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Forest Ecology and Conservation

Edited by Sumit Chakravarty and Gopal Shukla

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and **Gopal Shukla**

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Meet the editors

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Contents

Preface

Thirty percent of the earth's land area or about 3.9 billion hectares is covered by forests. Forests provide many environmental benefits including a major role in the hydrologic cycle, soil conservation, prevention of climate change, and preservation of biodiversity. They provide long-term economic benefits. Forest as one of the most important resources on this planet plays a pivotal role in the progress of human civilizations. The progress of civilization in perpetuity through forests needs sustainable forest management that will ensure that the goods and services derived from the forests meet today's needs and also secure their continued availability. This is only possible through our sound knowledge in forest ecology and our continued efforts for its conservation.

This book is therefore dedicated to forest ecology and conservation on ecological and conservation aspects of forest. The book is divided into two sections: the first section "Forest Ecology" with four chapters deals with forest ecological aspects, while the second section "Forest Conservation" with two chapters looks into new techniques for conserving the forests. The first chapter deals with stand classification. The demand for forest products and services had increased creating new approaches to stand composition, structure, and management. Stands were classified based on the number of trees, basal area, volume, and crown cover but unfortunately with no fixed standard. Consequently, now stand classification that incorporates not only the forest species and composition but also their horizontal and vertical arrangements came into existence. This new classification bridges the gap between composition and stand structure with an integrated approach and finds use in the National Forest Inventories and in several research studies.

The second chapter tried to establish a relationship between plant species diversity and productivity through a study of plant species diversity manipulation on abandoned arable fields. This study explained that most of the diversity effects are functioned by a successional shift from annuals to perennial plant species. The third chapter deals with soil macrofauna, especially earthworms because of their prominent role in soil engineering affecting physical, chemical, and biological components of the forest soil ecosystem. The chapter stressed the need for quantifying these effects as it would allow crucial improvement in biogeochemical budgets and modeling, predicting response of land use and disturbance, and could be applied to bioremediation efforts. Chapter 4 analyzed past fire occurrence records to quantify the relative importance of time, climate, land cover, and population to inform predictive models of fire occurrence for projecting future scenarios of fire risk to improve accuracy for fire management. The results of the study are particularly relevant to design fire management practice that can lead to successful forest conservation.

The second section of the book starts with an attempt to evaluate the possible use of a normalized difference vegetation index (NDVI) in local forest management and prevention of illegal logging and corruption. The authors were of the view that NDVI is very promising for countries that rarely perform forest inventories as it is relatively cheap and quick, is easy to implement, and also can significantly help in avoiding corruption and illegal logging. The last chapter highlights ways to finance forest conservation and offset through generating economic values of sequestered carbon. The author was of the view that forest conservation can be financed similarly like capital markets allowing the participating countries to choose between interest-bearing bonds from forest conservation (natural forests) and offset (forest plantations). The demand for the bonds can be generated out of carbon savings from forest conservation or offsetting forests, while bonds can be supplied from investments giving off carbon emissions that must be avoided through forest conservation or offset through forest plantations.

This book tried to bridge the gaps in the knowledge about some new emerging issues on forest ecology and conservation. It will be an interesting and helpful resource to all those in the field of forestry working for its sustainable use and conservation.

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Section 1

Forest Ecology

Multi-Species Stand Classification: Definition and Perspectives

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Additional information is available at the end of the chapter

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Abstract

The increasing demands for products and services from forests enhanced new approaches to stand composition, structure, and management, which encompass multiple use systems, frequently mixed either even aged or uneven aged. Stand classification is frequently based on one density measure (number of trees, basal area, volume or crown cover). As no standard criteria exist, the direct comparison between the different stand classifications is difficult. This created a need for a stand classification that incorporates not only the forest species and composition but also their horizontal and vertical arrangements. The four criteria stand classification incorporates the number of species and their proportion, their horizontal and vertical distribution. The application of this methodology enables an integrated approach, bridging the gap between composition and stand structure. Its use in the National Forest Inventories and in research studies is simple, as shown in the two cases of study presented. It also allows the evaluation of stands in a certain moment in time and their dynamics.

Keywords: density measures, composition, mixture degree index, horizontal distribution, vertical distribution

1. Introduction

Forests occupy vast areas of the world and were able to satisfy the human needs for a long time. They were at the same time a reserve and a resource, which provided shelter, wood, food and have been associated with culture and religion [1]. From the IX century onwards, the increase in human population and agriculture originated a reduction in the forest area. It was during the XIII century and following that an intensive use of wood occurred, which directed several countries in Europe to promote the protection of forests [2]. That gave rise to the development of the forest sciences in the XVII century. In the beginning, due to wood shortage, a pressure was put to create systems that were able to produce large quantities of wood. This led to pure even-aged stands, which were easier to manage. In the XIX century, the conservation issues started to arise. They were not only concerned with the maintenance of the forests and their production but also concerned with other biotic and abiotic components of the systems [1, 3, 4], which originated later the terms biodiversity and sustainability. In this context, it was considered that forests should provide several productions and be managed as multiple use systems. Management was driven to a set of practices that were associated with mimicking the natural development of forests. Many approaches, methods, and techniques are found in literature as well as terms to define them [1, 4–17]. Thought they are not entirely compatible they put a strong emphasis in pure or mixed uneven-aged stands and complex systems. This change of paradigm created new challenges, the first of which being the description of the composition of stands and forests.

Stand classification is constrained by the characteristics and the definition of pure and mixed stands (Section 2) as well as by the criteria used to define them. The most employed stand classifications use as criteria one density measure (number of trees, basal area, volume or crown cover), whereas only two classifications were found that used three criteria. Additionally, different thresholds are associated with each density measure, not enabling a simple and direct comparison between different stands (Section 3).

The aim of this study was the development of a methodology for stand classification with an integrated approach that: bridges the gaps between species composition and stand structure; give a better insight to diversity and stand dynamics; can be used regardless of the species, the stand development stage and the region; and can be implemented with data from National Forest Inventories or research studies. The stand methodology developed encompasses four criteria: species composition, their proportion, and their horizontal and vertical arrangements. Contrary to the other stand classifications, species proportion is evaluated through an index as function of three density measures (number of trees, basal area, and crown cover), enabling it to be independent of the species characteristics while discriminating different classes of mixed stands (Section 4). The application of the four criteria stand classification to both a National Forest Inventory and a research data set highlighted the difference between this classification and those with only one criterion, enabling also the stands dynamics evaluation (Section 5).

2. Pure stands vs mixed stands

The definition of stand composition exists for quite some time. It is based on the number of species and their proportions. Monospecies stands classification does not seem to have any ambiguity. Conversely, multi-species stands can be either pure or mixed, depending on each species proportion in the admixture, usually evaluated with one density measure (number of trees, basal area, volume, or crown cover). Literature puts in evidence the variability of the criteria and thresholds to distinguish stand composition. The number of trees is preferred in young stands, whereas volume, basal area, and crown cover in adult stands. Stands or forests are considered pure when the number of trees, basal area, or volume proportion of one species is equal of larger than 70%, with a varying threshold between 70 and 90%. For crown cover, there seems to be more uniformity with 75% being the most frequent one [18–21].

A frequent stand classification criterion used in research studies is often based on the number of species [22–37] with no reference to the species proportion in the mixture and their spatial arrangement. Few references are found with the proportion of the number of trees and basal area [38–39]. Thus, comparisons between the different stands or forests are rather difficult as one can be comparing different stand compositions and structures. Other question that can arise is the ecological difference between species. A stand of one broadleaved and one conifer specie, as long as the proportion of threshold is met, is considered mixed. The interpretation might be different when a stand is composed by two or more broadleaved species, especially when they belong to the same *genera* or have the same functionality. The stand classification will depend on whether the species or the *genera* or even other parameters are considered, and thus, the same stand can be classified as pure or mixed. A similar pattern is observed for two or more conifer species [40, 41]. Another two aspects to be considered in the classification of multi-species stands are the spatial horizontal and vertical distribution. Regarding the former if two species are individually mingled, the classification as mixed stand is obvious. On the contrary, when they are in groups an area threshold has to be set. Consider the example of a stand of 50 ha composed by two species A and B where the first occupies 30 ha and the second 20 ha. If the stems of the two species are mixed individually, then it is clearly a mixed stand. Conversely, if the spatial arrangement is a group 30 ha of species A and another 20 ha of the B, then it might mean that these groups are two pure stands. In between a wide span of group sizes, smaller or larger, can be found [18]. Thus for the groups' spatial distribution, its dimension has to be used to differentiate the pure and mixed stands. Reference [42] considered a maximum group area of 1 ha. As to the vertical distribution, if the species are casually distributed along the vertical profile, the classification as mixed stand does not seem to cause any doubt. Inversely, when each species mainly occupies one vertical layer, depending on the criteria, the stand can be classified as pure of one species (located in the upper layer) with an accessory stand of another (located in the inferior layer) or as mixed [43].

There seems to be a need to evaluate stand structural diversity not only to differentiate the number of species and their proportion but also to differentiate their horizontal and vertical arrangements. Structural diversity is frequently evaluated with diversity indices, which may or not require spatial information of the individual stems in a stand. Examples of the non-spatial indices are the Simpson, Shannon and Weaver, Sorenson, A profile, and uniform angle. Examples of the spatial indices are spatial mingling species, differentiation, dominance, Clark and Evans and Pielou [44–50] as well as composite stand indices, for example S index [51].

Bearing in mind the aforementioned considerations, there seems to be a need to find clear definitions and a set of criteria to make the clear distinction of stand composition, which enables the comparison between the stands regardless the species or the region of the world.

The advantages of mixed stands include the following: they provide several products [21]; are considered more resilient to disturbances [52, 53]; are more productive [20, 54–58], are frequently associated with positive interactions [55, 58, 59], especially if complementarity [3, 58, 60] and sociality principles are met [3]; have more biodiversity [13, 61–68]; and provide risk attenuation and dispersion [26]. But they are also more complex systems that encompass a wide variability of species (number and proportion) and horizontal and vertical distributions

[20, 43, 54, 69–72]. The different ecological and growth behaviours of a tree and its neighbours, the competitive effects [73–76], the species proportions and how they are calculated [77–79] may originate a reduction in the mixed stands productivity. Many definitions of mixed stands are found in literature as well as attempts to their standardisation. Reference [21] (p. 525) present a comprehensive description and definition. This definition is intended to be integrative of all the previous ones. The authors stress their broad character, underlining that in some situations, it might have to be adapted, considering the forest area, their development stage, the form of mixture, the time frame and the main relations being assessed.

3. Forest inventories and stand classification

Forest inventories had their start more than two centuries ago. Their initial objectives were focused in the evaluation of wood volume and forest planning. As described in the prior section, with the increasing demands for productions other than timber, there has been also an increase of its complexity. On one hand, parameters have to be found to evaluate an increasing number of variables to characterise the forest functions, especially those related with biodiversity for which assessment criteria are not easy to find [80, 81]. On the other hand, sampling designs and intensity for a given accuracy have to be set bearing in mind labour and costs [82] for which sample plot size and type are of crucial importance [83]. In forest stands, two interlinked measures are considered of interest to estimate forest canopies, the sum of the crowns horizontal projection area (in m^2) and the crown cover, which is the relative value of the former (in %) [84–86]. From all the variables assessed in National Forest Inventories, two variables are always assessed: area and crown cover [82, 87]. Two other variables are evaluated in the field plots: the number of trees and the diameter at breast height [36, 82]. Stand areas and crown cover are frequently estimated optical passive sensors. Species can also be identified with high spatial resolution images [82, 88–91].

As already referred, the most frequent criteria to identify mixed stands are using a density measure frequently associated with the identification of the species or *genera* in the mixture. For adult timber, producing stands volume is widely used, with a threshold for the secondary species varying between 10 and 30%. Frequently used is also crown cover for a threshold of 25% for the secondary species. In young stands, the number of trees is preferred for a threshold of 10–30% [20]. Commonly associated with those quantitative criteria, are the names of one to five of the most frequent species. References [40, 43], independently, presented two stand classifications using three criteria: form, type, and degree, which gave a contribution to the better knowledge of the multi-species stands dynamics.

Reference [43] defines texture as the way of the species group and interacts in the stand as function of: type, degree, and form (**Figure 1**). ***Type*** characterises the number of species. ***Degree*** evaluates the species abundance, as function of canopy closure, in four classes: (1) isolated, species individually mixed; (2) sparse, when the secondary species have <10% of canopy closure; (3) accompanying, when the secondary species have 10–40% of canopy closure; and (4) intimate, when the secondary species have more than 40% of canopy closure. ***Form*** refers to the spatial distribution of the individuals of the same species, in four classes: (1) individual, when the tree can be differentiated from the adjacent environment; (2) clump, little groups of trees up to a maximum of five trees in the mature stage; (3) group, set of trees that occupy an area of 0.05–0.1 ha; and (4) *bosquet*, set of trees that occupy an area of 0.1–0.5 ha.

Reference [40] presents a stand classification based in Langhammer scheme. The stands are classified using three criteria: type, degree, and form (**Figure 1**). ***Type***, the stand vertical pattern, is function of the vertical distribution of the species that is if each species is located in one or several layers; horizontal crown mixture and vertical species stratification, respectively. ***Degree***, the relative proportions of each species (defined as the percentage of the total volume (basic criterion), basal area, number of trees (especially in young stands) or crown size), considers three classes where the secondary species represent the following: (i) <10%, (ii) 10–20%, and (iii) more than 20%. ***Form***, the species horizontal spatial pattern, is defined in three classes: (1) isolated, species mixed individually; (2) line, species are arranged in lines or strips; and (3) group, species are arranged in groups of variable forms and sizes.

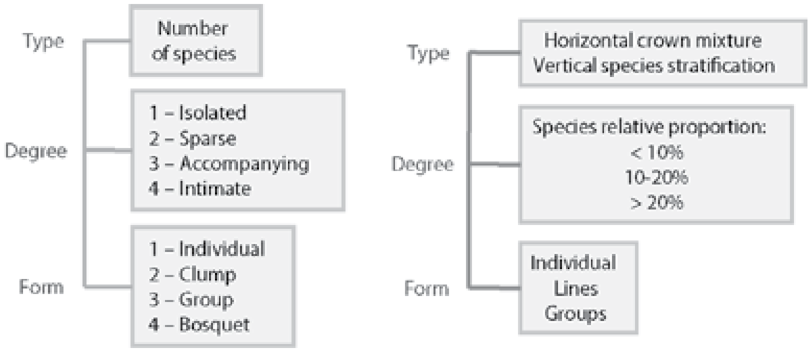

Figure 1. Representation of Schütz (left) and Leikola (right) stand classification.

4. Four criteria stand classification

Having stand composition defined the challenge is to develop a set of criteria that enables its evaluation. As already referred, especially in Europe, several methods to classify stands are found. The majority is based on one of the following density measures, number of trees, basal area, volume or crown cover, frequently associated with the species names or indicating only that the stands are composed by broadleaved and/or conifer species [19, 20]. The large number of methodologies associated with the wide span of forest species does not enable a straightforward comparison between different mixed stands. Also, no consideration is given to the horizontal and vertical distribution of the forest species in the stand, and these methods can hardly enable the analysis of the stand dynamics.

The four criteria stand classification will allow the differentiation of pure and mixed stands while discriminating different classes of the latter. The objectives are to give a better insight into the number of species, their proportions as well as their horizontal and vertical distribution in the stand. Thus, developing a tool enables stand classification with standard criteria that bridges the gap between existing ones and which gives a better insight into multi-species stands diversity as well as their dynamics. It is addressed to both National Forest Inventories and research studies. It can be easily implemented in the latter as frequently all parameters are evaluated as well as in the former with a very reduced, if any, increase in labour and costs. The stand classification was developed considering four criteria: composition, degree, form, and type (**Figure 2**). Composition evaluates the main species present in the mixture; degree their proportions, with three density measures (number of trees, basal area, and crown cover); form the species horizontal distribution; and type their vertical distribution.

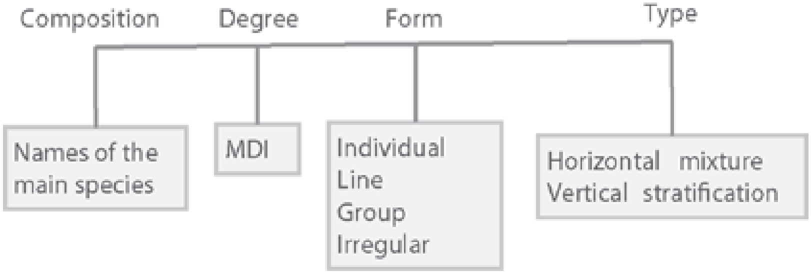

Figure 2. Representation of the four criteria stand classification.

Composition, characterising the species in the mixture, is a reflection of the site and the ecological and cultural characteristics of the species, thus a differentiating factor. In mixtures with many species, their full enumeration might be misleading as some species might have little significance in the mixture. Thus, for composition, it was considered that the two to six most representative species, considering the three density measures, should be indicated.

Degree was defined by the *mixture degree index* (*MDI*, Eq. (1)), incorporating the number of individuals as well as their dimensions. Three density measures were used as follows: number of trees (*N*), basal area (*G*), and crown cover (*CC*). Their thresholds are the most frequently used (**Table 1**). Each density parameter was reclassified as a binary variable (*Nrec, Grec, CCrec*), in which zero represents the pure and one represents the mixed stand (**Table 1**) [92]. Volume was not considered as a criterion as not all forest stands have timber as its main production, but other productions like bark (*Quercus suber*) or fruit (*Quercus ilex, Pinus pinea*). Nonetheless, an indirect evaluation of volume can be inferred from basal area and crown cover, as there is a positive relation between the former and the latter two.

$$MDI = 100\times Nrec + 10\times Grec + CCrec \tag{1}$$

	Main species			**Main species**	
Density measure	**Pure (%)**	**Mixed (%)**	**Reclassified density measure**	**Pure**	**Mixed**
N	75–100	0–75	*Nrec*	0	1
G	80–100	0–80	*Grec*	0	1
CC	75–100	0–75	*CCrec*	0	1

Table 1. Thresholds for the density measures used in *MDI*.

The evaluation of degree considering one density measure does not behave in the same way for the different stand compositions and structures. To illustrate the differences consider the examples of **Table 2**, for stands composed by two species (A and B), where the *N*, *G*, and *CC* are presented in percentage of the total. In case i using *N*, both stands are pure, while using *G* or *CC*, stand 1 is mixed and 2 is pure. In case ii for *G*, both stands are pure, whereas for N or *CC*, stand 1 is mixed and stand 2 is pure. In case iii for *N* and *G*, both stands are mixed, whereas for *CC*, stand 1 is pure and stand 2 is mixed. In case iv for *CC*, both stands are mixed, whereas for *N* or *G*, stand 1 is pure and stand 2 is mixed.

From the examples, it can be said that each density measure refers to the specie proportions, either in number or dimension, not allowing an integrated analysis. *N* evaluates only the number not giving any information about tree dimensions, thus not integrating the differences of the species development stages (young vs adult). *G* evaluates the tree dimensions yet it does not allow the distinction between species with different morphologic characteristics (large vs narrow crowns). *CC* evaluates the species-specific crown development but does not differentiate stem dimension (small vs large diameter). Though there is a direct relation between basal area and crown horizontal projection for individual trees, it varies per species and in a stand

	Stand				**Stand classification**	
	1		**2**			
Density measure	**A**	**B**	**A**	**B**	**1**	**2**
Case i						
N (%)	80	20	80	20	Pure	Pure
G (%)	40	60	80	20	Mixed	Pure
CC (%)	50	50	80	20	Mixed	Pure
Case ii						
N (%)	60	40	80	20	Mixed	Pure
G (%)	80	20	80	20	Pure	Pure
CC (%)	70	30	90	10	Mixed	Pure
Case iii						
N (%)	60	40	60	40	Mixed	Mixed
G (%)	50	50	50	50	Mixed	Mixed
CC (%)	80	20	50	50	Pure	Mixed
Case iv						
N (%)	80	20	50	50	Pure	Mixed
G (%)	80	20	40	60	Pure	Mixed
CC (%)	70	30	70	30	Mixed	Mixed

Table 2. Examples of stand classification with *N*, *G*, and *CC*.

due to competition phenomena, high and low shade, and branch abrasion might induce crown shyness resulting in a smaller crown when compared to individuals of the same species in open or free growth [3, 52]. *MDI* combines the three density measures and gives a better insight of different structures of mixed stands and their dynamics. It enables to distinguish the pure and mixed stands and in the latter differentiates in seven classes (**Table 3**). When *MDI* is 000 or 111, the classification is obvious; in the first case, it is a pure stand, and in the second case, it is a mixed one. If *MDI* = 001, the secondary species have high *CC* but low *N* and *G* indicative of adult or young trees with large crowns. When *MDI* = 010, the secondary species have high *G* but low *N* and *CC* indicative of secondary species adult trees with narrow crowns or subjected to strong competition. *MDI* = 011 corresponds to mixed stands where the secondary species proportion is low in *N* but high in *G* and *CC*, that is adult trees with large crowns. When *MDI* = 100, the secondary species have a high *N* but low *G* and *CC*, indicating young trees in the initiation development stage, thus with small diameters and crowns. *MDI* = 110 represents a stand where *N* and *G* of the secondary species are high but with low CC, as these species have narrow crowns or as a result of a strong competition. For *MDI* = 101, the secondary species have a high *N* and *CC* but low *G* indicating young stems with large crowns. *MDI* can be evaluated with inventory plot data, calculating the proportion of *N* and *G*. *CC* can be evaluated with the passive optical sensors either visually or with remote sensing classification methods. *MDI* can also be evaluated visually in the field by experienced foresters.

For ***form***, four classes were considered as follows: individual, line, group, and irregular. The first is similar to a chessboard; theoretically is the more elementary form of mixture of a community of plants [40]. The second, especially common in plantations, where species' spatial arrangement is in lines or strips. The third is defined by groups of variable forms and sizes. The fourth corresponds to a spatial distribution where individual and group distributions, or even line, have similar proportions. This criterion can be evaluated visually in the field or with optical sensors where the vegetation mask is attained per species. In research plots, where tree locations are known, diversity indices can be used for example the Pielou.

		Secondary species	
MDI	**Degree**	**Proportion**	**Characteristics**
000	Pure		
001	Mixed	Low	Young or adult with wide crowns
010	Mixed	Low	Adult with narrow crowns
011	Mixed	Low	Adult with wide crowns
100	Mixed	High	Young with narrow crowns
110	Mixed	High	Young or adult with narrow crowns
101	Mixed	High	Young with wide crowns
111	Mixed	Low/High	Young or adult with narrow or wide crowns

Table 3. *MDI* classes and secondary species proportion and characteristics.

Regarding ***type***, two classes were considered, the horizontal mixture and the vertical stratification, allowing the distinction between stands with one or more layers, thus even-aged from uneven-aged stands. This criterion can be evaluated visually in the field or with height distribution histograms. Though frequently total height is not measured in all trees in the field plots of the National Forest Inventories, it can be easily calculated with hypsometric functions that exist for almost all forest species. Again, in research plots, type can be evaluated with diversity indices, for example the profile A.

5. Application of the four criteria stand classification

5.1. Materials and methods

The four criteria stand classification was applied to two sets of data, the plots of the fifth Portuguese National Forest Inventory (NFI5) and to a set of research plots with two measurements to evaluate whether this classification can detect the stands' dynamics. *MDI* was also compared with *N*, *G*, and *CC*.

The NFI5 data set used is composed of 5435 plots, where the species were identified, diameter at breast height (1.30 m) was measured, and vertical distribution was evaluated visually. Crown cover was evaluated in aerial photographs. The representative forest species in Portugal are *Pinus pinaster, Eucalyptus* spp., *Quercus suber, Quercus ilex, Pinus pinea, Castanea sativa* and *Quercus robur* [93]. The second data set is composed of 28 research plots, with two measurements, one in Serra da Lousã (LO) and another in Herdade da Machoqueira do Grou (HM). LO is a mountain in central Portugal, about 250 km northeast from Lisbon (40°04′51″ N and 8°14′44″ W), where 16 plots were installed in adult stands of *Pinus pinaster*, located predominantly in the superior and intermediate layers, and broadleaved (mainly *Castanea sativa* and *Quercus robur* but also *Quercus rubra, Prunus avium, Fagus sylvatica*) and several conifers (*Pinus pinaster, Pseudotsuga menziesii*, C*hamaecyparis lawsoniana*) of natural regeneration in the intermediate and inferior layers. HM, located in Coruche, about 120 km east from Lisbon (39°06′59″ N and 8°21′05″ W), is mainly composed of *Quercus suber* and *Pinus pinea* with some *Pinus pinaster* individuals. The surveys took place in 2001 and 2009 in LO and in 1998 and 2008 HM. In these plots, diameter at breast height, total height and crown radii (North, South, East, and West directions) were measured for all trees with diameter at breast height ≥5 cm, and the tree coordinates recorded. The equality between each pair of density measures (*N, G, CC*, and *MDI*) was evaluated with McNemar test [94], implemented in R statistical software [95], for $\alpha = 0.05$.

5.2. Results and discussion

The plots of NFI5 have one to six species. Those with one species account for 63.8%, whereas two or more species represent 36.2%. In the latter, the most frequent have two (28.1%) and three (6.2%) species. In the two species plots, 113 combinations were found. The most frequent are *Pinus pinaster* × *Eucalyptus* spp. (15.7%), *Eucalyptus* spp. × *Pinus pinaster* (14.2%), *Quercus suber* × *Quercus ilex* (11.7%), *Quercus suber* × *Pinus pinaster* (10.9%), and *Quercus ilex* × *Quercus suber*

(7.3%), which correspond to 59.8% of their total number of plots. As to the three species plots, 261 combinations were found being the more frequent those of *Quercus suber* × *Pinus pinea* × *Pinus pinaster* (4.4%) and *Pinus pinaster* × *Eucalyptus* spp. × *Quercus robur* (3.0%). The number of plots of NFI5 classified has mixed vary according to the criterion used for degree (**Table 4**). *N* detects less 3.9% mixed stands than *G*, less 5.4% than *CC* and less 19.4% than *MDI*. *G* and *CC* detect less 15.5 and 14.0% plots than *MDI*. These results are confirmed by the significant differences between *N* and *G* ($\chi_1^2 = 69.5$, $p < 0.001$), *N* and *CC* ($\chi_1^2 = 71.2$, $p < 0.001$), *G* and *CC* ($\chi_1^2 = 4.5$, $p = 0.033$), *N* and *MDI* ($\chi_1^2 = 1052.0$, $p < 0.001$), *G* and *MDI* ($\chi_1^2 = 839.0$, $p < 0.001$), and *CC* and *MDI* ($\chi_1^2 = 760.0$, $p < 0.001$).

The analysis of *MDI* (**Table 5**) reveals that the most frequent class is 001, that is, where the secondary species have large crowns, but has a low number of individuals with small basal area. The second and third more frequent mixed plots are 010 and 110, respectively, and correspond to secondary species with large basal area and small or large number of individuals. From the above results, it can be said that *N* and *G* seem to detect less mixed plots, conversely to *CC* and *MDI*. One of the reasons might be that stands of species with large crowns (e.g. *Quercus* sp.) have frequently <200 tree ha^{-1} [96], and only a few trees are needed to reach the minimum threshold for *CC*, but not for *N* or *G*. From the plots classified as mixed with *N*, 44.5% were classified as mixed by *CC* (*MDI* 101 and 111) and from those mixed with *G* 38.3% were classified as mixed with *CC* (*MDI* 011 and 111). *MDI* has the advantage of being a flexible index enabling their use regardless the species ecological characteristic and growth habits. These results suggest that the NFI data sets [36, 82, 83, 87] can be used to enhance further detail on stand classification using *MDI* and they can be of help during regeneration phases or during transformation processes [4, 7].

As form was not evaluated in the NFI5, all mixed plots detected by *MDI* were surveyed in the corresponding ortophotomaps to evaluate form visually. Form was irregular for 63.1%, in

	Pure		**Mixed**	
Density measure	**Number**	**Proportion (%)**	**Number**	**Proportion (%)**
N	4626	85.1	809	14.9
G	4334	79.7	1022	18.8
CC	4334	79.7	1101	20.3
MDI	3572	65.7	1863	34.3

Table 4. Stand classification with different density measure criteria.

MDI	001	010	011	100	101	110	111
Number	624	313	117	131	86	318	274
Proportion (%)	33.5	16.8	6.3	7.0	4.6	17.1	14.7

Table 5. *MDI* mixed classes, number and proportion (in %) of the IFN5 plots.

groups for 22.3%, individual for 14.4%, and only residual (0.2%) for lines. The more frequent form varies according to *MDI* classes (**Table 6**). The irregular form prevails for 001, whereas for 010 group, irregular distributions occur with similar frequencies. Not considering line form, 011, 110, and 101 have predominantly an individual distribution. In the 110 predominates, the group distribution (not considering the line) as for 111 individual is the most frequent one. These results are expectable for two main reasons. Many forest species have heavy fruits (such as *Quercus* sp., or *Castanea sativa*), so it is expected that fruits fall near the seed bearer [96] and also as some forest species coppice very easily (*Eucalyptus* sp. and *Castanea sativa*) [69, 96] thus increasing the tendency to form groups. Contrary, other species have light fruits (*Pinus pinaster*) with fruit dispersal in large areas [52, 97]. Inversely, individual and group forms seem to be linked to management practices.

Type detected that more than half of the plots classified as mixed had vertical stratification (**Figure 3**), 62.4% with *N*, 57.8% with *G*, 61.5% with *CC*, and 52.9% with *MDI*. This is indicative of a successful natural regeneration for *MDI* classes 100, 101, and 111, in number of individuals,

Form	001	010	011	100	101	110	111
Group	26.4	17.8	7.7	7.5	4.8	19.0	16.8
Individual	24.3	12.3	10.1	9.0	7.1	17.9	19.4
Irregular	38.2	17.5	4.9	6.3	4.0	16.2	12.9
Line	0.0	0.0	0.0	66.7	0.0	33.3	0.0

Table 6. Form per *MDI* classes (in %).

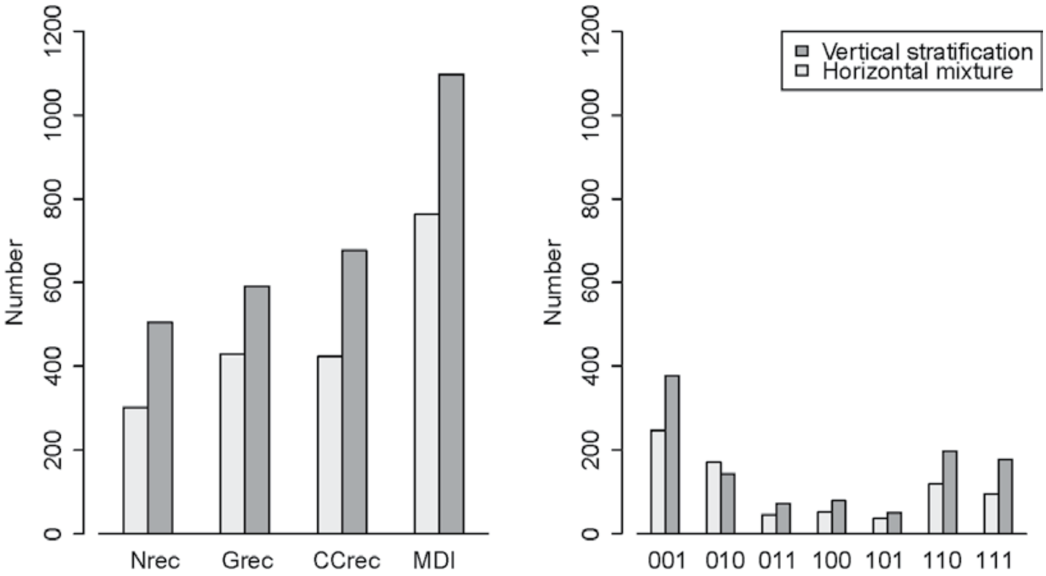

Figure 3. Number of plots per density measure (left) and per *MDI* classes for the total of *MDI* mixed plots (right).

which is one of the principles of uneven-aged silviculture [4, 7]. *MDI* classes 001, 010, and 011, with vertical stratification, correspond also to establish regeneration of species with large crowns (001), and for adult individuals with narrow crowns (010) or large crowns (011). This can be, at least partially, explained by the periodic recruitment [3, 4, 7, 52]. The horizontal mixture is frequent for 001 (32.2%) and 010 (22.2%). The former corresponds to stands where the secondary species have few individuals with large crowns and the latter to stands where the secondary species have few adult individuals with large stems.

In LO, the number of species per plot varies between 2 and 7. In all plots, *Pinus pinaster* and *Castanea sativa* are present, in 14 *Quercus robur*, and the other species occur in reduced numbers in one or two plots. In HM, three species are present in the plots; *Quercus suber* and *Pinus pinea* are present in all plots and *Pinus pinaster* is present in five plots. In these research plots, there seems to be a trend to a similar classification with *CC* and *MDI* for LO and all the density measures for HM (**Table 7**). The latter is not surprising as the plots are composed by adult stems with *Quercus suber* as main species and *Pinus pinea* and *Pinus pinaster* with *N*, *G*, and *CC* always higher than the threshold. Contrary, LO plots have adult *Pinus pinaster* stems and recruitment mainly of *Castanea sativa* and *Quercus robur*, two species of large crowns, thus it is not surprising that *CC* and *MDI* detected the same number of mixed plots. The largest difference is observed when *G* is used as a criterion, as the secondary species have quite small diameters. Conversely, *N* reflects a successful regeneration. The results for LO are confirmed by the statistical differences observed *N* and *G* ($\chi_1^2 = 10.5625$, $p < 0.001$), *N* and *CC* ($\chi_1^2 = 4.1667$, $p = 0.041$), *G* and *CC* ($\chi_1^2 = 18.05$, $p < 0.001$), *N* and *MDI* ($\chi_1^2 = 4.1667$, $p = 0.041$), and *G* and *MDI* ($\chi_1^2 = 18.05$, $p < 0.001$). For HM, no significant differences were found between each pair of the density measures (*all*, $p > 0.05$).

MDI classified 21 plots as mixed (**Table 8**), and four different classes of degree were identified as follows: 001, 011, 101, and 111. *MDI* = 001 identifies stands where the individuals of the secondary species have large crowns, and this was observed in one plot of LO, where a small number of *Castanea sativa* stems recruited developed large crowns [92]. *MDI* = 010 occurs in one plot of HM where the *Pinus pinaster* adult individuals have large basal area but, characteristic of the species, have narrow crowns [97]. *MDI* = 011 occurs in HM in four plots of *Quercus suber* and *Pinus pinea* where the latter are adult, with large basal area and crowns [96].

		N		***G***		***CC***		***MDI***	
Local	**Survey**	**Mixed**	**Pure**	**Mixed**	**Pure**	**Mixed**	**Pure**	**Mixed**	**Pure**
LO	2001	9	7	1	15	11	6	11	6
	2009	10	6	3	13	14	2	14	2
HM	1998	10	2	11	1	11	1	11	1
	2008	11	1	11	1	11	1	11	1

Table 7. Stand classification per density measure, local and survey.

MDI = 101 was observed in eight plots of LO with a large number of recruitment individuals with large crowns, mainly of *Castanea sativa*. *MDI* = 111 corresponds to established mixed uneven-aged (LO, HM) and even-aged (HM) stands. The proportion of plots classified as mixed in LO was 56.3% for *N*, 6.3% for *G*, and 62.5% for *CC* and *MDI*; and for HM, 83.3% for *N* and 91.7% for *G*, *CC*, and *MDI*. From the first to the second survey, some changes occurred in the former but not in the latter. In LO, four pure plots in 2001were classified as mixed in 2009 (**Table 8**). In fact, six plots have changed *MDI*, three moved from 000 to 001, one moved from 000 to 101 and two moved from 101 to 111. These dynamics can be explained by two factors. First, due to the growth of the secondary species individuals. Second, from the first to the second survey selection, cuttings were carried out to remove mainly *Pinus pinaster* individuals [98], which increased the relative proportion of *N*, *G*, and *CC* of the secondary species. In HM, one plot passed from 011 to 111 for the aforementioned reason due to the removal of some individuals of *Quercus suber* and *Pinus pinaster*. These small changes show a trend towards mixed stands, though no significant differences were found between *MDI* classes between the two surveys for both LO and HM (*all*, $p > 0.05$).

Local	Survey	000	001	011	010	100	101	110	111
LO	2001	6	1	0	0	0	8	0	1
	2009	2	4	0	0	0	7	0	3
HM	1998	1	0	1	0	0	0	0	10
	2008	1	0	0	0	0	0	0	11

Table 8. Stand classification per *MDI* classes, local and survey.

Form was evaluated using the crown maps and revealed for both surveys that for LO species, spatial arrangement was individual in eight plots, irregular in five and in groups in three, and for HM, irregular in eight plots and in groups in four. The results of LO are in accordance with Ref. [99] that refer that Pielou index showed for *Pinus pinaster, Castanea sativa* and *Quercus robur* a tendency to segregation.

Type was evaluated with profile A index. For all plots, the index was greater than zero indicative of species in several height layers. In LO, it is indicative of the presence of *Pinus pinaster, Castanea sativa*, and *Quercus robur* in three, two (inferior and intermediate), and one (inferior) layer, respectively. From the first to the second survey profile A index increased, corresponding to the presence of *Castanea sativa* in the superior layer and *Quercus robur* in the intermediate, which is in accordance to the results of Ref. [99]. In HM, the profile A index values for *Quercus suber* and *Pinus pinea* are indicative of their presence in two (inferior and intermediate) height layers, while for *Pinus pinaster* in one (superior) layer. The analysis of the height distribution histograms confirmed the trend attained with the diversity index. In HM in seven plots, there was a slight reduction of the profile A index values. This can be explained, at least partially, by the removal of *Pinus pinaster* that was predominantly in the superior layer, confirmed by the height distribution histograms.

6. Conclusions

In the four criteria stand classification, composition characterises the most representative species, degree their relative proportions, form their horizontal distribution and type their vertical distribution. Stand classification with *N, G* or *CC* makes it dependent on the species and their morphological patterns. For *MDI* 111 and 000, stand classification is similar whatever density measure is used. For the other *MDI* classes, the selection of the density measure influences stand classification. *MDI* advantage is that it aggregates, in a simple way, *N, G*, and *CC* can be used regardless the stand structure or the species morphological patterns. The analysis of each density measure suggests that *N* is suited for uneven-aged stands, where diameters and crowns have the same morphological pattern; *G* for even-aged stands; and *CC* for even-aged and uneven-aged stands for species with different morphological patterns. Also, *MDI* can provide further detail on the stands' dynamics, as shown in the LO plots from the first to the second survey. The four form classes enable the evaluation of the horizontal distribution of the species, and the two type classes enable the evaluation of their vertical distribution. These criteria are especially useful to prescribe silvicultural practices, such as the control of competition pressure between individuals as well as to promote growth both in stem and crown diameters, especially important for stands with bark (the former) and fruit (the latter) as their main production.

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References

[1] Ciancio O. La teoria della selvicoltura sistémica i razionalisti e gli antirazionalisti, le "sterili disquisizioni" e il sonnambulismo dell'intellighenzia forestale [The theory of systemic silviculture, the rationalists and non rationalists, the "sterile discussions" and the somnambulism of forest intelligentsia]. Firenze: Accademia Italiana di Scienze Forestali; 2010. 51 p. [in italian]

[2] Alves AAM, Pereira JS, Correia AV. Silvicultura. A gestão dos ecossistemas florestais [Silviculture. Management of forest ecosystems]. Lisboa: Fundação Calouste Gulbenkian; 2012. 597 p. [in portuguese]

[3] Schütz JP. Sylviculture 2. La gestion des forêts irrégulières et mélangées [Silviculture 2. The management of even-aged and mixed forests]. Lausanne: Presses Polytechniques et Universitaires Romandes; 1997. 178 p. [in french]

[4] Schütz JP. Close-to-nature silviculture: is this concept compatible with species diversity? Forestry. 1999;**72**:359–366.

[5] Perkey AW, Wilkins BL, Smith HC. Crop tree management in Eastern hardwoods. NA-TP-19-93. USDA: Forest Service; 1993. 57 p.

[6] Mason B, Kerr G. Transforming even-aged conifer stands to continuous cover management. Edinburgh: Forestry Commission; 2004. 8 p.

[7] O'Hara K. The silviculture of transformation—a commentary. Forest Ecol Manag. 2001;**151**:81–86.

[8] Schütz JP. Opportunities and stratagies of transforming regular forests to irregular forests. Forest Ecol Manag. 2001;**151**:87–94.

[9] Schütz JP. Silvicultural tools to develop irregular and diverse forest structures. Forestry. 2002;**75**:329–337.

[10] Ciancio O, Nocentini S. Bioviversity conservation and systemic silviculture: concepts and applications. Plant Biosystems. 2011;**145**(2):411–418.

[11] McEvoy TJ. Positive impact forestry. A sustainable approach to managing woodlands. Island PRESS; 2003. 288 p.

[12] Pommerening A, Murphy ST. A review of the history, definitions and methods of continuous cover forestry with special attention to afforestation and restocking. Forestry. 2004;**77**(1):27–4.

[13] Puettmann KJ, Coates KD, Messier C. A Critique of Silviculture. Washington: Island Press; 2009. 188 p.

[14] Diaci J, Kerr G, O'Hara K. Twenty-first century forestry: integrating ecologically based, uneven-aged silviculture with increased demands on forests. Forestry. 2011;**84**(5). doi:10.1093/forestry/cpr053

[15] Moravčík M, Sarvašová Z, Merganič J, Kovalčík M. Close to Nature Management in High-Mountain Forests of Norway Spruce Vegetation Zone in Slovakia. In Blanco JA, Lo YH, editors. Forest Ecosystems—More than Just Trees. InTech; 2012. pp. 375–414.

[16] Schütz JP, Pukkala T, Donoso P J, Gadow Kv. Historical Emergence and Current Application of CCF. In: Pukkala T, Gadow Kv, editors. Continuous Cover Forestry. Springer Science+Business Media B.V.; 2012. pp. 1–28.

[17] Bauhus J, Puettmann KJ, Kühne C. Is Close-To-Nature Forest Management in Europe Compatible with Managing Forests as Complex Adaptive Forest Ecosystems? In: Messier C, Puettmann KJ, Coates KD. Managing Forests as Complex Adaptive Systems: Building Resilience to the Challenge of Global Change. London and New York: Routledge; 2013. pp. 187–213.

[18] Rackam O. Mixtures, Mosaics and Clones: The Distribution of Trees Within European Woods and Forests. In: Cannell MGR, Malcolm DC, Robertson PA, editors. The Ecology of Mixed Species Stands of Trees. Oxford: Wiley-Blackwell; 1992. pp. 1–20.

[19] Köhl M, Päivinen R, Traub B, Miina S. Comparative Study. In: Study on European Forestry Information and Communication System. European Comission. Reports on Forest Inventory and Survey Systems. 1997;**2**:1265–1322.

[20] Gardiner JJ. Changing Forests, Management and Growing Conditions. In: Olsthoorn AFM, Bartelink HH, Gardiner JJ, Pretzsch H, Hekhuis HJ, Franc A, editors. Management of Mixed-Species Forest: Silviculture And Economics. DLO Institute for Forestry and Nature Research; 1999. pp. 17–19.

[21] Bravo-Oviedo A, Pretzsch H, Ammer C, Andenmatten E, Antón C, Barbati A, Barreiro S, Brang P, Bravo F, Brunner A, Coll L, Corona M, Den Ouden J, Drössler L, Ducey MJ, Kaynas BY, Legay M, Löf M, Lesinski J, Mason B, Meliadis M, Manetti MC, Morneau F, Motiejunaite J, O'Reilly C, Pach M, Ponette Q, Río Md, Short I, Skovsgaard JP, Souidi Z, Spathelf P, Sterba H, Stojanovic D, Strelcova K, Svoboda M, Valsta L, Verheyen K, Zlatanov T. European mixed forests: definition and perspectives. Forest Systems. 2014;**23** (3):518–533.

[22] Miles PD. Using biological criteria and indicators to address forest inventory data at the state level. Forest Ecol Manag. 2002;**155**:171–185.

[23] Takahashi K, Mitsuishi D, Uemura S, Suzuki JI, Hara T. Stand structure and dynamics during a 16-year period in a sub-boreal conifer–hardwood mixed forest, northern Japan. Forest Ecol Manag. 2003;**174**(1–3):39–50.

[24] Koch JA, Makeschin F. Carbon and nitrogen dynamics in topsoils along forest conversion sequences in the Ore Mountains and the Saxonian lowland, Germany. Eur J For Res. 2004;**123**(3):189–201.

[25] Bouchard M, Kneeshaw D, Bergeron Y. Mortality and stand renewal patterns following the last spruce budworm outbreak in mixed forests of western Quebec. Forest Ecol Manag. 2005;**204**:297–313.

[26] Knoke T, Stimm B, Ammer C, Moog M. Mixed forests reconsidered: A forest economics contribution on an ecological concept. Forest Ecol Manag. 2005;**213**:102–116.

[27] Feng Y, Tang S, Li Z. Application of improved sequential indicator simulation to spatial distribution of forest types. Forest Ecol Manag. 2006;**222**(1–3):391–398.

[28] Mast JN, Wolf JJ. Spatial patch patterns and altered forest structure in middle elevation versus upper ecotonal mixed-conifer forests, Grand Canyon National Park, Arizona, USA. Forest Ecol Manag. 2006;**236**:241–250.

[29] Dittmar C, Elling C. Dendroecological investigation of the vitality of Common Beech (*Fagus sylvatica* L.) in mixed mountain forests of the Northern Alps (South Bavaria). Dendrochronologia. 2007;**25**(1):37–56.

[30] Tabor J, McElhinny C, Hickey J, Wood J. Colonisation of clearfelled coupes by rainforest tree species from mature mixed forest edges, Tasmania, Australia. Forest Ecol Manag. 2007;**240**(1–3):13–23.

[31] Everett RG. Dendrochronology-based fire history of mixed-conifer forests in the San Jacinto Mountains, California. Forest Ecol Manag. 2008;**256**(11):1805–1814.

[32] Jian Z, Zhanqing H, Buhang L, Ji Y, Xugao W, Xiaolin Y. Composition and seasonal dynamics of seed rain in broad-leaved Korean pine (*Pinus koraiensis*) mixed forest in Changbai Mountain, China. Acta Ecol Sin. 2008;**28**(6):2445–2454.

[33] Rodríguez-Loinaz G, Onaindia M, Amezaga I, Mijangos I, Garbisu C. Relationship between vegetation diversity and soil functional diversity in native mixed-oak forests. Soil Biol Biochem. 2008;**40**(1):49–60.

[34] Thessler S, Sesnie S, Bendaña ZSR, Ruokolainen K, Tomppo E, Finegan B. Using k-nn and discriminant analyses to classify rain forest types in a Landsat TM image over northern Costa Rica. Remote Sens Environ. 2008;**112**(5):2485–2494.

[35] Zald HSJ, Gray AN, North M, Kern RA. Initial tree regeneration responses to fire and thinning treatments in a Sierra Nevada mixed-conifer forest, USA. Forest Ecol Manag. 2008;**256**(1–2):168–179.

[36] Lindenmayer DB, Wood JT, Michael D, Crane M, MacGregor C, Montague-Drake R, McBurney L. Are gullies best for biodiversity? An empirical examination of Australian wet forest types. Forest Ecol Manag. 2009;**258**:169–177.

[37] Moktan MR, Gratzer G, Richards WH, Rai TB, Dukpa D, Tezin K. Regeneration of mixed conifer forests under group tree selection harvest management in western Bhutan Himalayas. Forest Ecol Manag. 2009;**257**(10):2121–2132.

[38] Aikens ML, Ellum D, McKenna JJ, Kelty MJ, Ashton MS. The effects of disturbance intensity on temporal and spatial patterns of herb colonization in a southern New England mixed-oak forest. Forest Ecol Manag. 2007;**252**:144–158.

[39] Teklemariam T, Staebler RM, Barr AG. Eight years of carbon dioxide exchange above a mixed forest at Borden, Ontario. Agric For Meteorol. 2009;**149**(11):2040–2053.

[40] Leikola M. Definition and Classification of Mixed Forests, with Special Emphasis on Boreal Forests. In: Olsthoorn AFM, Bartelink HH, Gardiner JJ, Pretzsch H, Hekhuis HJ, Franc A, editors. Management of Mixed-Species Forest: Silviculture and Economics. DLO Institute for Forestry and Nature Research; 1999. pp. 20–28.

[41] Picard N, Kölher P, Mortier F, Gourlet-Fleury S. A comparison of five classifications of species into functional groups in tropical forests of French Guiana. Ecol Complexity. 2012;**11**:75–83.

[42] Davies O, Haufe J, Pommerening A. Silvicultural principles of continuous cover forestry. A Guide to Best Practice. Bangor: Bangor University, Forestry Commission Wales; 2008. 111 p.

[43] Schütz JP. Sylviculture 1. Principes d'éducation des forêts [Silviculture 1. Principles of tending the forests]. Lausanne: Presses Polytechniques et Universitaires Romandes. 1990. p. 243. [in french].

[44] Gadow Kv, Zhang CY, Wehenkel C, Pommerening A, Corral-Rivas J, Korol M, Hui GY, Kiviste A, Zhao XH. Forest Structure and Diversity. In: Pukkala T, Gadow Kv, editors. Continuous Cover Forestry. Springer Science+Business Media B.V.; 2012. pp. 29–83.

[45] Pommerening A. Approaches to quantifying forest structures. Forestry. 2002;**75**:305–324.

[46] Aguirre O, Hui GY, Gadow Kv, Jiménez J. An analysis of spatial forest structure using neighbourhood-based variables. Forest Ecol Manag. 2003;**183**:137–145.

