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New Perspectives in Forest Science

Edited by Helder Filipe dos Santos Viana
and Francisco Antonio García Morote

NEW PERSPECTIVES IN FOREST SCIENCE

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Meet the editors

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Francisco Antonio García Morote holds a BSc degree in Forestry Engineering (1996), an MSc degree in Agrarian Engineering (2006) and a PhD degree in Agricultural Science and Engineering (2012). He is a professor in the Higher Technical School of Agricultural and Forestry Engineering (University of Castilla-La Mancha (UCLM), Spain) and a researcher in the Renewable Energy Research Institute (IER; Environment and Forest Resources Research Group, Spain). Actually, he is the secretary of Agroforestry Science and Technology Department (UCLM). He is a lecturer (since 2001, UCLM) and teaches forest inventory, sustainable forest management, soil conservation and hydrologic planning. His current scientific activity focuses on (i) adaptive forest management (forest response to silvicultural treatments under climate change and severe drought), (ii) productivity and C cycle in Mediterranean forests and (iii) regeneration of Mediterranean forest species. He has been a research leader in 15 research projects related to forest management and a researcher in other 14 competitive projects. He is an author and a co-author of 23 papers in journals with impact factor (JCR), more than 60 conference proceedings, 15 ponences in Forestry Congress, 4 books and 13 book chapters, and he collaborated as a reviewer with 5 prestigious scientific journals (forestry and soil sciences). His international publications have received 247 citations.

Contents

Preface

Sustainable forest management can increase the benefits of forest, in a way that conserves and maintains forest ecosystems for the benefit of present and future generations. However, many of the world's forests and woodlands are still not managed sustainably. This is not only due to a lack of appropriate forest policies but also due to a lack of technical capacity and knowledge in forestry. The book *New Perspectives in Forest Science* can serve as a source of information on actual trends of forest research, forest management and forest landscape. The book provides diverse information about different types of forest management at the worldwide level, recompilation of basic concepts and novel case studies related to forest research, which can serve to promote the implementation of a sustainable forest management due to greater knowledge of forest science basis. Thus, the chapters included in the book are intended for forest students, forest managers and researchers wishing to increase their knowledge about some issues that are fundamental to enable monitoring and control of forest management, C cycle, biomass, forest soils, forest restoration, forest economic value, forest resources, forest landscape, forest models and biodiversity under the scenario of climate change. Due to this wide variety of topics, the book has been divided into three sections.

The first section, "C Cycle and Biomass", deals with the study of the forest as a sink of C. Forests represent the largest terrestrial sink of emissions of carbon dioxide (CO_2) and thus are globally the more significant C stocks. In this section, effective methods for the collection, processing, analysis and mapping data regarding C stocks and biomass are analysed. In this sense, this section could serve as a good reference to understand the major aspects related to the study of soil C and forest soils. The section includes the study of C cycle from the latest available information, and in the context of climate change, including a revision of predictive models in the function of multiple variables and projections to estimate changes in the land use, CO_2 emissions and biomass maps.

The second section "Forest Management and Restoration" presents a decision support tool (in a GIS) designed for management decisions in the context of climate change by capturing spatial information. This interactive tool integrates empirical models of the current and future forest structure and function in a structured decision framework, to customize multiple management objectives and visualize suitability outcomes across the landscape. In this section, the reader can also review the basic equations for determination of the rotation lengths under Wicksell and Faustmann models, focusing on the impact of timber price variations on rotation length and planting decisions. It is very interesting to know how it is possible to apply the models in two alternative tree species with different stochastic timber prices, with non-stochastic rising timber prices and with stochastic timber prices. Following the analysis of the economic value of forests, this section presents a completed review of scientific terminus related to forest valuation. Static and empirical methods are well discussed, indicating the role of scientists in the development of forest economics in terms of forest value. Finally, this section focusses on the analysis of diverse techniques applied to forest restoration. The potential for direct seeding and nucleation techniques as an approach to restoring stands analysing various environments and techniques for seedling/seed placement is remarkable.

Finally, the third section "Forest Resources and Biodiversity" presents a more detailed revision referred to human-nature interactions in different degrees of naturalness. The revision quantifies the services provided by urban forests and human-forest interactions in four selected study areas, and it is possible to apply these concepts in other areas. Later, this section reviews the scientific literature that integrates climate change and forest landscape. Diverse studies and predictive models in this context are well evaluated and discussed. On the other hand, it is well known that forests contain most of the world's terrestrial biodiversity. In this sense, it is necessary to integrate in the sustainable management the rest of living beings of the ecosystem, in addition to the plants and trees. In fact, the composition and the structure of vegetation influence the abundance and the distribution of birds. For this, the section finishes with a novel study relating to bird species richness and abundance, with vegetation structure (with classical vegetation variables) at different successional stages. In the context of sustainable forest management, it is necessary to address studies of how changes in vegetation affect animal biodiversity, and this can be done by modelling bird community attributes.

In short, we think that this book presents a wide variety of thematic needed to increase our knowledge in forest science and can be of great help to students, researchers or managers who are immersed in sustainable forest management.

