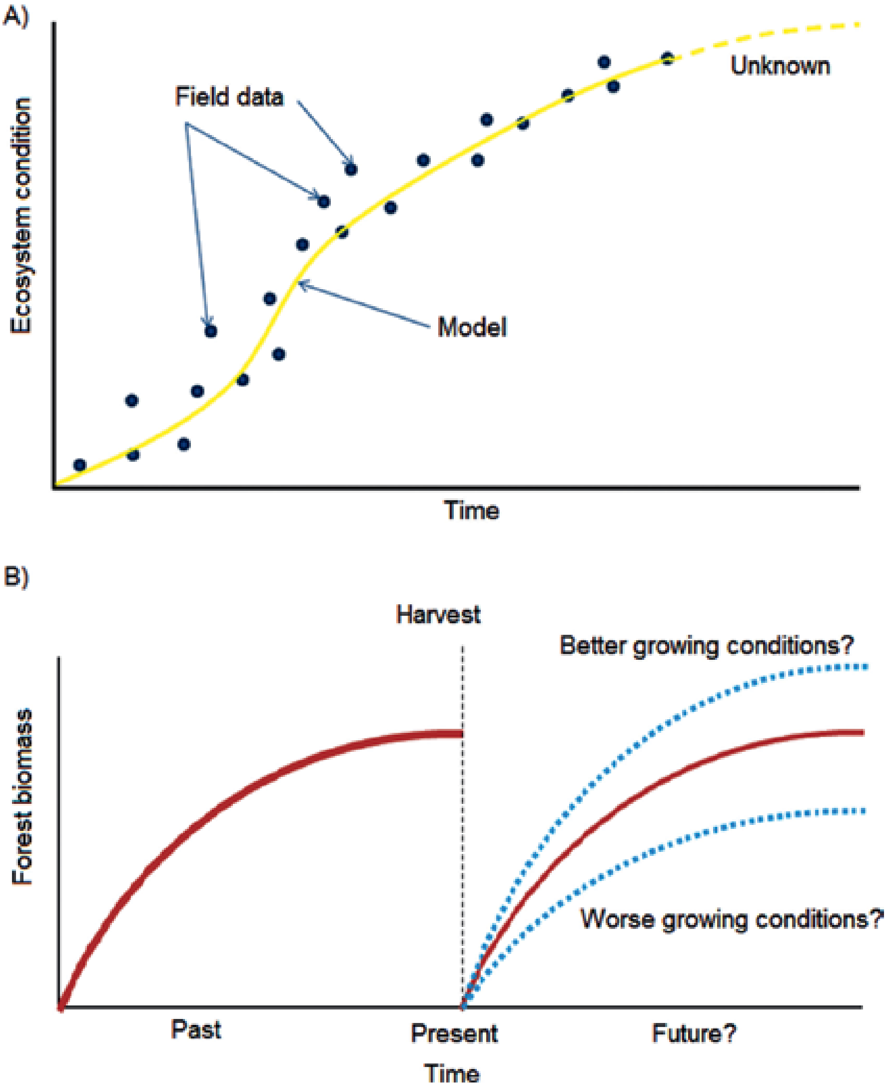


Figure 5. Conceptual development of an empirical model. A) The collection of available data is used to create a continuous equation to estimate growth for the whole range of data availability, but outside such range, tree growth patterns are unknown. B) Empirical models of forest biomass production are valid as predictive tools as long as future growing conditions are similar to the ones occurring when the data used for model creation were collected.

Then, the question to be asked is: will the future be similar to the past? The answer is that likely it won't be. Growing conditions that forest will experiment in the future will be different by changes in the biological factors (invasive species, understory competition, farming and grazing abandonment, etc.), as well as in the nonbiological factors (climate change) and human-related factors (management of mixed forests instead of monocultures, continuous cover forestry, new equipment and techniques, land use change, etc.). All those changes combined will modify growing resources (space, light, water, nutrients) and make them different from the ones that were available for trees when field data were collected. This phenomenon will reduce the utility of traditional growth and yield tables (and in general of any other empirical model based only on observations) to estimate future forest biomass production (**Figure 5**).

Therefore, more complex tools are needed to estimate the sustainability of forest biomass production for energy generation. Such tools already exist (see for example [44, 45]). They are the so-called process-based ecological models. Such models describe ecological processes using mathematical equations that relate tree growth with environmental variables such as rain, temperature, nutrients, etc. Although most of this type models were originally designed as research tools to better understand how ecological processes work at different scales (from individuals to ecosystems, from seconds to centuries), in several noticeable cases, they have reached applicability in forest management operations [44, 46, 47].