[47] McElhinny C, Gibbons P, Brack C, Bauhus J. Forest and woodland stand structural complexity: Its definition and measurement. Forest Ecol Manag. 2005;**218**:1–24.

[48] Pommerening A. Transformation to continuous cover forestry in a changing environment. Forest Ecol Manag. 2006;**224**:227–228.

[49] Sterba H, Ledermann T. Inventory and modelling for forests in transition from even-aged to uneven-aged management. Forest Ecol Manag. 2006;**224**:278–285.

[50] Merganič J, Merganičová K, Marušák R, Audolenská V. Plant Diversity of Forests. In: Blanco JA, Lo YH, editors. Forest Ecosystems-More than Just Trees. InTech; 2012. pp. 3–28.

[51] Pastorella F, Paletto A. Stand structure indices as tools to support forest management: an application in Trentino forests (Italy). J For Sci. 2013;**59**(4):159–168.

[52] Oliver CD, Larson BC. Forest Stand Dynamics. Update editions. New York: John Wiley & sons, Inc; 1996. 544 p.

[53] Spies T. Forest Stand Structure, Composition, and Function. In: Kohm KA, Franklin JF, editors. Creating a Forestry for the 21st Century. Washington: Island Press; 1997. pp. 11–30.

[54] Assmann E. The Principles of Forest Yield Study. Oxford: Pergamon Press; 1970. 506 p.

[55] Kelty MJ, Larson BC, Oliver CD, (eds). The Ecology and Silviculture of Mixed-Species Forests. Dordrecht: Kluwer Academic Publishers; 1992. 287 p.

[56] Pretzsch H, Schütze G. Transgressive overyielding in mixed compared with pure stands of Norway spruce and European beech in Central Europe: evidence on stand level and explanation on individual tree level. Eur J For Res. 2009;**128**:183–204.

[57] Pretzsch H, Block J, Dieler J, Dong PH, Kohnle U, Nagel J, Spellmann H, Zingg A. Comparison between the productivity of pure and mixed stands of Norway spruce and European beech along an ecological gradient. Ann For Sci. 2010;**67**:1–12.

[58] Forrester DI. The spatial and temporal dynamics of species interactions in mixed-species forests: from pattern to process. Forest Ecol Manag. 2014;**312**:282–292.

[59] Bertness MD, Callaway RM. Positive interactions in communities. Trends Ecol Evol. 1994;**9**:191–193.

[60] Forrester DI, Kohnle U, Albrecht AT, Bauhus J. Complementarity in mixed-species stands of *Abies alba* and *Picea abies* varies with climate, site quality and stand density. Forest Ecol Manag. 2013;**304**:233–242.

[61] Brockway DG. Forest plant diversity at local and landscape scales in the Cascade Mountains of southwestern Washington. Forest Ecol Manag. 1998;**109**:323–341.

[62] Lawesson JE, Blust Gde, Grashof C, Firbank L, Honnay O, Hermy M, Hobitz P, Jensen LM. Species diversity and area-relationship in Danish beech forests. Forest Ecol Manag. 1998;**106**:235–245.

[63] Pitkänen S. The use of diversity indices to assess the diversity of vegetation in managed boreal forests. Forest Ecol Manag. 1998;**12**:121–137.

[64] Gaines WL, Harrod RJ, Lehmkuhl JF. Monitoring biodiversity: quantification and interpretation. PNW-GTR-443. USDA: Forest Service; 1999. 27 p.

[65] Neumann M, Starlinger F. The significance of different indices for stand structure and diversity in forests. Forest Ecol Manag. 2001;**145**:91–106.

[66] Staudhammer CL, Lamey VM. Introduction and evaluation of possible indices of stand and structural diversity. Can J For Res. 2001;**31**:1105–1115.

[67] Gilliam FS. Effects of harvesting on herbaceous layer diversity of a central Appalachian hardwood forest in West Virginia, USA. Forest Ecol Manag. 2002;**155**:33–43.

[68] Nagendo G, Stein A, Gelens M, Gier A, Albricht R. Quantifying differences in biodiversity between a tropical forest area and a grassland area subject to traditional burning. Forest Ecol Manag. 2002;**164**:109–120.

[69] Boudru M. Forêt et Sylviculture. Le traitement des forêts [Forest and silviculture. The treatment of forests]. Tome 2. Presses Agronomiques de Gembloux; 1989. 344 p. [in french].

[70] Mathews JD. Silvicultural Systems. Oxford: Claredon Press; 1989. 284 p.

[71] Smith DM, Larson BC, Kelty MJ, Ashton PMS. The practice of silviculture. Applied forest ecology. 9th Edition. New York: John Wiley & Sons, Inc; 1997. 560 p.

[72] Bartelink JJ, Olsthoorn M. Introduction: mixed forest in western Europe. In: Olsthoorn AFM, Bartelink HH, Gardiner JJ, Pretzsch H, Hekhuis HJ, Franc A, editors. Management of Mixed-Species Forest: Silviculture and Economics. DLO Institute for Forestry and Nature Research; 1999. pp. 9–16.

[73] Río Md, Sterba H. Comparing volume growth in pure and mixed stands of *Pinus sylvestris* and *Quercus pyrenaica*. Ann For Sci. 2009;**66**:502.

[74] Bayer D, Seifert S, Pretzsch H. Structural crown properties of Norway spruce [*Picea abies* (L.) Karst.] and European beech [*Fagus sylvatica* (L.)] in mixed versus pure stands revealed by terrestrial laser scanning. Trees. 2013;**27**(4):1035–1047.

[75] Dieler J, Pretzsch H. Morphological plasticity of European beech (*Fagus sylvatica* L.) in pure and mixed species stands. For Ecol Manag. 2013;**295**:97–108.

[76] Metz J, Seidel D, Schall P, Scheffer D, Schulze E, Ammer C. Crown modeling by terrestrial laser scanning as an approach to assess the effect of aboveground intra- and interspecific competition on tree growth. For Ecol Manag. 2013,**310**:275–288.

[77] Ducey MJ, Knapp R. A stand density index for complex mixed species forests in the northeastern United States. For Ecol Manag. 2010;**260**(9):1613–1622.

[78] Dirnberger GF, Sterba H. A comparison of different methods to estimate species proportions by area in mixed stands. For Syst. 2014;**23**(3):534–546.

[79] Sterba H, Río Md, Brunner A, Condes S. Effect of species proportion definition on the evaluation of growth in pure vs. mixed stands. For Syst. 2014;**23**(3):547–559.

[80] Boutin A, Haughland DL, Schieck J, Herbers J, Bayne E. A new approach to forest biodiversity monitoring in Canada. For Ecol Manag. 2009;**258**:S168–S175.

[81] Corona P, Chirici G, McRoberts RE, Winter S, Barbati A. Contribution of large-scale forest inventories to biodiversity assessment and monitoring. For Ecol Manag. 2011;**262** (11):2061–2069.

[82] McRoberts R, Tomppo E, Naesset E. Advanced and emerging issues on national forest inventories. Scand For Res. 2010;**25**:368–381.

[83] Henttonen HM, Kangas A. Optimal plot design in a multipurpose forest inventory. For Ecosyst. 2015;**2**:31. doi:10.1186/s40663-015-0055-2

[84] Pretzsch H. Canopy space filling and tree crown morphology in mixed-species stands compared with monocultures. For Ecol Manag. 2014;**327**:251–264.

[85] Pretzsch H. Forest Dynamics, Growth and Yield. From Measurement to Model. Berlin: Springer-Verlag; 2009. 664 p.

[86] Burkhart HE, Tomé M. Modelling Forest Trees and Stands. Dordrecht: Springer Science +Business Media; 2012. 457 p.

[87] Vidal C, Lanz A, Tomppo E, Schadauer K, Gschwantner T, di Cosmo L, Robert N. Establishing forest inventory reference definitions for forest and growing stock: a study towards common reporting. Silva Fennica. 2008;**42**(2):247–266.

[88] Eisfelder C, Kuenzer C, Dech S. Derivation of biomass information for semi-arid areas using remote-sensing data. Int J Remote Sens. 2012;**33**(9):2937–2984.

[89] Ke Y, Quackenbush LJ. A review of methods for automatic individual tree-crown detection and delineation from passive remote sensing. Int J Remote Sens. 2011;**32**(17):4725–4747.

[90] Lu D, Chen Q, Wang G, Liu L, Li G, Moran EA. A survey of remote sensing-based aboveground biomass estimation methods in forest ecosystems. Int J Digit Earth. 2016;**9**(1):63–105.

[91] White JC, Coops NC, Wulder MA, Vastaranta M, Hilker T, Tompalski P. Remote sensing technologies for enhancing forest inventories: a review. Can J Remote Sens. 2016. doi:10.1080/07038992.2016.1207484

[92] Gonçalves AC. Modelação de povoamentos adultos de pinheiro bravo com regeneração de folhosas na Serra da Lousã [Modelling of maritime pine adult stand with natural regeneration of broadleaved in Serra da Lousã]. Lisboa: Instituto Superior de Agronomia, Universidade Técnica de Lisboa; 2003. [in portuguese].

[93] IFN5. Instruções para o trabalho de campo do Inventário Florestal Nacional–IFN 2005/2006 [Instructions to field work of natural forest inventory–IFN 2005/2006]. Lisboa: Autoridade Florestal Nacional. 2005; 82 p. [in portuguese].

[94] Maroco J. Análise Estatística com Utilização do SPSS [Satistic analysis with the use of SPSS]. 3rd Edition. Lisboa: Edições Sílabo; 2007. 824 p. [in portuguese].

[95] R Development Core Team. R: A language and environment for statistical computing. R Foundation for Statistical Computing [Internet]. 2012. Available from: http://www.R-project.org. (Accessed 2013-02-21).

[96] Correia AV, Oliveira AC. Principais espécies florestais com interesse para Portugal: zonas de influência mediterrânica [Main forest species with interest for Portugal: zones of Atlantic influence]. Lisboa: DGF; 1999. 119 p. [in portuguese].

[97] Oliveira AC, Pereira JS, Correia AV. A silvicultura do pinheiro bravo [The silviculture of maritime pine]. Centro Pinus; 2000. 102 p. [in portuguese].

[98] Gonçalves AC, Oliveira AC. Regeneration in multi-species in Serra da Lousã. For Syst. 2011;**20**(3):444–452.

[99] Gonçalves AC, Oliveira AC, Dias SS. Evolution in multi-species high forest stands in Serra da Lousã: Diversity analysis. Silva Lusitana. 2010;**n° especial**:79–90.

Chapter 2

Effects of Experimental Sowing on Agroforestry Ecosystem Primary Production during Recovery from Agricultural Abandonment in a Semi-Arid Region of Central Western Spain

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Additional information is available at the end of the chapter

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Abstract

We present a study of plant species diversity manipulation on abandoned arable fields and show that most diversity effects can be explained by a successional shift from annuals to perennial plant species. We tested the hypothesis that plant mixtures consisted of mid-successional plant species that were expected to occur on the site following secondary succession, and an increase in the initial plant species diversity at the beginning of secondary succession improves the amount of biomass produced. The main aim was to compare the aboveground biomass for the whole plant community and for different functional groups, using 7 years of field data at abandoned arable land in a semi-arid region of Central Western Spain. Significant differences were established for the treatment-year interaction, analysing the perennial-annual species ratio (P:A) in the HD and LD of sowing treatments. The differences were established at the start of the experiment, when the sown species were more effective in the HD-sowing treatment. There was a negative relationship among the productivity and mean richness of the natural colonization (NC) and the analysed sowing treatments (HD and LD). The coefficient of determination of this relationship was significant ($R2 = 0.307$, $F(1, 13) = 5.75$, $P = 0.\ 032$).

Keywords: aboveground biomass, biodiversity, land-use change, secondary succession, plant species richness, plant functional groups

1. Introduction

In some parts of the world, such as in North America and Europe [1], there is an opposing trend, towards set-aside policies and the abandonment of agricultural land. Land abandonment provides opportunities to restore ecosystem properties such as biodiversity and biogeochemical cycles. However, the recovery of pre-agricultural soil conditions can be very slow (e.g. about 200 years for soil carbon and nitrogen [2]). The restoration of former species diversity is often constrained by abiotic and biotic conditions, such as eutrophization or seed-bank depletion [3]. In the mean time, and especially in the years just after abandonment, unmanaged land may favour nutrient leaching, constitute reservoirs of aggressive weeds damageable to adjacent fields and alter the aesthetics of the landscape.

Sampling effects may be the result of the enhanced chance of including species with a specific trait (e.g. high productivity) in species-rich compared to species-poor plant mixtures [4]. Sampling effects are demonstrated by varying the plant species composition of low-diversity (LD) treatments [5]. A positive impact of species diversity on, for example, primary productivity may be the result of resource-use complementarity allowing more diverse communities to utilize a larger proportion of the ecosystem resources [6]. In order to establish whether such overyielding occurs in mixtures of plant species, information on the productivity of the monocultures of all individual plant species needs to be included in the comparison [7]. Using the performance of individual plant species in high-diversity (HD) mixtures, their performance in low-diversity mixtures may be predicted, but this is not sufficient to separate the sampling effect and from that of resource-use complementarity [8].

In the debate on the relationship between biodiversity and ecosystem function relationships, the current focus on experimental procedures and the statistical interpretation of experimental data [9] means that many questions on the response of biodiversity to global changes are still wide open. One of the types of global change having the most immediate effects on biodiversity is that of land-use change [10, 11]. Agricultural expansion and intensification is a major component of such changes resulting in low-diversity ecosystems and large losses of carbon and nitrogen, affecting global carbon balance and fresh water quality, respectively [12, 13].

The effects of increasing plant diversity often saturate at rather low number of species (in average 90% of the known cases, the productivity of the most diverse treatment is reached with mixtures of five species [14]; note that these cases may be biased). Since by definition, functional differences are larger between functional groups than between species, functional group diversity has been found to have a larger impact on ecosystem processes than species diversity [9, 15, 16]. In these experiments, plant functional groups have been identified on the basis of species physiology (C3 vs. C4 species, N fixers vs. non-N fixers, woody vs. non-woody species) or life history (early- vs. late-season species, annuals vs. perennials). However, in removal studies, plant functional traits were found to have little impact on soil communities [17].

We tested the hypothesis that plant mixtures consisted of mid-successional plant species that were expected to occur on the site following secondary succession, and an increase in the initial plant species diversity at the start of secondary succession improves the amount of biomass produced. The aim of this study was to compare the aboveground biomass for the entire plant community and for various functional groups, using 7 years of field data at abandoned arable land in a semi-arid region of Central Western Spain.

2. Materials and methods

2.1. Site description

The study area was located at 850 m a.s.l., 15 km to the west of Salamanca city, Spain (its coordinates: 40° 54′ 00" N, 5° 45′ 30" W), where a 1-ha experimental plot was chosen at the Muñovela experimental farm (C.S.I.C). The plot is edaphically homogeneous, with a dehesa-like woodland.

The climate of the zone features rainy winters and hot summers and may be classified as semi-arid Mediterranean (C_1 B'_1 S_2 b'_4). Long-term mean rainfall and temperature are 500 mm and 12.3°C, respectively, with November being the rainiest month (99 mm) and July the driest (17 mm). January is normally the coldest month (0.8°C).

The tree covering comprises *Quercus rotundifolia* Lam, with a density of 98 trees ha^{-1}, a mean height of 5.9 m and a mean diameter of 29.1 cm. Chronologically, the area lies in the Mediterranean Region, Carpetano-Ibérico-Leonesa province, Salmantino sector and Genisto hystricis-Querceto-rotundifoliae sigmetum series. The estimated mean age of the trees was 150 years. Other characteristic species defining the series are *Dorycnium pentaphyllum* Scop., *Thymus zygis* L., *T. Mastichina* L. and *Crataegus monogyna* Jacq.

The soil is a chromic Luvisol [18], developed over red clays and Miocene conglomerates. Soil texture A/B is loam/clay. The slope of the plot is 2 %.

2.2. Experimental design

A field experiment was carried out on abandoned arable land with sown low- and high-diversity treatments and natural colonization (NC) following typical farming practice for the site. In April-May 1996, experimental plots were installed on former agricultural land that had been cropped with (crop rotation) monocultures until the end of 1995. The experiment was organized according to a block design with five replicate blocks. Within each block, four plots measuring 10 × 10 m were marked out and each of the three treatments was randomly assigned to one plot. All plots were separated by 2-m walkways. The three treatments (low-diversity-sown, high-diversity-sown and natural colonization) were randomly allocated to the plots in each block.

As the initial vegetation development at abandoned land is usually highly unstable and unpredictable, late-successional types of functional groups of plants were experimentally sown in both low- and high-diversity mixture. Based on the specific characteristics of plants, the functional groups most widely recognized in tempered grassland communities and which were used in this study are (1) grasses, (2) legumes and (3) other forbs. The low- and high-diversity mixtures comprised the same amounts of seed (grasses: 2500 seeds m^{-2}, legumes: 500 seeds m^{-2} and other forbs also 500 seeds m^{-2}). Fifteen species (five per functional group) were sown as the high-diversity-sown treatment. For the low-diversity-sown treatment, low-diversity seed mixtures (two grasses, one legume and one other forb species) were used as random choices from the total set of plants available for each replicate in order to take account of the sampling effects. The plant mixtures used consisted of species typical of later-successional stages (**Table 1**).

2.3. Measurements

Each year at peak-standing biomass (May/June), vegetation sampling biomass was clipped at 5 cm from the soil surface. In each replicate, a plot of 10 × 10 m and 12 subplots of 25 × 25 cm were harvested and the clipped material was sorted into litters and living parts (the standing

		LD1	LD2	LD3	LD4	LD5	HD
Grasses							
Bromus inermis Leyss.	BromIner		1250	1250			500
Festuca rubra L.	FestRubr	1250				1250	500
Phleum pratense L.	PhlePrat	1250	1250				500
Poa pratensis L.	PoaPrat				1250	1250	500
Poa trivialis L.	PoaTriv			1250	1250		500
Legumes							
Lotus corniculatus L.	LotuCorn	500					100
Medicago lupulina L.	MediLupu			500			100
Trifolium fragiferum L.	TrifFrag					500	100
Trifolium pratense L.	TrifPrat		500				100
Trifolium subterraneum L.	TrifSubt				500		100
Forbs							
Achillea millefolium L.	AchiMill			500			100
Galium verum L.	GaliVeru				500		100
Matricaria chamomilla L.	MatrCham					500	100
Plantago lanceolata L.	PlanLanc	500					100
Sanguisorba minor Scop.	SangMino		500				100

$P < 0.05$

Table 1. Density of sown seeds (seeds.m^{-2}) in the five blocks of low-diversity treatments (LD1–LD5) and in the plots of high-diversity treatments (HD).

biomass). In addition, the standing biomass was sorted into grasses, legumes and other forbs. All standing biomass was dried to constant weight at 80°C and weighed. Each year, the clipping was done in different subplots. In addition, in 12 permanent subplots of 1 m^2 each, adjacent to the clipped subplots, the number of plant species was counted and grouped into two life history classes: annual and perennial species.

2.4. Statistical analyses

The effects of the treatment on aboveground biomass using analyses of variance (ANOVA) with treatments and years as factors were analysed. Also, mixed ANOVAs (treatment, year and interactions among them) were carried out to relate the dominance of total number of species, annuals and perennials and for each functional group, in the NC, LD and HD treatments. Additionally, regression analyses were performed correlating biomass to the perennial-annual plant species ratio (P:A) and the number of species (S). The relationship established between biomass and diversity (P:A species ratio and the total number of species S) was examined using a stepwise multiple regression analysis and an analysis of covariance (ANCOVA) to separate the possible influence of the different diversity parameters P:A and S on biomass production.

Analysis of correspondence (CA) on the matrix of frequency of dominant species in the natural colonization, low-diversity and high-diversity treatments was also performed. A step-by-step multiple regression analysis was conducted to test the variability in the biomass and to explain the model of biomass, total number of species and total cover.

3. Results

3.1. Aboveground biomass production

The aboveground biomass follows a similar pattern among natural colonization and sowing treatments (HD and LD); therefore, the treatment-year interaction was not significant (**Table 2**). However, the annual differences emerge when comparing HD and LD separately, with the year 1998 showing these differences. Generally, the aboveground biomass tends to reduce across years of succession, but with a significant increase in 1998 with respect to other years, being favoured by the precipitations in the previous years (**Figure 1**).

When focusing on the aboveground biomass as stratified to functional groups, grasses were the functional group which had the most biomass in each treatment, except the first years, when the forb group had the most amount of biomass. However, the great increase of biomass in 1998 was mainly due to the contribution of grasses, with their significant differences for the treatment-year interaction (**Table 2**). This was supported for the high relation between total biomass and grasses biomass ($R^2 = 0.72$; $F_{(1, 103)} = 262.99$; $P < 0.000$). This group began to be the most important from 1998 and had an influence on the total functionality of the ecosystem. The relationship between forb biomass and total biomass although significant was smaller ($R^2 = 0.37$; $F_{(1, 68)} = 40.12$; $P < 0.000$); this value shows that the forb species

Variables	Treatment	Year	Tr*Y	HD-LD	Year	HD-LD
		Aboveground biomass				
Total biomass	0.268	**0.000**	0.344	0.197	**0.000**	**0.012**
Grass biomass	0.553	**0.000**	0.526	0.369	**0.000**	**0.037**
Forb biomass	**0.998**	**0.000**	**0.019**	0.992	**0.000**	0.059
Legume biomass	0.645	**0.039**	0.976	0.619	0.126	0.961

Table 2. Mixed analysis of variance of three factors (treatment, year and interactions among them).

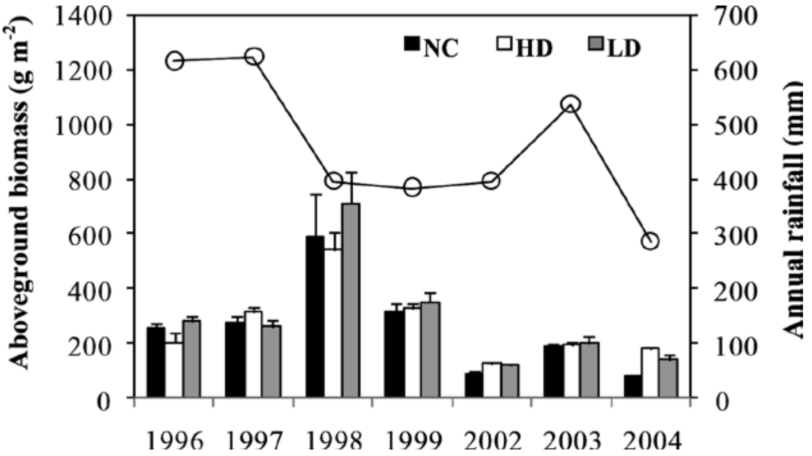

Figure 1. Aboveground biomass (g.m^{-2}) in the natural colonization (NC), low-diversity (LD) and high-diversity (HD) treatments for 7 years studies is compared.

were important through 1998, when they were substituted by better competitor grass species (**Figure 2**), grasses sowing in HD and grasses colonizing in LD and NC.

The cover of sown species was defined by 60% of the total biomass in HD ($R^2 = 0.60$; $F_{(1, 33)} = 48.54$; $P < 0.000$); meanwhile, this only explains the 15% in LD ($R^2 = 0.15$; $F_{(1, 33)} = 5.62$; $P < 0.024$). Comparing the relationship between total cover (sown species and colonizing species) and the total biomass in each treatment separately, there were important annual variations, which only explained the 37% of the biomass variability in HD ($R^2 = 0.37$; $F_{(1, 33)} = 19.74$; $P < 0.000$) and the 48% in LD ($R^2 = 0.48$; $F_{(1, 33)} = 30.70$; $P < 0.000$).

The sowing effect did not affect the total biomass, neither the functional groups nor the agreement of block performance suppressing colonizing species. The best block was observed in LD (blocks 2 and 3) (**Figure 3**), which did not establish an obvious increase in the biomass of the dominant sown species (*Bromus inermis*). In the same way, there was no significant correlation between the total biomass and the total number of colonizing species ($R^2 = 0.017$, $F_{(1, 68)} = 1.15$, $P = 0.28$).

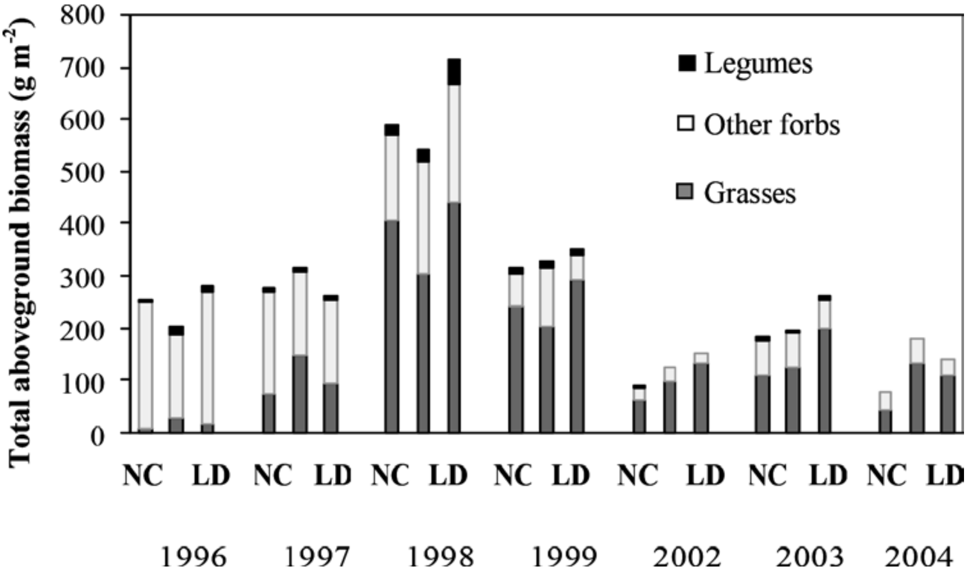

Figure 2. Aboveground biomass (g.m^{-2}) for different functional groups in the natural colonization (NC), low-diversity (LD) and high-diversity (HD) treatments for the 7 studied years.

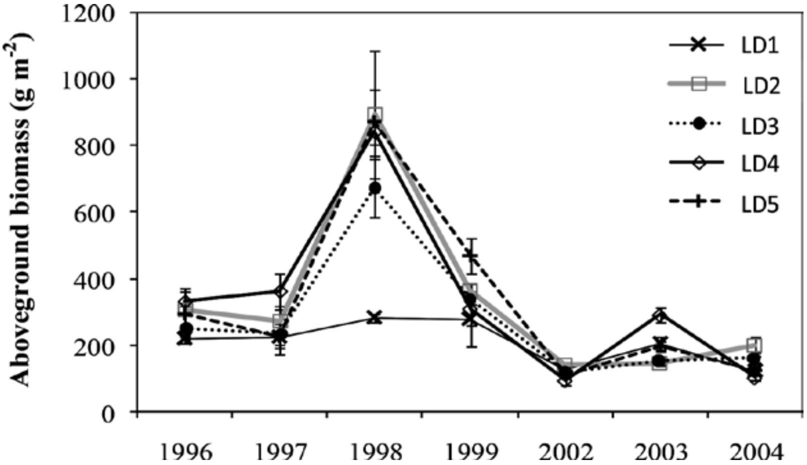

Figure 3. Aboveground biomass (g.m^{-2}) in the different blocks of the low diversity (LD) for the 7 studied years.

3.2. Achieved number of plant species and perennial/annual ratio on biomass production (P:A)

In the HD and LD of sowing treatments, significant differences were established for the treatment-year interaction, analysing the perennial-annual species ratio (P:A) ($F = 3.78$, 6 d.f., $P > 0.009$). The differences were established at the start of the experiment, when the sown species were more effective in the HD-sowing treatment (**Figure 4**). From 2002, when the cover and the number of sown species decrease, these differences disappear, whereas there was a relative importance of annual species in HD from 2002.

The annual species followed the same pattern in the two sowing treatments ($F = 1.75$, 6 d.f. $P = 0.15$), although there were annual differences at the beginning of the experiment (**Figure 5**).

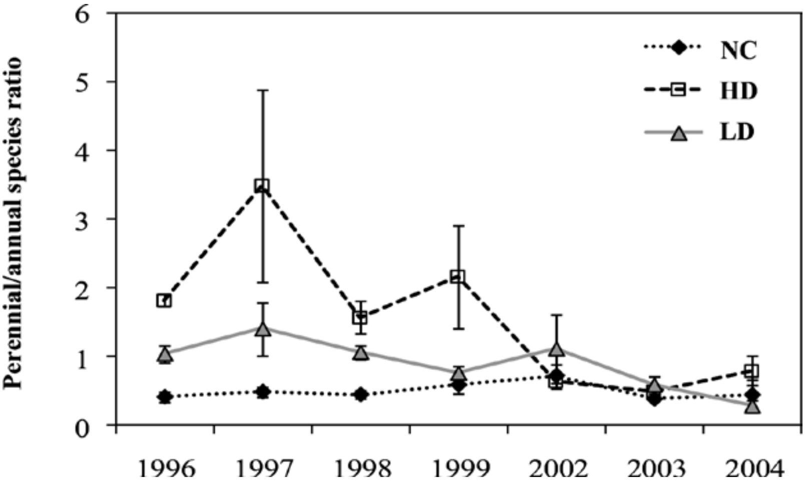

Figure 4. Perennial/annual species ratio in the natural colonization (NC), low-diversity (LD) and high-diversity (HD) treatments for the 7 studied years.

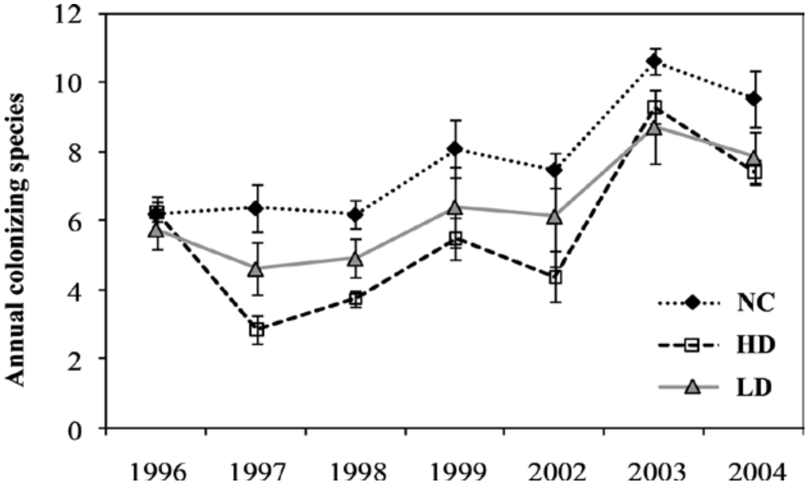

Figure 5. Annual colonizing number of species in the natural colonization (NC), low-diversity (LD) and high-diversity (HD) treatments for the 7 studied years.

3.3. Dominance model of experimentally established community

The dominance of the species in the natural colonization, low-diversity and high-diversity treatments is indicated in **Table 3** and **Figure 6**. Both sowing treatments (HD) and (LD) differed significantly from the natural colonization in terms of the total number of dominant species, except the dominance of annual species (**Table 3**). In 1999, there was a great increase in the number of annual dominant species, both in HD and in LD treatments, which prevent the significance of this interaction (**Figure 6**).

The dominance of the annual species followed the same pattern like that of the total dominance. In both variables, there were no significant differences between treatments or treatments-year interaction (**Table 3**). In the same way, the dominance of annual species explained the 74% of total dominance variability showed for the treatments during all sampling period ($R^2 = 0.74$, $F_{(1, 68)} = 196.60$, $P < 0.000$), while the dominance of perennial species only explained the 48% ($R^2 = 0.48$, $F_{(1, 68)} = 63.88$, $P < 0.000$) and showed significant differences among treatment-year; therefore, the sowing effect had no negative effect on beta-diversity, despite the fact that there were two sown species clearly dominant in both treatments (*B. inermis* and *Sanguisorba minor*). There was a positive relationship ($r = 0.83$) between the total dominance of the community of both treatments with the Shannon index, which shows that increasing this index will increase the beta-diversity ($R^2 = 0.69$, $F_{(1, 68)} = 149.11$, $P < 0.000$), while there was a negative relationship with dominance indexes of Berger-Parker ($r = -0.66$; $R^2 = 0.44$, $F_{(1, 68)} = 55.04$, $P < 0.000$) and Simpson ($r = -0.70$; $R^2 = 0.49$, $F_{(1, 68)} = 65.99$, $P < 0.000$), indicating that increasing the dominant species will not decrease the diversity.

The sowing effect was a little relevant in implanting or suppressing the perennial-colonizer species. Nevertheless, during the sowing period where there was a greater number of sown species in HD treatment, the perennial-colonizer species had a lower success than in LD or NC (**Figure 7**). Although there were a few perennial-colonizer species in natural conditions

Variables	Treatment	Year	Tr*Y	LD-HD	Y	LD-HD
Total dominance	0.308	**0.000**	**0.018**	0.813	**0.000**	0.506
Annual species dominance	**0.053**	**0.000**	0.204	0.185	**0.000**	0.526
Perennial species dominance	0.164	**0.000**	**0.000**	**0.006**	**0.003**	**0.013**
Grasses dominance	**0.047**	**0.000**	**0.000**	0.098	**0.000**	**0.000**
Forbs dominance	0.180	**0.000**	**0.008**	0.217	**0.000**	**0.041**
Legumes dominance	0.201	**0.003**	**0.000**	0.233	**0.000**	**0.000**

$P < 0.05$.

Table 3. Mixed analysis of variance of three factors (treatment, year and interactions among them).

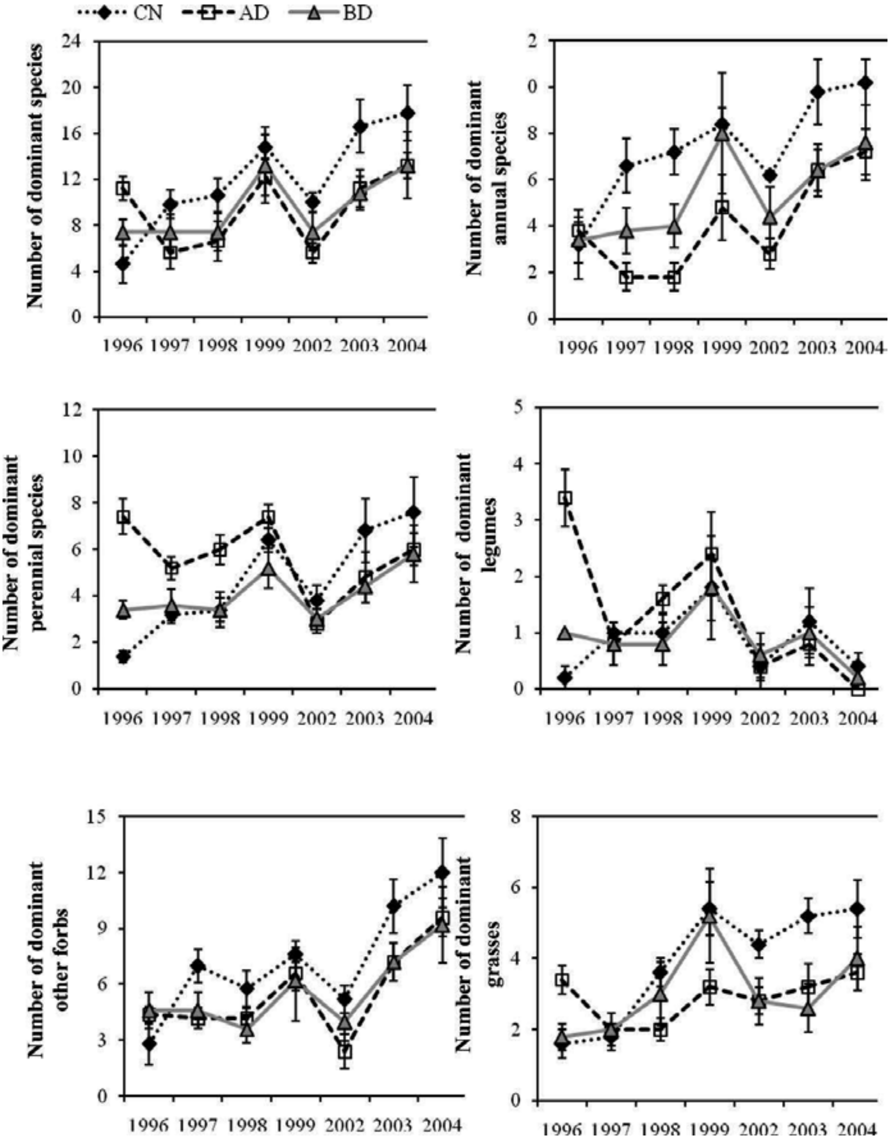

Figure 6. Total dominant number of species, annuals, perennials and for each functional group in the natural colonization, low diversity and high diversity treatments.

at the end of the experiment, all species belong to the forb group: C*arduus tenuiflorus* Curtis, *Chondrilla juncea* L., *Convolvulus arvensis* L. and *Echium plantagineum* L. (**Figure 7**).

The sowing effects on the dominance of colonizer species are indicated by a correspondence analysis (**Figure 8**). The first axis explains the 11% variability of species composition and there was a very high correlation ($r = 0.97$) among species and the explanatory variables of this composition. These values indicate that the treatment-year interaction may be explained for this horizontal axis with a high percentage, for the high number of species involved in this analysis.

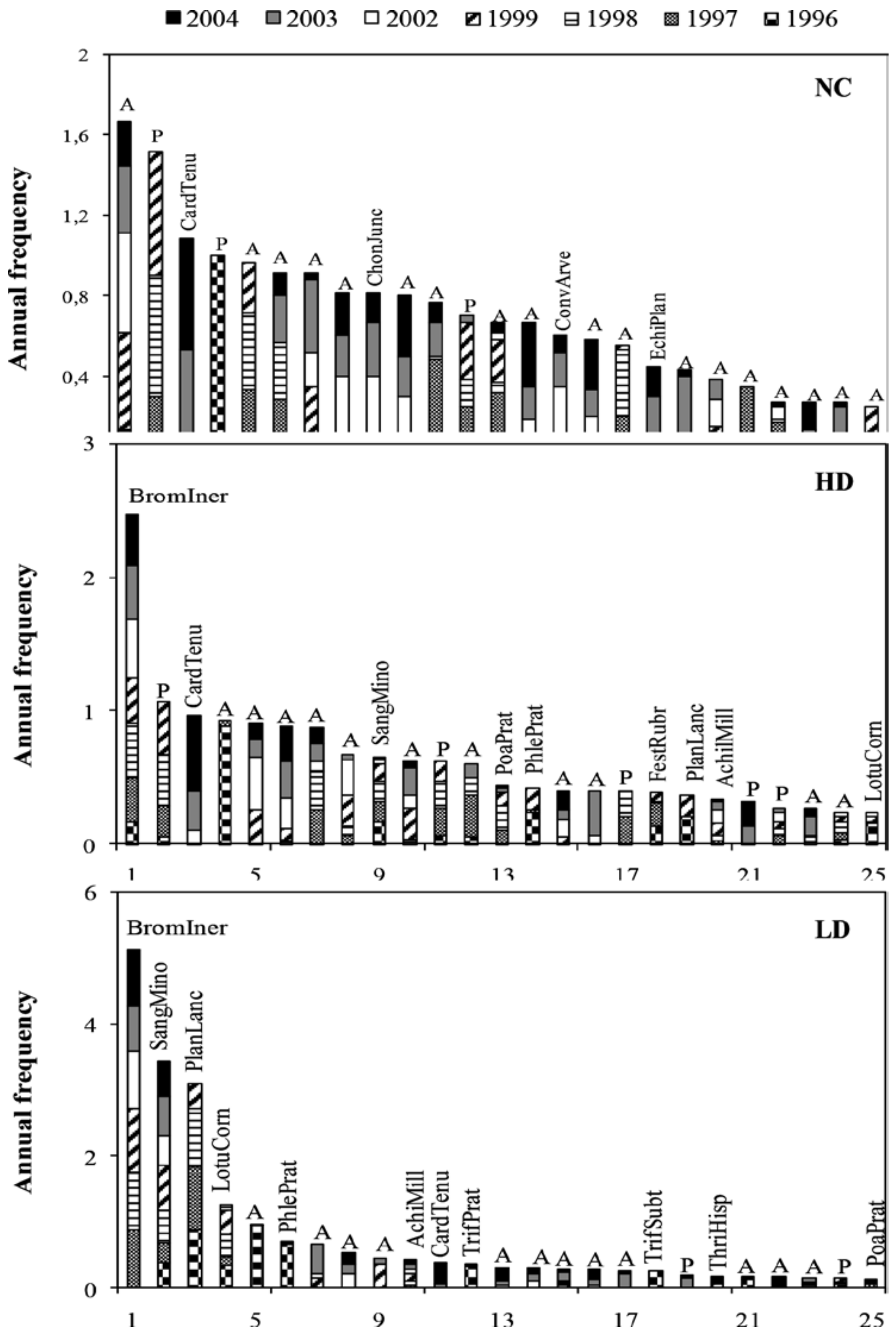

Figure 7. Annual frequency of species appearance in the natural colonization (NC), low-diversity (LD) and high-diversity (HD) treatments. Only the species with dominance >3% are indicated.

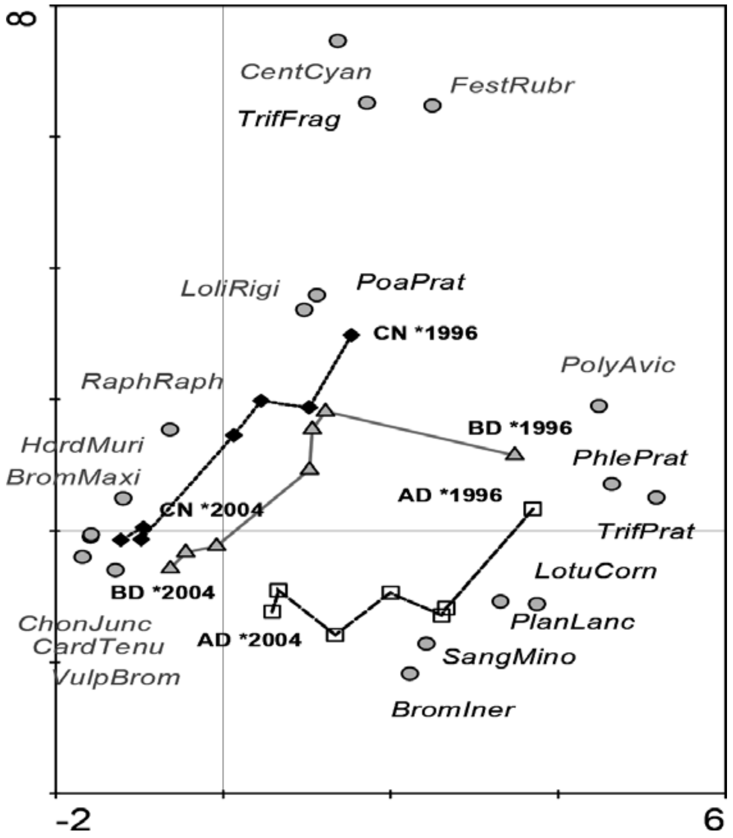

Figure 8. Analysis of correspondence (CA) on the frequency matrix of the dominant species in the natural colonization (NC), low-diversity (LD) and high-diversity (HD) treatments.

3.4. Temporal change of the aboveground biomass

There is a negative relationship between the productivity and mean richness of the natural colonization and the analysed sowing treatments (HD and LD). The coefficient of determination of this relationship was significant (R^2 = 0.307, $F_{(1,\ 13)}$ = 5.75, P = 0.032) (**Figure 9**). The species richness influenced with a very short percentage on the biomass production, with the species composition being more important, directly influencing certain functional groups. Thus, the biomass of forb group may be explained at about 55%, due to the total cover presented for all species in both sowing treatments (R^2 = 0.43, $F_{(1,\ 68)}$ = 52.11, P < 0.000), but the biomass of the grasses was independent of this variable (R^2 = 0.08, $F_{(1,\ 68)}$ = 5.79, P = 0.019).

In a step-by-step multiple regression analysis, these variables explain the 61% of the variability in the biomass of forb group (**Table 4**), and only two explain the model of biomass, total number of species and total cover. The contribution of species richness (value BETA) had a greater and negative effect on the biomass of forbs than on the total cover in natural conditions.

The biomass in HD treatment responded positively to the cover of sown species, while in LD treatment, this effect was null and clearly showed the effect of cover on all grouped species (**Table 5**).

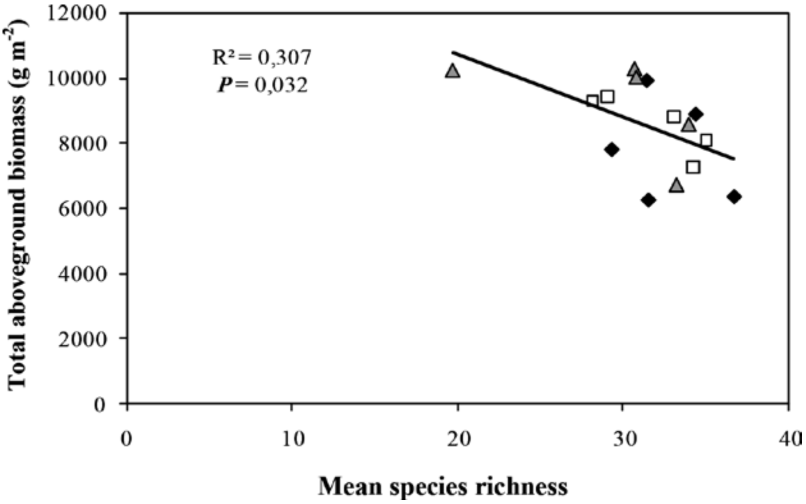

Figure 9. Correlation among mean species richness and aboveground biomass from natural colonization (NC), Low-diversity sown (LD) and high-diversity sown (HD) for the 7 studied years.

R = 0.78; R^2 = 0.61; R^2-adjusted = 0.59; $F_{(2, 32)}$ = 25.03; P < 0.000

MODEL	*B*	BETA	*t*	*P*
Total number of species	−22.850	−0.735	−6.224	**0.000**
Total number of species	−21.593	−0.694	−6.218	**0.000**
Total cover	1.910	0.268	2.396	**0.023**

BETA: Partial coefficients of standardized regression; *B*: regression coefficients not standardized; *t*: to prove the null hypothesis $B_i = 0$; *P* ($\alpha = 0.05$)

Table 4. Multiple regression model of the aboveground biomass for the forbs group, explained for the defined variables from the number and coverage species shown in the natural colonization treatment.

HD (R = 0.81; R^2 = 0.65; R^2 adjusted = 0.63; $F_{(2, 32)}$ = 29.49 P < 0.000)

MODEL	*c*	BETA	*t*	*P*
Shown species coverage	6.350	0.772	6.967	**0.000**
Shown species coverage	5.128	0.623	4.993	**0.000**
Total coverage	3.185	0.274	2.195	**0.036**

LD (R = 0.69; R^2 = 0.48; R^2 adjusted = 0.47; $F_{(1, 33)}$ = 30.70 P < 0.000)

MODEL		BETA	*t*	*P*
Total coverage	12.367	0.694	5.541	**0.000**

BETA: Partial coefficients of standardized regression; *B*: regression coefficients not standardized; *t*: to prove the null hypothesis $B_i = 0$; *P* ($\alpha = 0.05$)

Table 5. Multiple regression model of the aboveground biomass for the forbs group, explained for the defined variables from the number and coverage of species shown in the low and high treatments.

The high correlation among the sum of variance and the sum of covariance also indicated a stabilization effect in the temporal change where there was a greater amount of perennial species. In the HD treatment, when the sown species were more successful, the smaller fluctuations on the biomass change with the number of perennial species (**Figure 10**). There was a positive relationship between the total biomass and the sum of variance (**Figure 11**), indicating inter-annual changes inside and between treatments, due to fluctuations of dominant species.

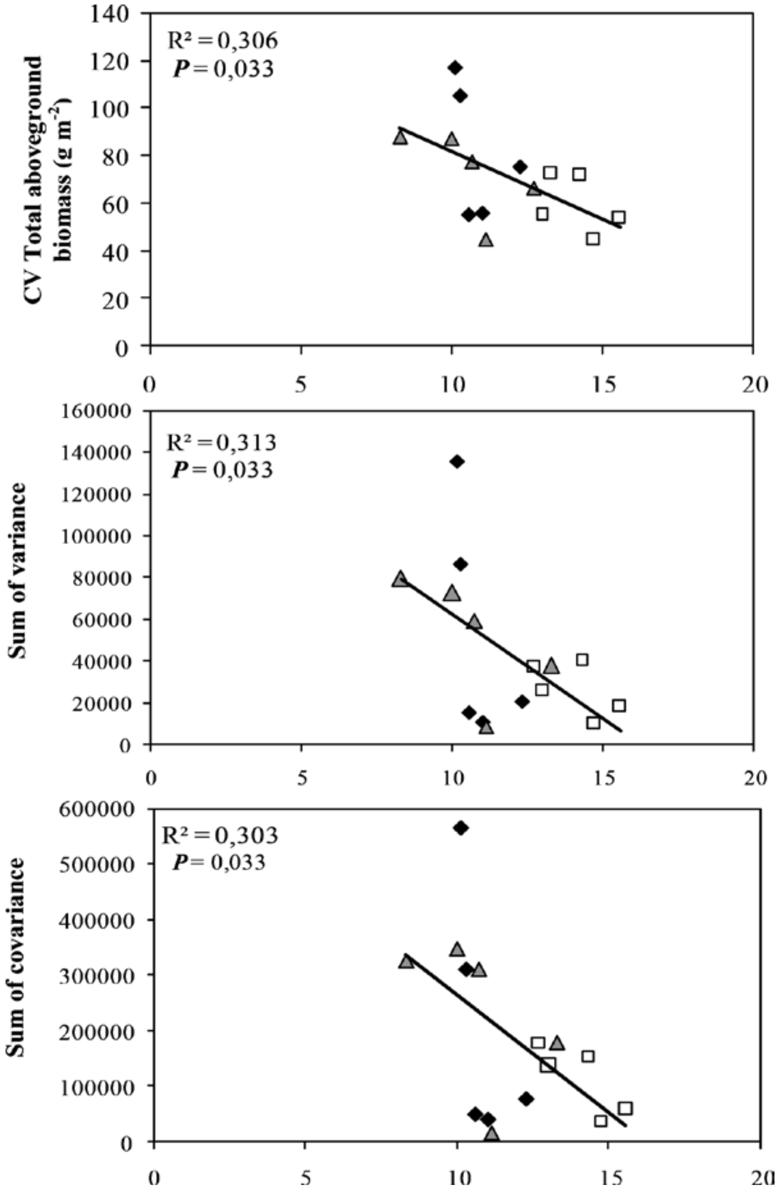

Figure 10. Correlation among the perennial number of species and the biomass temporal change and the sum of variances and covariances, in the natural colonization (NC), low-diversity (LD) and high-diversity (HD) treatments for the 7 studied years.