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Section 1

C Cycle and Biomass

Chapter 1

Modeling Deforestation and CO_2 Emissions in Tropical Forests (Western South Amazon)

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Abstract

Spatial modeling is a tool to represent deforestation and predict future scenarios according to different landscape change. Establishing 80% Legal Reserve Area (LR) in the Amazon since 90th, the Brazilian forestry code has made clear the biodiversity conservation profile of the largest tropical forest in the world. However, this mechanism did not prevent the advance of deforestation, which in recent years has increased again. This remote tool aims to monitor the deforestation, simulating its possible future trajectories, and thus generate information that can be used to assist in the management of deforestation reduction. The spatial modeling in the prediction of different deforestation scenarios based on public policies and their changes to the state of Acre (north of Brazil). Using the methodological processes of the Dinamica EGO software, three scenarios were projected up to the year 2050: (1) deforestation "Business as usual", (2) deforestation with 50% LR and (3) deforestation with 80% LR provided by law. Based on these results it was evident that maintaining and respect 80% LR, it's possible reduce the CO_2 emissions more than 76%, avoiding around 119,534,836 t of CO_2 and influences positively on reducing deforestation. Dinamica EGO proved to be an effective to represent the deforestation.

Keywords: legal reserve area, Dinamica EGO, state of Acre, Brazilian Amazon

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1. Introduction

Tropical forests represent the largest terrestrial sink of anthropic emissions of carbon dioxide (CO_2) and are globally responsible for significant carbon stocks (C). The path for the global reduction of terrestrial CO_2 is controlled by the equilibrium between C fixation via accumulation of plant biomass and emission by burning [1, 2].

Modeling deforestation represents a powerful tool in different type of natural forest worldwide. The magnitude of the impact is been attested by predict changes in futures scenarios. In Brazil, modeling mechanisms is use to gauge the efficiency of the environmental policy established in Brazilian forestry code. Brazilian Amazon, since 1996 each property located in the must maintain a Legal Reserve Area (LR) of 80% where the vegetation cover consists of forest phytophysiognomies.

After more than 20 years and recent amnesty for properties the change landscape until July 22nd 2008, recently, in 2016 Legal Amazon lost 786,900 km^2, corresponding to 15.7% of the entire region in 2016 [3]. Precisely, the most common illegal practice is deforestation beyond the limit defined by law of LR [4].

Many landowners in the Amazon region claim to be unable to comply with the Forestry Code. Thus, the dominant debate in the region has always been favorable to noncompliance with legislation and not to its improvement [5]. Such positions may contribute to the loss of forest cover and may also cause direct environmental impacts, such as increased CO_2 emissions into the atmosphere.

The ways to understand the impacts of different methods of using land can be better understood from the construction of scenarios, which may represent the spatial design of future deforestation and its effects on global CO_2 emissions. In view of this, tools for remote deforestation monitoring, spatial dynamics modeling and assessments of the use and occupation of deforested areas can be used to increase the effectiveness of the implementation of environmental policies. The Satellite Monitoring Project of the Amazon Forest (PRODES) and the Amazon Near Real Time Deforestation Detection System (DETER) developed by the National Institute of Space Research (INPE) are good examples of the use of these tools.

PRODES is a project that has been monitoring deforestation in the Amazônia Legal since 1988, estimating annual deforestation areas and rates [3]. DETER is a warning system of changes in forest cover, which has the capacity to detect altered areas larger than 25 ha, thus making public policies to combat and control deforestation more effective [6]. The modeling of landscape dynamics allows the prediction of disturbances and monitoring the dynamics of vegetation, as these have the capacity to explore the role of deforestation, fire, climate, and weather [7].

The modeling process is a useful and multi-application tool. The results of modeling facilitate the study and understanding of deforestation and CO_2 emission issues, allowing simple representation of the complex deforestation processes by simulating the possible future trajectories according to the scenarios to be elaborated.

Thus, for the present case study, a model of landscape simulation based on cellular automata was developed and applied to the state of Acre, located in the Western South Amazon, northern region of Brazil, using the software platform Dinamica EGO [8]. Scenarios were projected for the entire territory of the state of Acre, represented by analysis applied to the simulation process, in order to assess the impacts of deforestation and CO_2 emission, in space and time.

2. Overview of spatial and temporal modeling for environmental services

In order to carry out effective measures of forest management and protection, a basic demand is the construction of methods for the collection, processing, analysis and publishing of spatial data regarding vegetation cover of these areas [9]. It is also important to evaluate the use of soil and changes in ecosystem dynamics, as well as observe advances in legislation, and other factors assessed by simulation models.

The main programs used for spatial image processing in remote sensing worldwide are QGIS, ArcGIS, Udig and Spring in Brazil. FIA Map Maker is used in the making of tables and maps in the USDA Forest Service's Forest Inventory and Analysis Data Base (FIADB), and the ACORn (A Comprehensive Ozark Regeneration simulator) program is used to predict the number of tree species and their potential individuals [10, 11].

Currently, there are versions of mapping programs in mobile devices, such as ArcGIS mobile, ArcPad and Avenza Maps, which replaces PDF Maps. They have been used successfully in assessing forest inventories, ecology studies, and conifer restoration projects in the United States [12, 13].