The main strong points of such tools are their great flexibility to simulate tree growth under changing conditions, the improvement of ecosystem understanding and their guidance to research. On the other hand, unless there is a deep understanding on the ecological mechanism underlying tree growth, assumptions may need to be made. Particularly, some parameters are difficult to determine in the field without special scientific equipment, which usually is not part of the standard equipment of forest management crews. Finally, additional challenges are ecosystems' emerging properties, in which the addition of what is known about each part of the ecosystem is not the same as the observed ecosystem-level ecological behavior of the system. Such challenges cause that sometimes model estimations of forest biomass are not as reliable as needed for forest management planning.

The reality is that there is not a single model that can do everything, and if there was, it would be so complex and cumbersome to use that it would be useless [33]. Using different methods to estimate sustainability of forest biomass use for energy production ensures that different aspects are taken into account [30]. Hence, it is worth to remember that the aforementioned tools do not exclude each other, and they can be combined in the modern forester's toolkit. Sustainable forest management in the modern world requires the combined use of traditional and new tools, using each of them for specific situations and to solve particular questions, and making them transparent and available for review and harmonization [4]. In fact, empirical and process-based models have already been combined to estimate social, economic, and ecological sustainability of district heating systems fueled with forest biomass [25, 29].

Many models of ecosystem productivity and function exist with varying levels of organization, scope, temporal resolution (from seconds to years), and spatial explicitness (from leaves to global assessments) [33, 48]. Recent reviews of biogeochemical models indicate that accounting for interactions between C and N cycles [49], as well as other nutrients [50], is necessary to predict future tree growth. In addition, not only climate change but chemical changes in atmospheric deposition will also be a major influence on future changes on ecosystem functioning and tree growth [51, 52]. Therefore, to estimate future forest ecosystems responses to global change, an accurate analysis of the importance of nutrient cycling on growth and its natural variability range is becoming paramount. Focusing only on average projections may not give a complete picture of the variability on nutrient stocks and flows. Variability is intrinsic to ecosystems, and it indicates different possible ecological states. Those alternative ecological states are as realistic as the average condition typically reported, being different just in the probability of such state to happen. Such variability can only be characterized through analysis of historical changes in forest ecosystem condition.

In spite of all these evidences, some state-of-the-art forest models for management and research still assume nutrient availability as a fixed parameter [53], even though models that incorporate site quality change over time have been available for more than 20 years [54]. Therefore, the key to understand the impacts of forest management on sustainable biomass production is the dynamics of belowground processes (e.g., nitrogen mineralization rates, soil respiration rates, and labile carbon forms in soils) [51]. However, empirical models regularly used in forest management do not include such considerations, and attempt to address their lack of complexity by using operational adjustment factors (usually simple multipliers) that account for changes in nutrient availability and climate, but only in a static and linear way, perpetuating their limitations.

On the other hand, over the past two decades, several process-based biochemical and physiological models have been developed for research applications. Several models of forest growth have included soil moisture or rainfall data as a parameter in the calculation of forest production, but few consider the effects of forest management for biomass production on the key processes of nutrient cycling as they interact with temperature, and moisture, under management or natural disturbances. These more complex models have generally been proven impractical in forestry applications due to the quantity of data required for their calibration, and also for failing to represent some part of the ecosystem important in forest management, such as litter decomposition (i.e., PnET-CN, BIOMASS, SORTIE, TASS, FVS/PROGNOSIS), multilayered canopies (i.e., BIOME-BGC, G'DAY, 3-PG, ForCENT, TRIPLEX), or understory (all models mentioned). Similarly, other European models such as PrognAus, SILVA, BWIN, or MOSES are well suited for management applications but they lack an explicit representation of soil nutrient processes, although EFIMOD is an exception. On the other hand, regeneration niche models such as TACA or water circulation models such as GOTILWA+ have been proven very useful for the applications that they were designed for, but they do not include nutrient cycling or nontree organisms. Finally, CENTURY is a proven soil model but it has a simple tree simulator, and therefore, it cannot be considered as a forest research and management tool (see reviews by [44, 46, 47]).

Owing to this lack of adequate and truly ecosystem-level models, decision-support tools that incorporate a greater proportion of the key determinants of tree growth and climate change effects are needed for modern forest biomass production systems. To address such issues, the latest developments in forest modeling move toward incorporating more hybrid approaches to ecological modeling [33, 55, 56], which are a mixture of both causal and empirical elements at the same hierarchical level [57]. To solve some of these issues, new models are being developed to increase the application of truly ecosystem-level simulators that account for stand-level competition for light, nutrients, and water among trees and understory, such as FORECAST-Climate [33, 57] (**Figure 6**).