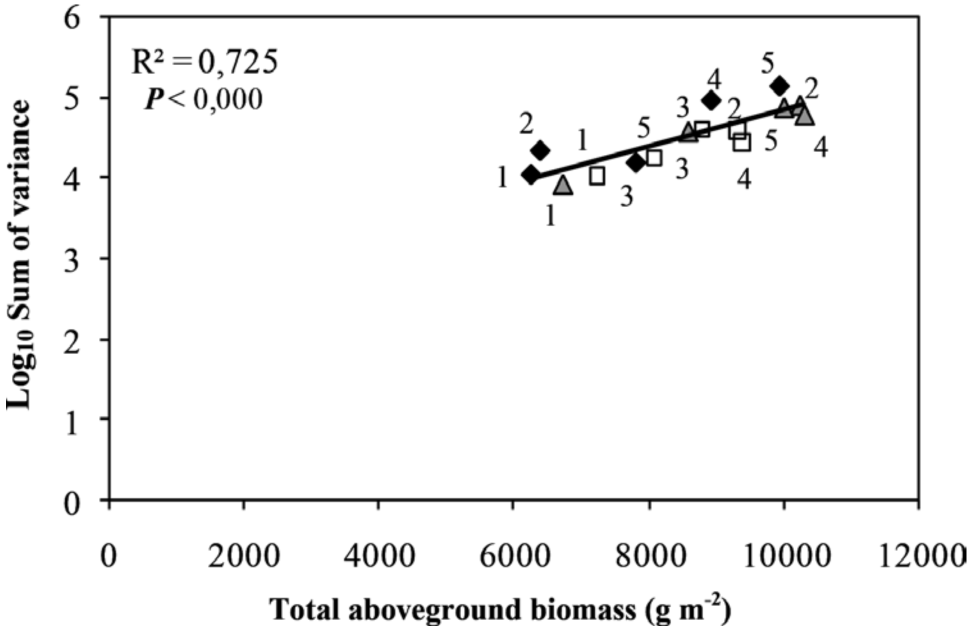

Figure 11. Correlation among the perennial number of species and the biomass temporal change of biomass and the sum of variances and covariances, in the natural colonization (NC), low-diversity (LD) and high-diversity (HD) treatments for the 7 studied years.

4. Discussion

4.1. Aboveground biomass production

The highest plant biomass in 1998 was found in low-diversity plots. All treatments had a similar pattern, displaying an important increase of both the grass and secondarily the forb biomass. The treatment-year interaction was not significant. The differences were established when considering HD and LD treatments separately. The forb biomass increased only under diversity treatments (LD and HD). The effect of the dominant species development could explain both the greater plant biomass yield in LD plots, where the number of sown plant species was less in HD plots, while the highest grass biomass yield was in NC (some sown and natural grass species are strong competitors). Our aboveground biomass values were greater than those obtained by Puerto et al. [19] in nine different types of dehesa, whose values ranged between 150 and 200 g.m^{-2}, and similar to values found by Pecó et al. [20], 150–250 g.m^{-2}, in Mediterranean semi-arid environments.

The contribution of grasses is known to increase as a result of sowing and temporal development when managing grassland communities [21]. The influence of functional group traits on productivity has been reviewed by Díaz and Cabido [22] who found a high number of field studies rather to be correlated to functional group traits than to species diversity. When comparing the two sown treatments, the aboveground biomass was not affected by the number of sown species in each functional group, like in naturally colonized treatments where there are higher probabilities of one functional group missing or being present in a low proportion.

A greater productivity shown in both qualitative ways was that plots with initial diversity were intently increased by sowing different species. Plots where more species were sown are those that eventually had higher biomass; however, the higher productivity for 1998 was not accompanied by the higher diversity. In addition to inherent production capabilities of each species, the evenness of the distribution (the relative proportion of the component species) can bear on productivity [23], especially the relative contribution of dominants and codominants. So, where fast-growing herbs are able to exercise competitive dominance, it is unusual to find exceedingly low floristic diversity; the resulting community may be closer to monoculture [24].

There were no significant differences considering treatment-year interaction when comparing HD and LD separately for the total number of dominant species or for the annual total of dominant species. Therefore, a similar pattern of both treatments during part and all period of experimentation could be expected. In this way, at the beginning of the experiment in HD, the dominance of annual species decreases significantly, but these differences disappeared with time. This behaviour is in connection with the dominance of perennial species, being highest in HD treatment through the year 1999, that is, in the same period of time, the dominant increase of perennial species (sown and colonizers) has a negative influence on the dominance of annual species, but only in this treatment. Then, the dominance of perennial species fell over minimum in HD, allowing the equalization of the same value than in LD, while the number of annual dominant species increases again. Firstly, this effect was important during the first period of experimentation when the sown species were performing better as cover. From 2002, many species disappeared or decreased its cover and only a few species were seen.

The effectiveness of both treatments in deleting annual species was clear, since there was a negative relationship ($r = -0.64$) between the cover of sown species and the number of annual dominant species ($R^2 = -0.41$, $F_{(1,\ 68)} = 47.68$, $P < 0.000$). However, the elimination of annual dominant species was not involved, at the same time the sowing has provided the implantation of colonizer perennial species. At the end of the experimentation, the HD, LD and NC seemed to converge again and the sowing effect tends to disappear with time. The sowing effect was comparatively different in HD treatment, due to the combined action of the greater number of sown species in the initial stages of experiment. During all periods, the LD and NC had a similar pattern, although the influence of sown species in those LD blocks is clear, where the performance of species in determined mixtures was successful. Considering total plant species richness, the observed patterns were in disagreement with the hypothesis that increased productivity results from higher species diversity in grasslands [9].

4.2. Enhancement of productivity and resource capture along secondary succession

The increase in aboveground biomass with grassland succession is in agreement with earlier results for mesic environments [25]. This trend is assumed to be related to successional changes in the traits of the dominant plant species. A gradual increase in the contribution of grasses as a result of sowing and temporal development was established when biomass was sorted into grasses, legumes and non-legume forbs.

During the course of secondary succession, there was a change in the number of species as well in productivity. In general, plant diversity decreases while productivity increases [26].

These trends are found in several studies [27]. The manipulations of the succession assay include the introduction of later-successional plant species, mainly perennial species. As a consequence of this manipulation, the course of succession has been affected as a result of the change in the dominance of vegetation pattern. Leps et al. [8] from a study on the first 4 years of the succession concluded that the floristic composition changed more rapidly than the performance of the dominant species. The effect of the sown diversity treatments was the rearrangement of most seasonally variable community [28] of early-successional species by weed suppression to a community when grasses and perennials dominated. The effects of species richness may be only effective in narrow degrees of the functional composition of communities [29], due to combination of some factors such as the species identity, effects of species attributes [30] and in the lower degree, as the complementary of resources used [31].

The poor relationship found between the perennial and annual species ratio (P:A) and the aboveground biomass in HD treatment ($R^2 = 0.18$, $F_{(1,\,33)} = 7.34$, $P = 0.011$) and in BD ($R^2 = 0.14$, $F_{(1,\,33)} = 5.32$, $P = 0.027$) indicated the limited contribution of perennial species to the aboveground biomass in the sowing treatments and the poor relationship between species richness and productivity.

4.3. Temporal change of the aboveground biomass

The diversity has no positive effect on aboveground biomass in the three treatments: HD, LD and NC. The results of temporal change in the aboveground biomass may confirm the null answer of the aboveground biomass to changes of diversity in the sowing treatments. However, the results confirmed the negative effect of diversity change on system productivity. There are sampling effects that may partially explain the total biomass [4]. This effect predicts that the total biomass was determined by the increase of probability which comprise a particular species in a community. Species such as the perennial grass *Lolium rigidum* or the forbs *Matricaria inodora, Polygonum aviculare* or *Plantago coronopus*, which were dominant in natural conditions, were removed from the sown species, particularly in LD treatment.

In general, the sowing effect caused destabilization in the system, increasing the number of species that may coexist in the established communities, at the expense of the biomass, that is, a negative productivity-diversity relationship. The HD treatment where there was more effective sowing effect exhibited a lower variation among blocks than the remaining treatments. Analysing the biomass by functional groups, a negative relationship was found in NC which was also valid for the forb group.

There was no significant relationship found between the temporal change of the aboveground biomass and mean species richness in each plot-treatment relationship ($R^2 = 0.012$, $F_{(1,\,13)} = 0.154$, $P < 0.701$). However, the temporal change of the aboveground biomass was positively related with the number of perennial present species (sown and natural colonizer) ($R^2 = 0.306$, $F_{(1,\,13)} = 5.723$, $P = 0.033$). This may indicate that this effect was mainly due to the group of perennial sown species.

The sown species negatively influenced the cover of colonizer-dominant species in HD treatment, allowing a redistribution of colonizer species taking a greater degree of competition.

This reflected a lower variability of the temporal change of the aboveground biomass because there was no dominant species in natural conditions. The sowing effect did not increase the species richness, but the uniformity of abundant colonizer species was increased. There was a diversity effect on the variability of established communities depending on the variation of biomass among dominant species or another species.

4.4. Vegetation composition effects: a negative productivity-diversity relationship

The biomass pattern cannot be explained by the hypothesis that more diverse plant communities are more productive [32]. Indeed, clear relationships have been found for biomass with the relative abundance of perennial species; this is taken as the mean number of perennial species/mean number of annual species ratio.

From the analysis of data, there were no great differences regarding the number of perennial species along the years. These considerations recommended the use of an index relating both groups of species (P:A); it was once observed that the relative abundance of perennial number of species is what changes along the years, and in doing so, the vegetation pattern observed could be explained satisfactorily. In fact, our data suggest that there was a change in the dominance pattern after sowing, so a higher P:A and a general trend towards decreasing richness were reached from the sown treatments.

The functional groups differ in both phenology and rooting depth implying a complementary exploitation of habitat; competitive interactions in mixture may have a strong effect on total plant biomass. In fact, the sampling effect and resource complementary, either in time or in space, may operate simultaneously [15].

As the productivity-diversity relationship was negative, the biomass of forb group was also influenced negatively by the richness species in the plots, showing that the functional diversity also had an effect on productivity and the efficiency of capture resources [33].

Sowing mixtures of later-successional plant species may enhance the initial functional diversity of plant species by stimulating the establishment of mid-successional perennials. This may have undesirable side effects through the introduction of alien genotypes, but it enhances the exclusion of arable weeds [5], and high-diversity mixtures enhance the reliability of ecosystem functions [8]. Introduction of mid-successional plant species stimulates the restoring of ecosystem processes, such as efficient light capture (LAI) and light utilization (photosynthesis), and it affects aboveground insect assemblages [34]. However, the short-term effects of the experimental treatments on different trophic groups of nematodes [35] and on microbial activity [36] are limited. In fact, in the first 2 years, the effect of ending agricultural practices alone had much larger effects on the soil community than any of the experimental manipulations of the plant assemblages [34, 36].

5. Conclusions

The aboveground biomass follows a similar pattern among natural colonization and sowing treatments (HD and LD); therefore, the treatment-year interaction was not significant.

However, the annual differences appear when comparing HD and LD separately, with the year 1998 showing these differences. In general, the aboveground biomass tends to decrease across years of succession, but with a significant increase in 1998 with respect to other years, being favoured by the precipitations in the previous years.

The sowing effect did not affect the total biomass, neither the functional groups nor the agreement of block performance suppressing colonizing species. The best block was observed in LD (blocks 2 and 3), which did not establish an obvious increase in the biomass of the dominant sown species (*B. inermis*). In the same way, there was no significant correlation among the total biomass and the total number of colonizing species.

Significant differences were established for the treatment-year interaction, analysing the perennial-annual species ratio (P:A) in the HD and LD of sowing treatments. The differences were established at the beginning of the experiment, when the sown species were more effective in the HD-sowing treatment. From 2002, when the cover and the number of sown species decrease, these differences disappear, whereas there was a relative importance of annual species in HD from 2002.

There is a negative relationship between the productivity and mean richness of the natural colonization and the analysed sowing treatments (HD and LD). The coefficient of determination of this relationship was significant ($R^2 = 0.307$, $F_{(1,13)} = 5.75$, $P = 0.032$). The species richness influenced with a very short percentage on the biomass production, being more important the species composition influencing directly on certain functional groups. Thus, the biomass of forb group may be explained at about 55%, due to the total cover presented for all species in both sowing treatments ($R^2 = 0.43$, $F_{(1,68)} = 52.11$, $P < 0.000$), but the biomass of the grasses was independent of this variable ($R^2 = 0.08$, $F_{(1,68)} = 5.79$, $P = 0.019$).

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References

[1] Richards JF. Land transformation. *In* "The earth as transformed by human action" (B. L. Turner II, W. C. Clark, R. W. Kates, J. F. Richards, J. T. Mathews and W. B. Meyer, eds.), 1990; pp. 163–178. Cambridge University Press, Cambridge.

[2] Knops JMH & Tilman D. Dynamics of soil nitrogen and carbon accumulation for 61 years after agricultural abandonment. Ecology, 2000; 81, 88–98.

[3] Bakker JP, & Berendse F. Constraints in the restoration of ecological diversity in grassland and heathland communities. Trends in Ecology and Evolution, 1999; 14, 63–68.

[4] Huston MA. Hidden treatments in ecological experiments: re-evaluating the ecosystem function of biodiversity. Oecologia, 1997; 110: 449–460.

[5] van der Putten WH, Mortimer SR, Hedlund K, Van Dijk C, Brown VK, Leps J, Rodriguez-Barrueco C, Roy J, Len TAD, Gormsen D, Korthals GW, Lavorel S, Regina IS, Smilauer P. Plant species diversity as a driver of early succession in abandoned fields: a multi-site approach. Oecologia, 2000; 124: 91–99.

[6] Loreau M. Separating sampling and other effects in biodiversity experiments. Oikos, 1998; 82, 600–602.

[7] Huston MA. Biological diversity, the coexistence of species on changing landscapes. 1994. Cambridge University Press, Cambridge.

[8] Leps J, Brown VK, Diaz-Len TA, Gormsen D, Hedlund K, Kailova J, Korthals GW, Mortimer SR, Rodriguez-Barrueco C, Roy J, Regina IS, van Dijk C, van der Putten WH. Separating the chance effect from other diversity effects in the functioning of plant communities. Oikos, 2001; 92(1): 123–134.

[9] Hector A, Schmid B, Beierkuhnlein C, Caldeira MC, Diemer M, Dimitrakopoulos PG, Finn JA, Freitas H, Giller PS, Good J, Harris R, Hogberg P, Huss-Danell K, Joshi J, Jumpponen A, Korner C, Leadley PW, Loreau M, Minns A, Mulder CPH, O'Donovan G, Otway SJ, Pereira JS, Prinz A, Read DJ, Scherer-Lorenzen M, Schulze ED, Siamantziouras ASD, Spehn EM, Terry AC, Troumbis AY, Woodward FI. Yachi S, Lawton JH. Plant diversity and productivity experiments in European grasslands. Science, 1999; 286: 1123–1127.

[10] Vitousek PM., Mooney HA., Lubchenco J & Melillo JM. Human domination of earth's ecosystems. Science, 1997; 277, 494–499.

[11] Sala OE, Chapin FS, Armesto JJ, Berlow E, Bloomfield J, Dirzo R, Huber-Sanwald E, Huenneke LF, Jackson RB, Kinzig A, Leemans R, Lodge DM, Mooney HA, Oesterheld M, Poff NL, Sykes MT, Walker BH, Walker M, Wall DH. Global biodiversity scenarios for the year 2100. Science, 2000; 287: 1770–1774.

[12] Houghton RA. The worldwide extent of land-use change. BioScience, 1994; 44, 305–313.

[13] Matson PA, Parton WJ, Power AG, & Swift MJ. Agricultural intensification and ecosystem properties. Science, 1997; 277, 504–509.

[14] Roy J. How does biodiversity control primary productivity ? *In* "Global terrestrial productivity: past, present and future" (H. A. Mooney, B. Saugier & J. Roy, eds.), Academic Press, San Diego. 2001; 169–186.

[15] Hooper DU. The role of complementarity and competition in ecosystem responses to variation in plant diversity. Ecology, 1998; 79: 704–719.

[16] Tilman D, Knops J, Wedin D, Reich P, Ritchie M, & Sieman E. The influence of functional diversity and composition on ecosystem processes. Science, 1997; 277, 1300–1302.

[17] Wardle DA, Bonner KI, Barker GM, Yeates GW, Nicholson KS, Bardgett RD, Watson RN, Ghani A. Plant removals in perennial grassland: Vegetation dynamics, decomposers, soil biodiversity, and ecosystem properties. Ecological Monographs, 1999; 69(4): 535–568.

[18] FAO. The revised legend: FAO/UNESCO: Soil map of the world. FAO, Rome 1989.

[19] Puerto A, Rico M, Matías MD, García JA. Variation in structure and diversity in Mediterranean grasslands related to trophic status and grazing intensity. Journal of Vegetation Science, 1990; 1: 445–452.

[20] Pecó B, Sánchez AM, Azcárate FM. Abandonment in grazing systems: Consequences for vegetation and soil. Agriculture, Ecosystems and Environments, 2006; 113: 284–294.

[21] Kosola KR, Gross KL. Resource competition and suppression of plants colonizing early successional old fields. Oecologia, 1999; 118: 69–75.

[22] Díaz S, Cabido M. Vive la difference: plant functional diversity matters to ecosystem processes. Trends in Ecology and Evolution, 2001; 16: 646–655.

[23] Nijs I, Roy J. How important are species richness, species evenness and interspecific differences to productivity? A mathematical model. Oikos, 2000; 88(1): 57–66.

[24] Grime JP. Biodiversity and ecosystem function: the debate deepens. Science, 1987; 277: 1260–1261.

[25] Gleeson SK, Tilman D. Allocation and the transient dynamics of succession on poor soils. Ecology, 1990; 71: 1144–1155.

[26] Montalvo J, Casado MA, Levassor C, Pineda FP. Species diversity patterns in Mediterranean grasslands. Journal of Vegetation Science, 1993; 4: 213–222.

[27] Brown VK, Southwood TRE. Secondary succession: patterns and strategies. In: "Colonization, succession, and stability" (Gray, A.J., M.J. Crawley y D.J. Edwards, eds.), pp. 315–337. Blackwell Publishing, Oxford, UK; 1987.

[28] Odum EP. The strategy of the ecosystem development. Science, 1969; 164: 262–270.

[29] Allison GW. The implications of experimental design for biodiversity manipulations. American Naturalist, 1999; 153(1): 26–45.

[30] Symstad AJ, Tilman D, Willson J, Knops JMH. Species loss and ecosystem functioning: effects of species identity and community composition. Oikos, 1998; 81(2): 389–427.

[31] McNaughton SJ. Biodiversity and function of grazing ecosystems. In: “Biodiversity and ecosystem function” (Schulze, E.D. & H.A. Mooney, eds.), pp. 361–383. Springer-Verlag, Berlin, Alemania. 1993.

[32] Grime JP. Benefits of plant diversity to ecosystems: immediate, filter and founder effects. Journal of Ecology, 1998; 86: 902–910.

[33] Jiang XL, Zhang WG, Wang G. Effects of different components of diversity on productivity in artificial plant communities. Ecology Restoration, 2007; 22: 629–634.

[34] Mortimer SR, Booth RG, Harris SJ, Brown VK. Effects of initial site management on the Coleoptera assemblages colonising newly established chalk grassland on ex-arable land. Biological Conservation, 2002; 104: 301–313.

[35] Korthals G W., Smilauer P, Van Dijk C & Van der Putten, W. H. Linking above- and below-ground biodiversity: abundance and trophic complexity in soil as a response to experimental plant communities on abandoned arable land. Functional Ecology, 2001;15: 506–514.

[36] Maly S, Korthals GW, Van Dijk C. et al. Effects of vegetation manipulation of abandoned arable land on soil microbial properties. Biology and Fertility of Soils, 2000; 31: 121–127.

Chapter 3

Methodological Considerations in the Study of Earthworms in Forest Ecosystems

Dylan Rhea-Fournier and Grizelle González

Additional information is available at the end of the chapter

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Abstract

Decades of studies have shown that soil macrofauna, especially earthworms, play dominant engineering roles in soils, affecting physical, chemical, and biological components of ecosystems. Quantifying these effects would allow crucial improvement in biogeochemical budgets and modeling, predicting response of land use and disturbance, and could be applied to bioremediation efforts. Effective methods of manipulating earthworm communities in the field are needed to accompany laboratory microcosm studies to calculate their net function in natural systems and to isolate specific mechanisms. This chapter reviews laboratory and field methods for enumerating and manipulating earthworm populations, as well as approaches toward quantifying their influences on soil processes and biogeochemical cycling.

Keywords: earthworms, lumbricids, soil fauna, ecosystem engineer, soil methods, faunal manipulations, arthropod exclusions, soil microcosms, electroshock

1. Introduction

The impact of earthworms on soil dynamics can be defined as changes in physical characteristics, microbial activity, and nutrient chemical conditions. However, these processes are interconnected to an extent while attempting to separate them can prove difficult. Physical effects of earthworms can be attributed to their feeding and burrowing behavior. Initial contact of earthworms with litter detritus or crop residue is often by comminution or fragmentation, which in effect reduces the size of both organic and mineral particles [1]. This increases the surface area for soil fauna and microbes to act upon. Individual soil microbes are limited by the inability to relocate and actively seek new substrates, thus their effect on the actual rate of chemical transformations may be more regulated by mechanisms that bring them in contact with new organic substrates than the total amount of substrate available [2], **Figure 1**.

While the soil microbial biomass is directly responsible for the majority of biogeochemical cycling and nutrient mineralization in soils (at least 90%), often the players that link such activities to higher spatial scales through organization and activation, such as roots and soil invertebrates, are largely ignored [2]. Earthworm casts are biogenic structures produced as a result of gut passage, mixing organic and mineral soils. The consequences of this aggregate formation can be physical in nature, including increased drainage and moisture loading capacity [1]. Both permanent and evanescent burrows, sometimes reinforced with protein-based mucus, can promote soil porosity and thus aeration, reducing anaerobic

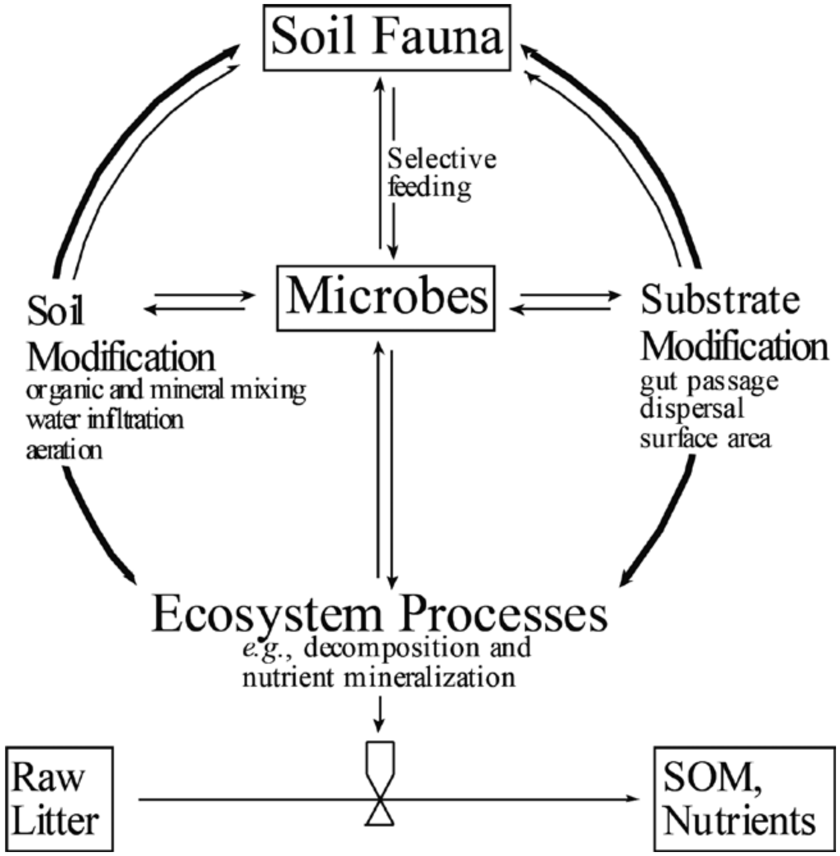

Figure 1. Conceptual model illustrating direct and indirect pathways of interactions between soil fauna, microbes, soil physical properties, substrate, and ecosystem processes [44].

conditions and increasing gaseous exchange, and thus promoting microbial activity, as well as infiltration [1, 3].

Earthworm populations differ significantly in terms of numbers, biomass, and diversity across the regions in which they are found. Population sizes are often determined by readily available organic matter, as well as soil type, pH, moisture capacity, precipitation, and ambient temperatures [4]. In most soils, earthworm biomass exceeds that of all other soil invertebrates [1].

There have been several efforts to standardize earthworm sampling across biomes [5]. The basic approach to identify the influences and mechanisms by which earthworms influence soil systems and detrital food webs is most often pursued via controlled experiments comparing earthworm containing soils to those void of them (controls). Most often this is achieved by expelling target fauna from a sample of soil. This can be no simple task working in the "black box" that is the domain of soils all the while striving for minimal disturbance in important soil properties such as structure, pedology, and faunal composition.

This chapter will serve as a review of the methodologies applied in past experiments, field work, and modeling efforts involving the influence of earthworms in forest soils. We will remind soil researchers of the plethora of challenges faced in soil research and argue that no singular method or tool is a panacea to the difficulties that may arise. This review provides a perspective into faunal experiments and a toolbox of techniques and approaches to evaluate and quantify the influences of lumbricids in terrestrial environments.

2. Microcosm laboratory and mesocosm field experiments

A traditional, effective method in studying the myriad of influences of earthworms on soil ecosystems is the laboratory microcosm. The concept of the microcosm is to recreate a miniaturized version of the ecosystem understudy in controlled lab settings in order to control all variables possible that are not those under question. Most commonly for soil and earthworm studies these microcosms consist of a plastic PVC cylinder ranging between 6 and 16 cm in diameter and 15–50 cm deep, but other materials such as plexiglass containers [6] or glass jars with perforated lids [7, 8] have been utilized to incubate between 75 and 150 g of soil substrates. These effectively act as experimental soil cores. Relatively larger controlled soil environs, known as mesocosms, can be placed in the field to subject the closed system to more natural climate conditions. These can be made of buckets with perforated bottoms (25 cm diameter, 8 kg soil [9]) or clay pots (4.5 L), which have the ability to maintain a desired moisture regime [10].

Usually a metal, plastic, nylon [11], or fiberglass [12] mesh of 1 mm [13] to 2 mm [6] is placed on the top and/or bottom of containers to prevent escape (or colonization in mesocosms) of earthworms as well as retain soil and moisture. For microcosms, these soil cores can be taken from the field intact [14, 15] by hammering the cylinder into the ground, removing, and returning to the lab for observation. Alternatively, microcosms can be filled with homogenized soils (often also gathered from the field) after being passed through a 2.5–4 mm sieve [12, 16–18]; however, this may destroy preexisting soil aggregate structure. Other substrates have been used in earthworm microcosms such as pig manure [8].

Soils used to fill microcosms may be gathered from a range of ecosystems in which earthworms may be found: tropical, temperate, agroecosystem, grassland/savannah, or forest. Recognizing the importance of pedology, Lachnicht et al. [19] sampled both the O- and A-horizon to incorporate into their mesocosms from the tropical forest soil they aimed to simulate. Alternatively, open bottomed microcosms can be returned to the field after manipulation [9, 11, 20, 21] and buried to retain field climate conditions in "field microcosms," more akin to the mesocosm approach.

In laboratory microcosms, a polyethylene film can be placed on the bottom of the cores to prevent leaching [14]. If leachate chemistry is under investigation, ceramic lysimeters can be installed and drained under semi natural conditions (−200 to −400 hPa [18]). Alternatively, if soil moisture chemistry is under study, microcosms can be capped with ceramic plates and atmospheric pressure reduced (−0.5 atm) to collect soil solution otherwise bound by capillary forces [17, 22]. Costello and Lamberti [23] use passive soil water percolation and collection of leachate to add to a stream mesocosm to assess periphyton growth in an elaborate simulated riparian study and effect of soil fauna on stream inputs.

A variant of the mesocosm experiment is the greenhouse pot or bag experiment, which involves established vegetation in a contained soil pedon or core; often exploited in investigating earthworm effects on plant growth or allocation [24, 25]. Similarly, plants can be added to the mesocosms previously described [13].

Most microcosm/mesocosm studies involve the inoculation of the substrate with the desired earthworm species (or functional group), community composition, density, and biomass for the study, often with multiple treatments. Climate conditions can be easily controlled in laboratory or greenhouse settings in addition to avoiding predation on earthworms. A common practice is to allow earthworms to void gut contents for 36–48 hours before being added to microcosms [15, 19, 25] to prohibit influences from outside origins or substrate, especially in studies involving isotopes.

Stable isotope (^{15}N and/or ^{13}C) labeled crop residue can be applied to mesocosms to track the assimilation of substrate and to discern from soil organic matter in earthworm tissues [7, 19, 26–29]. Microcosm experiments can range from 72 hours [8] to 12 days [7] to 16 weeks [12], to 120 days for the plant-pot experiments [24]. However, there is debate over the duration of microcosm experiments. Whalen et al. [26] argue less than a week for studies concerning excretion rates using ^{15}N to avoid reingestion and help discern between structurally incorporated N and excretions. Artificial earthworm burrows have even been created to compare abiotic and biotic influences of the burrows created by anecic species [12, 30].

To identify and isolate the effects solely of the internal gastrointestinal ecology of earthworms, Barois and Lavelle [31] dissected *Pontoscolex corethrurus* individuals sampled from an agricultural field, observing and comparing soil in the anterior, middle, and posterior thirds of the gut-deducing changes in physicochemical and respiration properties of soil during gut transit. In a laboratory setup, Horn et al. [32] sedated earthworms with 40% ethanol, laying them horizontally and partially embedded in 1.5% agarose to implant microsensors. They then could study the internal earthworm gut environment to analyze the activation of nitrous oxide-producing bacteria.

Various methods have been applied to manipulate preexisting earthworm communities in the soils used as substrates in the micro/mesocosm experiments before controlled inoculations are performed. Many of these are not possible in field experiments. Homogenized or sieved soils are usually depleted of earthworms and their cocoons via hand-sorting [14, 25]. However, with intact cores where preservation of soil structure is desired, different methods may be needed. Fonte et al. [11] use a modified electroshock technique (discussed here later, mainly for field experiments) for earthworm extraction. Willems et al. [14] use both the modified electroshock method and heat of 40°C for 48 hours to kill any remaining individuals or cocoons in intact cores. Defaunation by Butenschoen et al. [18] was achieved by freezing soil at −28°C for 2 weeks, followed by a week preincubation for microbial recovery. Similarly, after sieving (5 mm), Lenoir et al. [33] froze soil samples at −20°C for 3 days in an attempt to kill most meso- and macrofauna, noting that many microfauna (nematodes, rotifers, protozoans, tardigrades, and microarthropod eggs) survived. Huang et al. [20] froze intact PVC cores taken from the field (−30C°) for 48 hours to kill earthworms. After sieving, Alphei et al. [34] subjected soils for used in their mesocosms to two chloroform fumigation cycles, reestablishing microbiota with unfumigated soil slurries. Postma-Blaauw et al. [28] irradiated soil with γ-radiation (25 kGy) to sterilize, previous to reinoculating with microbiota and adding to mesocosms. Alternatively, soil can be sterilized with methyl bromide [34]. Soils can also be dried in the shade for a period of time [24] or autoclaved [25] to eliminate earthworm cocoons. In all of the above methods, effects on nontarget organisms must be considered and reinoculation of microbiota is often necessary. Soil organisms are extraordinarily diverse, spanning the three domains; the loss of key species or functional groups may affect interacting species and thus potentially large changes in the soil ecosystem processes. Additionally, the unintended contribution of dead earthworms or other fauna to soil organic matter must be considered in these methods that fail to remove fauna and kill them in place.

Barot et al. [35] criticize the use of micro- and mesocosms, as soils are usually homogenized—disrupting soil horizons, and partially defaunated. Carpenter [36] reviews microcosm experiments, arguing the usefulness and applicability of this approach; listing rapid results at relatively low costs, ease of replication and repetition, and ensuing statistical advantages. This allows enhanced power over experimental controls, testing specific mechanism hypotheses, and deriving rate estimates. Indeed high level of replication and control over abiotic factors can be desirable in variable heterogeneous medium such as soils. However, Carpenter [36] cautions of the danger of losing context and appropriate scale both spatially and temporally, leading to possible distortions in community and ecosystem considerations. In general, microcosms can be an indirect way to study ecology: “Without the context of appropriately scaled field studies, microcosm experiments become irrelevant and diversionary” [36]. Taking a computer model approach, Barot et al. [35] expand microcosm studies to predict long-term earthworm effects on primary production; however, they emphasize field studies must follow to confirm model-based conclusions.

3. Field manipulations

While laboratory experiments can be useful for examining specific mechanisms and chemical pathways, in lieu of the discussion above, it is apparent that field experiments are necessary to confer results before large-scale extrapolation. Different techniques can be utilized to remove or expel earthworms from manipulated plots. It must be recognized that no one method is likely to completely exclude earthworms, especially in longer-term studies, thus treatments can often be considered as reductions rather than eliminations or exclusions. To maintain earthworm treatments, investigators regularly install physical barriers to define experimental soil plots. Parmelee et al. [37] installed plexiglass enclosures to 25 cm depth and 30 cm above ground in their agroecosystem field experiment. A common practice is to bury plastic sheets (PVC) to 45–50 cm with 10–15 cm above ground to restrict lateral movements of earthworms in both agroecosystems [38–41] and forests [42]. Alternatively, plastic garden edging (to ~20 cm [23]) in a forest or corrugated plastic to 5 cm depth and 25 cm above ground [43] in an agriculture field can be used; however, this may be an inadequate depth to control all earthworm movement. González et al. [44] noted that the garden liner they used in combination with an aluminum fence (15 cm above ground) allowed both oxygen and water exchange to either side of the barrier; an important detail that other studies fail to address.

3.1. Faunacides

3.1.1. Naphthalene

Naphthalene is a general repellent for arthropod communities and has thus been widely applied to studies seeking fauna contributions to decomposition. However, it is unknown and rarely reported what effects naphthalene has on earthworms. González and Seastedt [45] do report that earthworms were found in plots that had been hand-sorted to 20 cm, lined with weed/garden liner and treated with naphthalene. Heneghan et al. [46] report over 58% reduction in arthropods in their temperate vs. tropical system study. Naphthalene was found not effective in reducing arthropod abundance compared to control plots in a tropical dry forest where numbers were already low [44]. In addition to the lack of specificity and knowledge concerning effectiveness with earthworms, naphthalene is known to affect microbial communities. For example, González et al. [44] found no net change in total microbial biomass, yet the abundance of salicylate mineralizers specifically was enhanced in naphthalene applications, having implications for lignin degradation and increased immobilization of nutrients in the microbial biomass. Work by Blair et al. [47] found nontarget effects of naphthalene where its application in soil-litter mesocosms directly affected both microbial abundance and activity. Additionally, it appeared that microbes utilized naphthalene as a carbon source, illustrated by increased soil respiration rates. Furthermore, naphthalene treatments drove net nitrogen mineralization compared to net immobilization in controls. Based on this brief collection of findings, naphthalene cannot be advised for application for earthworm reduction nor general faunal exclusion, especially when biogeochemical pools and fluxes are concerned.

3.1.2. Carbofuran

Carbofuran (or carbofuradan or carbamate, 2,3-dihydro- 2,2-dimethyl 1,7-benzofuranyl methylcarbamate) can be applied to eliminate soil fauna [48], in amounts and frequencies ranging from once at 0.41 g m^{-2} in field microcosms [39] to twice over 6 months at 25 g m^{-2} in 4 m^2 field enclosures [37] prior to earthworm inoculation. Alegre et al. [49] report natural elimination of the faunacide after 4 weeks, although questions of lasting nontarget effects and subsequent influences on investigative results remain.

Gilot [39] reports incomplete elimination of earthworms with a single application of carbofuran. While they specifically observed earthworm casts to assess significant differences in earthworm treatments and organic inputs on soil structure, they fail to caution interpretations in the light of nontarget effects of the carbofuran nor do they report earthworm establishment success.

Broadbent and Tomlin [50] report no significant differences in earthworm mean biomass between carbofuran-treated and control plots in an agroecosystem. However, their study found that a broadcast application of carbofuran was more effective than row application. They concluded that in this cultivated field experiment carbofuran application affected the short term, but not long-term community of soil decomposers.

Parmelee et al. [37] applied carbofuran as a vermicide in an agroecosystems field study. They report that initial earthworm reductions were greater in their no-till treatment (79%) compared to conventional till. However, after 286 days both no-till (98%) and conventional till (100%) had greatly reduced earthworm abundance in carbofuran-treated plots compared to controls. Conversely, this study, unlike many others in the past, assessed the nontarget effects of the vermicide treatment. Densities of microarthropod, enchytraeids, nematodes, and bacteria were reduced in at least some of the carbofuran-treated litterbags on some dates. However, these effects neither were neither consistent over time or till treatments nor reflected in litter decay rates, making them impossible to correct for in final calculations. The authors, therefore, stress that the OM processing rates contributed to earthworms in this study are a potential maximum and may be overestimated due to confounding effects on nontarget biota.

In a review and synthesis of the target and nontarget effects of applied biocides, Ingham [51] reports that in addition to earthworms, carbofuran reduces populations of beetles, weevils, assorted borers, nematodes (of various functional groups), springtails, and Rhizobium (at high concentration applications) and can alter fungal dominance 1 year after application. Recognizing that earthworms can play an ecological engineering role yet still remain a single component of the complex hierarchical detritivore food-web, when attempting to tease out the role of earthworms themselves researchers should strive to avoid the nontarget effects of such faunacides as carbofuran.

3.2. Passive methods

Relative to application of faunacides, more passive methods for obtaining soil within the sphere of earthworm influence (drilosphere) exist. The burrows of anecic earthworms by definition open to the soil surface, providing visual evidence allowing investigators to sample burrow soil [52, 53]. Alternatively, the presence of surface-casting species provides investigators with a visual cue to compare drilosphere and nondrilosphere soil by sampling the casts themselves (to compare microfauna of riparian and pasture soils [54]; or to subject to simulated rainfall [55]) or underlying soil [56]. Other visual indicators can be exploited in Northern temperate forests where patches of soil invaded with exotic earthworms contrast greatly with earthworm-free patches where thick organic horizons remain [57, 58]. A quite different passive approach employed by Lavelle et al. [59] is utilizing the difference in size of soil aggregates and the casts of geophagous earthworms by passing soil through a 2 mm mesh sieve to exclude casts. This technique worked in this study looking specifically at casts; however, the effect of earthworms extends beyond the casts themselves. For direct study, fresh casts can be obtained by lightly squeezing on the posterior end of a collected or raised worm [31, 59]. While working in a sensitive area that prohibited addition or extraction of any elements to the soil system, Nuzzo et al. [60] utilized the practice of artificial cover. Similar to herpetofaunal studies, placing "cover boards" made of untreated, rough-cut lumber, on top of the soil and checking them every 2–3 weeks allowed them to estimate community composition, returning individuals after sampling. This method is completely reliant on earthworm activity near the surface (thus biased towards epigeic species) but may be a better indicator of active biomass than total earthworm biomass derived with other methods. A similar passive method, pitfall traps are used for litter and detritus-dwelling mesofauna and have been seen limited application for the collection of epigeic species in Northern temperate forests [61]. While decreased disturbance on the soil system under study is often greatly desired, the aforementioned methods have limited applicability.

3.2.1. Litterbags

Litterbags are a commonly applied technique for decomposition studies. Along with the physical-chemical environment, biota, and substrate quality are the driving factors of decomposition [44, 62]. Soil ecologists have utilized litterbags filled with preweighed material and placed in the field to study these factors. Not only can litter of different chemistry be applied and placed in different sites in transplant experiments [63], but also the mesh size of litterbags can vary, limiting which size class fauna (body diameter) can access them. Indeed a functional definition of litter and soil micro-, meso-, and macrofauna is by body diameter (45 μm, 1 and 5 mm, respectively; [64]).

Filley et al. [65] determined the use of 1 mm mesh litterbags was sufficient to exclude all macrofauna in their study including earthworms in northern temperate deciduous forests on a successional spectrum. In a comparison study between Northern deciduous and conifer forests in Colorado, González et al. [63] used litterbags of two different mesh sizes to quantify the effects of different groups of soil fauna on the decay of aspen leaves and lodgepole pine needles. They found litterbags with the small mesh size (1.8 × 1.6 mm) did not inhibit the activities of litter microarthropods but excluded macroarthropod effects on decay. Therefore, by using this technique on comparing sites, they were able to show that both litter microarthropods and the macrofauna are important determinants of decay in the lodgepole pine forest. Although litter decomposition in these subalpine sites might be influenced differently by various groups of soil and litter fauna [63].

Barajas-Guzmán and Alvarez-Sánchez [66] used litterbags of 1 and 6 mm mesh in a tropical rainforest experiment to assess faunal contribution to decomposition rates; however, the authors do not specify which particular fauna were excluded. In a humid tropical forest in southwestern China, Yang and Chen [67] use litterbags of 2 mm mesh to allow most macro-, meso-, and microfauna, while 0.15 mm mesh was applied to exclude most macro- and mesofauna. In their application of litterbags in a subtropical wet forest canopy trimming experiment, Richardson et al. [68] found that relatively smaller (0.475 mm) compared to larger (1.8 mm) mesh sizes only influenced the abundance and biomass of microarthopods by excluding larger organisms, but not causing major changes in taxonomic composition. Still, in the same experiment, González et al. [69] found negative correlations between mesh size and percent mass loss in litterbags, and between the Margalef index of diversity for the litter arthropods contained in the litterbags and the percent mass loss, suggesting that functional complexity is an important determinant of decay in their forest.

Litterbags are effective in excluding fauna of different size classes, but not specifically earthworms when other fauna in the same size class (ants, termites, millipedes, centipedes) are present. In longer-term experiments, one could presume that the non-epigeic (litter-feeding) species could still obtain access to the substrate on the underside of the litterbag where products of humification have begun to accumulate or fungal hyphae have colonized. However, initial fragmenting and communition are often the central mechanisms addressed in these studies. González and Seastedt [45] and González et al. [62] use mesh of 1.8 × 1.6 mm, as to not inhibit indirect effects of earthworm casts. Suárez et al. [58] considered possibilities of faunal restriction in their litter decomposition experiment concerning earthworms by using litter boxes instead of bags to eliminate faunal constraints.

Litterbags can be used in conjunction with earthworm exclusion techniques discussed (carbofuran [37]; electroshock [42]; sieving [44]; utilizing mosaic landscape of earthworm-free, and invaded patches [64]). In a combination of the field placed mesocosm and the mesh litterbag for exclusion methods, Cortez et al. [70, 71] buried cylinders horizontally in the soil with two sized mesh treatments. Mesh of 0.5 cm allowed the entry and passage of earthworms, while 0.1 cm mesh prohibited earthworm access, feeding, and influence on the substrate contained inside.

Further criticism of the litterbag technique is the often lack of acknowledgement that some mass loss can be attributed to physical leaching and subsequent transportation, not just direct mineralization [64, 72]. However, this can partially be taken into account by sampling at day 0 of the experiment. Knowledge of the detritivore community present must be applied when considering use and mesh size of litterbags in field decomposition experiments [64].

3.2.2. Earthworm additions

The addition of earthworms to experimental field plots, similar to that of meso/microcosms, would appear a sound approach to unearth the influences of earthworms, either in plots previously purged of or still containing established populations. Butt [73] reviews earthworm addition methods for the purpose of bioremediation in the United Kingdom, listing several factors that may lead to unsuccessful establishment in earthworm addition treatments. When extracting individuals using a repulsion technique such as formalin (discussed here later) and then broadcasting on the surface of the soil, survival may be limited by the exposure of the animals to predation and desiccation. The possibility also exists of the extraction method harming the individuals (and thus adversely affecting survivability), being biased to anecic species, and/or the difficulty in transferring cocoons. Some of these potential restrictions can be ameliorated if earthworms are placed beneath the soil surface; however, this will introduce a soil disturbance that may be undesired in many studies considering soil structure and microbial biomass, especially in the short-term. Butt [73] stresses the importance of site history, origin, and life history of earthworm species used. He suggests the ideal method for earthworm inoculation involves a "starter culture" allowed to develop in a bag of soil for a few months so that inoculums include adults, cocoons, and hatchlings. However, his review is in consideration of harsh or disturbed environments that need remediation such as landfills.

After eliminating previously existing populations via electroshock (discussed later), Costello and Lamberti [23] removed surface litter and lightly aerated the upper 5 cm of soil to assist the introduction of earthworms into 0.25 m^2 plots. After a 30-day field incubation, the earthworms were again removed via electroshock, allowing assessment of addition success. They concluded that the endogeic *Aporrectodea caliginosa* (sometimes labelled "endoanecic" [60]) were more successful in establishment after addition than for the anecic *Lumbricus terrestris* and *Lumbricus* spp. juveniles. The removal of litter and disturbance of upper soil may have aided earthworm entry and establishment but may be undesirable in smaller scale studies (temporally or spatially) where soil structure and horizons are considered.

Subler et al. [43] applied earthworm addition treatments (100 m^{-1}) to two temperate agroecosystems, comparing them to controls. In this study, earthworms were collected for additions by formalin extraction, thus yielding primary *L. terrestris* individuals. Following the five-month duration of the experiment earthworm abundances were sampled via hand-sorting and formalin (both discussed here later). Despite heavy inoculation numbers, no significant difference in earthworm abundance was found between addition and control treatments at experiment termination. However, while no taxonomic nor functional group information was provided in this study, the addition treatments were found to have significantly fewer "surface-dwelling" species and greater "deep-burrowing" species compared to the control plots. Considering the behavior of anecic species to remove surface organic matter and incorporate it into lower soil horizons in their burrows and these findings, possible competition between anecics and epigeic earthworms is possible.

Similarly, Shuster et al. [74] employed earthworms extracted from nearby no-till cornfield via formalin repulsion in addition treatments and compared to ambient populations. Each spring and fall 100 individuals m^{-2} were added to each addition treatment. Epigeic earthworm abundances were reduced in response to additions, possibly due to removal of surface residue by anecics. These findings caused the investigators to question whether earthworm populations ever reached a steady state during their experiment. The authors discuss how added earthworms on top of an existing population could increase burrow and forage activity in the face of limited resources. Furthermore, they suggest much less anecics could be added to recreate a situation that occurs in natural systems of few anecics in coexistence with other functional groups. They also question the appropriateness of adding anecics to floodplain soil, as they may not be well adapted for the constant flooding and high water tables in these environments. This discussion should lead to caution and concern in the application of earthworm addition treatments in field experiments and in the interpretation of the results derived from these artificially forced communities, especially where OM resources can become competitive.

3.2.3. *Tagging*

Capture-mark-recapture programs have seen wide spread use for terrestrial, aquatic, and volant megafauna for decades, providing vital information on life-span, reproduction, site fidelity, migrations, and habitat use of individuals which can be extrapolated to the population level. While this type of data would be incredibly useful for soil macrofauna, it has not been feasible in the past considering the toxicity of some marking methods, or the size and thus impairment to locomotion of most tags. However, some subcutaneous marker dyes show promise. Recently, nontoxic marker tags made from visible implant elastomers (VIE) with fluorescent properties were applied to earthworm mark-recapture studies with success in *L. terrestris* in a laboratory microcosm [75], and with *P. corethrurus* in field mesocosms in a tropical pasture and wet forest [76]. This method has since been applied in field experiments to study the dispersal of earthworms [77] and shows great promise for spatial and seasonal distribution, age structure, longevity, and range expansion studies.

3.3. Physical extraction

Methods for extracting earthworms were first developed to assess annelid population densities and community composition but can equally be applied to field exclusion experiments. These methods can be divided into two categories: physical—those that depend on the physical examination by the researcher within a known volume of excavated soil, and behavioral—those that depend on the behavioral response of the worms to an irritant employed by the researcher, allowing collection at the soil surface [78].

3.3.1. *Hand-sorting*

Coleman et al. [78] give an overview of sorting soils by hand for earthworm sampling. Hand-sorting typically samples a 25 × 25 cm area. While small sample areas increase the fraction of fragmented worms and can be inefficient where population densities are low, larger areas can decrease efficiency purely due to the time required to process the amount of soil. Wet sieving or washing can be applied in addition to detect smaller species and cocoons; or instead of hand-sorting in grassland systems where fibrous roots are very dense [78].