Many databases used in these programs may be compatible with statistical programs such as R. In addition, most of these programs have tools and interfaces that can be accessed by a variety of programming languages such as C++, Java, Python, Perl, C# and Ruby.

The Dinamica EGO is a spatial simulator of landscape dynamics widely used in Brazil. This simulator allows the construction of explicit scenarios that simulate the dynamics of an environmental system from transition probabilities, reproducing the way its spatial patterns evolve and allowing the projection of the probable ecological and socioeconomic consequences of the dynamics of the systems [8, 14].

Based on previous linear regression analysis, another model used in South America and that presents satisfactory results is the Land Change Modeler. It was used in a study in Peru with the purpose of simulating the loss of primary forest cover and changes in the characteristics of the landscapes from 2015 to 2025 [15].

MELA is a widely used simulator in Finland developed by the Finnish Forest Research Institute. It aims to model predictions for forest indicators (biomass, C sequestration) and economic indicators (yield and wood supply) [16].

In Africa, the MODIS (Moderate Resolution Imaging Spectroradiometer) platform is used in conjunction with the Land Degradation Surveillance framework (LDSF) and is implemented as part of the Africa Soil Information Service (AfSIS) in a soil-quality monitoring project. It has been employed in several tropical landscapes around the globe and also by NASA observatories [17].

The LDSF model stands out for its ability to perform joint map evaluations of several variables such as soil organic matter (SOM), pH, sand, sum of exchangeable bases, quantitative occurrence and degree of root depth restriction in the first soil layers, water infiltration capacity, soil erosion levels, and vegetation cover data, as well as information on structure, distribution, diversity and even species richness. From the joint analysis of these variables, the identification of soil and ecosystem quality can be improved, as well as the contrasts between land uses over time and also the possibility of verifying management interventions at various scales, helping to identify the adoption of these techniques [18].

The LANDIS-II model has been used to simulate the potential of C sequestration of vegetation under different fire regimes and prediction of biomass. The LANDIS-Pro model performs simulations of biomass, forest succession, seed dispersal, species establishments and disturbances in general. The PHENIPS model can be used in phenological prediction in tropical forests, in addition to the productivity and cycling of C and nitrogen in coniferous forests as a result of multiple validated models [19].

The SIMILE and TROLL models have been used in the simulation of plant succession processes [20]. However, such models have a limitation in the validation of their predictions, since most of the studies were conducted in specific times and areas (short-term predictions), which is not compatible with the simulations carried out in the large temporal space used for the most part of these simulations. Such short-term predictions can be more realistic by calibrating the models employed with the use of field data in time series, such as forest inventory data, on an appropriate timeline scale, providing greater predictability credibility [19].

Finally, the DSS-Wuk model, widely used in Germany, follows the tendency of the models to gather several variables (complex dynamic simulation) [21]. Recently, the model's ability to project forest development in the period from 2011 to 2100, compared to the reference period (1971–2000) [22], was evaluated in the country based on two climatic scenarios.

The DSS-Wuk combines several submodels (regionalization of climate, allometric and phenological, nutrient rates, water in the soil, mortality due to drought stress, wind and insect damage, site index, and others) for the purpose of better describing the impacts of climate change, besides including a sub-model of economic evaluation. It is recognized as a very realistic and very promising approach in the modeling area [23].

In the modeling carried out in Geographic Information System (GIS) programs, the analysis known as Anselin Local Moran's I can also be used, a type of Cluster analysis that verifies outliers and which, combined with geostatistical and remote sensing data, is able to evaluate patterns of environmental variables related to forest ecosystems. This was done recently in the simulation of future C sequestration potential in soils in China from 2000 to 2300 [24], and in the analysis and simulation of forest deforestation in Iran from 1972 to 2010 due to residential growth [25].

According to the rapid changes in the landscape observed globally, it is perceived that the systems of earth science have been improved by the use of remote sensing data and models based on the conversion of land cover, especially in a temporal space resolution, starting from the promising use of several submodels and the union of GIS data with data obtained in the field.

Advances in the field of forest modeling present the capacity to improve the predictive ability of different scenarios and the possibility of understanding ecological processes. The results of these models enable the adoption of better practices for the conservation of natural resources, along with the economic aspects of their exploration, as well as predicting the potential of forests, especially tropical forests, to mitigate GHG emissions.

3. Deforestation scenarios for the western south Amazon (case study)

The state of Acre is located in the northern region of Brazil, in the South Western Amazon, covering an area of 164,215.98 km^2 [26]. It borders Bolivia and Peru, and the states of Rondônia and Amazonas in Brazil. About 50% of its territory is composed of protected areas, distributed and classified according to the Brazilian system named "System of Units of Conservation" (SNUC), Law 9985/2000 in 18 Units of Conservation of Sustainable Use, 3 Integral Protection Units and 36 Indigenous Lands (**Figure 1**).

The natural forest types that occurs in the study area are: Open Broadleaf Forest with Palm trees, Open Broadleaf Forest with Dominant Bamboo, Open Broadleaf Forest with Bamboo, Dense Rainforest, Open Alluvial Rainforest with Palm Tree, Open Alluvial Rainforest with Bamboo, Open Alluvial Rainforest with Palms + Secondary Vegetation, Open Alluvial Forest with Palms + Pioneering vegetation.