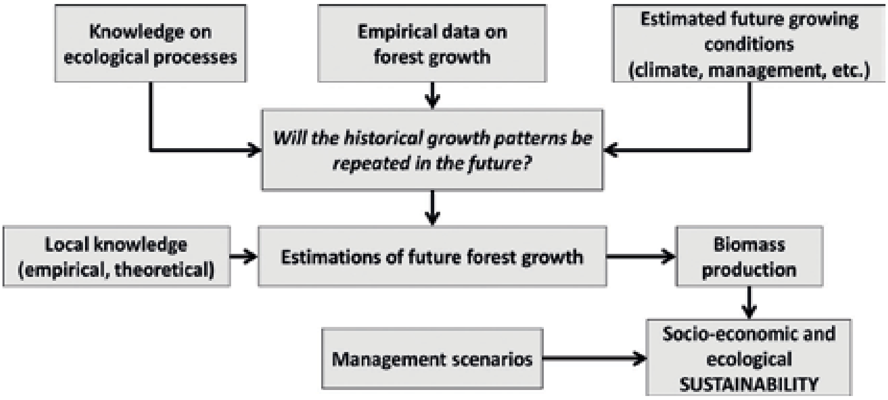


Figure 6. Conceptual development of hybrid empirical-theoretical models of forest biomass production and its use to assess sustainability of forest management practices.

These hybrid models are increasingly popular because they keep the reliability of field data but also add the scientific knowledge on ecological processes. Hybrid simulation models take advantage of the credibility of empirical models and use process simulation to account for changing conditions [57]. The feasibility of using this modeling approach to assess the sustainability of forest management focused on biomass production for bioenergy has been demonstrated for boreal [25, 29], temperate [42], and tropical forests [11, 58]. Such tools can therefore be used to explore the ecological consequences of both intensive forest management but also subsistence biomass harvesting in developing countries for cooking and heating [1]. The limitation for applying ecological models to developing countries is mostly the availability of data to accurately represent the soil, tree, and understory species in the region of interest.

In fact, hybrid models are still not widely used in forest management, as managers favor simpler experience-based empirical models such as growth and yield tables or inventory-based models. As a result, most hybrid models have been applied only as research tools. The reasons are several, but mainly, inventories of tree growth have been a good proxy for future growth as far as the conditions for tree growth remained stable. However, if changes in the forest composition, nutritional status, or climate produce a deviation from the recorded pattern (a situation dominant in European forests since the 1960s) [59], empirical models cannot predict these deviations as they lack the representation of ecological processes involved in tree growth. Therefore, using science-based models that allow for testing new management options such as ash recycling, slash and stump removal, or carbon balances and life cycle analysis should therefore be incorporated in sustainability certification schemes [60].

However, for such ecosystem-level process-based models to be valid management and research tools, they need to incorporate realistic interactions between ecological factors and be evaluated against independent data [33]. Model evaluation should be carried out by comparing model estimations with long-term observations, because even though models can show good agreement with short time series of empirical data, when projecting long-term forest dynamics, their estimations could be very sensitive to model architecture. Unfortunately, this is also a point where the lack of reliable long-term empirical data for model evaluation makes difficult the use of ecosystem-level models in developing countries. In conclusion, understanding how forests have changed during the forest's life span will allow evaluating the efficiency of current and future forest management practices to produce biomass for energy use: a crucial step to successfully adapt forest management to an uncertain future of climate change and atmospheric pollution [61].

3. Conclusions

As a take-home message, it should be remembered that sustainability of forest biomass management depends on the ecological variable being assessed and the time frame for which the assessment is carried out. As forest ecosystems are complex systems, multivariable sustainability analysis should be done, taken into account variables not only directly related to forest biomass (such as volume growth, tree productivity, etc.) but also other ecological parameters such as soil organic matter, biodiversity of plant, fungal and animal communities, levels of coarse and fine woody debris, etc. [7]. A good starting point to reach sustainable forest biomass management is the use of certification schemes, and particularly, the ones specifically designed for biomass and biofuels as energy sources. In addition, it is important to face the reality that managing complex ecological systems such as forests will always have some uncertainty associated [62]. Such uncertainty should not be considered a limitation, but a challenge that can be overcame by using an appropriate suite of ecological models (from empirical to theoretical, and particularly hybrid) that incorporate the latest scientific knowledge on past, present, and future relationships among ecological, geoclimatic, and human processes. Such tools already exist, and should be used and adapted to ensure the transition from fossil-based to renewable-based energy systems.

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Conflict of interest

The authors declare no conflict of interest. The funding sponsors had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript, and in the decision to publish the results.

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Forests grow and their biomass increases; they absorb carbon from the atmosphere and store it in plant tissue. Understanding the biomass of forest vegetation is essential for determining the storage of carbon in the dominant tree component and computing carbon cycling at a regional as well as global level. This book consisting of five chapters will give a comprehensive understanding of biomass production vis-à-vis carbon storage in relation to litter and nutrient dynamics of the forest by analyzing the mode and magnitude of biomass production and carbon storage as a function of various silvicultural factors.

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