This method is well and broadly applied in studies seeking relationships in environments with variation in earthworm abundance, community structure and diversity alone [79], or in correlation with other factors such as tree species or communities [80, 81], elevational gradients [82], land use [83], or chronosequences of succession or agricultural abandonment [84, 85], or more experimental manipulations such as litter exclusion [86] or litter addition [80].

Hand-sorting or sieving can be undesirable in field experiments as it destroys the pedology and soil texture of the site (thus porosity and hydrology) and likely disturbs many other classes of soil biota. While time and labor consuming, control (or ambient) earthworm treatments could also be sorted without earthworm removal to duplicate the disturbance; however, the resulting soil would not reflect any natural system, even rarely agroecosystems. Zhang et al. [21] applied a 1 cm sieve to sort out native earthworm cocoons, stating this mesh size was too large to disrupt soil aggregates, replacing the soil into field-installed mesocosms in a rubber plantation, inoculating 30 *P. corethrurus* per mesocosm.

3.4. Behavioral extraction

The behavioral response methods are often considered "non-destructive" and indeed they are in comparison to the disruption of soil horizons, texture, and aggregates likely resultant of the application of hand-sorting. Nonetheless, there are inherent biases and non-target effects in behavioral extraction techniques along with other methods discussed here.

3.4.1. Vermifuges

The first set and most common of the behavioral methods involve liquid earthworm irritants applied to soils for extraction at the soil surface. When effective, these liquid expellants can be considered vermifuges.

3.4.1.1. Formalin

First evaluated by Raw [87], the application of dilute formalin to a known area of soil to expel earthworms has since become widely used. Raw [87] found that formalin yielded the highest abundance count for *L. terrestris* (anecic) compared to counting burrow openings at the surface, hand-sorting, and the application of another more lethal irritant potassium permanganate. However, numbers for *Allolobophora chlorotica* (endogeic) and *Eisenia rosea* (epigeic) were much poorer. Reviewing this method, Coleman et al. [78] concluded that formalin is better for vertical burrowing (anecic) species, less for horizontal burrowing species, and ineffective for megascolecid species. In addition, climate restricts efficacy in cold (below 8°C), or very wet or dry soils. Furthermore, as the flow path of formalin cannot be determined, it is difficult if not impossible to determine the volume of soil sampled with this method. However, formalin extraction may be the best technique for *L. terrestris* and similar deep-dwelling species during times of highest activity (spring and fall in temperate regions). No mentions on non-target effects were listed.

Coleman et al. [78] suggest when both shallow and deep dwelling species are present; formalin can be applied to the bottom of a hand-sorted pit and included in total estimations (as done in [26, 86]). Gunn [88] found formalin killed clover ground cover where applied with inhibited recovery, suggesting residual effects on vegetation, thus organic matter and nutrient inputs to the soil system. It is apparent that formalin may have applicability in estimating earthworm populations but may be inappropriate in manipulation studies due to its carcinogenic qualities and possible lasting nontarget affects. Burtelow et al. [57] use formalin not for exclusion but to assess populations after termination of experiment.

3.4.1.2. Mustard

A non-toxic alternative to formalin that acts largely in the same way is "mustard flour" or "hot mustard" with the active ingredient allyl isothiocyanate. Gunn [88] concluded that mustard was an effective vermifuge, with better extracting efficiencies than formalin, potassium permanganate, and household detergent. Furthermore, unlike the other extractants tested, mustard had no phytotoxic effect on vegetation nor killed the earthworm specimens given they were rinsed soon after extraction. Chan and Munro [89] compare the effectiveness of mustard to formalin at different concentrations and hand-sorting for anecic and endogeic species in Australia. Through field tests, they found the optimal mustard solution was created by adding 106 g dry mustard powder to 1 L of 5% acetic acid and shaking over-night. This solution is then diluted with water to a 15 mL:1 L ratio. For the anecic *Anisochaetae sp.*, the mustard solution yielded a higher (67%) abundance than formalin. Hand-sorting to 10 cm depth proved to be less efficient than both repellents, as the anecics were able to retreat to lower depths. However, it is important to note that most earthworm studies that employ hand-sorting often sample beyond the 10 cm depth. For the endogeic species *Aporrectodea trapezoides*, all repellent treatments were deemed inefficient, formalin better than mustard with an extraction efficiency of 36%. However, subsequent sorting of repellent treatments revealed that the endogeics were dead. This could reflect that mustard may not be a good extraction technique for endogeics but is appropriate in effective reductions if *in situ* death and decomposition is not a concern. In addition, this study supports previous suggestions that higher concentrations of repellent are too strong for juveniles, preventing their surfacing and thus can cause an underestimation.

Lawrence and Bowers [90] evaluated the use of mustard solution for extracting earthworms by subsequent hand-sorting over a variety of soil and land-use types. They used a solution of 50 g hot mustard powder (ChamponTM −0.2% allyl isothiocyanate) mixed with 100 ml water, which sat for 4 hours and then was diluted with 7 L of water. They conclude mustard is an appropriate method as it explained 83% of the variation in total abundance and 98% of total biomass with no differences across land-use types or soil attributes. They do report a decreased efficiency in extracting the endogeic *Octolasion tyrtaeum* compared to other species. This may be due to the rare occurrence of surface openings to burrows of endogeics. It is clear that the extractants discussed so far are biased toward anecics or other species with burrow openings at the surface, with limited efficiency in extracting endogeics. Furthermore, the application of mustard and formalin depends on infiltration qualities of the soil to take effect and may be inappropriate in compact or low-porosity soils such as many clay-rich tropical soils.

Zaborski [91] was the first to take the active ingredient in mustard, Allyl isothiocyanate (AITC) and use it directly as a vermifuge, finding no difference in efficacy in total numbers or biomass compared to formalin. Gutiérrez-López et al. [92] took this expellant one step further, utilizing AITC solution and tested against hand-sorting, formalin, and a combination of methods in a Mediterranean climate in Central-Western Spain with historic Dehesa agroforestry land use. They found that hand-sorting alone had the potential to underestimate anecic species that may escape through burrows out of the sample area or to deeper soil horizons in response to the vibrations of digging researchers, while a combination of hand-sorting and an expellant minimized this effect. Similar to other studies, they found expellants to be efficient in sampling epigeics, and to an extent, anecic species, but lacking in efficacy in sampling endogeics which most commonly occupy horizontal burrows, making infiltration difficult.

3.4.1.3. Onion

Similar to mustard, cultivars of the genus *Allium* (onions) produce natural sulfur compounds in high densities that act as irritants to many animals including humans and earthworms. Steffen et al. [93] tested the application of an onion solution as a vermifuge in both a sandy Ultisol and a clayey Oxisol compared to formalin. The solution was prepared by the authors using white onions blended with water and then strained. Results indicated that 175g onion extract L^{-1} was the ideal concentration, with higher concentrations yielded less earthworms, and lower concentrations being less efficient than formalin. This study demonstrated the efficacy of a low cost homemade vermifuge. The authors advise that repeated tests of this expellant are needed in a variety of climate conditions, and that perhaps the compounds themselves could be isolated and utilized.

3.4.2. Grunting

Catania [94] reviews the practice of collecting the endemic *Diplocardia mississippiensis* earthworm in Florida's Apalachicola National Forest by locals for generations in a method known as "worm grunting" (or fiddling, snoring, or charming). This involves driving a wooden stake into the ground and then rubbing a metal bar across the top, sending vibrations down the stake and into the ground. Earthworms emerge up to 12 m away and thousands can be collected in hours. The study by Catania [94] supports the hypothesis that worm grunters unknowingly are mimicking the vibrations of American moles (*Scalopus aquaticus*) the earthworm's natural predator, which they exit their burrows to escape, rejecting an alternative hypothesis of mimicked raindrops. Other predators, wood turtles and herring gulls have been reported to exploit this relationship as well. An interesting question is whether this method is effective in ecosystems where moles or other fossorial predators have never been present (or other animals exploiting this relationship) for earthworms to co-evolve alongside.

3.4.3. Heat (Kempson apparatus)

Some researchers have utilized the concept behind the Berlese or Tullgren funnel, used widely to extract diverse groups of soil micro- and mesofauna in the lab, for earthworms, also known as a Kempson apparatus. These devices exploit the photophobic reaction of soil fauna to temperature, light, and moisture gradients, which move away from the heated surface of a soil sample and into a collection pan holding a euthanizing agent or fixation solution. Tuf and Tvardik [95] describe such a device and how to easily build one at low-cost that uses heat from a light bulb to extract fauna based of previous designs. Several research groups have used a modified Kempson apparatus successfully to assess earthworm communities, or as a parameter for land-use or rehabilitation [61, 96, 97]. However, it is obvious that adapting this technique to field settings is not feasible. Furthermore, even for microcosm and mesocosm studies, the nontarget effects on all other soil biota groups are well known, and consequences of drying on soil structure and aggregates are profound. Therefore, this method is only advised for when complete and destructive sampling is warranted, and suitable access to a lab is provided.

3.4.4. Electroshock

An earthworm extraction method that shows great potential in limiting both physical soil disturbance and nontarget species effects is that of electroshock extraction. Satchell [98] first proposed and tested the application of electric current to soils to extract earthworms in a temperate pasture in cases where both contamination by chemicals or physical disturbance might negate experimental factors in field investigations. His results showed that current greater than 0.5 A is necessary for earthworms to be expelled (even with voltages up to 50,000 V), and direct current to be ineffective compared to alternating current. Satchell [98] used a metal probe driven to 18 in (45.7 cm), which he further developed into an elaborate water-cooled double-tube copper electrode to eliminate over-heating and drying of the soil allowing "indefinite" operation. Concentric bands of aluminum to assess distance effects surrounded the probe. The use of 3 A resulted in death of earthworms near the electrode. He concluded that 50-cycle frequency AC was ideal (deviation from this gave no advantages) for 40 min (longer brought more worms but did not change community composition estimates).

Rushton and Luff [99] further evaluated the application of electrical current to sample earthworms in a temperate grassland. Their setup consisted 15 bars surrounding a central electrode in a circle and driven to 30 cm depth. They concluded that ideal current for extraction is between 0.2 and 0.4 A. Rushton and Luff [99] found that juveniles were extracted more efficiently than adults across species and that extraction efficiency was correlated with soil moisture content but not temperature. The latter may be explained by the behavior of earthworms to aestivate during dry conditions and that soil moisture is necessary for electrolytes and the passage of electric current. They note the difficulty in quantifying this method considering the inability of defining where the current flows and thus volume of soil sampled.

Schmidt [100] describes the use of "Worm-Ex III" based on German Thielemann design of 8 stainless steel electrodes (60 cm length, 0.6 cm diameter) arranged in opposing pairs and installed to a depth of 40 cm. Vegetation was clipped and litter removed in the 0.125 m^2 sample area. An auto battery (12 V, 90 Ah) and control unit was used to regulate output voltage and current between specific pairs of electrodes. Voltage was increased in a stepwise fashion from 200 to 600 V over 35 min. Following application, the top 5 cm of soil was sorted with a hand rake.

Schmidt [100] tested this octet design of electroshocking against formalin and hand-sorting in an agroecosystem in Ireland across conventional and direct till on soil of medium to heavy texture over 2 years. He found electroshocking to yield higher numbers and biomass of earthworms compared to formalin extraction, similar community size, and composition compared to hand-sorting, except where recently ploughed. However, electroshocking appeared to underestimate juvenile endogeic and the very small *Murchieona minuscule* (endogeic) compared to hand-sorting. This may be due to the lack of surface burrow openings and leads the author to suggest a subsequent shallow hand-sort following electroshocking. Schmidt [100] concludes formalin extraction may be better for large anecics (electroshock yielded relatively lower and more variable results for *L. terrestris*) in this temperate agriculture field. Electrical extraction is limited by soil moisture; however this does not present a problem in many of earth's biomes, including temperate and tropical wet forests. Despite its advantages, Schmidt [100] warns that this method is less straightforward, involving many factors that can be altered by individual investigators in both hardware and application, making it more difficult to standardize and/or compare across studies. For these reasons, we suggest all investigators exploiting this method report the electrical current (in amperes) to act as a common denominator making this technique and results comparable. Measurement of current reflects and is a function of such soil properties as moisture and resistance. The clipping of vegetation and postshock hand-sorting of Schmidt's method may introduce some undesirable disturbances.

Eisenhauer et al. [101] tested the same octet design as Schmidt [100] against mustard extraction in dry conditions in a seminatural grassland in Germany. Electrical current was applied for 35 min per treatment in step-wise incremental increase of voltage from 250 to 600 V with no report of electric current or distance between probes. Neither the mustard nor electroshock method improved by addition of water. Mustard extraction was found to be more efficient in sampling anecic earthworm species, even under dry conditions. Endogeic species were extracted in low numbers for both methods compared to hand-sorting, suggesting decreased activity or inactivity during dry seasons. These findings led Eisenhauer et al. [101] to conclude that the octet method was inappropriate in estimating earthworm community structure, however these conclusions may have limited applicability to dry or seasonally dry soils.

The majority of the literature concerning the octet design is in German and finding this device outside of Europe is difficult. However, Weyers et al. [102] have described in detail how to build an octet device with current less than 1 A, including a control panel with data logger capabilities. The authors describe a difference in their design relative to previous octet construction in that this design lacks a return path that would otherwise limit the field, thus earthworms may surface outside of the sample area (which should not be counted if doing an area estimate). Weyers et al. [102] tested their device on conifer soils in North Carolina and agroecosystem soils in Georgia (USA). They listed water as a limiting factor and that higher numbers were obtained in the spring and fall when moisture was optimum for earthworm activity. Furthermore, they note that compaction or thick root-mats can limit the installation of probes or the exit of earthworms. In addition, they list the appropriate safety precautions. They mention that the octet device did not operate under very high soil moisture conditions as soil conductivity limited the generated electrical field. This may suggest why the octet device has not been applied successfully in the humid or wet tropics.

The octet design appears to be a sufficient method for sampling earthworm populations, but the limited surface area affected makes it inefficient for large field exclusion experiments. Bohlen et al. [103] are the first to describe the application of electroshock for large field manipulations. Enclosures of 20.25 m^2 in an agroecosystem were reinforced with PVC walls. Eight steel probes (50 cm long, 33 cm apart) were applied 220 V (AC) for 45–60 min (current not reported). Application was during the known peak of earthworm activity of spring and fall (twice within 2 weeks for each plot) for 3 years. Results revealed earthworm abundance was reduced 25–75% of natural levels using this method. Bohlen [86] concludes removal by electroshock is much more effective at manipulating earthworm populations than additions in paired plots, possibly due to mortality from handling or resource limitation. Both Blair et al. [38] and Shuster et al. [40] utilized the same plots and methods of above [86]. Blair et al. [38] found no effect of this electroshock application on enchytraeids, nematodes, springtails, mites, or other microarthropods.

Based on methods of Bohlen [86] Costello and Lamberti [23] are the first to apply the electroshock method to earthworm exclusion treatment plots (0.25 m^2) in a natural system in their Northern temperate deciduous forest site (mixed coarse-loamy, superactive, nonacid soil). Electrodes were placed to 25 cm depth and supplied with 110 V (AC) for 40 min (20 min, then 90° rotation of probes). Upon termination of their experiment, they conclude that electrical reduction successfully excluded *A. caliginosa* and *L. terrestris*, yet was inefficient in expelling juveniles of *Lumbricus* spp., which promptly reverted to the electroshocked plots. Even though the reduced plots were not entirely void of earthworms, the authors believed the disparity between the two treatments was great enough to make a case that electroshocking was a valid method to illustrate the effect of invasive earthworms on forest and riparian soils.

Liu and Zou [42] are the first to report application of the electroshock method in tropical soils (clayey Oxisol of Zarzal series). A slightly different design than Bohlen [86] was applied using 9 steel rods connected in parallel, driven to 50 cm depth in 0.25 m intervals and supplied 240 V (AC, current not reported) for 1.5 hour every 3 months. Hand-sorting at termination of the experiment (one 25 × 25 × 50 cm deep subsample) allowed reporting of extraction efficiencies of 85% in their pasture site and 87% in the forest.

Rhea-Fournier [104] also applied electrical extraction in a wet subtropical forest with some modifications. Two strands of aluminum stakes connected in series were installed at intervals of 35 cm, and to 50 cm depth. Current was supplied by a 220 V AC gas-powered generator, controlled by a dimmer-switch, and direction alternated every application. In the first month plots were shocked six times, with voltage increased in a step-wise fashion every 10 min during hour-long treatments. For the remaining 13 months of the experiment plots were shocked monthly at maximum voltage. To calculate current passing through the circuit (soil), the voltage across a 1 Ω standardized resistor was measured and converted to amps using application of Ohm's Law. Hand-sorting of a 25 cm^2 subsample pedon of each plot at experiment termination was used to determine mean extraction efficiencies, calculated by dividing the total individuals or biomass extracted by the sum of the final hand-sort estimates and total extractions from each plot. Extraction efficiencies were greatest for the anecic *Estherella* sp. in terms of both abundance and biomass (86 and 97%, respectively) with the epigeic *Amynthas* sp. comparable (82 and 94%). Extraction efficiencies were notably lower for *P. corethrurus* (abundance: 60%, biomass: 83%), giving total earthworm extraction efficiencies of 60% in terms of abundance, and 80% biomass. These findings suggest difficulty in extraction of endogeic species compared to anecics and epigeics, or alternatively that the exotic invasive *P. corethrurus* has inherent physiological resistance to the treatment or life history traits that allow rapid recolonization [104]. No significant relationships were found between voltage or current, and biomass, abundance, and species of extracted earthworms.

Electrical extraction is not limited to field experiments and has been employed in laboratory microcosm experiments. As mentioned earlier, Fonte et al. [11] applied a modified electroshock method to soil cores (20 cm diameter, 30 cm deep) collected from a Mediterranean climate agriculture experimental site remaining in a PVC cylinder. Water was added to each core to reach field capacity. Four stainless-steel probes were administered 2 A of electric current for 8 min total (current switched between opposing probes every 2 min). Earthworms were then added to a subset of cores while the others were shocked every month. Importantly, unlike the majority of reports on the electroshock method, this study reports the current applied. Alike, Willems et al. [14] inserted two thin metal probes along the perimeter of a soil core collected from a temperate agriculture field and a PVC sleeve to apply electric current, however very little detail is included in the description of this method.

Staddon et al. [105] designed an experiment to directly test for nontarget effects in temperate European grasslands. The electroshock setup they used is more akin in size to the soil core experiments. It involves a stainless steel cylinder driven into the earth (40.5 cm diameter, 16 cm deep) with a copper electrode installed in the center to a depth of 30 cm, and electrical current was applied at 120 V AC for 4 min (current not reported). Previous trials indicated no greater numbers of earthworms were obtained with more time or voltage; however, this may have been due to the limited volume of soil affected by electric current. As for nontarget effects, the results of Staddon et al. [105] found no effect of electroshocking on canopy CO_2 exchange, root respiration nor mycorrhizal fungal abundance or vitality.

Szlavecz et al. [106] applied the electroshock method in a highly replicated field experiment in a deciduous temperate forest in Maryland, USA with mixed results. Their experiment involved trenched plots with aluminum and copper rods installed to 0.4 m, applied with 110/120 V AC electricity for 45 min per application, eight times each. Earthworm reductions of 50% in terms of abundance and biomass were achieved after 2 years of treatment. The authors discuss how the electroshock method was more successful in past experiments in grasslands and agroecosystems, citing such differences as woody underground biomass and spatial heterogeneity inherent in forest ecosystems as impairments for soil conductivity. They advise monitoring soil temperature and moisture, and that a dynamic schedule for electroshocking is adopted in future applications of this method to maximize efficacy. Additionally they recommend strong considerations given to safety, site access and maintenance, labor/effort, and trade-offs between site disturbance and treatments when designing a study using the electroshock method.

To summarize, the findings by Blair et al. [38] and Staddon et al. [105] suggest that any nontarget effects of this method are limited or undetectable, making it ideal for earthworm exclusion experiments that do not aim to reduce other soil fauna. Electrical extraction is the least destructive and thus more desirable among other methods with no reliance on hazardous materials. Furthermore, this approach appears applicable with mixed success in both temperate and tropical forests. It is ideal for study sites such as reserves, protected areas, or long term research sites where introduction of chemicals or interference with other research is undesired [98]. However, researchers who do not include applied electrical current should be strongly criticized against doing so. The actual electrical current felt by the earthworm in the soil is a result of the soil resistivity and the voltage applied. Knowing that each soil has a different moisture regime dependent on climate and life zone, reporting only the voltage is not very useful in comparing studies. Furthermore, measuring the actual current in the soil is relatively easy by connecting an in-series standardized resistor and measuring the voltage drop across it with a voltmeter.

3.5. Comparisons between methods

Looking at human impacts and disturbance on native vegetation and soil fauna community in a mixed subtropical wet forest in Brazil, Baretta et al. [107] compared hand-sorting of two different sizes of soil pedons to formalin extraction. They concluded that a combination of hand-sorting of larger soil monoliths and formalin extraction was the only proper technique to sample the surface-active and geophagous species.

In experimental meadow grassland in Austria, Čoja and others [108] test five of the methods discussed above in their ability to extract earthworms. They found that hand-sorting, and a modified Kempson apparatus were the two most effective in terms of earthworm abundance, yielding more than three times as many as an electrical octet method. However, it should be noted that for this method soil samples were removed from the field and processed in a laboratory setting. Comparisons between formalin and the mustard extract Allyl isothiocyanate (AITC) in this study yielded no difference, suggesting the nontoxic AITC be used as an alternative. They found an electrical octet method to be biased to juvenile earthworms, compared to other methods applied at their site, resulting in underestimations of biomass. Despite this, they suggest the octet method for sensitive sites, or where groundwater quality is of concern for chemical methods. They concluded that no one technique fulfilled all criteria of low-cost, nondestructive, efficient, and time-saving.

4. Quantifying influences of soil fauna on soil processes and biogeochemical cycling

Considering the myriad of approaches discussed above to passively monitor earthworms or manipulate their populations in the field or laboratory settings, we will now briefly discuss some approaches to quantifying the potential direct and indirect effects of earthworms on the soil ecosystem.

4.1. Physical soil properties

The influence of earthworm casts and burrows on soil porosity can be quantified using a soil infiltrometer between different earthworm treatments. Differences in infiltration rates can serve as corollaries to soil porosity and aeration, which in turn can be indicative of soil saturation rates and microbial processes. Quantification of the impact of earthworm casts on soil aggregate structure and size classes can be directly obtained by using different size sieves on soil samples from different earthworm treatments or communities [7].

4.2. Leaf litter consumption

As discussed earlier, certain functional groups of earthworms can drive the rate of forest leaf litter decomposition and/or incorporation into lower soil horizons. Comparing mass loss rates between different mesh size leaf litterbags or between earthworm treatments may allow deduction of the direct influence of the given earthworm community on these processes [45, 62–67].

4.3. Soil moisture and groundwater chemistry

The role of earthworms in leachate loss, groundwater, and soil moisture chemistry is a pertinent investigative question, especially when budgeting biogeochemical processes for a given forested watershed. Soil lysimeters can be employed to collect and measure soluble chemical species in soils [17, 18, 22]. For fully saturated soils simple pan lysimeters can be used to collect groundwater samples using gravitational properties. For unsaturated soils, suction lysimeters provide the means to sample soil moisture otherwise held in soil pores by capillary forces by applying negative pressure. These soil water samples can then be analyzed for concentrations of soluble chemical species that may have implications for soil biota, plants, and stream input/output budgets in forests.

4.4. Soil chemistry

In studies investigating changes or differences in soil chemistry (such as carbon or nitrogen content) over time, direct quantification of the elements under study can be determined using an elemental analyzer on soil samples at different time steps or at termination of the study across earthworm treatments or communities. Further contrast between earthworm influenced soils and control samples can be achieved by leaving aggregates intact during throughout processing and comparing to samples with aggregates disrupted before chemical analysis. Assuming earthworm casts plays a dominant role in aggregate formation, this method allows enumeration of the quantity of nutrients or carbon are protected within aggregates, and thus more stable, contributing to longer turnover times in soils [21, 59].

4.5. Soil microbiota

Recognizing the role of earthworms in regulating microbial activity and processes, it is often desired to quantify the amount of carbon and nitrogen contained within the microbial biomass. Chloroform-fumigation techniques can be applied in the laboratory setting to lyse microbial cell walls, making nutrients contained within the biomass available for measurement [57].

Microbial activity in soils can be measured *in situ* using soil respiration techniques. The most common class of soil respirometer instruments utilizes an infrared gas analyzer. Soil respiration allows for an easily obtainable proxy for heterotrophic metabolic rates in soils, which can be very useful to quantify differences between earthworm treatments and communities. While soil respiration may include the rates of tree root respiration, this can be avoided by trenching sample plots to the depth of existing roots, or otherwise removing plants of substantial size across plots prior to experiment initiation.

As methods for determining microbial communities and functional groups in soils advance through genetic barcoding and other DNA techniques, potential for determining the direct influence of earthworms and other soil fauna on soil microbial communities continues to grow.

To assess the fungal component of soil microbiota separately, a direct count method to determine biovolume can be employed. Creating a soil slurry with agar allows suspension of fungal hyphae fragments to be placed on a microscope slide and counted across a transect [109].

5. Conclusions

This review of various techniques and findings shows that there is no single method that can be applied across ecosystems for equally successful earthworm sampling. We suggest a combination of a behavioral extraction technique (such as electroshock or a nontoxic vermifuge) with limited hand-sorting as a viable method for manipulating populations in field experiments. When selecting an extraction (or exclusion) method to apply such site-specific conditions as earthworm community (or functional groups present), soil conditions, and previous land use must be considered. It must be recognized that complete exclusion is not a likely attainable goal in most circumstances. Furthermore, functional group bias may exist for all methods. Both large differences in size and behavior between earthworm species in a given community introduce greater complexity and thus difficulty in calibrating methods to varying ecosystems. For electrical extraction, continual and frequent application is suggested in heavy clay tropical soils, especially where invasive species exist. Sustained methodological development and standardization of these techniques (e.g., electrical current) are encouraged for its utility, particularly in forest ecosystems.

For the comprehensive study of earthworms' roles in forest soils, we advise a combination of field experiments, and laboratory microcosms or controlled mesocosm studies. Recent studies in the genetic structure of common earthworm species reveal that there are likely many undescribed cryptic species only identified through DNA verification, and thus further collaborative efforts to combine morphological traits, phylogenetics, and DNA-barcoding are needed to resolve a possible underestimation of earthworm biodiversity.

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References

[1] Edwards, C.A. Earthworm Ecology. Boca Raton, Forida. CRC Press; 1994.

[2] Lavelle, P., Decaens, T., Aubert, M., Barot, S., Blouin, M., Bureau, F., Margerie, P., Mora, P., and Rossi, J.-P. Soil invertebrates and ecosystem services. European Journal of Soil Biology. 2006;**42**:S3–S15.

[3] Lee, K.E., and Foster, R.C. Soil fauna and soil structure. Australian Journal of Soil Research. 1991;**29**:745–775.

[4] González, G., Huang, C.-Y., and Chuang, S.-C. Post-Agricultural Succession in the Neotropics . In: R.W. Myster, editor. Earthworms and Post-Agricultural Succession. New York City, New York: Springer; 2008. pp. 115–138.

[5] Römbke, J., Sousa, J.P., Schouten, T., & Riepert, F. Monitoring of soil organisms: a set of standardized field methods proposed by ISO. European Journal of Soil Biology. 2006;**42**:S61–S64.

[6] Sheehan, C., Kirwan, L., Connolly, J., and Bolger, T. The effects of earthworm functional group diversity on nitrogen dynamics in soils. Soil Biology and Biochemistry. 2006; **38**:2629–2636.

[7] Bossuyt, H., Six, J. and Hendrix, P. Protection of soil carbon by microaggregates within earthworm casts. Soil Biology and Biochemistry. 2005;**37**:251–258.

[8] Aira, M., Sampedro, L., Monroy, F., and Domínguez, J. Detritivorous earthworms directly modify the structure, thus altering the functioning of a microdecomposer food web. Soil Biology and Biochemistry. 2008;**40**:2511–2516.

[9] Coq, S., Barthès, B.G., Oliver, R., Rabary, B., and Blanchart, E. Earthworm activity affects soil aggregation and organic matter dynamics according to the quality and localization of crop residues—An experimental study (Madagascar). Soil Biology and Biochemistry. 2007;**39**:2119–2128.

[10] Bhattacharjee, G., and Chaudhuri, P.S. Cocoon production, morphology, hatching pattern and fecundity in seven tropical earthworm species— a laboratory-based investigation. Journal of Biosciences. 2002;**27**(3):283–94.

[11] Fonte, S.J., Kong, A.Y.Y., Kessel, C. van, Hendrix, P.F., and Six, J. Influence of earthworm activity on aggregate-associated carbon and nitrogen dynamics differs with agroecosystem management. Soil Biology and Biochemistry. 2007;**39**:1014–1022.

[12] Amador, J. A, Gorres, J.H., and Savin, M.C. Carbon and nitrogen dynamics in *Lumbricus terrestris* (L.) Burrow soil: relationship to plant residues and macropores. Soil Science Society of America. 2003;**67**:1755–1762.

[13] Laossi, K.-R., Ginot, A., Noguera, D.C., Blouin, M., and Barot, S. Earthworm effects on plant growth do not necessarily decrease with soil fertility. Plant and Soil. 2009;**328**:109–118.

[14] Willems, J.J.G.M., Marinissen, J.C.Y, and Blair, J.M. Effects of earthworms on nitrogen mineralization. Biology and Fertility of Soils. 1996;**23**:57–63.

[15] Callaham, M., and Hendrix, P.F. Impact of earthworms (Diplocardia: Megascolecidae) on cycling and uptake of nitrogen in coastal plain forest soils from northwest Florida, USA. Applied Soil Ecology. 1998;**9**:233–239.

[16] Scheu, S. Microbial activity and nutrient dynamics in earthworm casts (Lumbricidae). Biology and Fertility of Soils. 1987;**5**:230–234.

[17] Tiunov, A., and Scheu, S. Carbon availability controls the growth of detritivores (Lumbricidae) and their effect on nitrogen mineralization. Oecologia. 2004;**138**:83–90.

[18] Butenschoen, O., Marhan, S., Langel, R., and Scheu, S. Carbon and nitrogen mobilisation by earthworms of different functional groups as affected by soil sand content. Pedobiologia. 2009;**52**:263–272.

[19] Lachnicht, S.L., Hendrix, P.F., and Zou, X. Interactive effects of native and exotic earthworms on resource use and nutrient mineralization in a tropical wet forest soil of Puerto Rico. Biology and Fertility of Soils. 2002;**36**:43–52.

[20] Huang, C.-Y., González, G., and Hendrix, P.F. The Re-colonization ability of a native earthworm, *Estherella* spp., in forests and pastures in Puerto Rico. Caribbean Journal of Science. 2006;**42**:386–396.

[21] Zhang, M., Zou, X., and Schaefer, D.A. Alteration of soil labile organic carbon by invasive earthworms (*Pontoscolex corethrurus*) in tropical rubber plantations. European Journal of Soil Biology. 2010;**46**:74–79.

[22] Scheu, S., and Wolters, V. Influence of fragmentation and bioturbation on the decomposition of 14C-labelled Beech leaf litter. Soil Biology and Biochemistry. 1991;**23**:1029–1034.

[23] Costello, D.M., and Lamberti, G.A. Biological and physical effects of non-native earthworms on nitrogen cycling in riparian soils. Soil Biology and Biochemistry. 2009;**41**:2230–2235.

[24] Pashanasi, B., Melendez, G., Szott, L., and Lavelle, P. Effect of inoculation with the endogeic earthworm *Pontoscolex corethrurus* (Glossoscolecidae) on N availability, soil microbial

biomass and the growth of three tropical fruit tree seedlings in a pot experiment. Soil Biology and Biochemistry. 1992;**24**:1655–1659.

[25] González, G., and Zou, X. Earthworm influence on N availability and the growth of *Cecropia schreberiana* in tropical pasture and forest soils. Pedobiologia. 1999;**43**:824–829.

[26] Whalen, J.K., Parmelee, R.W., and Subler, S. Quantification of nitrogen excretion rates for three lumbricid earthworms using 15 N. Biology and Fertility of Soils. 2000;**32**:347–352.

[27] Butenschoen, O., Poll, C., Langel, R., Kandeler, E., Marhan, S., and Scheu, S. Endogeic earthworms alter carbon translocation by fungi at the soil–litter interface. Soil Biology and Biochemistry. 2007;**39**:2854–2864.

[28] Postma-Blaauw, M., Bloem, J., Faber, J., Vangroenigen, J., Degoede, R., and Brussaard, L. Earthworm species composition affects the soil bacterial community and net nitrogen mineralization. Pedobiologia. 2006;**50**:243–256.

[29] Huang, C. Y., González, G., & Hendrix, P. F. Resource Utilization by Native and Invasive Earthworms and Their Effects on Soil Carbon and Nitrogen Dynamics in Puerto Rican Soils. *Forests*, 2016;**7**:277.

[30] Amador, J.A., Gorres, J.H., and Savin, M. Effects of *Lumbricus terrestris* L. on nitrogen dynamics beyond the burrow. Applied Soil Ecology. 2006;**33**:61–66.

[31] Barois, I., and Lavelle, P. Changes in respiration rate and some physicochemical properties of a tropical soil during transit through *Pontoscolex corethrurus* (Glossoscolecidae, Oligochaeta). Soil Biology and Biochemistry. 1986;**18**:539–541.

[32] Horn, M.A., Schramm, A., and Drake, H.L. The earthworm gut: an ideal habitat for ingested N_2O-producing microorganisms. Applied and Environmental Microbiology. 2003;**69**:1662–1669.

[33] Lenoir, L., Persson, T., Bengtsson, J., Wallander, H., and Wirén, A. Bottom–up or top–down control in forest soil microcosms? Effects of soil fauna on fungal biomass and C/N mineralization. Biology and Fertility of Soils. 2007;**43**(3):281–294.

[34] Alphei, J., Bonkowski, M., and Scheu, S. Protozoa, Nematoda and Lumbricidae in the rhizosphere of *Hordelymus europaeus* (Poaceae): faunal interactions, response of microorganisms and effects on plant growth. Oecologia. 1996;**106**:111–126.

[35] Barot, S., Ugolini, A., and Brikci, F.B. Nutrient cycling efficiency explains the long-term effect of ecosystem engineers on primary production. Functional Ecology. 2007;**21**:1–10.

[36] Carpenter, S.R. Microcosm experiments have limited relevance for community and ecosystem ecology. Ecology. 1996;**77**:677–680.

[37] Parmelee, R.W., Beare, M.H., Cheng, W., Hendrix, P.F., Rider, S.J., Crossley Jr., D.A., and Coleman, D.C. Earthworms and enchytraeids in conventional and no-tillage agroecosystems: a biocide approach to assess their role in organic matter breakdown. Biology and Fertility of Soils. 1990;**10**(1):1–10.

[38] Blair, J.M., Bohlen, P.J., Edwards, C.A., Stinner, B.R., McCartney, D.A., and Allen, M.F. Manipulations of earthworm populations in field experiments in agroecosystems. Acta Zool. Fennica. 1995;**169**:48–51.

[39] Gilot, C. Effects of a tropical geophageous earthworm, *M. anomala* (Megascolecidae), on soil characteristics and production of a yam crop in Ivory Coast. Soil Biology and Biochemistry. 1997;**29**:353–359.

[40] Shuster, W., McDonald, L., McCartney, D., Parmelee, R., Studer, N., and Stinner, B. Nitrogen source and earthworm abundance affected runoff volume and nutrient loss in a tilled-corn agroecosystem. Biology and Fertility of Soils. 2002;**35**(5):320–327.

[41] Domínguez, J., Bohlen, P.J., and Parmelee, R.W. Earthworms increase nitrogen leaching to greater soil depths in row crop agroecosystems. Ecosystems. 2004;**7**:672–685.

[42] Liu, Z.G., and Zou, X. Exotic earthworms accelerate plant litter decomposition in a Puerto Rican pasture and a wet forest. Ecological Applications. 2002;**12**:1406–1417.

[43] Subler, S., Baranski, C.M., and Edwards, C.A. Earthworm additions increased short-term nitrogen availability and leaching in two grain-crop agroecosystems. Soil Biology and Biochemistry. 1997;**29**:413–421.

[44] González, G., Ley, R., Schmidt, S.K., Zou, X., and Seastedt, T. Soil ecological interactions: comparisons between tropical and subalpine forests. Oecologia. 2001;**128**:549–556.

[45] González, G., and Seastedt, T. Soil fauna and plant litter decomposition in tropical and subalpine forests. Ecology. 2001;**82**:955–964.

[46] Heneghan, L., Coleman, D.C., Zou, X., Crossley, D.A.J., and Haines, B.L. Soil microathropod community structure and litter decomposition dynamics: A study of tropical and temperate sites. Applied Soil Ecology. 1998;**9**:33–38.

[47] Blair, J.M., Crossley, D.A., and Rider, S. Effects of naphthalene on microbial activity and nitrogen pools in soil-litter microcosms. Soil Biology and Biochemistry. 1989;**21**(4):507–510. DOI: 10.1016/0038-0717(89)90122-3

[48] Hallaire, V. Soil structure changes induced by the tropical earthworm *Pontoscolex corethrurus* and organic inputs in a Peruvian ultisol. European Journal of Soil Biology. 2000;**36**:35–44.

[49] Alegre, J.C., Pashanasi, B., and Lavelle, P. Dynamics of soil physical properties in Amazonian agroecosystems. Soil Science Society of America. 1996;**60**:1522–1529.

[50] Broadbent, A.B. and Tomlin, A.D. Comparison of two methods for assessing the effects of carbofuran on soil animal decomposers in cornfields. Environmental Entomology. 1982;**11**(5):1036–1042.

[51] Ingham, E.R. Review of the effects of 12 selected biocides on target and non-target soil organisms. Crop Protection. 1985;**4**(1):3–32.

[52] Parkin, T., and Berry, E.C. Microbial nitrogen transformations in earthworm burrows. Soil Biology and Biochemistry. 1999;**31**:1765–1771.

[53] Gorres, J.H., Savin, M.C., and Amador, J.A. Soil micropore structure and carbon mineralization in burrows and casts of an anecic earthworm (*Lumbricus terrestris*). Soil Biology and Biochemistry. 2001;**33**:1881–1887.

[54] Aira, M., Monroy, F., and Dominguez, J. Effects of two species of earthworms (Allolobophora spp.) on soil systems: a microfaunal and biochemical analysis. Pedobiologia. 2003;**47**:877–881.

[55] Hong, N.H., Rumpel, C., Henry des Tureaux, T., Bardoux, G., Billou, D., Duc, T.T., and Jouquet, P. How do earthworms influence organic matter quantity and quality in tropical soils? Soil Biology and Biochemistry. 2011;**43**:223–230.

[56] Bohlen, P.J., Pelletier, D.M., Groffman, P.M., Fahey, T.J. and Fisk, M.C. Influence of earthworm invasion on redistribution and retention of soil carbon and nitrogen in northern temperate forests. Ecosystems. 2004;**7**:13–27.

[57] Burtelow, A.E., Bohlen, P.J., and Groffman, P.M. Influence of exotic earthworm invasion on soil organic matter, microbial biomass and denitrification potential in forest soils of the northeastern United States. Applied Soil Ecology. 1998;**9**:197–202.

[58] Suárez, E., Fahey, T., Yavitt, J., Groffman, P., and Bohlen, P. Patterns of litter disappearance in a Northern Hardwood forest invaded By exotic earthworms. Ecological Applications. 2006;**16**(1):154–165.

[59] Lavelle, P., Melendez, G., Pashanasi, B., and Schaefer, R. Nitrogen mineralization and reorganization in casts of the geophagous tropical earthworm *Pontoscolex corethrurus* (Glossoscolecidae). Biology and Fertility of Soils. 1992;**14**:49–53.

[60] Nuzzo, V.A., Maerz, J.C., and Blossey, B. Earthworm invasion as the driving force behind plant invasion and community change in Northeastern North American forests. Conservation Biology. 2009;**23**:966–74.

[61] Pižl, V. Earthworms (Lumbricidae) of the Bohemian Forest. Silva Gabreta. 2002;**8**:143–148.

[62] González, G. Soil Organisms and Litter Decomposition. In: Ambasht R.S., Ambasht, N.K., editors. Modern Trends in Applied Terrestrial Ecology. New York, NY: Kluwer Academic/Plenum Publishers; 2002. pp. 315–329.

[63] González, G., Seastedt, T.R., and Donato, Z. Earthworms, arthropods and plant litter decomposition in aspen (*Populus tremuloides*) and lodgepole pine (*Pinus contorta*) forests in Colorado, USA. Pedobiologia. 2003;**47**:863–869. Doi: 10.1078/0031-4056-00272.

[64] Swift, M.J., Heal, O.W., and Anderson, J.M., editors. Decomposition in Terrestrial Ecosystems. Berkely and Los Angeles, California, USA: University of California Press, Blackwell Scientific Publications; 1979.

[65] Filley, T.R., McCormick, M.K., Crow, S.E., Szlavecz, K., Whigham, D.F., Johnston, C.T. and Heuvel, R.N. van den. Comparison of the chemical alteration trajectory of *Liriodendron*

tulipifera L. leaf litter among forests with different earthworm abundance. Journal of Geophysical Research. 2008;**113**:1–14.

[66] Barajas-Guzmán, G., and Alvarez-Sánchez, J. The relationships between litter fauna and rates of litter decomposition in a tropical rain forest. Applied Soil Ecology. 2003; **24**:91–100.

[67] Yang, X., and Chen, J. Plant litter quality influences the contribution of soil fauna to litter decomposition in humid tropical forests, southwestern China. Soil Biology and Biochemistry. 2009;**41**:910–918.

[68] Richardson, B.A., Richardson, M.J., González, G., Shiels, A.B., and Srivastava, D.S. A canopy trimming experiment in Puerto Rico: the response of litter invertebrate communities to canopy loss and debris deposition in a tropical forest subject to hurricanes. Ecosystems. 2010;**13**(2):286–301.

[69] González, G., Lodge, D.J., Richardson, B.A. and Richardson, M.J. A canopy trimming experiment in Puerto Rico: the response of litter decomposition and nutrient release to canopy opening and debris deposition in a subtropical wet forest. Forest Ecology and Management. 2014;**332**:32–46.

[70] Cortez, J., Billes, G., and Bouché, M.B. Effect of climate, soil type and earthworm activity on nitrogen transfer from a nitrogen-15-labelled decomposing material under field conditions. Biology and Fertility of Soils. 2000;**30**:318–327.

[71] González, G., Seastedt, T., and Donato, Z. Earthworms, arthropods and plant litter decomposition in aspen (*Populus tremuloides*) and lodgepole pine (*Pinus contorta*) forests in Colorado, USA. Pedobiologia. 2003;**47**:863–869.

[72] Cleveland, C.C., Reed, S.C., and Townsend, A.R. Nutrient regulation of organic matter decomposition in a tropical rain forest. Ecology. 2006;**87**:492–503.

[73] Butt, K.R. Inoculation of earthworms into reclaimed soils: the UK experience. Land Degradation and Development. 1999;**10**:565–575.

[74] Shuster, W.D., Shipitalo, M.J., Subler, S., Aref, S., and Mccoy, E.L. Landscape and watershed processes: earthworm additions affect leachate production and nitrogen losses in typical midwestern agroecosystems. Journal of Environmental Quality. 2003;2132–2139.

[75] Butt, K.R., and Lowe, C.N. A viable technique for tagging earthworms using visible implant elastomer. Applied Soil Ecology. 2007;**35**(2):454–457.

[76] González, G., Espinosa, E., Liu, Z., and Zou, X. A fluorescent marking and re-count technique using the invasive earthworm, *Pontoscolex corethrurus* (Annelida: Oligochaeta). Caribbean Journal of Science. 2006;**42(**3):371–379.

[77] Grigoropoulou, N., and Butt, K.R. Field investigations of *Lumbricus terrestris* spatial distribution and dispersal through monitoring of manipulated, enclosed plots. Soil Biology and Biochemistry. 2010;**42**(1):40–47.

[78] Coleman, D.C., Blair, J.M., Elliott, E.T., and Wall, D.H. Soil Invertebrates. In: Robertson, G.P., Coleman, C.S., Bledsoe, D.C. and, Sollins, P, editors. Standard Soil Methods for Long-Term Ecological Research. New York: Oxford University Press; 1999. pp. 291–456.

[79] Borges, S., and Alfaro, M. The earthworms of Baño de Oro, Luquillo Experimental Forest, Puerto Rico. Soil Biology and Biochemistry. 1997;**29**:231–234.

[80] González, G. and Zou, X. Plant and litter influences on earthworm abundance and community structure in a tropical wet forest. Biotropica. 1999;**31**:486–493.

[81] Lugo, A.E., Abelleira, O.J., Borges, S., Colón, L.J., Meléndez, S., and Rodríguez, M.A. Preliminary estimate of earthworm abundance and species richness in *Spathodea campanulata* Beauv. forests in Northern Puerto Rico. Caribbean Journal of Science. 2006;**42**(3):325–330.

[82] González, G., García, E., Cruz, V., Borges, V., Zalamea, and M., Rivera, M. Earthworm communities along an elevation gradient in Northeastern Puerto Rico. European Journal of Soil Biology. 2007;**43**:S24–S32.

[83] Huerta, E., Fragoso, C., Rodríguez-Olan, J., Evia-Castillo, I., Montejo-Meneses, E., De la Cruz-Mondragon, M., and García-Hernández, R. Presence of exotic and native earthworms in principal agro- and natural systems in Central and Southeastern Tabasco, Mexico. Caribbean Journal of Science. 2006;**42**:359–365.

[84] Zou, X., and González, G. Changes in earthworm density and community structure during secondary succession in abandoned tropical pastures. Soil Biology and Biochemistry. 1997;**29**(3–4):627–629.

[85] Sánchez de León, Y., Zou, X., Borges, S., and Ruan, H. Recovery of Native Earthworms in Abandoned Tropical Pastures. Conservation Biology. 2003;**17**(4):999–1006.

[86] Dechaine, J., Ruan, H., Sanchez-de Leon, Y., and Zou, X. Correlation between earthworms and plant litter decomposition in a tropical wet forest of Puerto Rico. Pedobiologia. 2005;**49**:601–607.

[87] Raw, F. Estimating earthworm populations by using formalin. Nature. 1959;**184:**1661–1662.

[88] Gunn, A. The use of mustard to estimate earthworm populations. Pedobiologia. 1992; **36**:65–67.

[89] Chan, K.-Y., Munro, K. Evaluating mustard extracts for earthworm sampling. Pedobiologia. 2001;**45**(3):272–278.

[90] Lawrence, A.P., and Bowers, M.A. A test of the 'hot' mustard extraction method of sampling earthworms. Soil Biology and Biochemistry. 2002;**34**(4):549–552.

[91] Zaborski, E.R. Allyl isothiocyanate: an alternative chemical expellant for sampling earthworms. Applied Soil Ecology. 2003;**22**(1):87–95.

[92] Gutiérrez-López, M., Moreno, G., Trigo, D., Juárez, E., Jesús, J.B., and Cosín, D.J.D. The efficiency of earthworm extraction methods is determined by species and soil properties in the Mediterranean communities of Central-Western Spain. European Journal of Soil Biology. 2016;**73**:59–68.

[93] Steffen, G.P.K., Antoniolli, Z.I., Steffen, R.B., Jacques, R.J.S., and dos Santos, M.L. Earthworm extraction with onion solution. Applied soil ecology. 2013;**69**:28–31.

[94] Catania, K.C. Worm grunting, fiddling, and charming—humans unknowingly mimic a predator to harvest bait. PLoS One. 2008;**3**(10):e3472.

[95] Tuf, I., Dedek, P., Jandova, S., and Tvardik, D. Length of recovery of soil macrofauna communities (Coleoptera: Carabidae, Isopoda: Oniscidea) after an irregular summer flood. Peckiana. 2005;**5**:65–75.

[96] Tajovsky, K.A.R.E.L., Pizl, V., Stary, J., Balík, V., Frouz, J., Schlaghamersky, J., and Kalcik, J. Development of soil fauna in meadows restored on arable land: initial phases of successional development. In: Tajovsky, K., Schlaghamersky, J. & Pizl, V., editors. Contributions to Soil Zoology in Central Europe I. Czech Institute of Soil Biology, Czech Republic. pp. 181–186.

[97] Frouz, J., Prach, K., Pižl, V., Háněl, L., Starý, J., Tajovský, K., and Řehounková, K. (2008). Interactions between soil development, vegetation and soil fauna during spontaneous succession in post mining sites. European Journal of Soil Biology. 2008;**44**(1):109–121.

[98] Satchell, J.E. An electrical method for sampling earthworm populations. Soil Zoology. 1955;**24**:356–364. https://www.ars.usda.gov/ARSUserFiles/41695/Reprints/Weyersetal2008AEIA.pdf.

[99] Rushton, S.P., and Luff, M.L. A new electrical method for sampling earthworm populations. Pedobiologia. 1984;**26**:15–19.

[100] Schmidt, O. Appraisal of the electrical octet method for estimating earthworm populations in arable land. Annals of Applied Biology. 2001;**138**:231–241.

[101] Eisenhauer, N., Straube, D., and Scheu, S. Efficiency of two widespread non-destructive extraction methods under dry soil conditions for different ecological earthworm groups. European Journal of Soil Biology. 2008;**44**:141–145.

[102] Weyers, S.L., Schomberg, H.H., Hendrix, P.F., Spokas, K.A., and Endale, D.M. Construction of an electrical device for sampling earthworm populations in the field. Applied Engineering in Agriculture. 2008;**24**:391–398.

[103] Bohlen, P.J. Efficacy of methods for manipulating earthworm populations in large-scale field experiments in agroecosystems. Soil Biology and Biochemistry. 1995;**27**:993–999.