3.1. Database

The data sources used for the modeling were: Acre data—Economic Ecological Zoning of the state of Acre [26]; PRODES data—Project of Estimation of Deforestation in the Brazilian Amazon, carried out by the INPE [3]; and the National Water Agency (ANA[1]) data. The software used to manipulate and generate the results were ArcGIS®10.x and Dinamica EGO, version 1.8.9.

3.2. Data processing

The entry maps of the deforestation model were obtained by PRODES (land use and land cover map for the years 2006 and 2009) and converted to the matrix format with spatial resolution of 100 × 100 m. Then, the maps were reclassified to the following classes: forest, non-forest and deforestation, as hydrography was defined as null. The 2006 map represented the initial input landscape of the model and the 2009 map represented the final input landscape model.

The static variables used in the deforestation model are subdivided into categorical, continuous and dynamic. The set of categorical variables is composed of maps of vegetation, soils and protected areas. In the set of continuous variables are the maps of altitude, slope, distance to main rivers, urban attraction distance, distances to population nuclei, distance to all roads, and distance from main roads (**Figure 2**). The deforested area distance was used as a dynamic variable.

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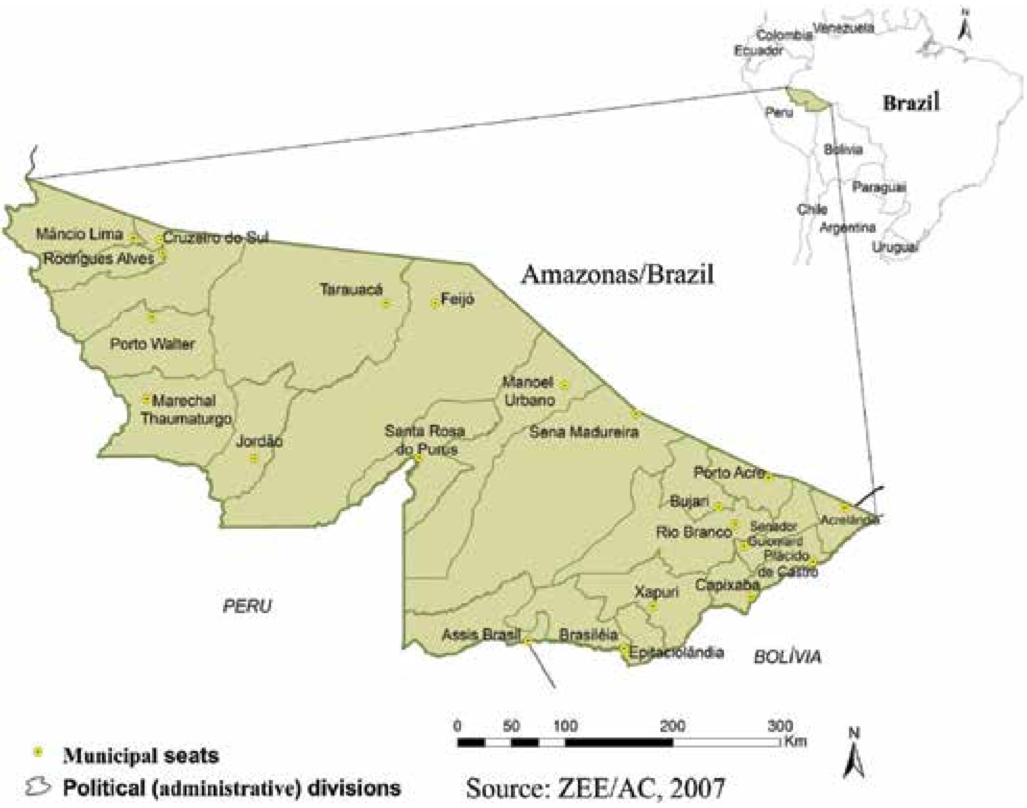
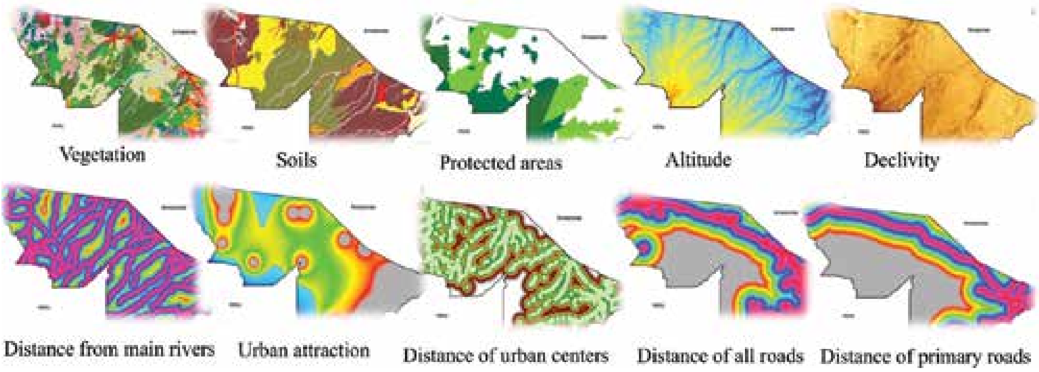
[1] *Agência Nacional de Águas.*



Source: ZEE/AC, 2007

Figure 1. Overview of the study area.