[104] Rhea-Fournier. The relationship of earthworms and soil carbon, nitrogen, and microbial biomass in a subtropical wet forest in Puerto Rico. MS thesis. University of Puerto Rico - Río Piedras. 2012.

[105] Staddon, P.L., Ostle, N., and Fitter, A.H. Earthworm extraction by electroshocking does not affect canopy CO_2 exchange, root respiration, mycorrhizal fungal abundance or mycorrhizal fungal vitality. Soil Biology and Biochemistry. 2003;**35**:421–426.

[106] Szlavecz, K., Pitz, S.L., Bernard, M.J., Xia, L., O'Neill, J.P., Chang, C.H., McCormick, M., Whigham, D.F. Manipulating earthworm abundance using electroshocking in deciduous forests. Pedobiologia. 2013;**56**(1):33–40.

[107] Baretta, D., Brown, G. G., James, S. W., & Cardoso, E. J. B. N. Earthworm populations sampled using collection methods in Atlantic forests with *Araucaria angustifolia*. Scientia Agricola. 2007;**64**(4):384–392

[108] Čoja, T., Zehetner, K., Bruckner, A., Watzinger, A., & Meyer, E. Efficacy and side effects of five sampling methods for soil earthworms (Annelida, Lumbricidae). Ecotoxicology and Environmental Safety. 2008;**71**(2):552–565.

[109] Lodge, D.J., and Ingham, E.R. A comparison of agar film techniques for estimating fungal biovolumes in litter and soil. Agriculture, Ecosystems & Environment. 1991;**34**:131–144.

Chapter 4

Characterizing Predictability of Fire Occurrence in Tropical Forests and Grasslands: The Case of Puerto Rico

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Additional information is available at the end of the chapter

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Abstract

Global estimates of fire frequency indicate that over 70% of active fires occur in the tropics, and the size and frequency of fires are increasing every year. The majority of fires in the tropics are an unintended consequence of current land-use practices that promotes the establishment of grass and shrubland communities, which are more flammable and more adapted to fire than forests. In the Caribbean, wildland fires occur mainly in dry forests and in grasslands and crop lands. Climate change projections for the Caribbean indicate increasing area of drylands and subsequent increasing potential for wildland fire. We assessed the last decade of fire occurrence records for Puerto Rico to quantify the relative importance of time, climate, land cover, and population to inform predictive models of fire occurrence for projecting future scenarios of fire risk. Kruskal-Wallis, generalized linear models, robust regression, simple and multiple regressions, and tree models were used. We found that hour of the day (time), mean minimum temperature (climate), and percent forest cover (land cover) significantly influenced fire occurrence, while population showed a weak effect. Many variable interactions showed to be important. These significant variables and interactions should be considered in fire-predicting models for the island.

Keywords: wildfire, tropical dry forests, wildfire predictability, climate change, Caribbean

1. Introduction

Wildfire is an important natural process that shapes many terrestrial ecosystems because it influences species composition and community structure and function [1–5]. The occurrence and extent of wildfires are controlled by climate, operating at regional to global spatial scales and at interannual to multidecadal temporal scales [6–10]. Fire occurrence and spread is also controlled by local factors such as ignition source, topography, local weather patterns, variations in fuels type and condition, and management actions [6, 7, 9, 11–15]. Because of the complex interplay among drivers of wildfires, predicting fire occurrence in the context of global change is a challenge [16–18].

Predicting fire occurrence in the tropics and subtropics is especially urgent as vulnerabilities are expected to increase in the coming decades as changing climate influences temperature and precipitation patterns [19]. Current global estimates of fire frequency indicate that over 70% of active fires occur in the tropics [20] and that the size and frequency of fires are increasing every year [10, 21]. Among climate variables found to be determinant for fire occurrence in the tropics and subtropics, daily and monthly precipitation and daily relative humidity were negatively associated with fire ignitions [22, 23]. In addition to climatic drivers, fires in the tropics are unintended consequence of current land-use practices [21, 24–26] that promote the establishment of grass and shrubland communities, which are more flammable and more adapted to fire than evergreen forests [27–31]. In the Caribbean, although there is a general lack of available data on fire occurrence it appears that wildfires occur mainly in grasslands and crop lands [25], and in dry forests [32]. Lowland moist and montane forests are less susceptible to fires, though they can burn in dry years [25]. Understanding the relationship of current climate, landscape, and population can help predict the likelihood of increasing risk of fire occurrence in light of climate change projections in tropical islands [33]. Puerto Rico is an example from a tropical insular region where rapid and complex changes in land-use and land-cover (LULC) are occurring and where fire-prone ecosystems are being created [34, 35]. Puerto Rico land-use and land-cover changes in managed and unmanaged reforestation of abandoned agricultural lands, and deforestation and fragmentation related to population increases and urban development [36, 37] have created a complex, fragmented landscape of wildland-urban interface [38]. This has produced a shift in the fire regime from one of *fewer* natural fires prior to human habitation of the island, to possible use of fire in pre-Columbian times, through a pattern of widespread agricultural burning during the sugarcane era, to the current regime of thousands of small to intermediate scale human-induced fires occurring in a wildland-urban interface of forests and grasslands. In Puerto Rico, the limited available data related to fire occurrence suggest that most fires occur in dry areas of pasture, sugarcane, or abandoned lands in the southern part of the island [24]. In addition, historical information suggests that only 5% of the unintended wildfires in Puerto Rico in 1999 were caused by lightning and 95% were human-caused [24]. Present data and historical and paleoecological evidence suggest that fire frequency is increasing in the island and that fires are beginning to occur in areas of humid forests never known to have been burned [39]. Grassland and forest fires are common during the dry season but there is little information regarding the short- and long-term effects of fires. The cumulative effect of the current wildfire regime in Puerto Rico, over 5000 fires yearly, is unknown and it is expected that even slight climatic warming and drying areas have the potential to increase fire frequency and fire-related economic and ecological effects in the island.

In this study we analyzed field-collected data on fire occurrence in Puerto Rico between 2003 and 2011 to answer the questions: (1) How did fire occurrence vary in time (daily, seasonally, and yearly)? (2) How was fire occurrence related to decadal climate means (temperature and precipitation)? (3) What was the effect of land cover type on fire occurrence? and (4) What was the effect of population per *barrio* on fire occurrence? Our results will help to understand the conditions driving fire patterns and dynamics and hence inform predictive models of fire occurrence for projecting future scenarios of fire risk in the island.

2. Methods

2.1. Study area

The island of Puerto Rico is located in the northeastern Caribbean Sea, at ~17°45′N–18°30′N, and ~65°45′W–67°15′W. Its area is about 8740 km^2 and has a predominantly maritime climate, with orography strongly controlling local patterns and variation in decadal means of temperature and precipitation [40]. Early rainfall season occurs from May through June and a late rainfall season occurs from August to November. Puerto Rico is topographically diverse in terms of elevation and slope. Elevation ranges from sea level to 1338 m above the sea level in the central mountains. Therefore, climatic conditions across the island are highly variable. Six life zones have been described, ranging from subtropical dry forests to subtropical rain forests [41]. Dry, open forests are located in the south, and wetter, more closed forest are located in the north, east, and in the central mountains. The landscape is a complex matrix of wildlands, developed areas, and agricultural lands [42, 43].

2.2. Data acquisition

We used the information on fire occurrence collected by the Fire Department of Puerto Rico between 2003 and 2011. Information related to fire location was available for all the 78 municipalities of Puerto Rico. A total of 46,955 fires were reported by the Puerto Rico Fire Department occurring in this time period and we could assign the *barrio* (smallest administrative unit) information to 34,636 (74%) fires. Only this subset could be located at the *barrio* level because many fire locations were described using general information (e.g., route number but not km). Likewise, it was not possible to convert fire location descriptions into map points with unique latitude and longitude values. Fire location descriptions were used to determine the number of fires at the municipality and *barrio* levels. Around 832 of the 902 *barrios* were represented across the island (92%), capturing all the variability in climatic conditions (wet to dry environments), elevation (high to low elevation), and degree of urbanization (urban or rural areas). In addition, information about hour of occurrence was available for 1682 fires (~5%) occurring between 2008 and 2010.

Climate variables (daily temperature and precipitation) were obtained from the National Weather Service Cooperative Observer stations for the period 2002–2011 and interpolated across the island. Land cover classes were obtained from the Puerto Rico 2000 GAP Land Cover [43]. Population data were obtained from the US Census Bureau (2010 Census; [44]).

2.3. Data analysis

Fire occurrence was summarized across the island in relation to hour, month, and year of occurrence. A Kruskal-Wallis test was used to test for differences among hours, and ANOVA test were used to test for differences among months and among years. Generalized linear models (GLM) were used to identify the time variables and interactions that contributed most to fire occurrence at the *barrio* level.

To identify the climate variables that contributed most to fire occurrence we ran a robust linear model (robustbase package; [45]). Robust regression was used to account for non-normal measurement errors in the data, given their nature; this analysis provides a statistical framework from which to both identify and limit the influence of extreme values or leverage points on parameter estimation [46]. The number of fires per *barrio* was analyzed as the response variable and the climate variables mean daily maximum temperature, mean daily minimum temperature, mean daily annual precipitation, and interactions were analyzed as explanatory variables.

To determine the effects of land cover on the occurrence of wildfires we performed linear model selection using number of fires per *barrio* as the response variable and percent forest, percent woodland/shrubland, percent nonwoody vegetation (including grasses), percent urban, and percent forest edge per *barrio* as explanatory variables. A regression was run to account for the effect of population on the occurrence of wildfires (number of fires) at the *barrio* level.

Number of fires per *barrio* was log transformed and quadratic terms of the explanatory variables were added when necessary. Tree models were used in combination with linear models to examine the order of importance of the variables when necessary. All data analyses were performed using the *R* statistical package [47].

2.4. Results

Fires were registered and managed in the 91 fire stations located along the 78 municipalities on the island (**Figure 1**). On average, fire extent was 1.52 ha with a standard deviation of 4.67 ($n = 2472$). The number of fires per municipality ranged from 69 to 2174. The municipalities that registered more fire episodes between 2003 and 2011 were located mainly south of the island while municipalities with less number of fires reported were located north. The number of fires per 1000 persons was higher (40–56) in seven municipalities located south and west of the island (**Figure 1a**). The same spatial pattern was found in relation to municipality area, the highest numbers of fires per km^2 (10–17) were found in six municipalities located *mainly in the south* and *located in the north* west area of the island (**Figure 1b**). At the *barrio* level, the number of fires ranged up to 783. Most of the *barrios* with high number of fires were located in the south of the island (**Figure 1c**).

Most of the fires were reported during the afternoon (KW chi-squared = 653.08, df = 11, p < 2.2e–16: **Figure 2a**). Specifically, fire events increased significantly from around 15 fires per *barrio* at

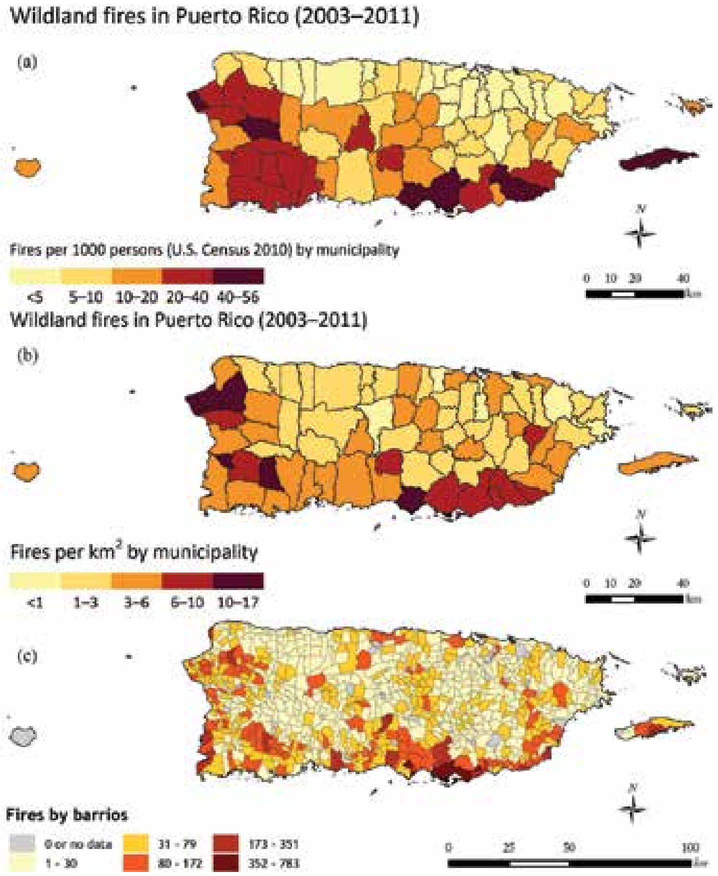

Figure 1. Maps of Puerto Rico showing the incidence of total number of fires (a) per 1000 persons by municipality, (b) per km^2 by municipality, and (c) per barrio where fire events were recorded from 2003 to 2011 by the Fire Department of Puerto Rico.

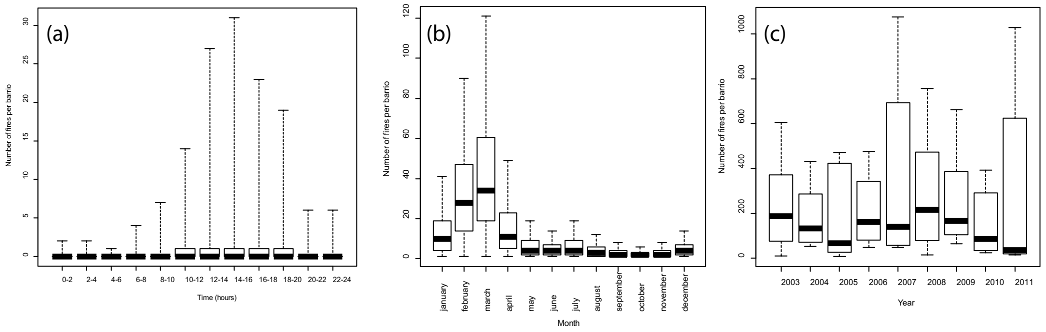

Figure 2. Box-and-whisker plots showing number of fires per barrio (a) throughout a day, (b) throughout a year, and (c) across years in Puerto Rico from 2003 to 2011. Each box shows the lower and upper quartiles, the black line within the box is the median, and the error bars are the minimum and maximum values, respectively.

10:00–12:00 hours ($n = 160$) to more than 30 fires at 14:00–16:00 hours ($n = 372$) and decreased to six fires at 20:00–22:00 hours ($n = 108$). Through the year, fires showed a peak between January and April ($F = 212.8$, df = 11, 2650, $p < 2.2\text{e}{-16}$; **Figure 2b**). Fire events increased from around 40 fires per *barrio* in January ($n = 3648$) to 120 fires in March ($n = 11674$) and then decreased to around 20 fires in May ($n = 1359$). In relation to year of occurrence, we found no significant differences among years from 2003 to 2011 ($F = 0.3663$, $df = 1, 106$, $p = 0.546$; **Figure 2c**). The GLMs combined with tree models showed that the interaction Hour*Month*Year was significant ($p = 2.466\text{e}{-16}$) and the order of importance of the three variables was Hour>Month>Year.

The number of fires per *barrio* was significantly influenced by all three climate variables, despite a high variability (Adj. R-squared = 0.062; **Table 1**, **Figure 3**). It was positively related to mean maximum daily temperature; the number of fires increased from 1 at 25°C to 783 at 30.7°C (**Figure 3a**). The number of fires per *barrio* was positively related to mean minimum daily temperature; it ranged from 1 at 16.2°C to 783 at 22.1°C (**Figure 3b**). The number of fires was negatively related to mean daily annual precipitation (**Figure 3c**); it decreased from 783 fires at 3.81 inches to 19 at 13.01 inches. All interactions performed in the robust regression were significant except for mean maximum daily temperature × Precipitation, which was marginally significant. The most significant relationship between number of fires per *barrio* and climate was mean minimum daily temperature (**Table 1**).

	Estimate	Std. error	t value	Pr(>|t|)
(Intercept)	135.79586	66.69509	2.036	0.0421*
Max.temp	−4.21917	2.23246	−1.890	0.0591.
Min.temp	−7.58724	3.39628	− 2.234	0.0258*
Precipitation	−19.46869	9.91169	−1.964	0.0498*
Max.temp: Min.temp	0.24307	0.11324	2.147	0.0321*
Max.temp: Precipitation	0.62015	0.33509	1.851	0.0646.
Min.temp: Precipitation	1.07691	0.50422	2.136	0.0330*
Max.temp: Min.temp: Precipitation	−0.03451	0.01697	− 2.034	0.0423*

Note: Max.temp: mean daily maximum temperature, Min.temp: mean daily minimum temperature, Precipitation: mean daily annual precipitation. Significance codes: 0.01 ‘*’; 0.05 ‘.’.

Table 1. Coefficients of the robust linear model using log(number of fires per *barrio*) as the response variable and climate variables as explanatory.

The best linear model explaining the effects of land cover on the occurrence of wildfires included simple terms, quadratic terms such as percent forest cover ^2, percent woodland and shrubland ^2, and percent nonwoody vegetation ^2, percent forest edge, and complex interactions (e.g., one five-term interaction) (Adj. R-squared = 0.2492, $F = 411.2$, $df = 28, 34582$, $p < 2.2\text{e}{-16}$; **Table 2**). According to the tree model, percent forest cover was the main variable explaining the number of fires per *barrio*, followed by nonwoody

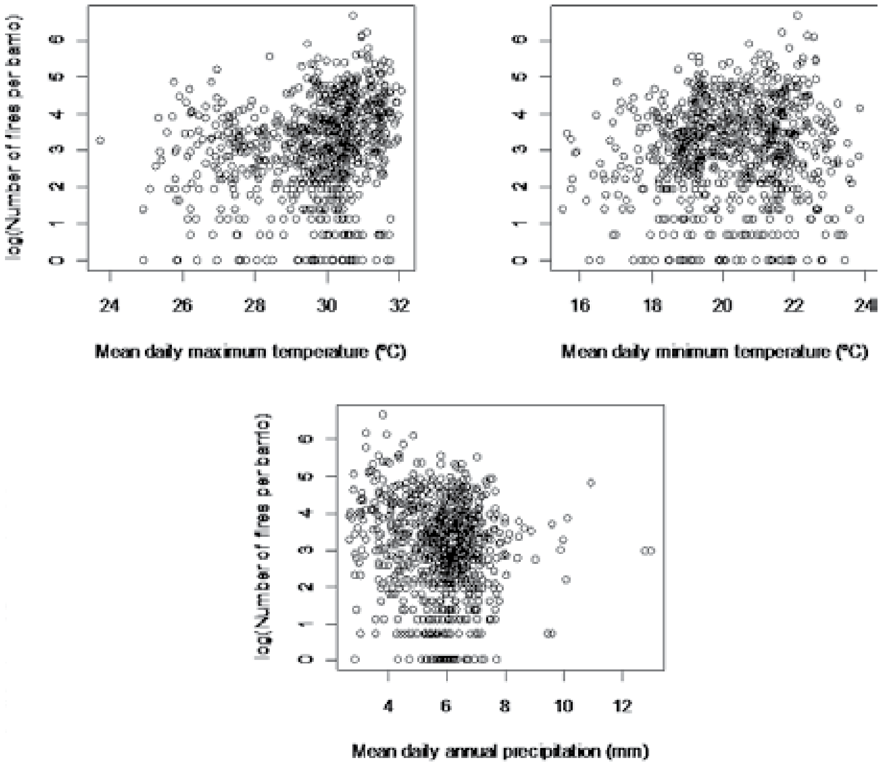

Figure 3. Scatterplots between the log(number of fires per barrio) and (a) mean daily maximum temperature, (b) mean daily minimum temperature, and (c) mean daily annual precipitation (see **Table 1** for coefficients).

Variable	Estimate	Std. error	t value	Pr(>|t|)
(Intercept)	3.009e+00	1.045e−01	28.810	<2e−16 ***
Forest	−1.589e−01	8.097e−03	−19.623	<2e−16 ***
Forest ^2	1.818e−03	7.659e−05	23.739	<2e−16 ***
Woodland and shrubland	2.430e−01	9.187e−03	26.455	<2e−16 ***
Woodland and shrubland ^2	−2.562e−03	9.672e−05	−26.494	<2e−16 ***
Non woody vegetation	8.168e−02	3.582e−03	22.803	<2e−16 ***
Non woody vegetation ^2	−5.936e−04	2.938e−05	−20.203	<2e−16 ***
Urban	9.217e−03	1.386e−03	6.650	2.97e−11 ***
Forest edge	1.391e+00	4.885e−02	28.483	<2e−16 ***
Forest: woodland and shrubland	−1.539e-03	1.145e−04	−13.435	<2e−16 ***
Forest: urban	1.786e−03	1.175e−04	15.202	<2e−16 ***

Variable	Estimate	Std. error	t value	Pr(>|t|)
Woodland and shrubland: urban	−1.725e−03	1.357e−04	−12.716	<2e−16 ***
Forest: non woody vegetation	7.496e−04	9.012e−05	8.318	<2e−16 ***
Woodland and shrubland: non woody vegetation	−2.633e−03	9.981e−05	−26.384	<2e−16 ***
Non woody vegetation: urban	−3.342e−04	5.014e−05	−6.666	2.66e−11 ***
Forest: forest edge	−1.519e−02	5.029e−04	−30.214	<2e−16 ***
Woodland and shrubland: forest edge	−2.205e−02	1.576e−03	−13.987	<2e−16 ***
Urban: forest edge	−2.413e−02	9.938e−04	−24.277	<2e−16 ***
Non woody vegetation: forest edge	−1.504e−02	6.052e−04	−24.843	<2e−16 ***
Forest: non woody vegetation: urban	−4.930e−05	3.473e−06	−14.195	<2e−16 ***
Woodland and shrubland: non woody vegetation: urban	−4.209e−05	4.967e−06	−8.473	<2e−16 ***
Forest: woodland and shrubland: forest edge	1.750e−04	1.880e−05	9.313	<2e−16 ***
Forest: urban: forest edge	2.064e−04	1.356e−05	15.225	<2e−16 ***
Woodland and shrubland: urban: forest edge	5.486e−04	6.695e−05	8.194	2.62e−16 ***
Forest: non woody vegetation: forest edge	1.467e−05	4.783e−06	3.066	0.00217 **
Woodland and shrubland: non woody vegetation: forest edge	1.704e−04	2.811e−05	6.061	1.37e−09 ***
Forest: woodland and shrubland: non woody vegetation: urban	5.076e−06	2.118e−07	23.973	<2e−16 ***
Forest: non woody vegetation: urban: forest edge	7.151e−06	1.059e−06	6.755	1.45e−11 ***
Forest: woodland and shrubland: non woody vegetation: urban: forest edge	7.991e−07	8.238e−08	−9.701	<2e−16 ***

Note: (Adj. R-squared = 0.2492, $p < 2.2e{-}16$). Significance codes: 0 '***', 0.001 '**'.

Table 2. Coefficients of the best linear model using log(number of fires per *barrio*) as response variable and percent land covers and percent forest edge as explanatory variables.

vegetation cover and urban cover. The log(number of fires) was negatively related to percent forest cover (**Figure 4a**). The number of fires decreased from 400–783 at 1–7% forest to 1–5 at 0–91% forest. The log(number of fires) was slightly positively related to percent urban cover (**Figure 4b**). Fires increased from 1–5 at 0–7% urban cover to 400–783 at 8–64% urban cover. The log(number of fires) was positively related to percent nonwoody vegetation cover (**Figure 4c**); fires increased from 1–10 at 0–78% cover to 436–783 at 28–84% cover. Lastly, the log(number of fires) was slightly negatively related to woodland and shrubland cover (**Figure 4d**). The number of fires decreased from 783 at 3% woodland-shrubland cover to 1 at 55% cover. A high variability was observed in the data in all cases.

The log(number of fires) was positive but weakly related to population (Adj. R-squared = 0.038, $p = < 2.2\text{e-}16$; **Figure 5**). The greatest incidence of fire occurred in *barrios* of intermediate population density.

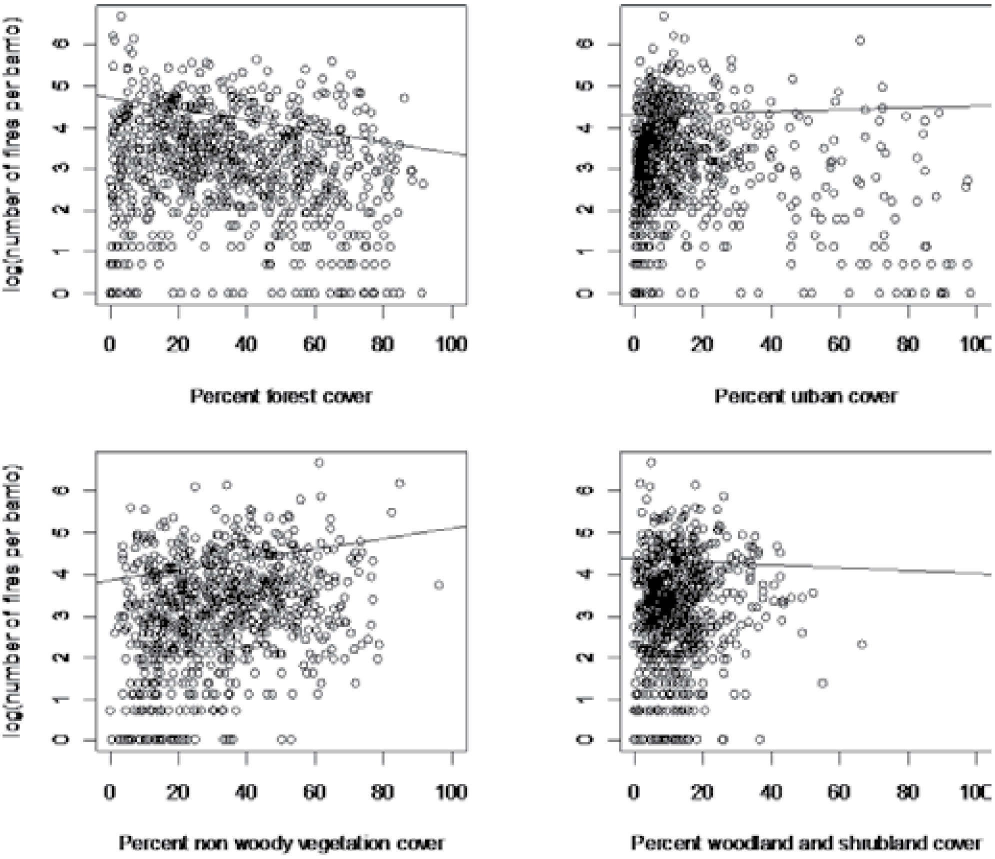

Figure 4. Scatterplots between the log(number of fires per barrio) and (a) percent forest cover, (b) percent urban cover, (c) percent non-woody vegetation cover, and (d) percent woodland and shrubland cover (see **Table 2** for coefficients).

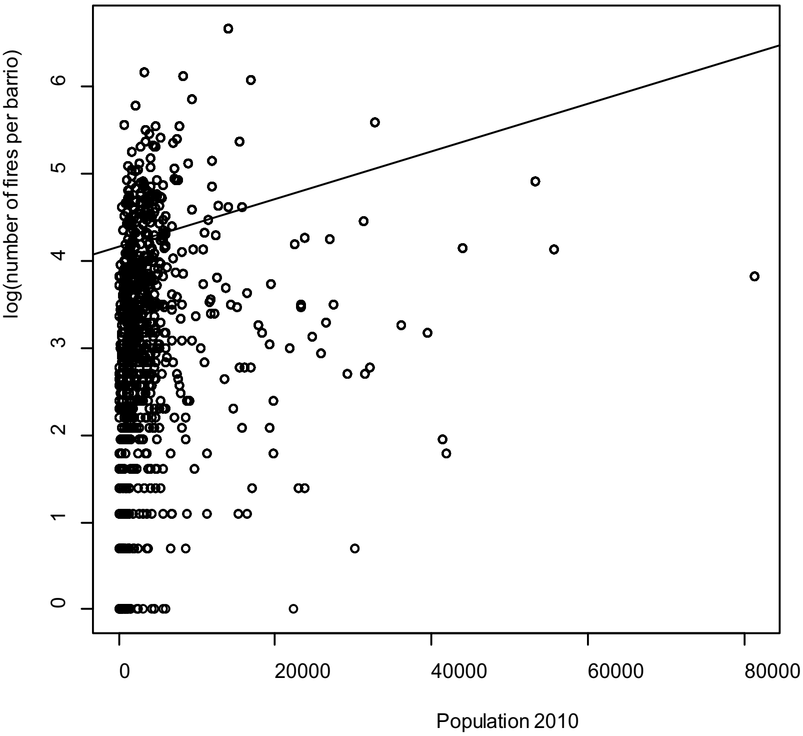

Figure 5. Scatterplot between the log(number of fires per barrio) and population according to the 2010 U.S. Census.

2.5. Discussion

We analyzed a total of 34,636 fires, most of them of small extent, reported by the Puerto Rico Fire Department from 2003 to 2011 for the entire island. Hour of the day (time), mean minimum temperature (climate) and percent forest cover (land cover) were found to significantly influence fire occurrence, while population showed a weak effect. In addition, many variable interactions showed to be important. These significant variables and interactions can be useful if considered in fire occurrence models for assessing fire risk under potential future conditions.

Our data support the idea that active fires occur in the tropics [25]. More fires were recorded at the municipalities in the southern, drier areas of the island than in the northern, more humid areas. Lowland moist and montane forests are less susceptible to fires than dry forests, but it was remarkable that in our study fires occurred in high frequency in many northern and central forests. These results support the idea that fires are beginning to occur in areas of humid forests never known to have been burned [39] and may be an indirect evidence of an increasing fire frequency through time, as suggested [21]. Alternatively, this may be a result of recording fires only recently in Puerto Rico, in contrast to temperate regions.

Time of the day was the most important determinant of fire occurrence among the time variables. It has been shown that "typical" fire weather conditions observed during the day can be abruptly worsened and fire risk increased when rapid increase in wind speed, decrease in relative humidity or both take place [48, 49]. This pattern is apparently independent of daily temperature thus we suggest to consider local conditions of wind and humidity in predicting models for Puerto Rico. Not surprisingly, fire occurrence was markedly seasonal and corresponded to the driest period of the year (January–April). Our results, though, did not show direct evidence of an increase in fire frequency through years, as suggested for other regions of the globe [21]. This result could change if the time frame for analysis is extended; alternatively, fire extent and/or intensity could be increasing over time but these aspects of fire were not evaluated in this study. The climatic variables mean daily maximum temperature, mean daily minimum temperature, and mean daily annual precipitation, averaged over the period 2003–2011, were significantly correlated with wildfire occurrence in Puerto Rico. In contrast to other tropical and subtropical countries [22, 23], mean annual precipitation was not the most important single influencing factor for fire occurrence. Instead, mean daily minimum temperature and daily thermal amplitude represented by the interaction between temperature maximum and minimum were more determinant for fire occurrence and of special consideration in predicting models. In another study using random forests and aggregating data into different day intervals we found that precipitation in fact explained fire occurrence better than temperature variables suggesting that precipitation variability rather than mean precipitation is a better predictor of fire occurrence in Puerto Rico [49]. In both cases, mean daily minimum temperature was more important than maximum temperature to explain fires. Given the projections that precipitation will decline in tropical islands including Puerto Rico and that this decline will cause increases in drought intensity [33], special care should be taken when selecting climatic variables for fire prediction models.

Land cover and forest edge were determinant for fire occurrence and the variability explained by the model was higher than that explained by climatic variables and population. This result contrasts with other studies that found stronger influences of climate than LULC on fire occurrence when using remote sensors [17, 22, 50]. An open question remains how different resolution fire data are explained by variables operating at different scales. Percent forest cover, negatively related to fire occurrence, was the main variable in our study explaining the number of fires per *barrio*, followed by nonwoody vegetation cover and to a lesser degree, urban cover, both positively related to fire occurrence. The negative relationship between percent forest and fire occurrence confirms patterns observed in other tropical regions [22]. From a subset of the Fire Department data we could determine that grass (GR4 and GR3 fuel models) was the dominant fuel type in Puerto Rico, followed by forest (TU and TL fuel models) (results not shown). Previous studies have shown that this is a general pattern in which fires occur mainly in dry forests, the most threatened tropical forest type [24, 32], and in grasslands and crop lands [25]. Grass and shrubland communities alter the vegetation cover [27–29] creating new fire-prone ecosystems and the land cover change favoring the establishment of grass would increase fire risk in Puerto Rico, as previously suggested [34, 35]. At the same time, the large percentage of wildland-urban interface (WUI) in Puerto Rico (36%) [38] could be promoting the occurrence of thousands of small to intermediate (<4 ha) scale human-induced fires as a result of closer interactions between humans, infrastructure, and natural lands. In this regard, in Puerto Rico and regions with similar climate and landscape mosaic characteristics, fire occurrence models may improve when they include climatic and LULC variables as both are significantly correlated with occurrence. Furthermore, integrating LULC variables in fire occurrence predictability is crucial in changing landscapes. In Puerto Rico, agriculture is slowly becoming an emergent economy in the island. Land use-land cover changes are expected to occur and the degree of land change would have a direct effect on fuel loads and hence on fire occurrence.

Population had the weakest effect on fire occurrence among the variables examined. A subset of the Fire Department data showed that for most of the fires reported in Puerto Rico between 2008 and 2010, the cause of ignition was unknown (results not shown). Given that historical information suggests that only 5% of the unintended wildland fires in Puerto Rico in 1999 were caused by lightning and 95% were human-caused [24], we conclude that those fires were the source of ignition was unknown were human caused. Given that nearly all fires are human caused, the action of people in combination with climate and LULC likely determines the ignition, spread and extent of fires in the island. The scenarios of population density and land use are changing throughout the tropics in complex ways, for example over the last decade in Puerto Rico, population has decreased while housing units have increased [51]. Thus, changes in fire occurrence increase may not be a consequence of population density per se, but can be responding to changing land use. Weak effects of population on fire occurrence have been reported in other regions of the world where humans influence variables were most strongly associated with fire size [50].

Additional factors not considered in this study could be added to predictive models in order to improve accuracy. In previous studies conducted in tropical and nontropical environments road density was determinant for fire occurrence [22, 52]. Roads increase the WUI, facilitating human access to shrubland, grassland and forest areas, increasing the probabilities of fire episodes. Furthermore, roads alter soil, microclimatic conditions and native vegetation [53] creating a fire prone environment. Puerto Rico is one of the countries with highest road density in the Caribbean [54] and including this variable in future models would provide a better understanding of fire occurrence based on climate and land local conditions.

Finally, we think that fire model accuracy can be improved by standardizing field data collection among fire managers and improving data acquisition such as recording exact fire location (i.e. latitude and longitude). This would increase data resolution and therefore models' accuracy. Furthermore, as it has been suggested that Random Forest models are better than Multiple Linear Regressions at predicting fire occurrence in some regions of the world [52], we recommend that different methods be implemented to compare their performance in tropical islands where the nature of data may be different.

2.6. Wildfires' effects on forest ecosystems and conservation

The occurrence of fires either in fire-adapted ecosystems (i.e., fire is a natural disturbance) or in non-fire-adapted ecosystems (i.e., fire is not a natural disturbance) produces changes in vegetation composition and structure, alters fuel loads and biomass, and shapes the landscape [3, 5, 6]. These changes are favorable in wildfire-prone ecosystems where fires are key in preserving ecosystem processes and promoting ecosystem resilience, while they can have adverse implications in ecosystems where fires do not occur naturally (e.g., [10]).

In temperate forests where fire is a natural disturbance native species show different adaptations or life-history strategies such as resprouting from below-ground despite top-kill, preventing top-kill through fire resistance by the increase of bark thickness, among others [55]. In these forests with fire history succession paths depend strongly on fire events and infrequent severe weather conditions can lead to the burning of relatively fire-resistant forests with consequent expansion of shrublands at the expense of forests [12]. The result of these processes is an increase in anthropogenic ignitions and a conversion from forest to shrubland. In Puerto Rico, fire is not a natural disturbance in modern times. Most of the native plant species in Puerto Rico dry forests have a bark structure that unlikely prevents top-kill in even low-intensity fires [55], and hence are not well-adapted to fire episodes. Therefore, even a single low-intensity fire has the potential to kill most of the trees [55] and to change the structure and composition of the forest. In this regard, fires have a negative effect on forest restoration success and this is critical for the conservation of unique forest ecosystems such as the dry forest. In dry forests, a single fire episode can eliminate years of forest regeneration [56]. In addition, secondary forests in Puerto Rico are characterized by the presence of a variety of exotic species. In southern dry forests of Puerto Rico, exotic species such as *Leucaena leucocephala* are fire-adapted, spread easily after fire episodes and can establish into previously forested areas. These exotic fire-adapted species can dominate the canopy forest and maintain a fire regime, preventing the establishment of native trees and shrubs [57].

Fire management policies in temperate, fire-adapted forests should be different from policies applied in tropical and subtropical forests with no fire history. Common management practices in fire-adapted forests include to schedule and conduct fire ignitions under a highly controlled regime and to use wildland fires (i.e., allow natural fires to burn). These practices are used to remove excess fuel and to stimulate native plant growth and regeneration. In contrast, in Puerto Rico this kind of fire management practices are scarce. The only example is in Guánica Forest where prescribed fires have been used since 1986 in grass-invaded areas along roadsides during the beginning of the dry season to reduce grass fuels and to limit the occurrence of uncontrolled fires into adjacent forest [57, 58]. In this regard, in Puerto Rico and other tropical forests the conservation of native forests will be successfully achieved if fires are not prescribed but suppressed and avoided. Effective fire suppression will be achieved by speeding the firefighter response especially between 14:00 and 16:00 and by improving personal training. Based on our results, special attention should be given to days when minimum temperatures are extremely high and to regions of the island where forest cover is low. In addition, our data showed a high incidence of human-caused fires in different regions of the island, especially in dry forests where species are not fire-adapted. Due to the fact that most of forest fires are human caused, and given the large extent of WUI in Puerto Rico, education and awareness about fires in high sensitive areas is an important strategy that is being implemented.

2.7. Conclusions

Including hour of the day, mean minimum temperature, and percent forest cover in models predicting fire occurrence will likely improve accuracy for fire management in Puerto Rico. Surprisingly, including population does not show a strong effect on fire occurrence but a question remains open about its effect on fire size. These results are particularly relevant to design fire management practices that lead to successful forest conservation.

Our results were based on one of the largest fire datasets in the Caribbean and other tropical regions. The variables analyzed in our study have been explored in other tropical environments and some differences have been found, especially in relation to the relative importance of climate vs. LULC variables. Predicting fire occurrence in a context of global change is a challenge; care should be taken when analyzing individual tropical islands especially when taking variables' interactions into account. A deep understanding of socioecological interactions in each case is necessary to incorporate relevant variables into fire predictive models and understand model's output to correctly translate them into management actions. We anticipate that standardizing field data collection and comparing different statistical methods in tropical islands will improve our understanding of fire occurrence in these environments.

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References

[1] Ryan KC. Vegetation and wildland fire: implications of global climate change. Environ Int [Internet]. 1991;17(2–3):169–78. Available from: http://dx.doi.org/10.1016/ 0160-4120 (91) 90099-c

[2] Kauffman JB, Boone Kauffman J, Cummings DL, Ward DE. Relationships of fire, biomass and nutrient dynamics along a vegetation gradient in the Brazilian Cerrado. J Ecol [Internet]. 1994;82(3):519. Available from: http://dx.doi.org/10.2307/2261261

[3] Sauvajot R. Conservation Science in fire-prone natural areas. In: Keeley J E, editor. Brushfires in California wildlands: Ecology and Resource Management. Fairfield, WA: International Association of Wildland Fire; 1995. p. 11–9.

[4] Shackleton CM, Scholes RJ. Impact of fire frequency on woody community structure and soil nutrients in the Kruger National Park. Koedoe [Internet]. 2000;43(1):75–81. Available from: http://dx.doi.org/10.4102/koedoe.v43i1.210

[5] Arseneault D, Dominique A. Impact of fire behavior on postfire forest development in a homogeneous boreal landscape. Can J For Res [Internet]. 2001;31(8):1367–74. Available from: http://dx.doi.org/10.1139/cjfr-31-8-1367

[6] Veblen TT, Thomas K, Joseph D. Climatic and human influences on fire regimes in ponderosa pine forests in the Colorado front range. Ecol Appl [Internet]. 2000;10(4):1178. Available from: http://dx.doi.org/10.2307/2641025

[7] Venevsky S, Sergey V, Kirsten T, Stephen S, Wolfgang C. Simulating fire regimes in human-dominated ecosystems: Iberian Peninsula case study. Glob Chang Biol [Internet]. 2002;8(10):984–98. Available from: http://dx.doi.org/10.1046/j.1365-2486.2002.00528.x

[8] Westerling AL, Gershunov A, Brown TJ, Cayan DR, Dettinger MD. Climate and wildfire in the western United States. Bull Am Meteorol Soc [Internet]. 2003;84(5):595–604. Available from: http://dx.doi.org/10.1175/bams-84-5-595

[9] Iniguez JM, Swetnam TW, Baisan CH. Spatially and temporally variable fire regime on Rincon Peak, Arizona, USA. Fire Ecol [Internet]. 2009;5(1):3–21. Available from: http://dx.doi.org/10.4996/fireecology.0501003

[10] Jolly WM, Cochrane MA, Freeborn PH, Holden ZA, Brown TJ, Williamson GJ, et al. Climate-induced variations in global wildfire danger from 1979 to 2013. Nat Commun [Internet]. 2015 Jul 14;6:7537. Available from: http://dx.doi.org/10.1038/ncomms8537

[11] Donnegan JA, Veblen TT, Sibold JS. Climatic and human influences on fire history in Pike National Forest, central Colorado. Can J For Res [Internet]. 2001;31(9):1526–39. Available from: http://dx.doi.org/10.1139/x01-093

[12] Mermoz M, Mónica M, Thomas K, Veblen TT. Landscape influences on occurrence and spread of wildfires in Patagonian forests and shrublands. Ecology [Internet]. 2005;86(10):2705–15. Available from: http://dx.doi.org/10.1890/04-1850

[13] Johnson SD, Randy B. Seasons within the wildfire season: marking weather-related fire occurrence regimes. Fire Ecol [Internet]. 2006;2(2):60–78. Available from: http://dx.doi.org/10.4996/fireecology.0202060

[14] Bajocco S, Sofia B. Modelling fire occurrence at regional scale: does vegetation phenology matter? Eur J Remote Sens [Internet]. 2015;763-775. Available from: http://dx.doi.org/10.5721/eujrs20154842

[15] Gallardo M, Marta G, Israel G, Lara V, Javier M-V, Martín MP. Impacts of future land use/land cover on wildfire occurrence in the Madrid region (Spain). Regional Environ Change [Internet]. 2015;16(4):1047–61. Available from: http://dx.doi.org/10.1007/s10113-015-0819-9

[16] Hantson S, Stijn H, Salvador P, Emilio C. Global fire size distribution is driven by human impact and climate. Glob Ecol Biogeogr [Internet]. 2014;24(1):77–86. Available from: http://dx.doi.org/10.1111/geb.12246

[17] Huang Y, Yaoxian H, Shiliang W, Kaplan JO. Sensitivity of global wildfire occurrences to various factors in the context of global change. Atmos Environ [Internet]. 2015;121:86–92. Available from: http://dx.doi.org/10.1016/j.atmosenv.2015.06.002

[18] Liu Z, Wimberly MC. Direct and indirect effects of climate change on projected future fire regimes in the western United States. Sci Total Environ [Internet]. 2016 Jan 15;542(Pt A):65–75. Available from: http://dx.doi.org/10.1016/j.scitotenv.2015.10.093

[19] van der Werf GR, Randerson JT, Louis G, Nadine G, Dolman AJ. Climate controls on the variability of fires in the tropics and subtropics. Global Biogeochem Cycl [Internet]. 2008;22(3). Available from: http://dx.doi.org/10.1029/2007gb003122

[20] Dwyer E, Gregoire JM, Malingreau JP. A global analysis of vegetation fires using satellite images: spatial and temporal dynamics. Ambio [Internet]. 1998;27(3):175–81. Available from: http://publications.jrc.ec.europa.eu/repository/handle/JRC14990

[21] Cochrane MA. Fire science for rainforests. Nature [Internet]. 2003;421(6926):913–19. Available from: http://dx.doi.org/10.1038/nature01437

[22] Armenteras D, Dolors A, Cerian G, Carla V, Juan E, Wania D, et al. Interactions between climate, land use and vegetation fire occurrences in El Salvador. Atmosphere [Internet]. 2016;7(2):26. Available from: http://dx.doi.org/10.3390/atmos7020026

[23] Guo F, Futao G, Zhangwen S, Guangyu W, Long S, Fangfang L, et al. Wildfire ignition in the forests of southeast China: identifying drivers and spatial distribution to predict wildfire likelihood. Appl Geogr [Internet]. 2016;66:12–21. Available from: http://dx.doi.org/10.1016/j.apgeog.2015.11.014

[24] Glogiewicz J, Baez J. Vegetation fire dynamics in Puerto Rico: a report about its incidence, cause, and danger, with emphasis on the urban-rural interface. International Institute of Tropical Forestry, USDA Forest Service; 2001.

[25] Robbins AMJ, Claus-Martin E, Maya Q. Forest Fires in the Insular Caribbean. AMBIO: J Human Environ [Internet]. 2008;37(7):528–34. Available from: http://dx.doi.org/10.1579/0044-7447-37.7.528

[26] Goldammer JG. Fire in the tropical biota: ecosystem processes and global challenges [Internet]. Vol. 84. Springer-Verlag, Berlin-Heidelberg-New York. 1990;497 p. Available

from: https://www.google.com/books?hl=es&lr=&id=7h7yCAAAQBAJ&oi=fnd&pg=PA1&dq=Goldammer,+J.G.+(2012).+Fire+in+the+Tropical+Biota:+Ecosystem+Processes+and+Global+Challenges+(Springer+Science+%26+Business+Media).&ots=swwu1fpPIc&sig=1LHUMlxVhG1fpIwqb-7GsuXGUDY

[27] Uhl C, Christopher U, Boone Kauffman J. Deforestation, fire susceptibility, and potential tree responses to fire in the eastern Amazon. Ecology [Internet]. 1990;71(2):437–49. Available from: http://dx.doi.org/10.2307/1940299

[28] Cochrane MA, Schulze MD. Fire as a recurrent event in tropical forests of the eastern Amazon: effects on forest structure, biomass, and species composition. Biotropica [Internet]. 1999;31(1):2. Available from: http://dx.doi.org/10.2307/2663955

[29] Cochrane MA. Synergistic interactions between habitat fragmentation and fire in evergreen tropical forests. Conserv Biol [Internet]. 2001;15(6):1515–21. Available from: http://dx.doi.org/10.1046/j.1523-1739.2001.01091.x

[30] Hoffmann WA, Birgit O, do Nascimento PKV. Comparative fire ecology of tropical savanna and forest trees. Funct Ecol [Internet]. 2003;17(6):720–6. Available from: http://dx.doi.org/10.1111/j.1365-2435.2003.00796.x

[31] Chuvieco E, Emilio C, Louis G, Chris J. Global characterization of fire activity: toward defining fire regimes from Earth observation data. Glob Chang Biol [Internet]. 2008;14(7):1488–502. Available from: http://dx.doi.org/10.1111/j.1365-2486.2008.01585.x

[32] Miles L, Lera M, Newton AC, DeFries RS, Corinna R, Ian M, et al. A global overview of the conservation status of tropical dry forests. J Biogeogr [Internet]. 2006;33(3):491–505. Available from: http://dx.doi.org/10.1111/j.1365-2699.2005.01424.x

[33] Khalyani AH, Gould WA, Eric H, Adam T, Maya Q, Collazo JA. Climate change implications for tropical islands: interpolating and interpreting statistically downscaled GCM projections for management and planning*. J Appl Meteorol Climatol [Internet]. 2016;55(2):265–82. Available from: http://dx.doi.org/10.1175/jamc-d-15-0182.1

[34] Aide TM, T.Mitchell A, Zimmerman JK, Luis H, Maydee R, Mayra S. Forest recovery in abandoned tropical pastures in Puerto Rico. For Ecol Manage [Internet]. 1995;77(1-3):77–86. Available from: http://dx.doi.org/10.1016/0378-1127(95)03576-v

[35] Lugo AE, Eileen H. Emerging forests on abandoned land: Puerto Rico's new forests. For Ecol Manage [Internet]. 2004;190(2–3):145–61. Available from: http://dx.doi.org/10.1016/j.foreco.2003.09.012

[36] Grau HR, Mitchell Aide T, Zimmerman JK, Thomlinson JR, Eileen H, et al. The ecological consequences of socioeconomic and land-use changes in postagriculture Puerto Rico. Bioscience [Internet]. 2003;53(12):1159. Available from: http://dx.doi.org/10.1641/0006-3568(2003)053[1159:tecosa]2.0.co;2

[37] Parés-Ramos IK, Gould WA, Aide TM. Agricultural abandonment, suburban growth, and forest expansion in Puerto Rico between 1991 and 2000. Ecol Soc [Internet]. 2008;13(2):1. Available from: http://www.academia.edu/download/8408122/10092.pdf

[38] Martinuzzi S, Sebastián M, Gould WA, Ramos González OM. Land development, land use, and urban sprawl in Puerto Rico integrating remote sensing and population census data. Landsc Urban Plan [Internet]. 2007;79(3–4):288–97. Available from: http://dx.doi.org/10.1016/j.landurbplan.2006.02.014

[39] Burney DA, Burney LP, MacPhee RDE. Holocene charcoal stratigraphy from Laguna Tortuguero, Puerto Rico, and the timing of human arrival on the island. J Archaeol Sci [Internet]. 1994;21(2):273–81. Available from: http://dx.doi.org/10.1006/jasc.1994.1027

[40] Daly C, Christopher D, Helmer EH, Maya Q. Mapping the climate of Puerto Rico, Vieques and Culebra. Int J Climatol [Internet]. 2003;23(11):1359–81. Available from: http://dx.doi.org/10.1002/joc.937

[41] Ewel JJ, Whitmore JL. The ecological life zones of Puerto Rico and the US Virgin Islands. 1973; Available from: http://www.treesearch.fs.fed.us/pubs/5551

[42] Van Beusekom AE, Hay LE, Viger RJ, Gould WA, Collazo JA, Khalyani AH. The effects of changing land cover on streamflow simulation in Puerto Rico. JAWRA J Am Water Resourc Assoc [Internet]. 2014;50(6):1575–93. Available from: http://dx.doi.org/10.1111/jawr.12227

[43] Gould WA, Alarcón C, Fevold B, Jiménez ME, Martinuzzi S, Potts G, et al. The Puerto Rico Gap Analysis Project volume 1: land cover, vertebrate species distributions, and land stewardship [Internet]. U.S. Department of Agriculture, Forest Service, International Institute of Tropical Forestry; 2008. Available from: http://www.treesearch.fs.fed.us/pubs/38430

[44] 2010 Census Data [Internet]. United States Census 2010. 2010 [cited 2016 Oct 31]. Available from: http://www.census.gov/2010census/data/

[45] Website [Internet]. [cited 2016 Oct 31]. Available from: Rousseeuw, P., C. Croux, V. Todorov, A. Ruckstuhl, M. Salibian-Barrera, T. Verbeke, M. Koller, M. Maechler. 2016. robustbase: Basic Robust Statistics. R package version 0.92-6. URL http://CRAN.R-project.org/package=robustbase).

[46] Bowlby HD, Gibson AJF. Environmental effects on survival rates: robust regression, recovery planning and endangered Atlantic salmon. Ecol Evol [Internet]. 2015;5(16):3450–61. Available from: http://dx.doi.org/10.1002/ece3.1614

[47] Website [Internet]. [cited 2016 Oct 31]. Available from: R Core Team (2016). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

[48] Fox-Hughes P, Paul F-H. Characteristics of some days involving abrupt increases in fire danger*. J Appl Meteorol Climatol [Internet]. 2015;54(12):2353–63. Available from: http://dx.doi.org/10.1175/jamc-d-15-0062.1

[49] Van Beusekom A, Gould WA, Monmany AC, Henareh Khalyani A, Quiñones M, Fain SJ, et al. Fire weather and likelihood: modeling climate space for fire occurrence and extent in Puerto Rico. unpublished ms

[50] Liu Z, Wimberly MC. Climatic and landscape influences on fire regimes from 1984 to 2010 in the western United States. PLoS One [Internet]. 2015 Oct 14;10(10):e0140839. Available from: http://dx.doi.org/10.1371/journal.pone.0140839

[51] Castro-Prieto J, Martinuzzi S, Radeloff VC, Helmers D, Quiñones M, and Gould WA. Declining human population but increasing residential development around protected areas in Puerto Rico. In press. Biological Conservation.

[52] Oliveira S, Sandra O, Friderike O, Jesús S-M-A, Andrea C, Pereira JMC. Modeling spatial patterns of fire occurrence in Mediterranean Europe using multiple regression and random forest. For Ecol Manage [Internet]. 2012;275:117–29. Available from: http://dx.doi.org/10.1016/j.foreco.2012.03.003

[53] Olander LP, Scatena FN, Silver WL. Impacts of disturbance initiated by road construction in a subtropical cloud forest in the Luquillo Experimental Forest, Puerto Rico. For Ecol Manage [Internet]. 1998;109(1–3):33–49. Available from: http://dx.doi.org/10.1016/s0378-1127(98)00261-8

[54] López TM, Aide TM, Thomlinson JR. Urban expansion and the loss of prime agricultural lands in Puerto Rico. Ambio [Internet]. 2001 Feb;30(1):49–54. Available from: https://www.ncbi.nlm.nih.gov/pubmed/11351793

[55] Wolfe BT, Saldaña Diaz GE, Van Bloem SJ. Fire resistance in a Caribbean dry forest: inferences from the allometry of bark thickness. J Trop Ecol [Internet]. 2014;30(02):133–42. Available from: http://dx.doi.org/10.1017/s0266467413000904

[56] Aide TM, Mitchell Aide T, Zimmerman JK, Pascarella JB, Rivera L, Marcano-Vega H. Forest regeneration in a chronosequence of tropical abandoned pastures: implications for restoration ecology. Restor Ecol [Internet]. 2000;8(4):328–38. Available from: http://dx.doi.org/10.1046/j.1526-100x.2000.80048.x

[57] Wolfe BT, Van Bloem SJ. Subtropical dry forest regeneration in grass-invaded areas of Puerto Rico: understanding why Leucaena leucocephala dominates and native species fail. For Ecol Manage [Internet]. 2012;267:253–61. Available from: http://dx.doi.org/10.1016/j.foreco.2011.12.015

[58] Thaxton JM, Van Bloem SJ, Whitmire S. fuel conditions associated with native and exotic grasses in a subtropical dry forest in Puerto Rico. Fire Ecol [Internet]. 2012;8(3):9–17. Available from: http://dx.doi.org/10.4996/fireecology.0803009

Section 2

Forest Conservation

Chapter 5

Remote Sensing and Forest Conservation: Challenges of Illegal Logging in Kursumlija Municipality (Serbia)

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Additional information is available at the end of the chapter

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Abstract

Evidence convincingly shows that illegal and corrupt activities are the major underlying cause of deforestation—illegal logging contributes up to 30% of the global market, in excess of US $20 billion a year. Since so much deforestation is a result of illegal logging, we cannot rely on official production statistics to capture deforestation. Given the importance and complexity of forest preservation, an attempt was made to evaluate the possible use of a normalized difference vegetation index (NDVI) in local forest management and prevention of illegal logging and corruption. We used the example of southern Serbian municipality Kursumlija that in the 2006–2011 periods experienced a 10% loss in forest area, as the obvious result of abrupt illegal logging. This process was very easy to locate and quantify (because illegal logging produced large canopy gaps that extend from the border of Kosovo to approximately 3–4 km into the Kursumlija's territory). In short, NDVI is very promising for countries like Serbia (that rarely perform forest inventories): It is relatively cheap and quick, and it can provide forest managers with essential information; it is easy to implement; the objectivity of these methods can significantly help in avoiding corruption and illegal logging.

Keywords: illegal logging, deforestation, normalized difference vegetation index (NDVI), local forest management, Serbia, Kursumlija

1. Introduction

Forests are under severe threat in many parts of the world. An average of almost 15 million hectares of forest was lost every year in 1990–2000 periods [1]. Forest decline consists of deforestation, forest degradation, and/or a combination of both. The Food and Agriculture

Organization of the United Nations defines deforestation as the sum of all transitions from natural forest classes (continuous and fragmented) to all other classes [2]. The loss of forest cover attributed to these transitions must be the reduction of tree crown cover to less than 10% of the total area for the phenomenon to qualify as deforestation [3]. While forest degradation events only partially and temporarily remove forest canopy cover, in deforestation there is near-complete removal of the original forest cover. Deforestation is an ongoing process of converting forested land to other land uses, such as pastures, agricultural fields, mining, or regional urbanization [4].

There is enough available evidence that convincingly shows that *illegal and corrupt activities* constitute a major underlying cause of forest decline worldwide [3, 5–7] and that illegal use of forests is rampant [3, 8]. Hence, illegal use of forests and deforestation is at the top of the current global policy agenda, especially understanding how to counter illegal extraction, since illegal logging contributes up to 30% of the global market, in excess of US $20 billion a year [9–13]. For some countries, such as Cambodia, Indonesia, Bolivia, indicative estimates of illegal logging even exceed 80% [14–16].

In post-socialist European countries, also, forest management changed drastically during the transition period—an increase in forest timber extraction, *including substantial illegal logging*, became common throughout the region. The illegal logging was particularly evident during the early transition years when poverty was at its peak and institutional oversight of forests was at its weakest [17]. Illegal logging is also very pronounced in southern Serbian municipalities bordering Kosovo (which is ranked as the one of the worst offenders in the world with Indonesia by Transparency International (TI) based on the percentage of illegal logging) [18].

Actually, the main problem concerning forest policies and management stems from the fact that ignoring common goods—like forests—which are difficult to produce and easy to deplete [19–21], leads to tragic results, since it is very difficult to restrict the rate at which they are consumed [22–24]. The size of many forests and the inevitable complications involved in monitoring the use of the forest and balancing one use against another, make exclusion or restrictions on access intrinsically problematic [25, 26].

Forest conservation, as a concept, has evolved from simple preservation ideas how natural resources should be used (essentially that their rate of use should not exceed the ability of the resource to replenish itself), to the very complex conservation concept that covers the equitable sharing of benefits derived from the resources, in the present and in the future (similar to the definition of sustainable development adopted by the World Commission on Environment and Development in 1987). Unfortunately, conservation managers are often faced with making decisions without access to reliable scientific information about what the potential outcomes of alternative management actions might be [27]. Such information is often lacking, partly because the scientific community has traditionally failed to address research questions of direct relevance to management practice, and partly because the scientific information that is available is often not readily accessible by conservation managers.

Chakravarty et al. point out that although it is not possible to properly manage a forest ecosystem without first understanding it, it is common that even the most basic information about forests is not always available … but new remote sensing technologies make it feasible and affordable to identify hotspots of deforestation [22]. Nevertheless, remote sensing (detection, recognition, or evaluation of objects by means of distant sensing or recording devices) that has recently emerged to support data collection and analysis methods in forest management [28–30] is still rarely used, not well understood, and probably not well suited by forest managers who might best use it [31].

It has been often pointed out that successful remote sensing applications in forestry were made only: (a) over an area too large or (b) otherwise difficult to survey on the ground [31]. But, there is now a pronounced shift around the globe of forest management authority from central government to municipalities (in Bolivia, Zimbabwe, Tanzania, Indonesia, Philippines, India, USA, Canada, China, etc.) [32], since forest policies are not likely to work when imposed on a country as a whole [23]. Actually, the aim of good *local* forest management is to: (1) strengthen the local rule of law, (2) improve local accountability and transparency (especially, through establishing clear mechanisms for the provision of and access to information and mechanisms and procedures for reporting grievances and misbehavior), (3) strengthen local participatory planning and decision-making, and (4) improve local governance effectiveness and efficiency through development of effective monitoring and evaluation systems at local and central levels [33]. Two of these four key dimensions of "good forest management and governance" have been taken up and pursued by many countries on national and international levels. Unfortunately, accountability and transparency, as well as governance effectiveness and efficiency, have not received equally broad recognition. Especially at the local level, which plays a crucial role in good governance, it has received comparatively little or no attention [33].

Obviously, forest conservation and management are entering a period of new challenges and greater uncertainty. Forest ecosystems supply services are crucial to human well-being, but the delivery of these services is diminishing globally. While agencies ranging from national governments to community organizations struggle to develop policies to effectively secure the conservation and sustainable management of forests resources, at the same time illegal and corrupt activities constitute a major underlying cause of forest decline worldwide.

2. Illegal logging

Most reasons for deforestation are due to market imperfections [20]. Market imperfections arise when property cannot be clearly defined, when property cannot be freely transferred, when the use of goods cannot exclude others from such use, and when private rights cannot be protected [21, 24]. Evidence convincingly shows that illegal and corrupt activities are a major underlying cause of forest decline [3, 5]. The main reason for this is that governments and private landowners cannot control these illegal operations. In addition, this lack of control may be deliberate, is often corrupt, or may be determined by the limitations of administrative capacity. One way or another, illegal use of forests is rampant [3, 8].

In principle, a distinction must be made between two types of illegal logging. On the one hand, wood may be stolen by the local population due to their poverty to satisfy their living requirements. This mainly comprises firewood. *Poverty-driven illegal logging* emerges where poor people have little other choice. The harvested quantities are typically small. The greater proportion of illegal logging, however, is carried out by companies dealing with *industrial* timber which occasionally have mafia-style structures and are part of organized crime. This form of illegal logging is closely tied to other criminal activities such as corruption, violence, and money laundering [7, 34].

Since so much deforestation is a result of illegal logging, we cannot rely on official production statistics to capture deforestation.

For example, with its extremely high percentage of illegal logging, Kosovo is ranked as the one of the worst offenders in the world with Indonesia [18]. The annual *illegal* fuelwood harvesting in Kosovo represents a market of up to 21.6 million euro and is done mostly by well organized groups of individuals, with market-oriented behavior, acting rather in state forests than in private forests. To put this into perspective, the domestic market demand for fuelwood in Kosovo is estimated at more than 1.5 hm^3, while the legal supply, including imports, is slightly higher than 0.3 hm^3. Hence, illegal logging for satisfying Kosovo population fuelwood needs is therefore widespread [35]. However, the need for auto-consumed firewood resulting from the high level of poverty does not represent the major issue compared to the well spread and costly commercial illegal logging crimes. These crimes are committed by the well-off. All institutions, from the MAFRD, KFA, police to the Ministry of Justice and politicians but also the forest products industry, are responsible for the present high level of illegal logging taking place in Kosovo [18].

Also, in Serbia, illegal logging is most intense exactly in the areas adjacent to the territories of Kosovo (which are formally under Serbian forest estates Vranje, Kursumlija, Leskovac, Raska, and Leposavic) to which Serbian authorities have limited access. According to Public Enterprise for Forest Management "Srbijašume," well-organized groups of Albanians from Kosovo, which usually have several tractors and whole professional equipment and mechanization, organize large illegal loggings in Serbian municipality Kursumlija. Given extremely high percentage of illegal logging in Kosovo, it is not surprising that they extended illegal logging across the Serbia-Kosovo administrative border, approximately 3–4 km into the Kursumlija's territory.

There has been, also, considerable concern about the reliability of official forest resource statistics in Central and Eastern Europe, not only from the socialist period, but also after the breakdown of socialism [17]. Latest forest resource data from these countries often do not consider forest degradation and illegal logging. For example, 2005 World Bank study estimated that unrecorded, illegal logging in some of Central and Eastern European countries, like Albania, exceeded the legal harvest by a factor of ten [36].

The extent of illegal forest-related activities is notoriously difficult to estimate. Since deforestation and forest degradation stemming from illegal practices are inherently hard to measure, attempts to quantify illegal actions are often "guesstimates" [14].

Also, in Serbia there is no established uniform system of monitoring of illegal activities in forestry, and there are no unique records, based on which all information about illegal activities in forestry could be monitored. Forests inventory (national and stand inventory) is not registering stumps of illegally logged trees. The main body responsible for the control of legal regulations implementation in the field of forestry, and therefore illegal logging, is the forest inspection service. Problems that the control services have are mainly related to the lack of equipment, vehicles, fuel, etc., *as well as lack of jurisdictions* [37]. Sometimes one forest inspector covers 1.000 km^2 but has no vehicle (nor public transport) in the very poorly developed areas [36]. In spite of all these shortcomings, public forests (that represent 50% of all forest territories) are relatively well protected. Obviously theft from privately owned forests constitutes a more complex problem in Serbia, since the real amount of logged wood in private forests is six times as high as the registered amount. Private owners are not an organized group, the average area of forest plots is very small, and owners tend to live at a distance from their property, and thus, there is no security service for private forests.

Obviously, more accurate data on illegal logging could be provided if a ground inventory is undertaken combined with using remote sensing technology. Such an approach could be much cheaper, and the series of data would be quite useful for monitoring the forest cover [18]. Although forest conservation policymakers in the developing countries still have limited financial, human, and political resources over the past two decades, publicly available, remotely sensed satellite data on deforestation and degradation have dramatically reduced evaluation costs [38]. These advances in conservation best practices and remotely sensed data availability have created significant new opportunities to enhance understanding of the effectiveness and efficiency of forest conservation policy. It is for these reasons that in-depth case studies, such as the ones that use remote sensing to map changes in land cover and forest patterns, in addition to using qualitative analyses, are crucial to understanding forest trends in Central and Eastern Europe.

3. Normalized difference vegetation index (NDVI) and local forest management

Remote sensing is the detection, recognition, or evaluation of objects by means of distant sensing or recording devices. Historically, digital remote sensing developed rapidly from aerial photography and photo interpretation. Information extracted visually from remote sensing is widely used in forestry [31].

According to Lu et al. [39], there are high, medium, and coarse spatial resolution images. High spatial resolution images such as IKONOS, QuickBird, and Worldview have recently become important data sources for change detection analysis at a local scale [40]. Medium spatial resolution images, especially Landsat images due to their long history of data availability and suitable spectral and spatial resolutions, have become a common data source for regional change detection [41, 42]. At a continental or global scale, coarse spatial resolution data such as AVHRR, MODIS, and SPOT VGT (VEGETATION) may be used [10, 43–46] but present challenges in developing suitable techniques to extract changed features from coarse spatial resolution data [39].

In our study, we are going to use normalized difference vegetation index (NDVI). In our previous research, we compared NDVI and Corine land cover (CLC), and NDVI proved to be much more precise than CLC [47]. Since both NDVI and CLC used the same Landsat satellite images and the same (NDVI) methodology, these major differences in the data obtained were due to the different spatial resolution of NDVI and CLC. Whereas CLC does not go below the range of 4–5 ha, NDVI easily deals with minimum space units of 25 m^2. This proved to be decisive for Serbia, where privately owned forest parcels, which account for half of the total forest area of the country, usually cover much smaller areas (the average private holding is 0.3 ha; [48]). In addition, apart from the obvious CLC imprecision for studies at the local level, CLC data are not available for every year. In short, CLC proved not to be very suitable for local forest management in Serbia.

Normalized difference vegetation index (NDVI) [49] is one of the most widely used vegetation indices (VIs), which focuses on the vegetation cover and its status [50]. NDVI is actually a simple graphic indicator that can be used to analyze remote sensing measurements, whether the target observed contains live green vegetation or not [51]. NDVI was one of the most successful of many attempts to simply and quickly identify vegetated areas and their »condition«, and it remains the best-known and most-used index for detecting live green plant canopies in multispectral remote sensing data [52, 53]. NDVI also has the advantage of allowing comparisons between images acquired at different times [54]. It belongs to the VIs related to vegetation cover and its status. VIs have a direct correlation with leaf chlorophyll content and leaf area index (LAI) and vary in relation to vegetation cycle and phenology [55, 56]. They are also sensitive to other external factors, such as the contribution of the soil and background optical behavior where the vegetation does not completely cover the ground, the geometry of view due to sensor angle of acquisition and to Sun position, atmospheric effects, and other factors [31, 50, 52, 57]. NDVI, like all VIs, relates the spectral absorption of chlorophyll in the red with a reflection phenomenon in the near infrared, influenced by the leaf structure type [58].

Given the importance and complexity of forest preservation and sustainable forest management [1, 59, 60], an attempt was made to evaluate the possible use of a normalized difference vegetation index (NDVI) [49] in local forest management and preventing illegal logging and corruption in southern Serbian municipality Kursumlija, adjacent to the territories of Kosovo.

In Serbia around 30% of land is forested (of which 48% is state-owned forests and 52% privately owned). Forest management (of both privately owned and state-owned forests) is also very poor [61].

In the study, it was not possible to make a reliable long-term comparative analysis between NDVI and official forest inventories because national forest inventories have very rarely been carried out in Serbia. Such inventories were carried out at roughly 20-year intervals: in 1961, 1979, and 2003–2006. Since 2007 until 2011, official estimates of forest areas have been made annually. The study was carried out for the municipality of Kursumlija (where illegal logging and deforestation are extremely pronounced) and also for municipality of Topola (where it is not) (**Figure 1**). The municipality of Topola is located in central Serbia, and the municipality of Kursumlija lies in southern Serbia and is adjacent to the territories of Kosovo.

Figure 1. Location of the municipalities of Kursumlija and Topola (Serbia).

Data obtained using NDVI for spring/summer 2006 and 2011 were analyzed and compared to official forest area estimates for 2006 and 2011 created at the end of the same year.

NDVI data for both municipalities are based on Landsat-5 Thematic Mapper (TM) satellite images for 2006 and 2011, which were created during spring/summer (August), with minimum clouds (10–20%; [62]). In order to remove atmospheric effects from the NDVI final results, Idrisi software was used for data preprocessing. For calculating NDVI, satellite (Landsat) imagery (which has a resolution of approximately 30 m) and pan-sharpening images (with 15 m resolution) were used to obtain more precise results.

NDVI was used, and necessary corrections/transformations were applied for visible red in constellation with the infrared spectrum of satellite images using the following procedure: GIS analysis/mathematical operation/image calculator, and then the equation $NDVI = (NIR - RED)/(NIR + RED)$, in which NIR is the near-infrared channel and RED is the red channel from the visible part of the spectrum [63, 64]. Basic tasks included analysis and photo interpretation of elements, occurrences, and processes detected on images using specialized GIS software (Idrisi 15 Andes) for processing remotely sensed images through application of NDVI.

Shadows can cause NDVI values to be lower than their actual values. In this sense, "empirical topographic corrections have proven only marginally successful" [31]. Because shadow areas were less than 5% in the municipality of Kursumlija and less than 3% in the municipality of Topola, no topographic corrections were made.

Characteristic NDVI signatures are as follows: NDVI of dense vegetation canopy tends to have positive values (0.3–0.8); clouds and snowfields are characterized by negative values of this index; bodies of water (e.g., oceans, seas, lakes, and rivers) have rather low reflectance in both spectral bands (at least away from shores), thus resulting in very low positive or even slightly negative NDVI values; soils generally exhibit a near-infrared spectral reflectance somewhat larger than the red and thus also tend to generate rather small positive NDVI values (0.1–0.2); very low values of NDVI (0.1 and below) correspond to barren areas of rock, sand, or snow; moderate values represent shrub and grassland (0.2–0.3); and high values (0.6–0.8) indicate temperate and tropical rainforests [65, 66]. Negative values of NDVI ranging from 0 to –0.3 are displayed in orange shades (**Figure 2**). These low negative values are detected in arable agricultural land (without vegetation) and are shown in shades of light orange. On the other hand, vegetation areas are presented with values between 0 and 1. Grassy areas, meadows, and pastures have values that range from zero up to 0.13 (shades of yellow, due to more intense reflectance of infrared radiation). Shrub vegetation has an NDVI value of 0.25 because reflectance of infrared rays decreases (light green). All forest vegetation (shades of dark green), with maximal positive NDVI values of 0.85 (due to minimal reflectance of infrared rays), is easily observed. Coniferous forest has an NDVI value above 0.5, mixed forest between 0.35 and 0.5, and broad-leaved forest between 0.3 and 0.4 [57, 67].

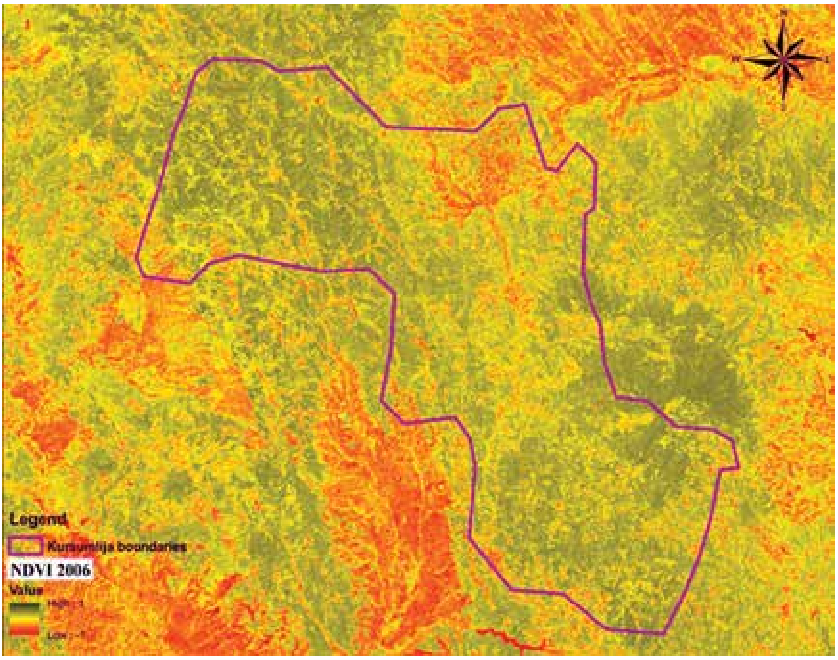

Figure 2. Vegetation cover in the municipality of Kursumlija for 2006 obtained from NDVI.

After image processing, it was determined (**Table 2**) that forest areas encompass 529.83 km^2 or 55.7% of the total area of the municipality of Kursumlija, much higher than the average 30% for Serbia, and 50.73 km^2 or 14.2% of the total area of the municipality of Topola, which is approximately half of the Serbian average.

When compared with official forest area estimates [68], the NDVI results show a mere 0.12 km^2 (0.23%) difference for the municipality of Topola's forest area and a -27.01 km^2 (-4.47%) difference for the municipality of Kursumlija (**Table 1**). Not only do these results completely fit within the ±5% margin of error allowed for this method [44, 69], but they also allow room for further analysis and investigation.

Because the NDVI aerial photos were taken during spring/summer, whereas official forest area estimates are made at the end of the year, NDVI values would be expected to be higher, not lower—at least for the municipality of Kursumlija (known for its illegal logging). Moreover, because additional NDVI forest area estimates were made for 2011 (**Table 2**), it seems that even for 2006, this study's NDVI results better fit the forest area trajectory of Kursumlija for the 2006–2011 period than do the official statistics (the official forest inventory for 2006 is 604.41 km^2 [70] and NDVI results 577.4 km^2; and the official forest inventory for 2011 is 544.3 km^2 [68] and NDVI results 529.8 km^2).

The main reason that the (slightly smaller) NDVI results possibly better fit the forest area trajectory of Kursumlija than the official inventory (**Tables 3** and **4**) is that this municipality is known for illegal logging. According to the state-owned forest management company Srbijašume, Kursumlija experienced a 10% loss in forest area in the last few years alone [61, 68, 71, 72].

Municipality	Municipality total area for 2006 (km^2)	Forest area		Forest area	
		Official statistics for 2006 (km^2)	Calculated on the basis of NDVI for 2006 (km^2)	NDVI—official statistics difference (km^2)	NDVI—official statistics difference (%)
Topola	356	52.00	52.12	−0.12	0.23
Kursumlija	950	604.41	577.40	−27.01	−4.47

Table 1. Forest areas according to official statistics and calculated on the basis of NDVI for the year 2006.

Municipality	Municipality total area for 2011 (km^2)	Forest area		Forest area	
		Official statistics for 2011	Calculated on the basis of NDVI for 2011	NDVI—official statistics difference (km^2)	NDVI—official statistics difference (%)
Topola	357	52.0494	50.73	−1.3194	−2.53
Kursumlija	952	544.2856	529.83	−14.4556	−2.66

Table 2. Forest areas according to official statistics and calculated on the basis of NDVI for the year 2011.

Municipality	Official statistics for 2006 (km²)	Official statistics for 2011 (km²)	2006–2011 difference (%)
Kursumlija	604.41	544.2856	−9.95
Topola	52.00	52.05	+0.96
Serbia	19845.13	19623.35	−1.11

Table 3. Kursumlija and Topola—forest areas according to official statistics.

Municipality	Calculated on the basis of NDVI for 2006 (km²)	Calculated on the basis of NDVI for 2011 (km²)	2006–2011 difference (%)
Kursumlija	577.40	529.83	−8.24
Topola	52.12	50.73	−2.67

Table 4. Kursumlija and Topola—forest areas calculated on the basis of NDVI.

4. Kursumlija: challenges of illegal logging and rapid deforestation

Since, according to official statistics, Kursumlija experienced a 10% loss in forest area in the 2006–2011 period, which is 60 km^2 loss of forest cover (or 48 km^2, calculated on the basis of NDVI), and this is very clear case of alarmingly rapid process of *deforestation*, in very sharp contrast with very modest rates of deforestation in Serbia and even slight process of reforestation in municipality of Topola. This extremely quick process of *deforestation* in Kursumlija is the obvious result of illegal logging. In order to solve this problem, Public Enterprise for Forest Management "Srbijasume" regularly informed all levels of the state administration, Ministry of Interior Affairs, Army, representatives of KFOR and UNMIK, which is also well documented in various World Bank and REC studies [36, 37, 72].

Again, it seems that NDVI results possibly better fit the forest area trajectory of Kursumlija than official inventory (for 2006 official forest inventory shows 27 km^2 more forest areas, for 2011, 14 km^2 more). This is probably due to the notoriously imprecise official inventory of illegal loggings in private forests in Serbia. The real amount of logged wood in private forests is six times as high as the registered amount. This is the reason why NDVI gives more precise results than official inventory [6].

It is important to underline that the number of shortcomings that are usually addressed/related to the satellite imagery use, regarding its possibilities to truly capture illegal logging, proved to be completely irrelevant in the case of Kursumlija. For example, Khai et al. [73] state that the illegal nature of timber harvests makes it difficult to locate and quantify overall amounts of timber harvested, largely because illegal logging frequently does not produce large canopy gaps visible on satellite images. Khai et al. [73] also stress that it is very important to clearly distinguish illegal cutting from legal cutting based on hammer marks and size/height of stumps in the field. Although it can be time-consuming to estimate illegal logging from field surveys of stumps, recent forest inventories of many countries include stump measurements for carbon stock and biodiversity evaluation. Thus, such stump measurements in regular forest inventories can be readily used for estimating the extent of illegal logging if there is a clear difference between legally and illegally cut stumps.

Also, Lawson and Larry MacFaul [74] emphasize that, although comparing satellite imagery with official concession maps and harvesting plans could be expected to capture illegal logging more completely and precisely than existing indicators, they are still far from perfect …. First of all, because satellite images cannot easily detect whether a company harvests more trees than permitted within the area in which it is licensed to cut in a given year, the method also fails to capture illegalities regarding concession allocation.

Nevertheless, none of the above-mentioned possible shortcomings of satellite imagery proved to be relevant in the case of Kursumlija: First, it is very easy to locate and quantify overall amounts of timber harvested in the case of Kursumlija municipality, because illegal logging produces large canopy gaps that go/extend from the border of Kosovo to approximately 3–4 km into the Kursumlija's territory; second, from the field survey (hammer marks and size/height of stumps in the field), it is obvious that it is the clear case of illegal cutting …; and third, since illegal logging in Kursumlija is organized by groups of individuals, with market-oriented behavior [35], which is part of organized crime and closely tied to other criminal activities such as corruption, violence, and money laundering [34], it is, of course, not any sort of concession allocation issue.

Obviously, governments often cannot efficiently control these illegal operations. As Contreras-Hermosilla [3] points out: "This lack of control can be either deliberate, often corrupt, or determined by the limitations of administrative capacity. One way or the other, illegal use of forests is rampant in most forested countries. By their very nature, the true extent of illegal operations in the forestry sector cannot be known with precision, but evidence suggests that such activities are important and that they constitute an important underlying cause of forest decline."

Since this research strongly implies that illegal logging in Kursumlija is not properly covered by current official forest area estimates, further NDVI research on the extent of illegal logging in southern Serbian municipalities is of the utmost importance.

In short, because the municipality of Kursumlija has a large territory (952 km^2), with more than 544 km^2 (or 55.7%) of its total area covered by forests, and because NDVI can be performed very quickly, it is obvious that NDVI can provide local forest managers in Kursumlija with much essential annual information about the forest inventory [75–80]. This is of crucial importance for preventing illegal logging, which is very prevalent in this southern Serbian municipality [61, 71, 72].

Finally, we should further investigate two important issues: (a) how the remote sensing can be used as a management tool for forest management in Serbia? and (b) how it will prevent illegal logging or help the forest managers to fight this menace in Serbia?

As Potapov et al. [81] point out, information derived from satellite imagery is not equivalent to inventory data collected by forest managers. Optical remote sensing data are suitable for mapping land cover (tree canopy cover, dominant tree species composition), while national forest inventory data focus on land-use (e.g., forest land). This means that while tree canopy cover change can be readily observed with remote sensing data, it is not directly comparable to harvested timber volumes reported by the national forest statistics [81]. Remote sensing (RS) data can provide an alternative data source to quantify forest cover and change independent of official governmental data sources.

This is extremely important, since there has been considerable concern about the reliability of (very rarely performed) official forest resource inventories/statistics in the Central and Eastern Europe [17] and especially in Serbia [47, 80]. Also, deforestation and forest degradation stemming from legal practices are inherently hard to measure, and attempts to quantify illegal actions are often "guesstimates" [14]. Information system on illegal activities in forestry in Serbia has not been established yet, and within the sector, it operates several (very inefficient) internal systems for collecting data on illegal activities. Since municipality of Kursumlija experienced a 10% loss in forest area in the 2006–2011 period only (and official forest resource statistics seem to be completely unreliable since 2011), information about this very quick process of deforestation in Kursumlija (on the yearly basis, at least) is of the utmost importance.

Also, concerning (generally still rather inefficient) forest management in Serbia, state-owned forests (48% of the nation's forest resources) are managed mainly by the state forest enterprises, according to the management plans prepared on the 10-year basis/cycle. The emphasis is narrowly focused on timber production. The process of forest certification, extremely important for combating illegal logging, has only recently begun (although state forests have adopted the Forest Stewardship Council certification scheme, only 200,000 ha of state forests have been certified by now).

Forest management of private forests is even in a much worst condition. Private forests represent/constitute 52% of the nation's forest resources and are characterized by very small plots (average size: 0.3 ha). Interestingly enough, the (previous) forest census completed in 1979 covered only state forests and national parks and did not include private forests at all (new inventory completed in 2007, finally included private forests also). Although (at list in theory) private forests in Serbia should be managed by the private landowners according to the general plan and forest management program (i.e., financed, prepared, developed, and delivered to the private forest owners by the *municipal forest enterprise* every 10 years), since municipal forest enterprises do not financially assist the management program of private forest owners, actually only less than 10% of all private forests have any sort of management programs. Hence, no surprise that private forests suffer not only from extremely high degree of fragmentation but also from very inefficient management. Also, although no official statistics on illegal logging exist, it is clear that timber theft is greatest in private forests. The estimated value of illegally harvested wood from private forests was US$ 2.4 million (in 2003). According to UN Economic Commission for Europe (UNECE), illegal logging on public lands is 1–5% of the total harvest, while illegal practices in private forests are greater than 50% in Serbia [82].

As Verstraete and Pinty [83] point out, the use of spectral indexes is most recommended: (1) when the desired information is required as soon as possible (like in the case of forceful illegal logging in Kusumlija) or (2) when the conclusions of the study do not depend critically on the accuracy of the information (e.g., detection of significant changes—quick deforestation in Kursumlija).

With recent progress in aerial photography, satellite imagery, and remote sensing, the possibilities of rapid analysis increase [84], which are the essential prerequisite for combating illegal logging. Also, the objectivity of these methods can significantly help in avoiding corruption in forest management because corruption is one of the main weaknesses of Serbia's economy.

Also, remote sensing can be extremely useful, because illegal logging in Serbia is by far most intense exactly in the Serbian municipalities adjacent to the territories of Kosovo, in the extremely sensitive Serbia-Kosovo border area (formally under Serbian forest estates Vranje, Kursumlija, Leskovac, Raska, and Leposavic) to which Serbian authorities have limited access … and where (according to Public Enterprise for Forest Management "Srbijašume"), well-organized groups of Albanians from Kosovo, organize large illegal loggings. It is illegal logging organized by groups of individuals, with market-oriented behavior [35] as part of organized crime and closely tied to other criminal activities such as corruption, violence, and money laundering [34] that simply extended illegal logging from Kosovo (ranked as the one of the worst illegal logging offenders in the world with Indonesia [18]) across the Serbia-Kosovo administrative border, approximately 3–4 km into the Kursumlija's territory.

It is exactly objectivity of remote sensing that can be of the greatest help in resolving extremely quick and forceful process of illegal logging in this very sensitive southern Serbian area. Obviously, remote sensing (RS) data can be very efficient tool for forest management in Serbia and help the forest managers to fight illegal logging, especially in these most extreme cases of galloping deforestation and illegal logging (like in Kursumlija), by providing an reliable, alternative data source to quantify forest cover and change (independent of official governmental data sources). As Chakravarty et al. point out, it is not possible to properly manage a forest ecosystem without first understanding it [22] or to prevent illegal logging without frequently updated, objective information about deforestation and forest degradation.

5. Conclusion

Despite certain shortcomings [31, 52, 73, 74], classification and area estimation of various land-cover types based on remote sensing have obviously advanced to a point where it surpasses old wood inventory techniques, especially in the case of Serbia.

Specifically:

- It is relatively cheap and quick, and it can provide forest managers with essential information;
- It is easy to implement, which is of crucial importance for Serbia, where national forest inventories have been carried out very rarely. The last three national forest inventories were carried out at roughly twenty-year intervals; however, since the last national forest inventory (2003–2006), necessary updates have been made every year until 2011, but only at the municipality level.
- The objectivity of these methods can significantly help in avoiding corruption in forest management because corruption is one of the main weaknesses of Serbia's economy.

Through this analysis of NDVI results for the municipalities of Kursumlija and Topola, it is evident that NDVI, especially in southern Serbian municipalities with prevalent illegal logging (like Kursumlija), can provide local forest managers with much annual information about forest areas. This is of crucial importance for monitoring (and consequently preventing) illegal logging.

NDVI is also very promising for countries like Serbia, which very rarely carry out national forest inventories. It is easily implemented, and it has objectivity that can greatly help avoid corruption and illegal logging in forest management.

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References

[1] Pagiola S, Landell-Mills N, Bishop J. Selling forest environmental services: market-based mechanisms for forest conservation and development. London: Earthscan; 2002. p. 299.

[2] FAO. State of the World's Forests—1997 [Internet]. 2003. Available from: http://www.fao.org/docrep/W4345E/w4345e00.htm#Contents%5Cnhttp://www.fao.org/DOCREP/003/X6953E/X6953E00.HTM%5Cnhttp://www.fao.org/forestry/site/sofo/en/%5Cnhttp://www.fao.org/DOCREP/003/Y0900E/Y0900E00.HTM

[3] Contreras-Hermosilla A. The underlying causes of forest decline [Internet]. Vol. 62, Centre for Forestry Research: Occasional Paper No. 30. 2000. Available from: http://www.cifor.org/online-library/browse/view-publication/publication/626.html

[4] Carlos M, Souza J, Siqueira JV, Sales MH, Fonseca AV, Ribeiro JG, Numata I, et al. Ten-year landsat classification of deforestation and forest degradation in the Brazilian Amazon. Remote Sens. 2013;5:5493–513.

[5] Brack D. Illegal logging and the illegal trade in forest and timber products. Int For Rev [Internet]. 2003;5(3):195–8. Available from: http://sfx.library.wur.nl:9003/sfx_local?sid=SP:C

ABI&id=pmid:&id=&issn=1465-5489&isbn=&volume=5&issue=3&spage=195&pages=195-198&date=2003&title=International Forestry Review&atitle=Illegal logging and the illegal trade in forest and timber products.&aul

[6] Markus-Johansson M, Mesquita B, Nemeth A, Dimovski M, Monnier C, Kiss-Parciu P. Illegal logging in South Eastern Europe. Szentendre:The Regional Environmental Center for Central and Eastern Europe; 2010.

[7] Blaser J, Contreras A, Oksanen T, Puustjärvi E, Schmithüsen F. Forest Law Enforcement and Governance (FLEG) in Eastern Europe and Northern Asia (ENA). In: Forest Law Enforcement and Governance Process for Europe and Northern Asia (ENA-FLEG); Nov 22-25, 2005; St Petersburg; 2005.

[8] Amacher G, Ollikainen M, Koskela E. Economics of forest resources [Internet]. 2010. pp. 1–423. Available from: http://erae.oxfordjournals.org/cgi/doi/10.1093/erae/jbq028

[9] Burgess R, Hansen M, Olken BA, Potapov P, Sieber S. The political economy of deforestation in the tropics. Q J Econ. 2012;127(4):1707–54.

[10] Hansen MC, DeFries RS. Detecting long-term global forest change using continuous fields of tree-cover maps from 8-km advanced very high resolution radiometer (AVHRR) data for the years 1982-99. Ecosystems. 2004;7(7):695–716.

[11] Stern N. The economics of climate change—the Stern review. Cambridge: Cambridge University Press; 2007.

[12] IPCC. Mitigation of climate change: contribution of working group III to the fourth assessment report of the Intergovernmental Panel on Climate Change. 2007. p. 851.

[13] Kindermann G, Obersteiner M, Sohngen B, Sathaye J, Andrasko K, Rametsteiner E, et al. Global cost estimates of reducing carbon emissions through avoided deforestation. Proc Natl Acad Sci USA [Internet]. 2008;105(30):10302–7. Available from: http://www.ncbi.nlm.nih.gov/pubmed/18650377

[14] FAO. Best practices for improving law compliance in the forestry sector [Internet]. vol. 145. 2005. p. 110. Available from: http://books.google.com/books?hl=en&lr=&id=GAUWZzBMCDEC&oi=fnd&pg=PP9&dq=Best+practices+for+improving+law+compliance+in+the+forestry+sector&ots=YSjWwid6w8&sig=7iYeYRW_ncPTEJGj16dYaycFESs

[15] European Forest Institute. Impact of the reduction of illegal logging in European Russia on the EU and European Russia Forest Sector Trade. Joensuu: European Forest Institute; 2005.

[16] Lee JH, Sigmund K, Dieckmann U, Iwasa Y. Games of corruption: how to suppress illegal logging. J Theor Biol. 2015;367:1–13.

[17] Taff GN, Müller D, Kuemmerle T, Ozdeneral E, Walsh SJ. Reforestation in Central and Eastern Europe after the breakdown of socialism. In: Nagendra H, Southworth J, editors. Reforesting Landscapes. New York: Springer; 2010. p. 121–47.

[18] Harou P, Hajredini E. Ilegal logging in Kosovo. Washington: USAID; 2009.

[19] Hanley N, Shogren J, White B. Introduction to environmental economics. Oxford: Oxford University Press; 2001. pp. 20–24.

[20] Jovanovic M. Kuznets curve and urban transport the scope of I+M programs. Glas Srp Geogr Drus [Internet]. 2012;92(4):127–42. Available from: http://www.doiserbia.nb.rs/Article.aspx?ID=0350-35931204127J

[21] Tietenberg T, Lewis L. Environmental and natural resource economics. Upper Saddle River, N.J: Pearson; 2012.

[22] Chakravarty S, Ghosh SK, Suresh SP, Dey AN, Shukla G. Deforestation: causes, effects and control strategies. In: Okia CA, editor. Global perspectives on sustainable forest management. Rijeka: InTech; 2012. pp. 3–28.

[23] Gibson C, McKean M, Ostrom E. People and forests. Cambridge, Massachusetts: The MIT Press; 2000.

[24] McKean M, Ostrom E. Common property: what is it, what is it good for, and what makes it work? People and forests. Cambridge: The MIT Press; 2000.

[25] Dale VH. Management of forests as ecosystems: a success story or a challenge ahead? Success, limitations, and frontiers in ecosystem science. New York: Springer-Verlag; 2008.

[26] Monteiro A, Souza C. Remote Monitoring for Forest Management in the Brazilian Amazon. In: Garsia J, Diez-Casero J, editors. Sustainable forest management—current research. Rijeka: InTech; 2012. p. 67–86. DOI: 10.5772/30126.

[27] Newton AC. Forest ecology and conservation—a handbook of techniques. Oxford: Oxford University Press; 2007.

[28] Kuvan Y, Erol YS, Yildirim HT. Forest managers' perceptions of the foremost forestry issues and functions in Turkey. Polish J Environ Stud. 2011;20(2):393–403.

[29] Paivinen R, Brusselan JV, Schuck A. The growing stock of European forests using remote sensing and forest inventory data. Forestry. 2009;82(5):479–490.

[30] Costantini ML, Zaccarelli N, Mandrone S, Rossi D, Calizza E, Rossi L. NDVI spatial pattern and the potential fragility of mixed forested areas in volcanic lake watersheds. For Ecol Manage. 2012;285:133–41.

[31] Franklin SE. Remote sensing for sustainable forest management [Internet]. New York. 2001. p. 407. Available from: http://www.amazon.ca/exec/obidos/redirect?tag=citeulike09-20&path=ASIN/1566703948

[32] Edmunds D, Wollenberg E. Local forest management—the impacts of devolution policies. London: Earthscan; 2004.

[33] Rametsteiner E. Improving governance for sustainable forest management at local and regional levels. In: Enabling sustainable forest management. New York: Unite Nations; 2007.

[34] WWF. Illegal logging & the EU. An analysis of the EU export & import market of illegal wood and related products. Brussels: WWF European Policy; 2008.

[35] Bouriaud L, Nichiforel L, Nunes L, Pereira H, Bajraktari A. A property rights-based analysis of the illegal logging for fuelwood in Kosovo. Biomass Bioenergy. 2014;67:425–34.

[36] World Bank. Ensuring sustainability of forests and livelihoods through improved governance and control of illegal logging for economies in transition. Helsinki: Savcor indufor oy; 2005.

[37] The Regional Environmental Center for Central and Eastern Europe. Illegal logging activities in the Republic of Serbia-B. DIAGNOSTIC AUDIT. Belgrade: The Regional Environmental Center for Central and Eastern Europe; 2009.

[38] Blackman A. Evaluating forest conservation policies in developing countries using remote sensing data: an introduction and practical guide. For Policy Econ. 2013;34:1–16.

[39] Lu D, Li G, Moran E. Current situation and needs of change detection techniques. Int J Image Data Fusion [Internet]. 2014;5(1):13–38. Available from: http://www.tandfonline.com/doi/abs/10.1080/19479832.2013.868372

[40] Lu D, Hetrick S, Moran E, Li G. Detection of urban expansion in an urban-rural landscape with multitemporal QuickBird images. J Appl Remote Sens. 2010;4:41880(10).

[41] Xian G, Homer C, Fry J. Updating the 2001 National Land Cover Database land cover classification to 2006 by using Landsat imagery change detection methods. Remote Sens Environ. 2009;113(6):1133–47.

[42] Hansen MC, Loveland TR. A review of large area monitoring of land cover change using Landsat data. Remote Sens Environ. 2012;122:66–74.

[43] Bergen KM, Brown DG, Rutherford JF, Gustafson EJ. Change detection with heterogeneous data using ecoregional stratification, statistical summaries and a land allocation algorithm. Remote Sens Environ. 2005;97(4):434–46.

[44] Lunetta RS, Knight JF, Ediriwickrema J, Lyon JG, Worthy LD. Land-cover change detection using multi-temporal MODIS NDVI data. Remote Sens Environ. 2006;105(2):142–54.

[45] Hansen MC, Shimabukuro YE, Potapov P, Pittman K. Comparing annual MODIS and PRODES forest cover change data for advancing monitoring of Brazilian forest cover. Remote Sens Environ. 2008;112(10):3784–93.

[46] Bontemps S, Langner A, Defourny P. Monitoring forest changes in Borneo on a yearly basis by an object-based change detection algorithm using SPOT-VEGETATION time series. Int J Remote Sens. 2012;33(15):4673–99.

[47] Jovanovic M, Milanovic M, Zorn M. The use of NDVI and CORINE Land Cover databases for forest management in Serbia. Acta Geogr Slov. DOI:10.3986/AGS.818.

[48] Glavonjic B, Jovic D, Vasiljevic A, Kankaras R. Forest and forest products country profile: Serbia and Montenegro. Geneva: United Nations; 2005.

[49] Weier J, Herring D. Measuring vegetation (NDVI & EVI) [Internet]. Feature Articles. 2000. pp. 1–4. Available from: http://earthobservatory.nasa.gov/Features/MeasuringVegetation/measuring_vegetation_1.php

[50] Jensen JR. Remote sensing of the environment: an earth resource perspective. Upper Saddle River, NJ: Prentice Hall,; 2007.

[51] Chen CH. Image processing for remote sensing. Boca Raton: CRC Press; 2008.

[52] Campbell J, Wynne R. Introduction to remote sensing. New York: Guilford Press; 2011.

[53] Win RN, Suzuki R, Takeda S. Remote sensing analysis of forest damage by selection logging in the Kabaung Reserved Forest, Bago Mountains, Myanmar. J For Res. 2012;17(2):121–8.

[54] Lillesand TM, Kiefer RW, Chipman JW. Remote sensing and image interpretation [Internet]. Vol. 3rd, Nev York Chichester Brisbane Toronto 6IS s. 2004. P. 756. Available from: http://www.osti.gov/energycitations/product.biblio.jsp?osti_id=6028047

[55] Montandon LM, Small EE. The impact of soil reflectance on the quantification of the green vegetation fraction from NDVI. Remote Sens Environ. 2008;112(4):1835–45.

[56] Vohland M, Stoffels J, Hau C, Schüler G. Remote sensing techniques for forest parameter assessment: multispectral classification and linear spectral mixture analysis. Silva Fenn. 2007;41(3):441–56.

[57] De Jong SM, Van der Meer FD. Remote sensing image analysis: including the spatial domain. Dordrecht: Springer; 2005.

[58] Wang Q, Tenhunen JD. Vegetation mapping with multitemporal NDVI in North Eastern China Transect (NECT). Int J Appl Earth Obs Geoinf. 2004;6(1):17–31.

[59] Lee B, Kim SY, Chung J, Park PS. Estimation of fire severity by use of Landsat TM images and its relevance to vegetation and topography in the 2000 Samcheok forest fire. J For Res. 2008;13(4):197–204.

[60] Ojea E, Ruiz-Benito P, Markandya A, Zavala MA. Wood provisioning in Mediterranean forests: a bottom-up spatial valuation approach. For Policy Econ. 2012;20:78–88.

[61] FAO. Forestry development strategy for Serbia. Belgrade: Ministry of Agriculture Forestry and Water Management Directorate of Forests; 2006.