For the analysis of future deforestation, the behavior of subregions was considered, for which a micro-basin map (OTTO basins level 12) was used. The regionalization method allows defining sequences of specific operations for each region (microbasin), modeling as a result the local context influenced by particular phenomena [27].


Vegetation

Soils

Protected areas

Altitude

Declivity

Distance from main rivers

Urban attraction

Distance of urban centers

Distance of all roads

Distance of primary roads

Figure 2. Static variables used to estimate deforestation in different scenarios.

With the input data of the model, maps of the initial and final landscape plus the explanatory variables to deforestation, the calibration, validation and future projection of deforestation were carried out. The procedures for developing the model using the software Dinamica EGO can be visualized in **Figure 3**.

Then, the model was performed generating a simulated map for the year 2009 and a probability map (which indicated the areas most susceptible to deforestation) based on the weights of evidence from the variable maps. From the simulated map, the validation process was carried out to verify the similarity between the simulated map and a reference map—land use and land cover map for year 2009 of the PRODES classification. For the validation, the model that

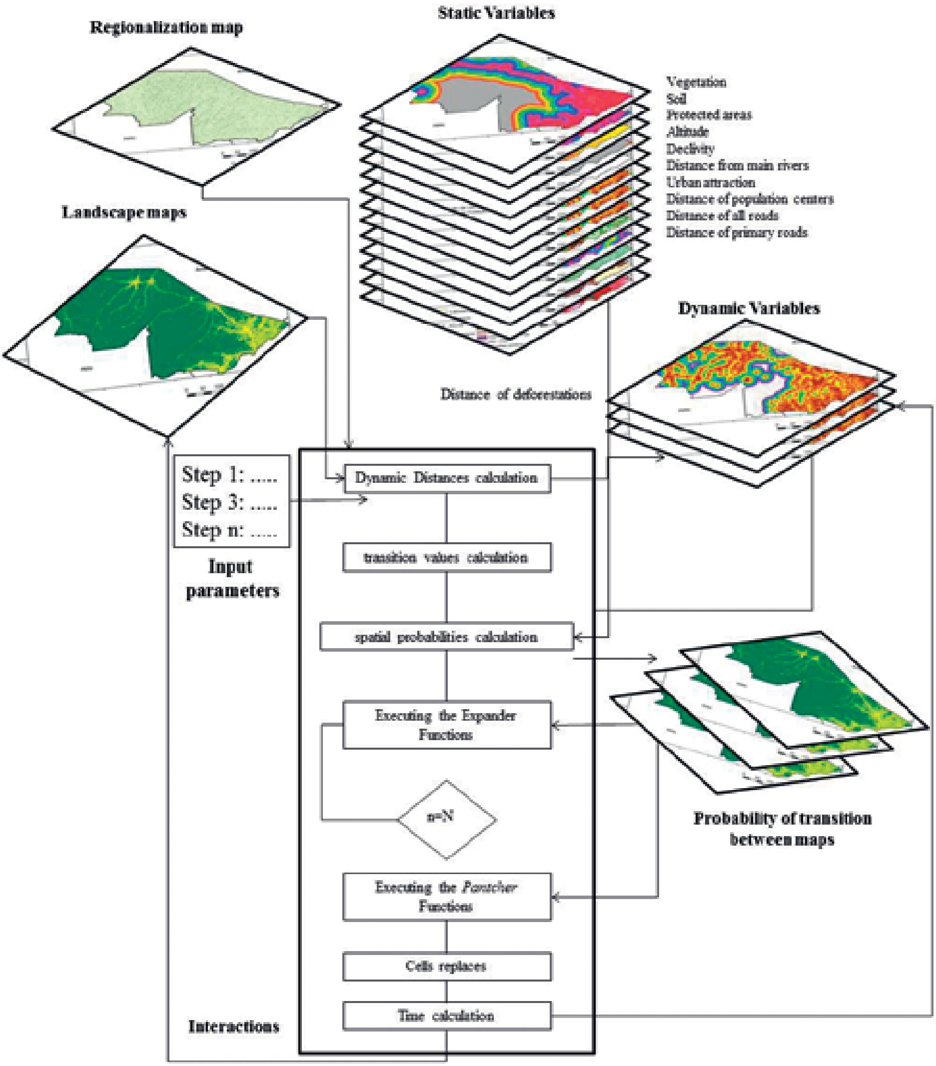


Figure 3. Flow diagram of the Dinamica EGO program. Source: Adapted from Soares-Filho [28].

uses the method of the constant decay function was applied, with windows of variable sizes from 1 × 1 to 13 × 13 pixels. The method is termed as "Fuzzy Similarity" in a familiar local context established by Hagen [29].

3.3. Generation of future scenarios

The study of future deforestation in the state of Acre was evaluated under the following scenarios: Scenario 1—deforestation "Business as usual"; Scenario 2—deforestation with LR of 50%; Scenario 3—deforestation with LR of 80% as provided by law. To evaluate the influence of LR size in scenarios 2 and 3, the useful areas of each basin (OTTO basins level 12) were used, with an average size of 4800 ha. In all scenarios, deforestation was simulated by the year 2050.