[62] Chávez PSJ. Image-based atmospheric corrections—revisited and improved. Photogramm Eng Remote Sensing. 1996;62(9):1025–36.

[63] Hájek F. Process-based approach to automated classification of forest structures using medium format digital aerial photos and ancillary GIS information. Eur J For Res. 2008;127(2):115–24.

[64] Johnson LF, Trout TJ. Satellite NDVI assisted monitoring of vegetable crop evapotranspiration in california's san Joaquin Valley. Remote Sens. 2012;4(2):439–55.

[65] Finelli M, Gelli G, Poggi G. Multispectral-image coding by spectral classification. In: International conference on image processing 2. New York: The Institute of Electrical and Electronics Engineers; 1996.

[66] Schmitt U, Ruppert G. Forest classification of multispectral mosaicking satellite images. Arch Photogramm Remote Sens. 1996;31.

[67] Hord RM. Digital image processing of remotely sensed data [Internet]. Enschede; 1982. Available from: https://books.google.co.in/books?id=e-tRAAAAMAAJ

[68] Statistical Office of the Republic of Serbia. Municipalities and regions of the Republic of Serbia 2012. Belgrade: Statistical Office of the Republic of Serbia; 2013.

[69] Eastman R. IDRISI32 Release 2—Tutorial. Worcester: Clark Labs; 2001.

[70] Statistical Office of the Republic of Serbia. Municipalities and regions of the Republic of Serbia 2007. Belgrade: Statistical Office of the Republic of Serbia; 2008.

[71] Anfodillo T, Carrer M, Giacoma E, Lamedica S, Pettenella D. Challenges and priority for adapting the management of Carpathians forests to new environmental and socio-economic conditions [Internet]. 2008. p. 202. Available from: http://www.carpathianconvention.org/NR/rdonlyres/EB908676-7393-43A8-BCB1-9A7CEF9225A2/0/Report2_ForestPolicy.pdf

[72] Regional Environmental Center. Illegal logging activities in the Republic of Serbia—A. Fact-finding study. Belgrade: The Regional Environmental Center for Central and Eastern Europe; 2009.

[73] Khai TC, Mizoue N, Kajisa T, Ota T, Yoshida S. Stand structure, composition and illegal logging in selectively logged production forests of Myanmar: comparison of two compartments subject to different cutting frequency. Glob Ecol Conserv [Internet]. 2016;7:132–40. doi:10.1016/j.gecco.2016.06.001

[74] Lawson S, MacFaul L. Illegal logging and related trade [Internet]. London: Chatham House; 2010. pp. 1–154. Available from: papers2://publication/uuid/C7E061BB-61C3-4A58-A30D-DF4FFDA3CBD2

[75] Bellone T, Boccardo P, Perez F. Investigation of vegetation dynamics using long-term normalized difference vegetation index time-series. Am J Environ Sci [Internet]. 2009;5(4):460–6. Available from: http://www.scopus.com/inward/record.url?eid=2-s2.0-70349452068&partnerID=40&md5=b04213c8bafee51977e388ed374c18b8

[76] Fensholt R, Rasmussen K, Nielsen TT, Mbow C. Evaluation of earth observation based long term vegetation trends—intercomparing NDVI time series trend analysis consistency of Sahel from AVHRR GIMMS, Terra MODIS and SPOT VGT data. Remote Sens Environ. 2009;113(9):1886–98.

[77] Martínez B, Gilabert MA. Vegetation dynamics from NDVI time series analysis using the wavelet transform. Remote Sens Environ. 2009;9(113):1823–42.

[78] Corral-Rivas JJ, Wehenkel C, Castellanos-Bocaz HA, Vargas-Larreta B, Diéguez-Aranda U. A permutation test of spatial randomness: application to nearest neighbour indices in forest stands. J For Res. 2010;15(4):218–25.

[79] Alessandrini A, Vessella F, Di Filippo A, Salis A, Santi L, Schirone B, et al. Combined dendroecological and normalized difference vegetation index analysis to detect regions of provenance in forest species. Scand J For Res [Internet]. 2010;25:121–5. Available from: http://content.ebscohost.com/ContentServer.asp?T=P&P=AN&K=52237277&S=R&D=a9h&EbscoContent=dGJyMMvl7ESep7I4zOX0OLCmr0meprJSrqa4TLWWxWXS&ContentCustomer=dGJyMPGss0yyp7RPuePfgeyx44Dt6fIA%5Cnhttp://search.ebscohost.com/login.aspx?direct=true&db=a9h&AN=52237277&

[80] Jovanović M, Milanović M. Normalized Difference Vegetation Index (NDVI) as the basis for local forest management. Example of the Municipality of Topola. Polish J Environ Stud. 2015;24(2):529–35.

[81] Potapov P V., Turubanova SA, Tyukavina A, Krylov AM, McCarty JL, Radeloff VC, et al. Eastern Europe's forest cover dynamics from 1985 to 2012 quantified from the full Landsat archive. Remote Sens Environ. 2015;159:28–43.

[82] UNECE. Illegal logging in Serbia. Geneva: United Nations; 2004.

[83] Verstraete MM, Pinty B. Designing optimal spectral indexes for remote sensing applications. IEEE Trans Geosci Remote Sens. 1996;34(5):1254–65.

[84] Sundström A. Understanding illegality and corruption in forest governance. J Environ Manage. 2016;181:779–90.

From Paper to Carbon Money: Financing Forest Conservation and Offset

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Additional information is available at the end of the chapter

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Abstract

As a result of the 21st Conference of the Parties (CoP-21) in 2015, the Paris Agreement formally recognised the importance of finance and forests to tackle climate change. However, Article 9 of the convention calls for the leadership of developed countries in mobilising climate finance, while encouraging other parties to provide financial support voluntarily. This is rather an unstable mechanism, since it is strongly affected by political and economic hardships. Forest finance could be established instead that, just like capital markets, might allow for countries to choose between interest-bearing bonds from forest conservation (natural forests) and/or offset (forest plantations). Bonds demand comes out of carbon savings from forest conservation or offsetting forests, whereas bonds supply arises from investments giving off carbon emissions that must be avoided through forest conservation or offset through forest plantations. A Loanable-Forest Funds (LFF) model is developed which shows that forest conservation scenarios require lower rates of interest on forest bonds than forest offsetting ones. Then, unlike the Kyoto Protocol, which emphasises forest offset (forestry-CDM), the formal inclusion of forest conservation (REDD+) in the Paris Agreement might lower the real rates of return to long-term forest investments.

Keywords: forest assets, forest financing, loanable forest funds, natural capital markets, climate policy

1. Introduction

Recent estimates of the planet's carbon (C) budget found an unaccounted imbalance of about 1.8 Gt (1 Gt = 10^9 tonnes), whose wanting absorption points to the existence of a missing carbon sink that probably lies in forests. Out of the 7 GtC given off yearly by the combustion of fossil fuels and land-use changes around the globe, oceans absorb some 2 GtC, whereas the atmosphere takes up other 3.2 GtC [1]. The amount of atmospheric carbon transformed into forest biomass has been estimated at 25–30 Gt per year. The world's forests can store 283 GtC in their biomass alone. If the carbon held in dead wood, litter and soil is added, carbon storage can reach 762 GtC, which is more than the amount of carbon in the atmosphere [2].

Nonetheless, forests have been cautiously considered by the ongoing climate policy as a safe way to sequester and store carbon emissions. A 'forestry-CDM' (Clean Development Mechanism) was first put forward at the CoP-5,[1] in Bonn, Germany, in 1999, by African, Asian and Latin American forest-rich countries to allow for the inclusion of afforestation and reforestation projects in tackling climate change. These projects, however, concerned unnatural forests or forest plantations. Later on, at the CoP-13, in Bali, Indonesia, 2007, it was argued that 'avoiding deforestation'[2] was the cheapest way to reduce carbon emissions, thereby spurring forest protection rather than forest offset [4]. Thence, the CoP-14, in Poznań, Poland, 2008, approved a mechanism to incorporate forest protection into the efforts of the international community to combat climate change [3].

Although the forest protection feature of the 'avoiding deforestation' approach has meant a step forward regarding carbon offsetting from forestry-CDM, it does not properly encourage forest conservation. It works as a reward for 'not doing something' (not deforesting or not degrading) that is legally forbidden, instead of fostering the production of something additional, such as the storage of new carbon, brought about by forest conservation techniques[3] [5, 6].

Unlike common-sense understanding, conservation does not mean non-use, but rather saving for the future, which amounts to investment [7]. In this sense, it is an economically productive activity that implies transformation over time, and through which goods and services available today are also made available in the future [8]. Forest stocks are thus natural capital assets, whose treatment belongs in the theory of capital and investment [7].

In climate negotiations, such an approach started being outlined at the CoP-15, in Copenhagen, Denmark, 2009. Next, at CoP-16, in Cancún, Mexico, 2010, the Green Climate Fund (GCF) formally allowed for deforestation avoidance (REDD),[2] forest conservation and enhancement of forest stocks (REDD+)[4] [3]. Then, at CoP-21, in Paris, France, 2015, the Paris Agreement stated the

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[1] The Conference of the Parties (CoPs) are formal meetings of the UNFCCC (United Nations Framework Convention on Climate Change) that are yearly held to assess progress in dealing with climate change. They began in the mid-1990s to negotiate the Kyoto Protocol and legally binding obligations for the reduction of greenhouse gas (GHG) emissions by developed countries. As of 2005, the CoPs turned out to bring together the parties to the Kyoto Protocol (Annex I countries) and those that were not parties to it (Non-Annex I countries). As of 2011, the CoPs had also been being used to negotiate the Paris Agreement, concluded in 2015 and meant to be a general path for climate action [3].

[2] This strategy has got to be known by the acronym REDD (Reduced Emissions from Deforestation and forest Degradation). But, as set out at CoP-16, in Cancún, Mexico, 2010, when it is added forest conservation, sustainable management of forests (SFM) and enhancement of forest carbon stocks, a plus (+) sign is attached.

[3] These techniques are encompassed by a wide range of practices labelled under SFM (Sustainable Forest Management). 'Forest management practices to conserve and sequester C (carbon) can be grouped into four major categories: (i) maintain existing C pools (slow deforestation and forest degradation), (ii) expand existing C sinks and pools through forest management, (iii) create new C sinks and pools by expanding tree and forest cover, (iv) substitute renewable wood-based fuels for fossil fuels' [1].

[4] See Footnote 2.

importance of financial resources to encourage deforestation avoidance, forest conservation, sustainable management of forests and enhancement of forest stocks[5] [10].

Anyway, neither for natural nor for unnatural (planted) forests has a forest bond market soundly been attempted yet. So far, the bulk of climate finance comes from financial assistance,[6] flowing from developed to developing countries [11–13]. Moreover, the existing *green bonds* are mostly devoted to funding clean energy, water, low-carbon transport and building[7] [14–16]. Land use, including sustainable forestry and agriculture, is covered by only 0.1% of the green bonds market[8] [17].

In order to stand out as financial assets, forests, as any capital asset, must get their value out of the periodical income flow they are able to yield. Arguably, in the climate policy for a low-carbon economy, that income flow ought to correspond to the amount of carbon a forest can yearly store, that is, to the carbon flux (in GtC or $GtCO_2$ per year) during the time over which the forest removes any given deal of emissions.[9] As the bridge or linkage between income and capital is the interest rate [18], the greater the carbon removal flux,[10] the higher the rate of return (interest) of the forest stand—thereby implying that it can provide larger income (carbon removal) flows per year. Accordingly, an interest-bearing bond grounded in the carbon stock of a forest would pay higher yields, thus driving its rate of return (interest) to rise.

Although usually thought of as a percentage reward upon an amount of money traded off between present and future, the interest rate holds for any other goods or commodity [18], such as carbon storage. Therefore, forest finance had better draw on a commodity money standard [10, 21]. Unlike *paper* or *fiat money*, which is intrinsically useless, commodity money requires an object that is

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[5] This statement sets down, after decades of struggle, the explicit recognition by both developed and developing countries of the role that (particularly, natural) forests have in addressing climate change [9].

[6] During the initial resource mobilisation period (2015–2018), fast-start finance (FSF) for climate (10.3 billion US dollars) in the Green Carbon Fund (GCF) comes from developed countries only. They have agreed to mobilise, until 2020, 100 billion US dollars per year to meet mitigation and adaption needs in developing countries [11, 12]. So far, annual climate finance flows from developed to developing countries have been estimated to lie roughly between 10% (40 billion US dollars) and 25% (175 billion US dollars) of estimated global total climate finance. About half of this share corresponds to grants, with mitigation receiving the largest part, whereas one-third of it is accounted of by ODA (Official Development Assistance) loans provided by multilateral climate funds, whose resources come virtually in full from developed countries' national governments [13].

[7] *Green bonds* have been establishing an increasingly attractive niche in the financial market. In 2014, the issuance of green bonds skyrocketed to a record of 37 billion US dollars, driven by a surge in corporate self-labelled issuance—that is, bonds issued by corporations with proceeds ring-fenced for green investments—as well as by volumes from large international and supranational banks. Regardless of its fast growth, however, the global green bonds market accounts for about 2.5% only of the issuance of corporate bonds in the USA alone, which was worth 1.4 trillion US dollars in 2013 [14, 15].

[8] Approved standards are still missing to set out which land use projects are applicable for bond issuing and certification [16]. Meanwhile, forest funds, which have been essential to tackling deforestation and to laying the groundwork for more sustainable management and governance of the natural assets of countries, rely on results-based finance (RBF), whereby direct payments are made upon delivery of pre-defined climate outcomes, such as verified greenhouse gas (GHG) emission reductions [9].

[9] Even though forest bonds have long been recognised as a potential financing instrument, their use has usually been called for avoiding deforestation, where income flows or revenue streams are not obvious [17].

[10] Sometimes [19, 20], data on net carbon *removals* are represented by a *negative* flux, whereas those on net carbon *emissions* are expressed by a *positive* flux. However, provided that the capital value of a forest is given by its carbon storage, *positive* fluxes throughout this chapter stand instead for net *removals*.

intrinsically useful as an input to production or consumption. A claim to (loan of) long-lived capital, like forests, contains an option to consume a predetermined service flow, such as the storage of carbon emissions, that can be used, like commodity money, as a medium of exchange [21, 22].

Provided that carbon storage is an actual source of forest income, *carbon money* conveys the expected uptake of carbon emissions by a forest. Whereas conservation of natural forests either enhances it or avoids carbon losses from deforestation and forest degradation, forest plantations (unnatural forests) can offset carbon emissions given off by the economic activity. Therefore, either natural or unnatural forests can be used as removal sinks (carbon savers) to carbon-consuming investments (emitting sources). By issuing and supplying carbon-laden bonds, emitting sources may meet carbon-saving sinks whose offering of removal capacity corresponds to a demand for carbon emissions enclosed in bonds. The demand for carbon conservation, through natural forests, or carbon offsetting, through forest plantations, will ride on the rate of return (interest) each forest bond potentially offers to its holder.

The major objective of this chapter is then to find the real rate of return (interest) (r) on natural ($k = n$) and unnatural ($k = u$) forest bonds in both Annex I ($j = 1$) and Non-Annex I ($j = 0$) countries.[11] In this regard, six scenarios have been set out, in which emissions from Annex I and Non-Annex I countries' economies demand removal being supplied by either natural or unnatural (planted) forests. Scenarios vary according to either the removal sinks called in (natural forests, unnatural forests or both) or the emitting sources seeking carbon removal (Annex I, Non-Annex I countries or both). But in none scenario, the rate of interest is supposed to vary according to the quantity or value of money.

The reason for this classical, nonmonetary assumption, discussed in the following sections, is twofold. Theoretically, changes in the rate of interest are ultimately triggered by changes in the demand for real commodities, whose movements affect the demand for money and cause prices to alter [24]. Empirically, carbon is a commodity whose emerging bond market includes a great deal of currencies. At present, there are 25 currencies represented in the labelled green bond market [17]. Therefore, the determination and comparison of money rates of interest would not come without tackling disturbances caused by monetary phenomena affecting each currency (e.g. inflation rates, budgetary imbalances, money and credit supply).

2. Historical background

Long-term options on climate financing started being taken up at CoP-15, in Copenhagen, Denmark, 2009, following the Copenhagen Accord and the Copenhagen Green Climate Fund.

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[11] *Annex* I Parties comprise the industrialised countries that were members of the OECD (Organisation for Economic Cooperation and Development) in 1992, when the UNFCCC emerged out of the Rio Earth Summit, plus countries with economies in transition (the EIT Parties), including the Russian Federation, the Baltic States and several Central and Eastern European States. *Non-Annex* I Parties are mostly developing countries. Certain groups of developing countries are considered to be especially vulnerable to the adverse impacts of climate change; others, whose income is heavily reliant on fossil fuel production and commerce, feel more vulnerable to the potential economic impacts of climate change response measures [23].

At CoP-16, in Cancún, Mexico, 2010, the Green Climate Fund (GCF) was formally put forward to assist developing countries in adaptation and mitigation practices to counter climate change. Concerning mitigation, issues such as avoidance of deforestation and forest degradation (REDD), forest conservation, sustainable management of forests (SFM)[12] and enhancement of forest carbon stocks in developing countries (REDD+) were addressed. Developed countries then committed themselves to provide 30 billion US dollars of fast-track finance in 2010–2012 and to jointly mobilise 100 billion US dollars per year by 2020 [3, 25].

At CoP-17, in Durban, South Africa, 2011, the GCF became, along with the existing Global Environmental Facility (GEF) of the World Bank, another operating entity of the financial mechanism of the UNFCCC (United Nations Framework Convention on Climate Change). However, at CoP-18, in Doha, Qatar, 2012, little progress was made towards the funding of the GCF. Developing countries became suspicious of the provenance of the money pledged to it. They feared that this money could be raised from private sector's wealthy investors who would deny channelling it to poorer regions in need of climate finance resources [3, 10, 26].

At CoP-19, in Warsaw, Poland, 2013, the financing of renewable energy and of technology transfer to developing countries was brought up. Accordingly, climate finance and capitalisation of the GCF were considered the most important milestone. Yet, financial commitments made before by developed countries melted into talks about alternative sources of funding and a rebuttal to support any loss or damage payments to developing countries [3].

A binding[13] and global agreement to reduce climate change was finally reached at CoP-21, in Paris, France, 2015, after a somewhat faint CoP-20, held in Lima, Peru, 2014. The *Paris Agreement*, the legal instrument ruling climate policy as of 2020 in place of the Kyoto Protocol, restated the leadership of developed country parties in providing financial resources to developing country parties for both mitigation of and adaption to climate change (Article 9). These latter countries, though, were encouraged, as part of a global effort, to provide support voluntarily [3, 10, 26].

Despite its shortcomings, the Paris Agreement recognises the need of finance flows to tackle climate change (Articles 2 and 9). However, for reasons just mentioned, there is some controversy about the extent to which capital markets should assist in raising the funds to the GCF. On one hand, by pooling savings, free capital markets are said to provide planned investments with the money needed to carry them out [27, 28]. Put differently, capital markets allow for the so-called

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[12] See Footnote 3.

[13] The Paris Agreement calls forth its signatory parties to set their own emission reduction targets, thereby making their individual contribution towards the worldwide goal of reaching 'global peaking of greenhouse gas emissions as soon as possible' (Article 4). According to Article 2 of the agreement, the achievement of this goal implies holding the increase in the global average temperature below 2°C above pre-industrial levels and limiting the temperature increase to 1.5°C above pre-industrial levels. Each party must, therefore, establish its 'nationally determined contribution' (NDC) on a voluntary, non-binding basis. These national targets are required to be ambitious and to follow close the 'principle of progression', whereby each further contribution must be more ambitious than the previous one. So, in this narrow sense, that the parties' determined contributions must demonstrate progression over time, the agreement can be said to be binding. Yet, in a wider sense, the contributions themselves are not binding. There is no enforcement mechanism to set them, to phase them out, nor for non-compliance with them. Hence, regardless of its outreach coverage, bringing together 217 signatories plus 85 ratifying or acceding countries, the Paris Agreement rests on a fragile consensus. Countries failing to meet their commitments can easily withdraw, which might as well encourage other unsuccessful parties to do the same, thereby bringing about the total collapse of the agreement [10, 26].

finance demand for liquidity—the demand for money arising during the period between the date when the entrepreneur arranges his finance (cash) and the date when he actually makes his investment [28]. Such an investment finance is a special case of finance required by any productive process and lies halfway between the active and the inactive balances [29]. On the other hand, savings are thought to withhold money that could otherwise finance investments. Accordingly, money so withdrawn causes the rate of interest to rise, thereby impairing the investments [28].

The former standpoint shares with the classical *loanable funds* (LF) theory the view that savings support investments, whereas the latter stance draws on the *liquidity-preference* (LP) theory, to which savings lessen investments.

3. Theoretical background

After all, how helpful might capital markets actually be with mobilising climate finance? The answer lies in the extent to which money balances withhold or encourage investments. There are two opposed theoretical views concerning these propositions.

3.1. Nonmonetary and monetary theories of interest

Nonmonetary theories of interest, put forward by the classical economists, argue that it is not money lending or borrowing that regulates the rate of interest,[14] but rather the rate of profits (return on capital), which is totally independent of the quantity or value of money, yet dependent, instead, on the time length of production and the *real* forces of productivity [24, 30–32]. According to nonmonetary assumptions, any decision on saving (not consuming) implies another decision on spending on capital goods (investing). Provided that money is a medium of exchange, there can be no hoard[15] of idle monetary balances. Therefore, savings are turned into an available fund to be loaned for investment. The *real* interest rate is then the price that rewards the lenders of funds (savers) for their postponement of consumption until a certain moment into the future, provided that commodity prices would remain constant [32].

If it is true that people have positive time preferences,[16] thereby preferring the present over the future consumption, then the higher their income, the easier for them to put off their

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[14] Classically, the two nonmonetary reasons underlying the rate of interest are *time preference* and the *physical productivity of capital goods*. On the demand side, *time preference* implies that people generally value present goods more highly than future goods, chiefly because the means to meet present needs are thought to be scarcer than those to meet future needs. The future is believed to be more plentiful, either because people assume that their earning capacity will be greater then, or because their current possession of a durable asset gives them advantage to choose between using it either now or in the future, whereas future possession gives only the advantage of the latter use. On the supply side, the growing productivity of capital goods is, although riddled with controversy, ascribed to some technical superiority—capital goods reproduce more of themselves over time. However, there is at least one aspect of technical superiority on which there is no confusion. The technical superiority of present goods (either capital or consumption goods) is in part due to the fact that the present investment of resources has a greater present value than next year's investment of those same resources [24].

[15] Hoard is defined as the quantity of money supplied *less* the quantity of money demanded by the public to meet its transactions of goods and services or precautionary behaviour [33].

[16] See Footnote 14.

consumption further into the future. Savings, therefore, are encouraged by increasing incomes, rather than by higher rates of interest. This argument bears the gist of the attack of *monetary* on nonmonetary theories of interest, regardless of the latter's reply that, under the assumption of full employment in the long run, neither income, nor the economic output, nor commodity prices are supposed to vary [32]. Yet, monetary theorists insist that there can be savings irrespective of the rate of interest, which is not a reward for waiting or not consuming, but, rather, for not hoarding.[14] Likewise, the rate of return on loans or on investments is not a reward for the wait itself, but, rather, for the preference towards risk. Since, in the contemporary monetary economies, all transactions are carried out in money rather than in commodities, there is a direct and objective relationship between the quantity of money and the rate of interest [33]. The rate of interest responds therefore to changes in the supply and demand of *money,* instead of funds (loans) [24].

3.2. Money rate and natural rate of interest

Nonetheless, nonmonetary theorists maintain that, even in an economy using money, the relation between capital and its yield, or between rent and interest, has no connection with the borrowing and lending of money [24]. 'Money does not itself enter into the process of production' [34]. The borrower of money does not intend to keep it but to exchange it at the first suitable opportunity for goods and services [34]. Moreover, money could, theoretically, be substituted for any other commodity. Yet, in practice, only money is traded off for between present and future. For this reason, in contemporary economies, the rate of interest is often misleadingly defined as the 'price of money' [18].

Actually, this *money* rate of interest is but a kind of aberration from the *real, natural* or *normal* rate, which depends on the efficiency of production, on the availability of fixed and liquid capital[17] and on the supply of labour and land[18] [30]. Accordingly, the *natural* rate

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[17] *Fixed* capital is the one bearing a very *high* and sometimes unlimited *durability,* such as houses, streets, railways, canals, certain improvements in land and certain kinds of machines. They are *rent-earning* rather than capital goods. Unlike *real* capital goods, which are due the payment of interest, rent-earning goods are not, because they contribute to output either with or without the assistance of further labour and land. Instead, they earn for their owners a certain rent, analogous to the rent of land [30]. The determination of the *real, natural* or *normal* rate of interest, therefore, does not rely on this kind of capital, which is more or less fixed or tied up in production. It rather hinges on *liquid* capital, that is, *mobile* capital in its *free* and uninvested form. Unlike commonly thought, this kind of *real* capital consists neither of stocks of manufactured, semi-manufactured and consumption goods, nor of stocks of raw materials. Actually, *free* capital does not have any material form at all. It is accumulated by those who save and abstain from consumption a part of their income. 'Owing to their diminished demand, or cessation of demand, for consumption goods, the labour and land which would otherwise have been required in their production is set free for the creation of fixed capital for future production or consumption and is employed by entrepreneurs for that purpose with the help of the money placed at their disposal by savings' [34]. Thus, *the natural* or *normal real* rate of interest corresponds to the expected yield on the newly created capital.

[18] One landmark assumption in classical theories of interest, like the so-called Stockholm theory of savings and investment, draws on Böhm-Bawerk's proposition that there are only two 'original' factors of production: land and labour. Capital just comes into existence because production takes *time* [31, 35]. Or, as Wicksell states it, 'the characteristic of capitalist production lies simply in the fact that … the main portion of the available labour and land is employed for the purposes, not of *current* consumption, but of consumption in the more or less distant *future*' [30]. Therefore, apart from decreasing returns, the lengthier the period of production, the larger the output. Too lengthy periods of production, however, would be held off by positive time preferences (see Footnote 14). That is, people prefer present over future goods, because of their present needs and uncertainty about the future [31].

follows suit the rate of return on capital and, at best, holds, as it were, only an indirect relationship with commodity prices[19] [30, 34]. It is determined by supply and demand as if no use of money were made and all lending were carried out in the form of real capital goods [24, 30]. Thus, in these circumstances, the use of a *money* rate of interest does nothing more than serving as a cloak to cover a procedure, which could have been carried on equally well without it [30].

3.3. Money rate and own rate of interest

The *money* rate of interest is technically defined as the percentage excess of a sum of money contracted for forward delivery over the *spot* or cash price of that sum. Its analogue for every kind of capital asset is the *own* rate of interest on commodities, which is the rate of interest for every durable commodity in terms of itself [24]. The former rate, set in terms of money, and the latter, set in terms of commodities, indicate a relationship between present and future values of assets (including money), whose most fundamental meaning is that provided by Fisher's marginal rate of return over cost [18, 24], namely: (future income − present income) ÷ present income.

In either case, income can be replaced by the amount of an asset (including money) that could be secured at some future time in return for a given present amount. Lending is, in any event, involved, because futures are bought in exchange for spot claims or, likewise, present assets are converted into future assets. However, if money is lent, the lender sells an immediate claim to buy a future claim. Conversely, when a commodity is borrowed, the borrower buys a spot claim and sells a future claim. Therefore, the relation between *money* and *own* rates of interest builds upon the type of asset used to work them out [24]. Whereas the *money* rate of interest is sensitive to commodity prices, the commodity (*own*) rate of interest will

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[19] As Wicksell remarks, interest on money and profit (return) on capital are not the same thing. Yet, interest and profit connect to one another through the *effect on prices* caused by their difference. Hence, when the rate of interest is lower than the rate of profit, prices must rise. Such a difference between the two rates turns credit easier, thereby bringing about the excess of demand over supply of raw materials, labour, land, and the like, as well as, directly and indirectly, of consumption goods. In the opposite situation, when the money rate of interest is higher than the rate of return on capital, prices fall [30, 36]. Thus, Wicksell's unusual proposition is that rates of interest and prices run in opposite directions. It frontally clashed with the ambiguous view that the money rate of interest depended not only on the excess or scarcity of money but also on the excess or scarcity of real capital. This opposing proposition followed from the usual definition that interest is the compensation paid for the use of capital, not of money. Money is only one of many forms of capital that can be transferred through loans. Therefore, under a system of credit, business men could get money to buy the capital goods (investment) needed to production. On one hand, this would result in an increased output (supply), thereby, given an unchanged output demand, causing commodity prices to fall; on the other hand, the growing demand for capital goods would raise the rate of interest [30]. Yet, followers of the *Banking School* and even of the *Currency School* of money suggested that a low rate of interest cheapened one of the elements of production, thereby bringing commodity prices down, whereas a high rate of interest raised the costs of production, thereby driving commodity prices up [36]. In any event, the rate of interest and prices moved in the same direction. Wicksell, then, wonders: 'How can a scarcity of goods be regarded as a cause of a rise in the rate of interest or a fall in prices?' And he, himself, keeps on to give the answer: 'On the contrary, the smaller the available amount of commodities, the smaller … is the demand for money. It follows that the rate of interest will fall rather than rise and that prices will go up still further' [30].

only be equal to the money rate if the spot price of the commodity is the same as its forward price[20, 21] [24, 30, 37].

Monetary theorists reply that, in contemporary economies, capital is, notwithstanding, lent in the form of money [30, 37]. Because barter is unwieldy, seldom are real and present commodities exchanged for real and future ones [37]. Not even merchandise credit involves any lending of commodities. Rather, it is carried on through a sale where payment is temporarily postponed or where a cash transaction is combined with a money loan [30]. At best, intertemporal trade corresponds to the exchange of present commodities for a pledged cash payment in the future (postponed payment), or, vice-versa, to the exchange of cash for a pledged delivery of commodities in the future (anticipated payment). Hence, in either case, any credit transaction comes down to a *money* loan combined with a spot or forward delivery of goods [37].

3.4. Loanable funds (LF) and liquidity-preference (LP) theories

Unsurprisingly, in the context of monetary economies, nonmonetary theories of interest have been deemed inadequate. They deny that changes in the quantity of money or the desire to hoard can set off but temporary, short-run effects throughout the economy. Drawing on classical concerns, nonmonetary approaches turn instead to *long-run* problems and against money as the exclusive determinant of the rates of interest [24].

Whereas pre-classical writers were men of affairs, concerned with daily events and thus with the *short-run* forces affecting the rate of interest, the classical writers were mostly philosophers of political economy whose concern was less with daily changes than with *long-run* movements [24]. Even though changes in the quantity of money might as well have a lasting effect on the rate of interest, the classical economists were primarily concerned with brushing aside any confusion that might be implied by mixing up *monetary* and *real* capital. Whereas the former refers to financing funds and to a certain deal of money, the latter includes concrete goods and certain amount of them [37]. Which, after all, eventually determines the rate of

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[20] The *own-rate of interest/return* (or the commodity rate of interest/return), r, is given by $r = (Q_2 - Q_1)/Q_1$, where Q_1 is the quantity of the commodity in the present, and Q_2 is the quantity of the commodity in the future. The *money rate of interest/return, i*, is given by $i = (P_2Q_2 - P_1Q_1)/ P_1Q_1$, where P_1 is the spot price of the commodity, and P_2 is the forward price of the commodity. Clearly, if $P_2 = P_1$, then $i = r$. An adjustment factor α can be found which represents the difference between the money rate of interest/return (i) and the own-rate of interest/return (r). This factor measures the influence of the price change $((P_2 - P_1)/P_1)$ on the future quantity of the commodity (Q_2), as if it should continue to be valued at its cash price (P_1). Algebraically: $[(P_2 - P_1)/P_1)]\cdot P_1\cdot Q_2$. In order to express this value as a proportion of the original value, it must be further divided by P_1Q_1. Therefore: $\alpha = \{[(P_2 - P_1)/P_1)]\cdot P_1\cdot Q_2\}/ P_1Q_2 = (P_2 - P_1)/P_1)\cdot (Q_2/Q_1)$ [24].

[21] *Forward* prices must not be mistaken for *future* prices. *Forward* prices are current prices with an addition for interest. They refer to prices accepted today for an immediate delivery of goods which will not be paid for until some point in the future. Therefore, they have nothing in common with prices that will have to be paid in the future for goods or services supplied *in the future*. The level of these *future* prices will be determined by the relation existing in the future between the conditions of supply and demand [30].

interest, the level of savings and investment: the demand for *money* or for *goods*? The *loanable funds* (LF) theory[22] claims that the right answer lies in the excess demand for *goods*; the monetary *liquidity-preference* (LP) theory maintains, on the other hand, that it rests on the excess demand for *money*[23] [40, 41].

Building on the demand and supply of loans, the LF theory holds that *securities* determine the rate of interest [39, 41, 42]. Regulating the supply and demand of 'claims' or interest-bearing securities, the rate of interest becomes the driver of the investors' supply and savers' demand of (loanable) funds that can be borrowed and lent [37, 39]. So, the supply of loanable funds (S) may be thought of as being the demand for claims or securities (B^D), whereas the demand for loanable funds (I) may be regarded as the supply of claims or securities (B^S)[24] [24]. The LF theory, however, emphasises savers' behaviour, because the rate of interest is taken rather as the *cause* than as the effect of saving.

In sharp disagreement with monetary assumptions, this proposition maintains that the rate of interest can neither be a reward for not hoarding or waiving liquidity (i.e. demanding money, the most liquid asset), nor can it be determined by the desire to keep money idle [39]. Savings are directed either to idle balances, through the demand for money, or to active investments, through the demand for securities [24, 27, 39]. Idle balances in the hands of consumers constitute but one of the alternative destinations of savings and absorption of cash, the others being investment market, banks and circulating capital of industry [39].

More generally, 'the rate of interest is simply the price of credit, and it is therefore governed by the supply of and demand for credit [or finance]' [35]. The supply of credit (or finance) is

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[22] Monetary theories of interest have had three major roots: (1) the Swedish approach provided by the Stockholm theory of savings and investment [30, 34, 35, 38, 39], initiated largely by Knut Wicksell and followed by Bertil Ohlin, Eric Lindahl, Gunnar Myrdal and Bent Hansen; (2) the English neoclassical tradition, most fully represented by D. H. Robertson; and (3) the school founded by John Maynard Keynes [29, 33]. The Swedish and English approaches are conveniently grouped together under the head of *loanable-funds theories* (LF); the Keynesian approach is best known as the *liquidity-preference theory* (LP). Yet, it is disputed whether the LF theories tune in to the monetary or to the nonmonetary frequency of the spectrum. As shown in Section 3.2, Wicksell's dynamic analysis builds on the divergence between the *natural* rate and the *money* rate of interest. Whereas the *natural* rate of interest owes to the (classical) nonmonetary tradition, the *money* rate of interest springs from the monetary branch. Therefore, at worst, the LF thinking represents a transitional linkage between one and another theoretical tradition [24]. Nonetheless, its underlying assumptions and analytical framework recall, to a large extent, those of nonmonetary theories.

[23] Assuming that financial wealth (W) may be split into *monetary* assets (money) and *nonmonetary* assets (e.g. bonds), it can, from the point of view of *income*, be expressed as $W = M + B^S$ (where M is money supply and B^S is bonds supply), and, from the point of view of spending, as $W = L + B^D$ (where L is the demand for money and B^D is the demand for bonds). By getting both expressions together, it comes out: $M + B^S \equiv L + B^D$, which, rearranging, gives $(B^S - B^D) \equiv (L - M)$. This macroeconomic identity is said to describe the static partial-equilibrium analysis of the financial market. The left-hand side accounts for *nonmonetary* assets, in which B^S corresponds, in the loan market, to the demand for loanable funds springing from investments (I), and B^D, to the supply of loanable funds brought about by savings (S). Therefore, in terms of goods market, the left-hand side would translate into $(I - S)$. Likewise, the right-hand side accounts for *monetary* assets. The excess demand for goods occurs when $(I - S) > 0$, whereas the excess demand for money comes about when $(L - M) > 0$. In the LF theory, the money market is supposed to be in equilibrium $[(L - M) = 0]$, so that the rate of interest is fully determined in the bonds market. Conversely, in the LP theory, the bonds market is supposed to be in equilibrium $[(B^S - B^D) = 0]$, so that the rate of interest is entirely set by the money market [40].

[24] See **Figure 1**, in Section 4.

given by people's willingness to hold different interest-bearing claims (bonds or equities) and other kinds of assets, whereas the demand for credit (or finance) is governed by the total supply of claims [35]. Yet, credit is closely related to savings and investment, because any saver (supplier of funds) must decide as well on whether to invest (demand for funds), to lend (demand for bonds or equities) or, even, to increase the quantity of cash (demand for money) instead of lending [39]. As loanable funds (or balances) come out of the discrepancies between income and expenditures, changes in idle stocks arising from new hoard (savings) or dishoard (investment) define a *flow* that will respectively give rise to a demand for claims (bonds or equities) and a supply of claims. Therefore, the LF theory is said to take up a *flow*, rather than a stock, approach[25] [24].

The *stock* (portfolio) approach is taken up by the LP theory, according to which the demand for money (or *liquidity preference*) determines the stock of cash held by society. If its individuals prefer to hold money over other assets (e.g. bonds), they will be hoarding and accumulating idle balances instead of increasing the working capital or active investment. Thus, their savings are diverted from the investment market to increase idle balances at the expense of the active ones. So long as the supply of money is assumed to be rigidly fixed, the consequence of an increased propensity to hoard is the rise of the rate of interest, which is, therefore, typically a monetary variable [39].

Since it is set by the supply and demand of money (cash) *solely*, the rate of interest in the LP theory is, unlike in the LF theory, rather the *effect* than the cause of savers' behaviour, which is assumed to be driven by liquidity preference (i.e. demand for money). Given this public's propensity to hold money, the LP theory argues that investments rely less on foregone consumption—that is, existing savings—than on financing—that is, access to money [24, 27, 33, 37]. Thus, investments can never be constrained by the lack of savings, but, rather, by the lack of money [27].

In some instances of the LF theory, on the other hand, the only effect of money is causing prices to change. The money rate and the natural (real) rate of interest mostly differ, because the transfer of capital and the remuneration of factors of production are not made in kind, but 'in an entirely indirect manner as a result of the intervention of money' [30]. So, instead of being lent or borrowed, real capital goods are now bought or sold. Therefore, 'an increase in the demand for real capital goods is no longer a *borrowers'* demand which tends to raise the rate of interest, but a *buyers'* demand which tends to raise the prices of commodities' [30]. Since the LF theory minds the effects of money on the real factors of production, it is often said to link nonmonetary and monetary theories of interest[26] [24].

To wind up, what monetary theories seem to have overlooked is that, although money is credit, credit is not necessarily money. Only when a debt pledge can be transferred to or traded with a third party, does credit get close to money. Then, transferrable debt and money blend into each other to mean the same [43]. Yet, money comes after. There must have been credit or some sort of transferrable fund—whether a debt claim (bond) or an ownership claim (equity)—before. Moreover, the public's holdings of cash (money) and credit, plus what it receives during a period, define its ability to spend or its total (unused) purchasing power. Whether it is exerted in the

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[25] This split, however, is often disputed. A thorough discussion can be found in [24].

[26] See Footnote 22.

present (consumption) or delayed until the future (investment), the purchasing power cannot be said to have diminished whatsoever but simply transferred over time [38]. Unlike in monetary theories, in nonmonetary theories, money does not itself function as a store of value. Therefore, any deferment of purchasing power can only be done by means of nonmonetary assets, like securities (bonds or equities), which are pieces of property that can store value [44].

4. Methodological assumptions

What, after all, does this theoretical discussion have to do with financing long-term productive investments, such as forest conservation or plantations? Natural capital, like forests, is, of

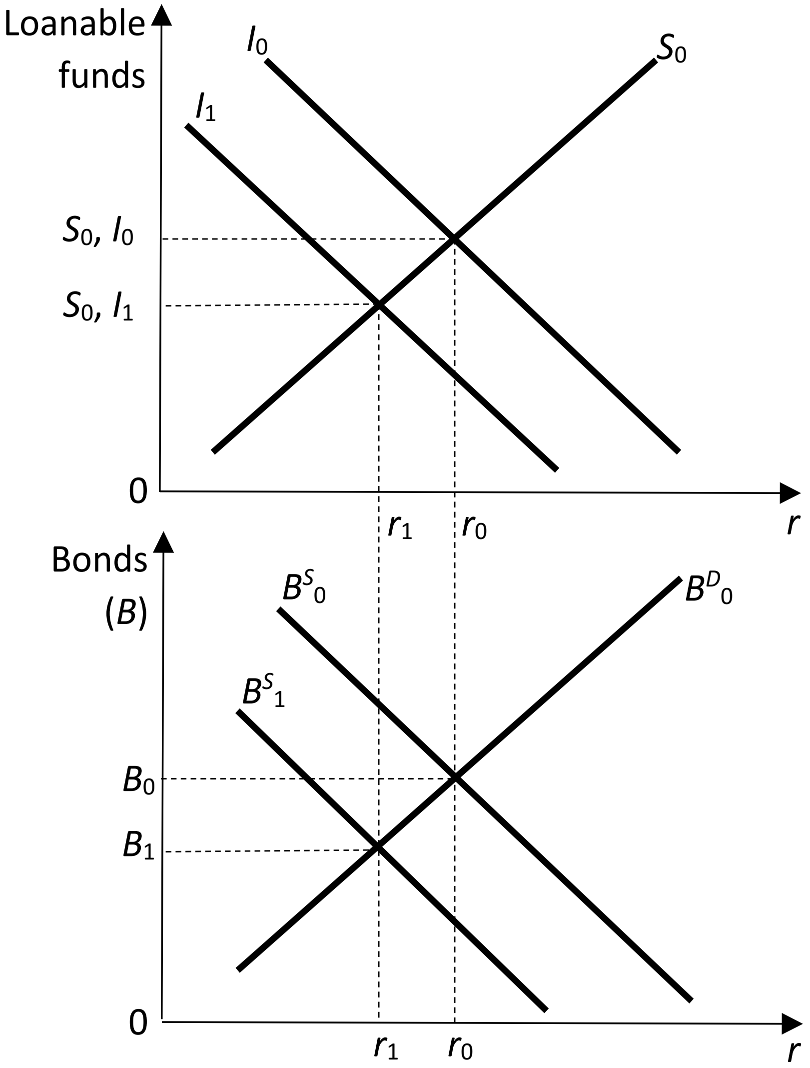

Figure 1. Loanable funds market.

course, a *real* rather than a monetary capital. It delivers concrete goods, namely carbon stocks, whose monetary valuation is laden with uncertainty concerning their future prices. Hence, alternatively, carbon stocks can be thought of as intrinsically useful objects that might serve as media of exchange, since they deliver environmental protection and offsetting services.

4.1. Commodity money

Unlike *paper* or fiat money, which is intrinsically useless, carbon stocks enclose a service flow that can be optionally consumed [21]. Of course, because, as yet, they are neither legal tender tools nor do fully function as unit of account, medium of exchange and store of value, they cannot be taken as money proper. However, they virtually fit in an economy with a *commodity* money standard.[27]

A commodity money system provides an anchor to the price level, so that prices, as claimed by the LF theory, do not affect the rate of interest (see Sections 3.3 and 3.4). In this system, when the value of the commodity-bearing money falls, it becomes preferable to exercise the option and convert it into other, nonmonetary uses, thus reducing the quantity of money and preventing its value from falling further. Conversely, when the value of the commodity-bearing money rises, it becomes preferable to hold more money [21].

4.2. Carbon money and loanable funds model

If carbon stocks are the commodity standard, their outflow (say, because of deforestation) lessens a country's (natural) assets and, therefore, the supply of *carbon* money [45]. For a given demand for carbon removal stocks, the rate of interest on carbon stocks must rise. This upward movement, however, concerns the rate of interest on commodity money. The rate of return on forest assets (i.e. the *natural* rate of interest) is supposed to remain unaffected, provided the removal capacity of forests has not changed yet. Then, if, as in **Figure 1** and in the Wicksell's version of the LF theory (see Section 3.4), the 'commodity money' rate of interest (i) becomes higher than the *natural* rate of interest[28] (r), the demand for loanable forest funds (I) to finance offsetting forest investments (plantations) will fall (from I_0 to I_1). As no further production of carbon offsetting stocks will take place, there will be no additional demand for raw materials or factors of production. Thus, the prices of unnatural forests will go down, thereby diminishing the supply of carbon-laden forest bonds (from $B^S{}_0$ to $B^S{}_1$). So, just like in the Wicksell's system, an initial increase in the (commodity) money rate of interest causes prices to fall.[29]

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[27] The *gold standard* is certainly the most known historical example. For more details, refer to [45, 46].

[28] Ohlin disagrees with Wicksell in respect of the distinction between those rates [27, 35]. 'The distinction between "normal" [natural] and "not normal" interest rates and savings depends on arbitrary assumptions that one kind of economic development, e.g. a constant wholesale price level, is "normal". Besides, it is far from certain that there is always one interest level which guarantees the existence of this normal development. On the one hand, it is possible that *no* interest level can do this. On the other hand, a great many and rather different interest levels may satisfy the condition of being compatible with this development' [35]. Ohlin, therefore, concludes that, in a dynamic analysis, such ideas have to be given up, although, on static assumptions, it is possible to define a certain interest level and the corresponding volume of savings which is compatible with the maintenance of static equilibrium. In this case, savings and interests diverging from them could then be called 'abnormal' or 'artificial' [35].

[29] See Footnote 19.

Eventually, *cœteris paribus*, the rate of interest on loans will end up falling too (from r_0 to r_1). Conversely, because of the inverse relationship between the price of bonds and the rate of interest (Eqs. (1) and (5)), prices of unnatural forest bonds (P_{uj}) will rise. The opposite movements hold for an inflow of carbon stocks (say, because of growing forest conservation).

5. Loanable-Forest Funds (LFF) model

Long-term climate financing laid down in the Paris Agreement is particularly attempted now for forests by applying a loanable funds model. The structure of financial markets in general is shown in **Figure 2**, while that of the *Loanable-Forest Funds* (LFF) model is displayed in **Figure 3**.

Data on net removal of carbon emissions per year by the world's forests come from FAO for the 1990–2015 period[30] [47]. These data are used to estimate the world's supply of carbon removal stocks by forestland (S), whether it is covered by natural or unnatural forest sinks. These carbon stocks are the *funds* that forest sinks of kind k will loan to carbon emission sources j (Annex I and Non-Annex I countries) by demanding carbon-laden forest bonds (B^D). These bonds are supplied (B^S) by the emission *sources*, which demand carbon stocks to prevent or offset the emissions brought about by their industrial[31] investments (I). The issuance of bonds, however, must reckon how long it will take for either kind of sink to fully meet the sources' removal needs. These needs are set by the total emissions of carbon dioxide (CO_2) at *each* source. Thus, the bond price (P) depends not only on the yearly carbon removal flux (Ψ) at each sink but also on the emissions per source. Data on CO_2 emissions per source are provided by the World Bank [48]. Yearly carbon fluxes in natural forests were inferred from [49], whereas, in unnatural forests, they were estimated by [1].

All data related to emissions and removal fluxes have been converted from carbon (C) into CO_2 at the physical-chemically defined rate of 3.67 tonnes of CO_2 per tonne of C. Since all emissions are measured in real CO_2 units, they are more intuitive to the general public than the corresponding C units, preferred by scientists and governments [50]. Data on carbon emissions by countries are statistically regressed on annual real rates of interest informed by the World Bank [48] at each emission source (r_j) to obtain the demand function for carbon funds per source (I_j). The corresponding supply function of carbon funds (S) is arrived at by statistically regressing data on carbon removal by forests on average real rates of interest per annum at sources [$\bar{r} = (1/j)\Sigma_j r_j$].

In line with the LF theory, it is assumed that, if money influences are set aside, S figures fully translate into the demand for bonds (B^D) by *all* carbon-saving sinks k (both natural and unnatural forests), so that $S \equiv B^D$ [24]. The same holds for I_j, which will correspond to the quantity of bonds of kind k supplied by each source j (Annex I and Non-Annex I countries), so that $I_j \equiv \Sigma_k B^S_{jk}$ and then $\Sigma_j \Sigma_k B^S_{jk} \equiv \Sigma_j I_j \equiv \Sigma_k \Sigma_j B^S_{jk} \equiv B^S$.

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[30] These data refer to net emissions/removals by forests, but leave out net emissions/removals from deforestation (forest conversion). This is because deforestation is an emission source (demand for carbon removal stocks), rather than an emission sink (supply of carbon removal stocks).

[31] The adjective 'industrial' here is employed in an as broad meaning as to comprise any industry or productive activity. In this sense, sectors like agriculture and commerce, for example, are also considered 'industries'.

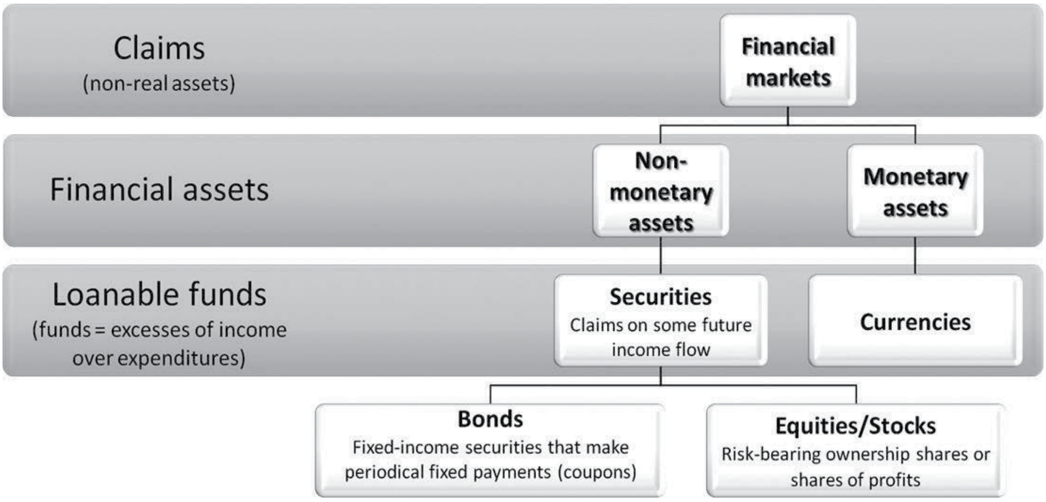

Figure 2. General structure of financial markets. Sources: Refs. [24, 42].