For all scenarios, the total deforestation in hectares and the amount of CO_2 emitted in tons were calculated. For scenarios 2 and 3, the amount of deforestation and CO_2 emissions that would be reduced under the following circumstances was calculated: (a) if there was a containment of the increase of liabilities in the micro-watershed already with liabilities up to the year 2010, and (b) if there was no generation of liabilities in micro-basins that, until the year 2010, were with forest assets. Environmental liabilities are deforested Legal Reserve Areas and environmental assets are units of areas that exceed LR size.

3.4. Model for accounting for CO_2 emissions

In order to account for CO_2 emissions, an annual biomass loss map was generated from areas that would be deforested annually, overlapping with a forest biomass map.

The biomass map used in the analysis was made by Saatchi et al. [30], where it was considered that 50% of forest biomass corresponds to C [31]. The CO_2 values resented were obtained by multiplying C by the ratio between the molecular weights of CO_2 and C, which is 44/12 (3.67 t CO_2 equivalent (CO_2 eq)).

4. Future scenarios modeled

In scenario 1, with the "Business as usual" pattern, from 2011 to 2050 more than 550,400 ha will be deforested, representing 42,324,000 t of C originally contained in these deforested forests and released into the atmosphere in the form of CO_2 of 155,329,080 t. The amount deforested by clear-cutting in the Western South Amazon (Acre State) will exceed 20.5% in 2010 [3] to more than 27% in the year 2050.

The areas of liabilities and assets already consolidated up to the year 2010 in the micro-basins and how these areas would remain until the year 2050 are presented in **Table 1**. It also shows the difference of the liabilities that would increase and the assets that would be reduced, based on the scenario of land cover change, considering the "Business as usual" prospects of deforestation with LR of 50 and 80%.

	Forest assets and liabilities (ha)					
	Until 2010		**Projection until 2050**		**Difference 2010–2050**	
	Liabilities	**Assets**	**Liabilities**	**Assets**	**+ Liabilities**	**− Assets**
LR 80%	988.700	900.100	1.422.200	783.200	433.500	116.900
LR 50%	312.800	2.750.900	545.500	2.433.300	232.800	317.600

Table 1. Table of liabilities and forest assets (LR of 50% and 80%) considering "business as usual" deforestation until 2010 and with the projection of deforestation up to 2050.

Considering the hypothesis that the legislation is not fulfilled, containing deforestation as described in the Law, deforestation trends in hectares and CO_2 emissions in tons would behave as shown in **Figure 4**. Deforestation, maintaining an LR of 80% (Scenario 3), would show a reduction of around 79%, with some 433,500 ha being deforested in relation to "Business as usual" deforestation patterns (Scenario 1). However, if LR changes back to 50% (Scenario 2), there will be an increase in deforestation by more than 170% when compared to an LR of 80%, with about 200,700 hectares being deforested (**Figure 4**).

One of the consequences of deforestation is the emission of CO_2, and the modeling shows that CO_2 emissions, with an LR of 80%, would decrease by more than 76%, avoiding around 119,534,836 t of CO_2 to be released into the atmosphere when compared to "Business as usual" deforestation patterns. However, when considering a LR of 50%, the CO_2 emissions would increase by more than 169% compared to the LR of 80% (**Figure 5**).

In view of the projections made, although the state of Acre (Western South Amazon) presents an even greater part of its territory in forest, it is evident that maintaining and respecting an LR of 80% would be of great importance for maintaining the environmental quality and, consequently, avoiding the reduction of biodiversity.

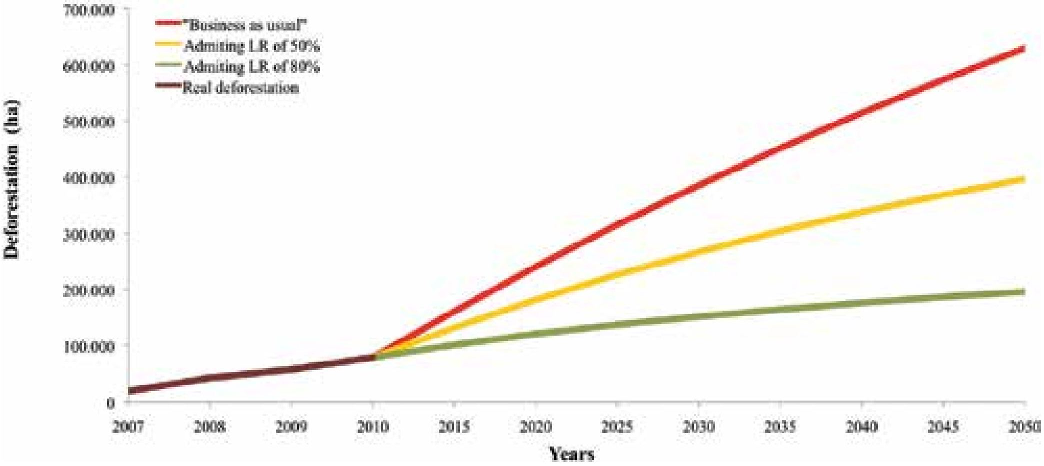


Figure 4. Trend of deforestation under scenario up to the year 2050 in three aspects: "Business as usual", 50% LR and 80% LR.