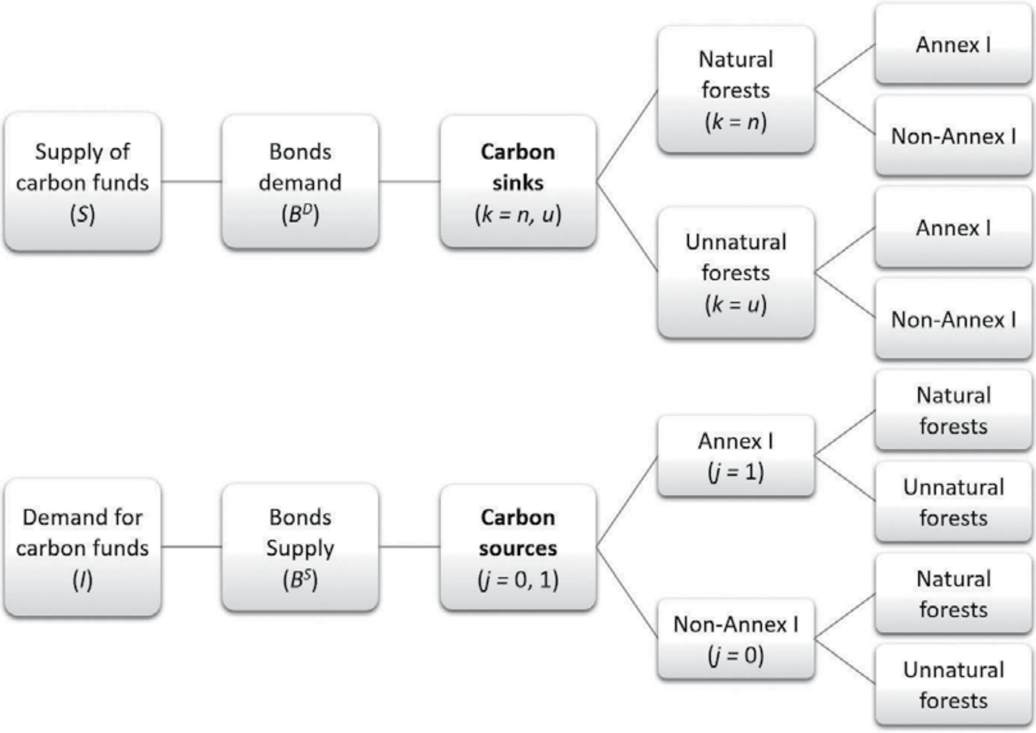

Figure 3. Loanable-Forest Funds (LFF) model.

The LFF model's interest is twofold. First, it is concerned with the total supplies of bonds made up in the last column of **Table 1**, that is, $B^S = \Sigma_k \Sigma_j B^S_{jk} = B^S_n + B^S_u$, which corresponds to the *row* sums. These *row* sums highlight the potential contribution of each forest sink to taking up the emissions

		Sources (j)		
		Non-Annex I (j = 0)	**Annex I (j = 1)**	**Total**
Sinks (k)	Natural (k = n)	$B_0^{S,n}$	$B_1^{S,n}$	$\boldsymbol{B^S}_{\boldsymbol{n}} = I^n$
	Unnatural (k = u)	$B_0^{S,u}$	$B_1^{S,u}$	$\boldsymbol{B^S}_{\boldsymbol{u}} = I^u$
	Total	$\boldsymbol{B}_0^{\boldsymbol{S}} = I_0$	$\boldsymbol{B}_1^{\boldsymbol{S}} = I_1$	$\boldsymbol{B^S} = I$

Table 1. Bonds-supply matrix.

given off by whichever bond-supplying source. Second, the LFF model is concerned with the *column* sums ($B^S = \Sigma_j \Sigma_k B^S_{jk} = B^S_0 + B^S_1$), which split up the total supply of bonds between the emitting sources. To hold on to the LF theory's assumptions, forest sinks are like 'capital goods' (or carbon savers) producing carbon stocks, whereas emitting sources are like consumers that cannot save for increasing those stocks.

One last assumption of the LFF model is that bringing forest assets under bonds rather than equities (**Figure 2**) appears more financially sound. Since bond issuers (emission sources j) owe 'money' (carbon stocks) to bond purchasers (forest sinks k), bonds, in general, and forest bonds, in particular, stand for *debt securities* [42].

Because forests are long-standing assets, the LFF model considers forest bonds as *perpetuities*, a special kind of coupon bond that does not repay its face value (principal), but makes fixed periodic payments (coupons) indefinitely[32] (Eq. (1)) [44]. In carbon-money forest finance, however, coupon payments are made in the form of commodity money (carbon stocks). Therefore, the sink k (bond purchaser) to which the forest bond refers is required to periodically deliver a uniform income flow that makes available, as close as possible, the quantity of carbon stocks that meets the removal needs of the emitting source j (bond seller/issuer). Then, ultimately, the coupon payments correspond to the total quantity of bonds annually supplied by each source j (Eqs. (14) and (15)), in order to finance and meet its demand for carbon removal stocks.

$$P_{kj} = B^S_k / r_j, \tag{1}$$

where P is the bond price, B^S is the annual coupon payment in the form of commodity money (carbon stocks), r is the rate of return, j is the emitting source and k is the forest sink.

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[32] This might well be assumed away to allow for bonds with repayment of face value and with a definite maturity date. However, by Eq. (4), for a fixed carbon removal flux (Ψ_k), the closer the maturity date (T_{jk}), the lower the coupon payments (B_j^S) made by the forest bond. In order for the bond price (P_{kj}) to remain unchanged, the rate of interest (r_j) paid by the security must, by Eqs. (1) and (5), go up. Therefore, the shorter (smaller T_{jk}) unnatural forests are supposed to live, the higher the rate of interest paid on their bonds as compared with the too much lower rates on those of longer-lasting natural forests. This would, at the onset, bring on an unequal competition between natural and unnatural forest bonds, heavily favouring the latter (scenario 2, in **Table 6**, shows how high the rate of interest on unnatural forest bonds would be if natural forests were left out of climate finance). Thus, the lengthier the lifespans of forest bonds, the smoother the forest financial market.

As the coupon payment hinges on the sink of kind k and on the rate of interest holding at the emitting source j (r_j),[33] the bond price will vary accordingly and will correspond to the present value of all the future income flows (yearly carbon removal fluxes) delivered by the forest sink k (Ψ_k) [18]. Thus, the bond price comes out of the solution (Eq. (3)) of the integral in Eq. (2), in which the terminal time (lifespan) of sink k at source j (T_{jk}) is given by Eq. (4).

$$P_{kj} = \int_{t=1}^{t=T_{jk}} \Psi_k e^{-r_j t} dt \tag{2}$$

$$P_{kj} = \frac{\Psi_k}{r_j} e^{-r_j} - \frac{\Psi_k}{r_j} e^{-r_j T_{jk}} \tag{3}$$

$$T_{jk} = B_j^S / \Psi_k, \tag{4}$$

where $B^S{}_j$ is the total annual supply of forest bonds by the emitting source j, and Ψ_k is the yearly flux of carbon removal by sink k. Finally, by bringing together Eqs. (1) and (4), the resulting Eq. (5) clearly shows the variables which the bond price hinges on. Eq. (5) also proves that, ideally, $B^S{}_k = B^S{}_j$, thereby implying that the supply of bonds (demand for carbon stocks) assigned to the sink k should meet the issuance of bonds by the emitting source j.

$$P_{kj} = T_{jk} \Psi_k / r_j \tag{5}$$

5.1. Model data

Table 2 presents the empirical data used to estimate the functions of supply and demand of forest bonds. **Table 3** displays the relevant variables and estimates for the bonds demand function, while **Tables 4** and **5** show them for the bonds supply functions.

The function of *demand for forest bonds* (Eq. (6)) is estimated by taking out of **Table 2** the figures of the total removal of CO_2 by forest sinks (7th column) and statistically regressing them on the values of the average real interest rate (4th column). The functions of *supply of natural* (Eqs. (7) and (8)) and *unnatural* (Eqs. (10) and (11)) *forest bonds* are estimated in a similar way, but now the figures of CO_2 emissions by sources (in the third from last and next-to-last columns of **Table 2**) are statistically regressed on the corresponding rates of return on *natural* (**Table 4**) and *unnatural* (**Table 5**) forest bonds. This means that the source's observed emissions ($B^S{}_j$ in **Table 2**) must be associated with its corresponding rates of return on either natural or unnatural forest sinks. It is not possible for a source to remove its own emissions by considering the rates of return rendered by another's forest bonds. Thus, no source can issue forest bonds that biophysically yield the returns of another's.

5.2. Model equations

Based on the data displayed in **Table 2**, the following equations have been estimated by SPSS Statistics 17.0. All of them have proven to be statistically significant at a 5% level (or within a 95% confidence interval).

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[33] For calculus reasons, the value of r_j in Eqs. (2) and (3) must be divided by 100.

Year	Real interest rate[a,b] (r_j)		Average[c] real interest rate ($\bar{r}$)	CO_2 removals by forest sinks[d]		Total CO_2 removal by forest sinks (B^D)	CO_2 emissions by sources[a,e] ($B^S{}_j$)		Total CO2 emissions by sources (B^S)
	% p.a.	% p.a.	% p.a.	$GtCO_2$	$GtCO_2$	$GtCO_2$	$GtCO_2$	$GtCO_2$	$GtCO_2$
	Non-Annex I	**Annex I**		**Non-Annex I**	**Annex I**		**Non-Annex I**	**Annex I**	
1990	5.53	7.39	6.46	1.5630	1.1642	2.7273	6.30	9.74	16.04
1991	7.36	5.40	6.38	1.5747	1.1651	2.7398	6.62	10.69	17.31
1992	9.99	8.70	9.34	1.5784	1.1817	2.7600	7.46	13.79	21.25
1993	9.62	0.27	4.94	1.5886	1.1825	2.7711	7.84	13.62	21.46
1994	5.74	0.09	2.92	1.6008	1.1833	2.7841	8.21	13.36	21.57
1995	4.05	3.74	3.90	1.6134	1.1841	2.7975	8.54	13.42	21.97
1996	9.15	10.88	10.01	1.6263	1.1850	2.8113	8.91	13.68	22.59
1997	12.69	3.07	7.88	1.6397	1.1858	2.8255	9.07	13.57	22.64
1998	17.76	6.30	12.03	1.6535	1.1866	2.8401	9.04	13.56	22.60
1999	10.39	4.39	7.39	1.6678	1.1874	2.8552	9.30	13.68	22.98
2000	7.39	3.86	5.63	1.6825	1.1882	2.8707	9.71	13.95	23.66
2001	12.87	4.47	8.67	0.6905	1.2436	1.9340	9.95	13.90	23.84
2002	11.10	4.82	7.96	0.6898	1.2445	1.9344	10.28	13.93	24.21
2003	10.27	3.37	6.82	0.6892	1.2455	1.9347	11.37	14.20	25.57
2004	7.08	2.85	4.96	0.6888	1.2464	1.9352	12.54	14.32	26.86
2005	5.64	2.08	3.86	0.6883	1.2474	1.9357	13.25	14.35	27.60
2006	5.61	2.22	3.92	0.7917	1.7635	2.5552	14.28	14.34	28.61
2007	6.84	1.75	4.30	0.7892	1.7656	2.5548	14.95	14.43	29.38
2008	3.61	1.45	2.53	0.7867	1.7677	2.5544	15.74	14.17	29.91
2009	13.61	6.83	10.22	0.7842	1.7698	2.5539	16.65	13.18	29.83
2010	5.12	3.91	4.52	0.7815	1.7719	2.5534	17.55	13.69	31.23
2011	3.99	0.23	2.11	0.5798	1.2732	1.8530	18.79	13.52	32.31
2012	7.10	2.16	4.63	0.5772	1.2735	1.8507			
2013	8.76	4.12	6.44	0.5746	1.2737	1.8483			
2014	9.50	2.78	6.14	0.5719	1.2740	1.8459			
2015	12.06	2.87	7.46	0.5692	1.2743	1.8434			

Sources: Refs. [47, 48].

[a] Ref. [48].

[b] Geometric mean of all countries' real rate of interest.

[c] Arithmetic mean between Non-Annex I's and Annex I's real interest rates (r_j).

[d] Ref. [47].

[e] Until 2011 only.

Table 2. Observed forest and financial values.

Year	Average real interest rate[a] ($\bar{r}$)	Total CO_2 removal by forest sinks[a] (B^D)	Estimated CO_2 removal by forest sinks (demand for forest bonds) Eq. (6)
	% p.a.	$GtCO_2$	$GtCO_2$
1990	6.46	2.7273	2.6324
1991	6.38	2.7398	2.6219
1992	9.34	2.7600	2.6312
1993	4.94	2.7711	2.3411
1994	2.92	2.7841	1.6398
1995	3.90	2.7975	2.0233
1996	10.01	2.8113	2.5269
1997	7.88	2.8255	2.7223
1998	12.03	2.8401	1.9766
1999	7.39	2.8552	2.7110
2000	5.63	2.8707	2.4971
2001	8.67	1.9340	2.6965
2002	7.96	1.9344	2.7221
2003	6.82	1.9347	2.6715
2004	4.96	1.9352	2.3465
2005	3.86	1.9357	2.0104
2006	3.92	2.5552	2.0305
2007	4.30	2.5548	2.1564
2008	2.53	2.5544	1.4640
2009	10.22	2.5539	2.4868
2010	4.52	2.5534	2.2235
2011	2.11	1.8530	1.2618
2012	4.63	1.8507	2.2554
2013	6.44	1.8483	2.6294
2014	6.14	1.8459	2.5876
2015	7.46	1.8434	2.7140

[a] Obtained from **Table 2**.

Table 3. Demand for forest bonds.

a. Demand for forest bonds:

$$B^D = -0.04365r^2 + 0.6894r \tag{6}$$

b. Supply of *natural* forest bonds ($k = n$) by the j emitting sources ($j = 0$ = Non-Annex I countries; $j = 1$ = Annex I countries)

Year[a]	Forest lifespan (T_{jk})[b] Eq. (4)		Forest bond price (P_{kj})[b] Eq. (3)		Rate of return on natural forest bonds (r_j)[g] Eq. (1)		Estimated supply of natural forest bonds ($\hat{B}_j^{S,n}$)	
	years	years	$GtCO_2$/%	$GtCO_2$/%	% p.a.	% p.a.	$GtCO_2$	$GtCO_2$
	Non-Annex I[c]	Annex I[d]	Non-Annex I[e]	Annex I[f]	Non-Annex I	Annex I	Non-Annex I	Annex I
							Eq. (7)	Eq. (8)
1990	8.80	13.61	4.29	5.45	1.47	1.79	9.38	14.36
1991	9.25	14.94	4.11	6.64	1.61	1.61	9.99	14.16
1992	10.42	19.28	3.95	6.01	1.89	2.30	11.03	13.44
1993	10.96	19.03	4.17	12.56	1.88	1.08	11.02	11.98
1994	11.47	18.67	5.32	12.53	1.54	1.07	9.71	11.86
1995	11.94	18.76	6.07	8.94	1.41	1.50	9.10	13.90
1996	12.45	19.11	4.64	5.08	1.92	2.69	11.16	11.19
1997	12.68	18.96	3.84	9.58	2.36	1.42	12.37	13.63
1998	12.64	18.94	2.95	7.22	3.07	1.88	13.28	14.36
1999	13.00	19.11	4.42	8.56	2.10	1.60	11.71	14.14
2000	13.57	19.50	5.44	9.10	1.78	1.53	10.67	13.99
2001	13.90	19.42	3.96	8.59	2.51	1.62	12.67	14.17
2002	14.36	19.47	4.46	8.34	2.30	1.67	12.24	14.26
2003	15.89	19.84	4.93	9.65	2.31	1.47	12.25	13.82
2004	17.52	20.01	6.49	10.21	1.93	1.40	11.18	13.58
2005	18.52	20.05	7.53	11.02	1.76	1.30	10.58	13.17
2006	19.95	20.03	7.89	10.87	1.81	1.32	10.76	13.25
2007	20.89	20.16	7.26	11.45	2.06	1.26	11.58	12.97
2008	21.99	19.80	10.16	11.61	1.55	1.22	9.73	12.77
2009	23.27	18.42	4.37	6.81	3.81	1.94	12.85	14.32
2010	24.52	19.13	9.29	8.94	1.89	1.53	11.04	13.99
2011	26.26	18.89	10.94	12.51	1.72	1.08	10.42	11.95

[a] Until 2011 only, because emissions data are not available beyond (see **Table 2**).
[b] $\Psi_{k=n}$ = 195 MtC.yr^{-1} = 0.195 GtC.yr^{-1} = 0.7157 $GtCO_2$.yr^{-1} [49].
[c] Using Annex I's emissions informed in **Table 2**.
[d] Using Non-Annex I's real rates of interest informed in **Table 2**.
[e] Using Annex I's real rates of interest informed in **Table 2**.
[f] Using Annex I's real rates of interest informed in **Table 2**.
[g] In which the numerator ($B^S_k = B^S_j$) corresponds to the CO_2 emissions by sources *j*, informed in **Table 2**.

Table 4. Supply of *natural* forest bonds ($k = n$).

Year[a]	Forest lifespan (T_{jk})[b] Eq. (4)		Forest bond price (P_{kj})[b] Eq. (3)		Rate of return on unnatural forest bonds (r_j)[g] Eq. (1)		Estimated supply of unnatural forest bonds ($\hat{B}_j^{S,n}$)	
	Years	Years	$GtCO_2$/%	$GtCO_2$/%	% p.a.	% p.a.	$GtCO_2$	$GtCO_2$
	Non-Annex I[c]	Annex I[d]	Non-Annex I[e]	Annex I[f]	Non-Annex I[h]	Annex I	Non-Annex I[h]	Annex I
							Eq. (10)	Eq. (11)
1990	0.86	1.33	−0.99	2.21	−6.35	4.42	5.27	9.56
1991	0.90	1.46	−0.67	3.14	−9.85	3.41	5.79	11.20
1992	1.02	1.88	0.11	5.70	70.28	2.42	7.05	13.36
1993	1.07	1.86	0.46	6.26	17.22	2.18	7.59	14.03
1994	1.12	1.82	0.82	6.01	10.06	2.22	8.15	13.90
1995	1.16	1.83	1.15	5.77	7.42	2.33	8.65	13.61
1996	1.21	1.86	1.42	5.42	6.27	2.52	9.02	13.10
1997	1.24	1.85	1.50	5.96	6.04	2.28	9.12	13.75
1998	1.23	1.85	1.40	5.68	6.47	2.39	8.95	13.46
1999	1.27	1.86	1.74	5.95	5.34	2.30	9.46	13.69
2000	1.32	1.90	2.17	6.25	4.47	2.23	10.07	13.88
2001	1.36	1.89	2.24	6.15	4.44	2.26	10.09	13.79
2002	1.40	1.90	2.57	6.15	4.00	2.27	10.52	13.78
2003	1.55	1.93	3.54	6.53	3.21	2.18	11.67	14.04
2004	1.71	1.95	4.73	6.69	2.65	2.14	13.05	14.14
2005	1.81	1.95	5.46	6.80	2.43	2.11	13.86	14.23
2006	1.94	1.95	6.39	6.77	2.24	2.12	14.71	14.21
2007	2.04	1.97	6.86	6.91	2.18	2.09	15.01	14.29
2008	2.14	1.93	7.94	6.69	1.98	2.12	16.21	14.20
2009	2.27	1.80	7.47	5.31	2.23	2.48	14.74	13.20
2010	2.39	1.86	9.36	6.00	1.87	2.28	17.03	13.74
2011	2.56	1.84	10.67	6.16	1.76	2.20	18.05	13.98

[a] Until 2011 only, because emissions data are not available beyond (see **Table 2**).

[b] $\Psi_{k=u} = 2\ GtC.yr^{-1} = 7.34\ GtCO_2.yr^{-1}$ [1].

[c] Using Non-Annex I's emissions informed in **Table 2**.

[d] Using Annex I's emissions informed in **Table 2**.

[e] Using Non-Annex I's real rates of interest informed in **Table 2**.

[f] Using Annex I's real rates of interest informed in **Table 2**.

[g] In which the numerator ($B^S_k = B^S_j$) corresponds to the CO_2 emissions by sources j, informed in **Table 2**.

[h] Negative figures have been left out for the estimation of Eq. (10).

Table 5. Supply of *unnatural* forest bonds ($k = u$).

$$B_0^{S,n} = -1.287r^2 + 8.277r \tag{7}$$

$$B_1^{S,n} = -4.284r^2 + 15.693r \tag{8}$$

$$B_n^S = B_0^{S,n} + B_1^{S,n} \tag{9}$$

c. Supply of *unnatural* forest bonds ($k = u$) by the j emitting sources ($j = 0$ = Non-Annex I countries; $j = 1$ = Annex I countries)

$$B_0^{S,u} = \exp\left(1.9286 + \frac{1.6994}{r}\right) \tag{10}$$

$$B_1^{S,u} = 18.956 - 6.3272\ln r \tag{11}$$

$$B_u^S = B_0^{S,u} + B_1^{S,u} \tag{12}$$

d. Supply of forest bonds of *all* k kinds ($k = n$ = natural forests; $k = u$ = unnatural forests)

$$B^S = B_n^S + B_u^S \tag{13}$$

e. Supply of forest bonds at the emitting source j ($j = 0$ = Non-Annex I countries; $j = 1$ = Annex I countries)

$$B_0^S = B_0^{S,n} + B_0^{S,u} \tag{14}$$

$$B_1^S = B_1^{S,n} + B_1^{S,u} \tag{15}$$

f. Supply of forest bonds by *all* j emitting sources ($j = 0$ = Non-Annex I countries; $j = 1$ = Annex I countries)

$$B^S = B_0^S + B_1^S \tag{16}$$

g. Objective-function

$$\max Z = B^D - B^S \tag{17}$$

h. Optimisation scenarios

1. $B^D \leq B_n^S$: emissions removal takes place in *natural* forests only;
2. $B^D \leq B_u^S$: emissions removal takes place in *unnatural* forests only;
3. $B^D \leq (B_n^S + B_u^S)$: emissions removal takes place in *both* kinds of forest sinks;
4. $B^D \leq B_0^S$: emissions removal is sought by *Non-Annex I* countries only;

5. $B^D \leq B_1^S$: emissions removal is sought by *Annex I* countries only;
6. $B^D \leq (B_0^S + B_1^S)$: emissions removal is sought by *both* Non-Annex I and Annex I countries.

The condition $B^D \leq B^S{}_{k,j}$ in all optimisation scenarios is required because, in reality, the supply of carbon removal stocks (demand for forest bonds) is by far smaller than the quantity demanded (supply of forest bonds) by the economic activities.

5.3. Model results and discussion

The results yielded by the LFF model sound consistent with the LF theory of the rate of interest. Because the rate of interest is considered a reward for saving (i.e., demanding bonds), there should be a *positive* relationship between the rate of interest and the demand for forest bonds. **Figure 4** does confirm this hypothesis, although there is a maximum value ($dB^D/dr = 0$) for the rate of interest ($r = 7.89\%$ per year), beyond which savings of carbon stocks (demand for forest bonds) will decrease until get vanished ($B^D = 0$, $r = 15.77\%$ per year). Certainly, this owes to the biophysical limits to *carbon* money, as opposed to losing or missing limits to *paper* (fiat) money.

Somewhat paradoxically, **Figure 5** shows that the expected *negative* relationship between investments (supply of bonds) and the rate of interest does not holdfully true for natural forests, although it does, as shown by **Figure 6** for unnatural forests. Again, natural forests are more heavily affected by ecological constraints and irreversibilities than unnatural forests. Therefore, the search for financing for carbon emitting investments through issuance of forest bonds cannot exceed certain biophysically established limits to conservation or offsetting of carbon stocks.

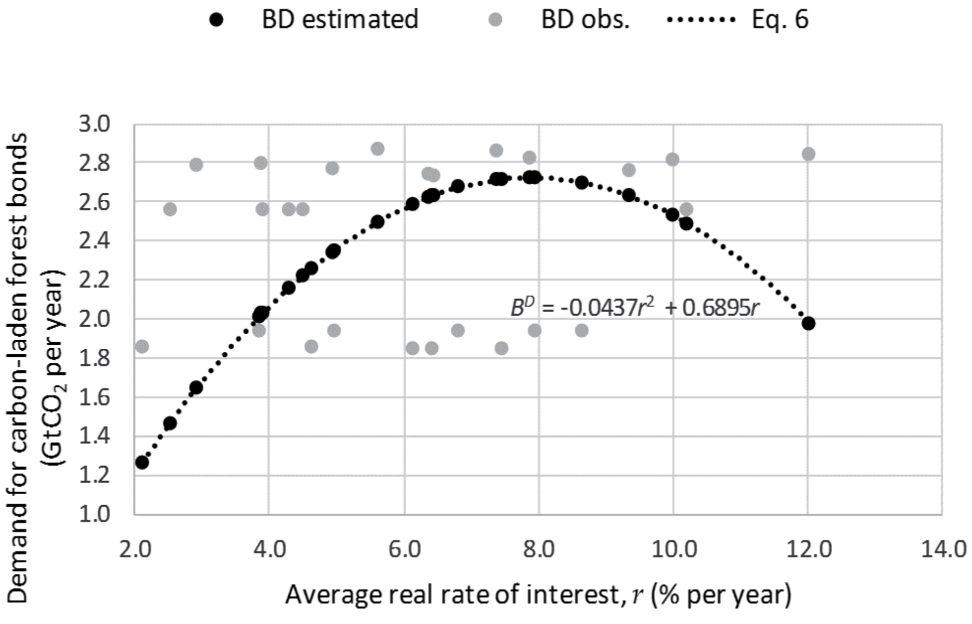

Figure 4. Demand for forest bonds.

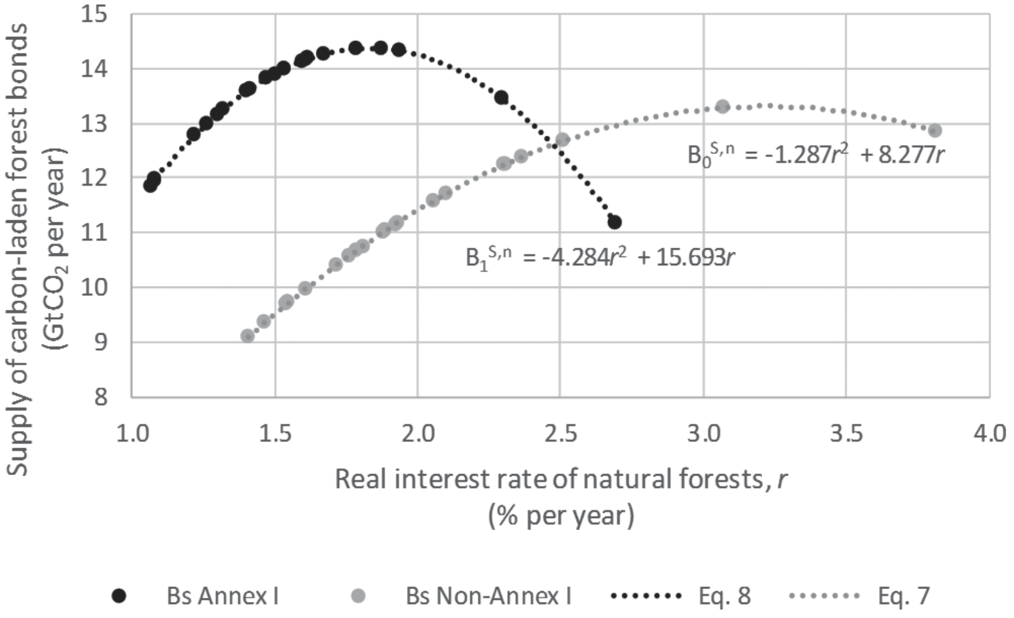

Figure 5. Supply of *natural* forest bonds by Non-Annex I and Annex I emission sources.

In **Figure 5**, the supply of *natural* forest bonds unexpectedly increases with the rate of interest as far as it reaches a maximum of 3.22% per year in Non-Annex I countries ($dB_0^{S,n}/dr = 0$) and 1.83% per year in Annex I countries ($dB_1^{S,n}/dr = 0$). From there on, it then behaves as expected, just like it does with respect to *unnatural* forest bonds. As **Figure 6** shows, the supply of unnatural forest bonds goes down as the rate of interest goes up. Yet, in either natural or unnatural forest stands, the supply of forest bonds in Non-Annex I countries changes more *slowly* with (is more *inelastic* to) the interest rates, whereas it does it *faster* (more *elastic*) in Annex I countries.

Next, a scenario analysis, carried on in **Table 6**, finds the optimal rate of interest on the world's market for loanable forest funds that meets, for every scenario described in Section 5.2, the objective-function established by Eq. (17). The results from **Table 6** allow for the estimation of Eqs. (18) and (19), depicted in **Figure 7**, which not only set down the finance boundaries for a forest bond market but also show the optimal path of the real rate of interest on forest bonds in the long run.

$$B^{D^*} = 1.3293 + 0.1982r^* \tag{18}$$

$$B^{S^*} = 154.35 - 31.4842r^* \tag{19}$$

From **Table 6**, Eqs. (18) and (19), the demand for forest bonds equals the supply when $r^* = 4.90\%$ per year. This amounts, in **Table 6**, to scenarios 3 and 6, which, as expected from **Table 1**, had to actually yield the same results. Next, by setting Eqs. (18) and (19) equal to zero, the optimal range for r^* is found to be $-6.707 < r^* < 4.934$, within which a financial market for forest bonds might really come to existence. However, negative values for r^* mean a supply of bonds so larger than

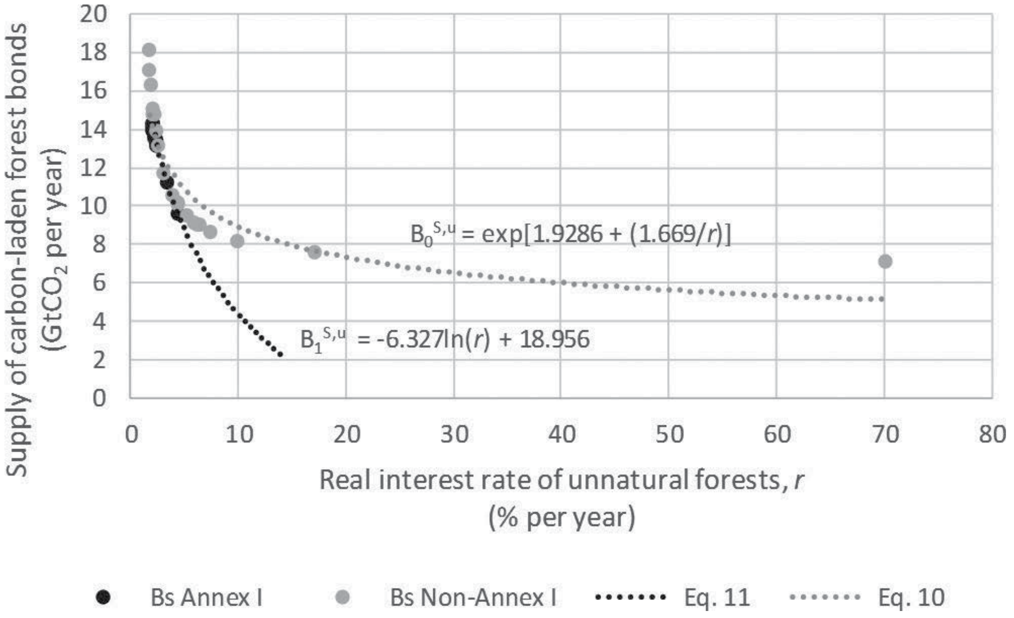

Figure 6. Supply of *unnatural* forest bonds by Non-Annex I and Annex I emission sources.

	B_k^{S*}		B_j^{S*}				
	Natural forests ($k = n$)	**Unnatural forests ($k = u$)**	**Non-Annex I ($j = 0$)**	**Annex I ($j = 1$)**	B^{S*}	B^{D*}	r^*
	Eq. (9)	**Eq. (12)**	**Eq. (14)**	**Eq. (15)**	**Eq. (13)** **Eq. (16)**	**Eq. (6)**	
Scenarios	**GtCO$_2$ yr^{-1}**	**GtCO$_2$ yr^{-1}**	**GtCO$_2$ yr^{-1}**	**GtCO$_2$ yr^{-1}**	**GtCO$_2$ yr^{-1}**	**GtCO$_2$ yr^{-1}**	**% per annum**
5	4.360	20.410	22.673	2.097	24.770	2.097	4.112
1	0.959	20.033	22.148	-1.156	20.993	2.145	4.262
3	-16.303	18.633	19.389	-17.059	2.330	2.330	4.900
6	-16.303	18.633	19.389	-17.059	2.330	2.330	4.900
4	-110.302	15.302	2.694	-97.694	-95.000	2.694	7.094
2	-1×10^{10}	-41.575	-2.31×10^{9}	-7.69×10^{9}	-1×10^{10}	-7.83×10^{7}	42369.708

Notes: (a) Calculations performed in GAMS-IDE version 24.7.1 (March 2016). (b) Negative figures mean carbon emissions, whereas positive values mean carbon removals.

Table 6. Scenarios and optimal rates of interest on forest bonds (r^*)[a,b].

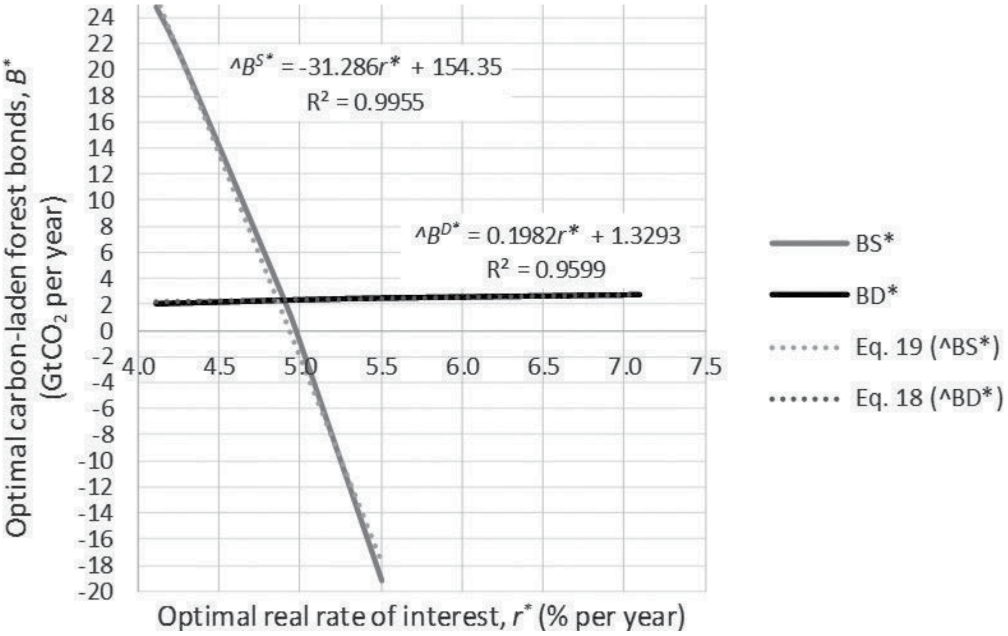

Figure 7. Optimal real rate of interest on long-term forest bonds (forest perpetuities).

the demand that the costs to remove emissions would outstrip the benefits, thereby yielding negative returns.

As long as the value found for r^* falls outside this optimal range, some scenarios (like scenarios 2 and 4 of **Table 6**) are likely to keep any market for forest bonds from thriving. Whereas scenario 2 encloses a *biophysical* restriction, so that emissions removal is assigned to *unnatural* forests only, scenario 4 takes on a *geographical* restriction, requiring that only Non-Annex I countries seek for emissions reduction by forest sinks. Clearly, none of these restrictions favours forest finance.

In other scenarios (1 and 5), although biophysical and geographical restrictions do not impair forest finance, they lower the rate of interest on forest bonds as compared with that of unrestricted scenarios (3 and 6). The lowest rate of return (r^* = 4.112% per year) occurs in scenario 5, whose *geographical* restriction allows for emissions reduction by only Annex I countries. A lower rate of return (r^* = 4.262% per year) also comes out of the *biophysical* restriction of scenario 1, in which only natural forests are committed to emissions reduction.

6. Conclusion

In light of the results of the LFF model, displayed in **Table 6**, the Kyoto Protocol resembles scenarios 2 and 5, in which forestry-CDM allowed for *unnatural* forests only to take part in emissions removal, required from Annex I binding countries. As shown in **Table 6**, the biophysical restriction of scenario 2 is more stringent to forest finance than the geographical restriction of scenario 5. They respectively yield the unrealistically highest (42,369.708% per year) and the lowest (4.112% per year) rate of interest on forest bonds. In between, the Paris Agreement has formally called in Non-Annex I countries and natural forests (REDD+) to assist in reducing emissions. When neither biophysical nor geographical restrictions are in place, the LFF results (scenarios 3 and 6 in **Table 6**) show that demand and supply of forest bonds would even off (2.33 GtCO2 per year), and the rate of interest on them would lie between the extremes yielded by scenarios 2 and 5. Although this would favour carbon finance, natural forests would behave as carbon sources (emitting BS(k = n) = 16.303 GtCO2 per year) rather than sinks. Nonetheless, if too a heavy burden is placed upon REDD+ and upon the carbon sink role of natural forests (scenario 1), this biophysical restriction would be less stringent to a forest bond market than the geographical restriction under which Non-Annex I countries would solely commit themselves to emissions reduction (scenario 4). As shown in section 5.3, scenario 4 yields a rate of interest on forest bonds (r^*= 7.094% per year) that exceeds the acceptable upper bound (r^*= 4.934% per year). This is, though, a likely scenario, provided that Annex I countries, as argued in Section 2, withdraw the Paris Agreement (see Footnote 13).

Moreover, in all scenarios, unnatural forests are required more emissions removal than natural forests ($B^S{}_u > B^S{}_n$). Non-Annex I countries issue more forest bonds—that is, demand more carbon removal stocks—than Annex I countries do ($B^S{}_0 > B^S{}_1$). All this suggests that, since deforestation is high in Non-Annex I countries, they are driven to offset carbon emissions by demanding carbon removal stocks from unnatural forests, whereupon BS(u) > BS(n) in **Table 6**. Therefore, according to the LFF model outcomes, the efforts of the Paris Agreement towards forest conservation (REDD +) point to scenarios 1 and 5, in which emission reductions by natural forests are positive ($B^S{}_n > 0$) and the rates of interest on forest bonds are lower (respectively, r^*= 4.262 and r^*= 4.112% per year). Higher rates of return/interest mean less forest conservation and more forest offset (plantations).

Although this outcome appears to disagree with the static one in **Figure 1**, it dynamically means that, in a carbon-storing economy, conservation amounts to an excess supply of carbon money (forestland), whereas deforestation corresponds to an excess demand for carbon stocks. Whenever deforestation outstrips conservation, the rate of interest is supposed to go up, because the actual supply of forestland (carbon money) is not enough to meet the demand for commodities (carbon stocks) throughout the economy [51]. Then, the consumption of carbon stocks is currently discouraged, thereby increasing the demand for forest bonds (upcoming supply of forest stocks). Conversely, when conservation is expected to outbalance deforestation, the rate of interest is supposed to go down. Since there is too much forestland (carbon money) in relation to the demand for carbon stocks, the supply of forest stocks is currently withheld, thereby increasing the supply of forest bonds (upcoming demand for forest stocks).

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References

[1] Dixon RK, Brown S, Houghton RA, Solomon AM, Trexler MC, Wisniewski J. Carbon pools and flux of global forest ecosystems. Science. 1994;**263**(5144):185–190. doi:10.1126/science.263.5144.185.

[2] Food and Agriculture Organisation of the United Nations (FAO). The role of forests in sequestering and storing carbon. In: Food and Agriculture Organisation of the United Nations (FAO), editors. Impact of the global forest industry on atmospheric greenhouse gases. FAO Forest Paper No. 159. Rome: FAO; 2010, pp. 7–12.

[3] Wikipedia. United Nations Climate Change Conference [Internet]. (Updated: 2016, September 23). https://en.wikipedia.org/wiki/United_Nations_Climate_Change_conference (accessed on September 26, 2016).

[4] Giacomelli Sobrinho V. Carbon bio-economics and forests: getting the BESF out of climate policy. In: Blanco J, Kheradmand H, editors. Climate change: socioeconomic effects, 1st ed. Rijeka: InTech; 2011, pp. 401–424. doi:10.5772/25056.

[5] Giacomelli Sobrinho V. REDD roses for a green lady: target setting for deforestation in the Brazilian Amazon. In: Morgado CRV, Esteves VPP, editors. CO_2 sequestration and valorization, 1st ed. Rijeka: InTech; 2014, pp. 1–33. doi:10.5772/57288.

[6] Martin RM. Deforestation, land-use change and REDD. Unasylva. 2008;**59**(230):3–11.

[7] Clark CW. Mathematical bioeconomics: the mathematics of conservation, 3rd ed. Hoboken: Wiley; 2010, 368 p.

[8] Napoleoni C. Curso de Economia Política, 4th ed. Rio de Janeiro: Graal; 1988, 405 p.

[9] The World Bank. Carbon finance for sustainable development: 2015 annual report, 1st ed. Washington, DC: The World Bank; 2015, 23 p.

[10] United Nations Framework Convention on Climate Change. Paris Agreement. In: FCCC, editor. Conference of the Parties Twenty-First Session (CoP-21); 30 November–11 December 2015; Paris, France. Paris: United Nations; 2015, pp. 1–32.

[11] Green Climate Fund (GCF). Status of pledges and contributions made to the Green Climate Fund [Internet]. 2016, October 17. http://www.greenclimate.fund/documents/20182/24868/Status_of_Pledges.pdf/eef538d3-2987-4659-8c7c-5566ed6afd19 (accessed on October 27, 2016).

[12] Green Climate Fund (GCF). Contributors: resources mobilized [Internet]. http://www.greenclimate.fund/partners/contributors/resources-mobilized (accessed on October 28, 2016).

[13] UNFCCC Standing Committee on Finance. 2014 biennial assessment and overview of climate finance flows report, 1st ed. Bonn: United Nations Framework Convention on Climate Change (UNFCCC); 2014, 94 p.

[14] Bloomberg New Energy Finance. Green bonds market outlook 2014: blooming with new varietals, 1st ed. New York: Bloomberg Finance L. P.; 2014, 22 p.

[15] The World Bank, Public-Private Infrastructure Advisory Facility (PPIAF). What are green bonds?, 1st ed. Washington, DC: The World Bank; 2015, 58 p.

[16] Climate Bonds Initiative. Climate bond certified: standard [Internet]. https://www.climatebonds.net/standards/about (accessed on October 28, 2016).

[17] Climate Bonds Initiative. Bonds and climate change: the state of the market in 2016, 1st ed. London: HSBC Climate Change Centre of Excellence; 2016, 20 p.

[18] Fisher I. A teoria do juro: determinada pela impaciência por gastar renda e pela oportunidade de investi-la, 1st ed. São Paulo: Abril Cultural; 1984, 356 p.

[19] Food and Agriculture Organisation of the United Nations. FAOSTAT [Internet]. 2015. http://faostat3.fao.org/download/G2/GF/E (accessed on August 1, 2016).

[20] Federici S, Tubiello FN, Salvatore M, Jacobs H, Schmidhuber J. New estimates of CO_2 forest emissions and removals: 1990–2015. Forest Ecology and Management. 2015;**352**:89–98. doi:10.1016/j.foreco.2015.04.22.

[21] Velde FR, Weber WE. Commodity money. In: Durlauf SN, Blume LE, editors. Monetary economics, 1st ed. New York: Palgrave MacMillan; 2010, pp. 27–33.

[22] Wallace N. Fiat money. In: Durlauf SN, Blume LE, editors. Monetary economics, 1st ed. New York: Palgrave MacMillan; 2010, pp. 66–75.

[23] United Nations Framework Convention on Climate Change (UNFCCC). Parties & observers [Internet]. 2014. http://unfccc.int/parties_and_observers/items/2704.php (accessed on October 30, 2016).

[24] Conard JW. An introduction to the theory of interest, 1st ed. Berkeley and Los Angeles: University of California Press; 1963, 375 p.

[25] International Institute for Sustainable Development (ISSD). Summary of the Durban Climate Change Conference: 28 November–11 December 2011. Earth Negotiations Bulletin. 2011;**12**(534):1–34.

[26] Wikipedia. Paris agreement [Internet]. (Updated: 2016, October 25). https://en.wikipedia.org/wiki/Paris_Agreement (accessed on October 25, 2016).

[27] Valente FSPF. A relação entre investimento, poupança e taxa de juros: um panorama do debate sobre financiamento de longo prazo. Leituras de Economia Política. January 2006–December 2007;**9**(12):83–99.

[28] Tsiang SC. Keynes's "finance" demand for liquidity, Robertson's loanable funds theory, and Friedman's monetarism. The Quarterly Journal of Economics. 1980;**94**(3):467–491.

[29] Keynes JM. Alternative theories of the rate of interest. The Economic Journal. 1937;**47** (186):241–252.

[30] Wicksell K. Interest and prices: a study of the causes regulating the value of money, 1st ed. New York: Sentry Press; 1936, 239 p.

[31] Hunt EK. Teorias neoclássicas da firma e da distribuição de renda: as obras de Marshall, Clark e Böhm-Bawerk. In: Hunt EK, editor. História do pensamento econômico: uma perspectiva crítica, 7th ed. Rio de Janeiro: Campus; 1989, pp. 315–344.

[32] Howells P, Bain K. Economia monetária: moeda e bancos, 1st ed. Rio de Janeiro: LTC; 2001, 319 p.

[33] Keynes JM. A teoria geral do emprego, do juro e da moeda, 1st ed. São Paulo: Atlas; 1992, 328 p.

[34] Wicksell K. Lectures on political economy: money, 1st ed. Fairfield: Augustus M. Kelley; 1978, 238 p.

[35] Ohlin B. Some notes on the Stockholm theory of savings and investments II. The Economic Journal. 1937;**47**(186):221–240.

[36] Wicksell K. The influence of the rate of interest on prices. The Economic Journal. 1907;**17** (66):213–220.

[37] Hicks JR. Valor e capital: estudo sobre alguns princípios fundamentais da teoria econômica, 1st ed. São Paulo: Abril Cultural; 1984, 276 p.

[38] Ohlin B. Some notes on the Stockholm theory of savings and investment I. The Economic Journal. 1937;**47**(185):53–69.

[39] Ohlin B, Robertson DH, Hawtrey RG. Alternative theories of the rate of interest: three rejoinders. The Economic Journal. 1937;**47**(187):423–443.

[40] Rose H. Liquidity preference and loanable funds. The Review of Economic Studies. 1957;**24**(2):111–119.

[41] Läufer NKA. Liquidity preference and loanable funds: the theory of neutral revision behaviour. Zeitschrift für Nationalökonomie. 1970;**30**:291–308.

[42] Ball LM. Money, banking, and financial markets, 2nd ed. New York: Worth; 2012, 608 p.

[43] Martin F. Dinheiro: uma biografia não autorizada – da cunhagem à criptomoeda, 1st ed. São Paulo: Portfolio-Penguin; 2016, 392 p.

[44] Mishkin FS. The economics of money, banking, and financial markets, 7th ed. New York: Addison-Wesley; 2004, 679 p.

[45] Officer LH. Gold standard. In: Durlauf SN, Blume LE, editors. Monetary economics, 1st ed. New York: Palgrave Macmillan; 2010, pp. 96–107.

[46] Bordo MD. The concise encyclopedia of economics: gold standard [Internet]. 2008 (Updated: 2008). http://www.econlib.org/library/Enc/GoldStandard.html (accessed on November 4, 2016).

[47] Food and Agriculture Organisation of the United Nations (FAO). FAOSTAT: emissions – land use [Internet]. 2015. http://faostat3.fao.org/download/G2/GF/E (accessed on August 2, 2016).

[48] The World Bank. World data bank: world development indicators [Internet]. 2016. http://databank.worldbank.org/data/reports.aspx?source=world-development-indicators (accessed on October 6, 2016).

[49] Food and Agriculture Organisation of the United Nations (FAO). Global forest resources assessment 2015: how are the world's forests changing?, 2nd ed. Rome: FAO; 2016, 45 p.

[50] Thinking Progress. The biggest source of mistakes: C vs. CO_2 [Internet]. 2008, March 25. https://thinkprogress.org/thebiggestsourceofmistakescvsco2c0b077313b (accessed on August 5, 2016).

[51] Lutz FA. Liquidity preference theory vs. theory of loanable funds. In: Lutz FA, editor. The theory of interest, 2nd ed. New Brunswick, US, and London, UK: Aldine Transaction; 2010. pp. 165–185.

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This book is dedicated to forest ecology and conservation on ecological and conservation aspects of forest. The book is divided into two sections: the first section "Forest Ecology" with four chapters deals with forest ecological aspects, while the second section "Forest Conservation" with two chapters looks into new techniques for conserving the forests. This book will bridge the gaps in the knowledge about some new emerging issues on forest ecology and conservation. It will be an interesting and helpful resource to all those in the field of forestry working for its sustainable use and conservation.

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