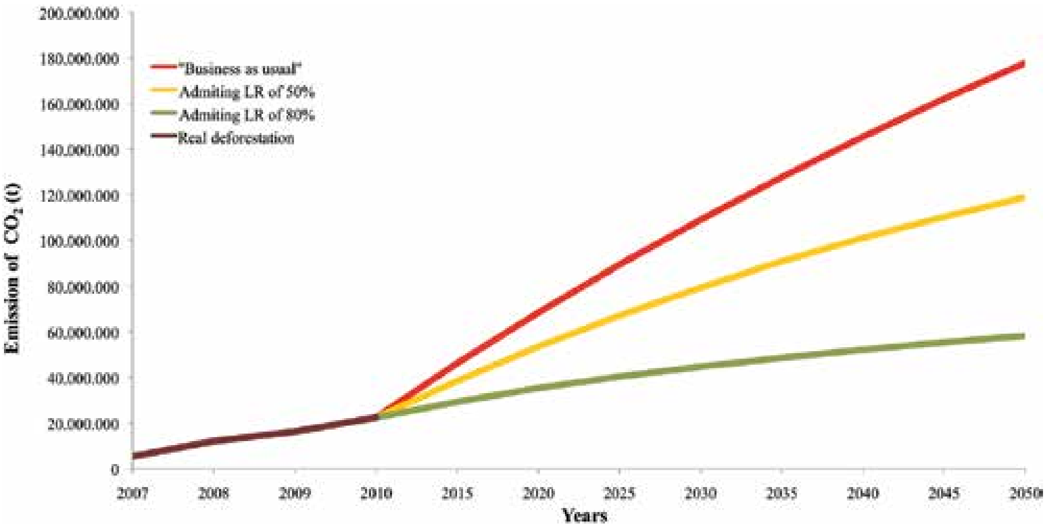


Figure 5. Trend of CO_2 emissions under scenario up to the year 2050 in three aspects: "Business as usual", 50% LR and 80% LR.

The set of variables used in this study presented a peculiar performance in the representation of deforestation, showing that it is mainly concentrated near the areas already deforested and the areas of influence of the current road network of the State. Because it is precisely in areas that are affected by the implantation of agricultural activities and logging, the latter has great importance in the expansion of deforestation in Acre [26]. In this sense, it is worth noting that the scenario presented for the state of Acre may take on another configuration as new roads and branches are deployed in the region. According to Nogueira [32], the influence of static variables on deforestation does not result from the individual action of each one, but from a complex relation of different factors of several causal variables, which has its origin in complex social, political and economic conditions.

The scenario "Business as usual" (Scenario 1) projects the historical evolution of deforestation in the region. The set of policies prohibiting land acquisition and implementation of rural settlement projects in areas of primary forest cover, together with the commitment of land use with adequate techniques and the prevention of clear cutting of vegetation [26] function as a hypothesis that reflects its effects on deforestation rates from 2011 to 2050.

The rates of deforestation on this period are smaller when compared to the deforestation rates of the 1980s, 1990s and the beginning of the twenty-first century when the state's growth rate was still under strong effects of the policy of incentive to occupy and deforest for production. However, deforestation reduction is already a reality throughout Amazonia [3, 26], even if due to the smaller amount of forest available for this activity.

As expected, the 80% LR size positively influenced the decrease in deforestation over a 50% LR, presenting approximately 63% reduction. The difference between the scenarios in maintaining an LR of 80% and reducing it to 50% implies, at the end of 41 years, the emission of around 60,610,417 t of more CO_2 into the atmosphere. This is equal to nearly 7 years of CO_2 emissions from the solid waste and domestic effluent sector in the state of São Paulo in 2008 [33].

The commercialization of CO_2 emission certificates avoided by deforestation is still an uncertain market. However, if it was to be regularized and implemented *de facto*, the Clean Development Mechanism (CDM) market, established in Article 12 of the Kyoto Protocol, the emission difference with an LR of 50% in relation to that of 80% may represent a loss of about $606.00 million if we consider an average value of $10.00 a ton of CO_2. And the prospect can be even worse when considering the 80% LR difference with the value obtained from the "Business as usual" scenario, reaching a loss of up to $1.19 billion.

The CO_2 considered in these accounts is only that contained in the C above the ground. Thus, this loss would only be related to the emission of CO_2, without taking into account other benefits offered by forests, such as the conservation of scenic beauty, socio-biodiversity, water and its services, climate regulation and soil conservation.

In the areas of the Brazilian Amazon, one of the elements that facilitate the dynamics of deforestation is the implantation of roads, since this practice allows easy access to forest remnants areas with greater potential for deforestation [34, 35]. In addition to the roads, rivers are also a source of easy access that enables deforestation, without taking into account that their use is often cheaper than transport by road. Ferreira et al. [36] argue that the process of deforestation usually begins with the official or clandestine opening of roads that allow the irregular occupation of land to the predatory exploitation of timber. Normally, after the exploitation of the forest, this area is converted to agriculture or pasture for the extensive creation of livestock, especially in large estates, being responsible for about 80% of deforestation in the legal Amazon.

Some researchers claim that the causes of deforestation in the Amazon are complex and often interrelated. Among them, there is the absent role of states, which in many situations do not restrict illegal deforestation [14, 37]. Added to this, there is still the ambiguity of public policies that stimulate, on one hand, the deforestation by great landowners, who thus assure the land ownership in the face of the "threat" of agrarian reform [37].

The state of Acre follows the model described, especially in the southeast region of the state. In this region, there is the highest concentration of deforestation, as is the case for the municipalities of Senador Guiomar, Acrelândia, Plácido de Castro, Rio Branco, Xapuri, Brasiléia, Epitaciolândia, Bujari and Porto Acre. However, it is in these municipalities that most of the population of Acre is found, along with most of the agricultural activity of the state and the greater demand for water resources. The region has hundreds of properties with 50–100% of its areas deforested [38]. This is a situation that may qualify these properties as illegal when facing the law, regardless of the size of the LR kept inside the property.

The other regions of the state still maintain most of their green areas preserved or, at least, conserved [26, 39]. Law No. 4771 of 1965 which refers to the Forest Code determines, among other obligations, the protection of sensitive areas and that landowners must maintain a portion of the native vegetation inside the rural properties, as is the case of the LR and Permanent Preservation Areas (APPs[2]). In the latter case, in general, forest management is prohibited, except in specific cases. Thus, one of the well-debated points is the size of the portion of the LR area to be maintained within each rural property, since from the implementation of

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[2] *Áreas de Preservação Permanente.*

Provisional Measure No. 1511 of 1996 it was determined that the LR in the Amazon region would increase from 50 to 80% of each rural property [5].

The changes in legislation aimed to restrict the rampant devastation that was taking place within the Amazon from 1994 to 1995, time and place in which the highest deforestation rate in history was recorded, with more than 29,000 km^2 of the forest devastated [40]. The size of the LR of a property functioned as a legislative component, which has its direct implications under the prospect of reducing deforestation.

Modification of the property use limit has curtailed deforestation for some years. However, deforestation rates returned to exponential growth starting in 1998 and had its peak in 2004, when, through public policies and more efficient geospatial monitoring, the deforestation scenario in the Amazon was again reduced.

Forests are of fundamental importance for balancing CO_2 emissions globally. When these are suppressed, most of the C stored in them is released into the atmosphere by burning or, more slowly, by decomposition [41].

CO_2 emissions due to changes in soil cover from land use have become a major component in the last decades, already considered as "dangerous interferences" in the global climate system. Approximately 0.8 ± 0.2–2.2 ± 0.8 billion tons of CO_2 from tropical forest deforestation is released annually, accounting for about 10–35% of global CO_2 emissions [5].

Deforestation in Brazil is the process that contributes most to CO_2 emissions, accounting for 70% of national emissions [27]. The stock of C in the state of Acre present in the aerial and underground plant biomass and the C contained in the soil varies from 75 to 270 Mg ha^{-1}, according to data published by the Environment State Secretariat (SEMA[3]) [42].

5. Final considerations

From a modeling point of view, the methodology and concepts used in the Dinamica EGO software were effective in representing deforestation under the three projections for the Western South Amazon. The variables selected and used in the scenarios clearly showed their influences both in the sense of restricting, as well as in potentializing the conversion of forest areas.

About the results of the deforestation of the projected scenario up to the year 2050, and considering the prospects of maintaining the current "business as usual" deforestation patterns, a severe intensification of deforestation is predicted over the years. As a consequence, the South Western Amazon is expected to reach very high levels of degradation, increasing CO_2 emissions to 155 million tons.

In the simulation of scenario 2, a reduction of CO_2 emissions of approximately 59 million tons would occur, when compared to the permanency of the current scenario. In this case, the reduction of the Legal Reserve (LR) to 50% must be considered, as it would lead to a loss, within the legality, of approximately 200,700 ha of native forests in relation to an LR of 80%. In addition, CO_2 emissions would increase by about 609,119,217 t. Based on the results of this

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[3] *Secretaria de Estado do Meio Ambiente.*

study, maintaining the size of LR in 80% of rural properties would prevent the emission of millions of tons of CO_2 into the atmosphere.

Thus, modeling tools are important mechanisms for forecasting and decision-making that can be used in order to assist in environmental conservation practices.

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Biomass Accumulation and Carbon Storage in *Pinus maximinoi*, *Quercus robur*, *Quercus rugosa*, and *Pinus patula* from Village-Forests of Chiapas, Mexico

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Abstract

The Frailesca region (Chiapas, Mexico) presents a lack of forest studies and its environmental contribution. This chapter displays a first case study with preliminary research information regarding the identification of main forest trees and rural villages with best potential for biomass production and carbon storage management. Twenty two plots of 500 m^2 were selected in 11 villages of the region, in order to identify the main and dominant forest trees species and then to estimate the biomass production and carbon storage in pine (*Pinus maximinoi*), oak (*Quercus robur*), holm oak (*Quercus rugosa*) and Mexican weeping pine (*Pinus patula*) species. This study shows that the largest accumulation of both biomass and carbon occurred in the pine forests and the lowest in the oak forests. Pine trees showed carbon storage of 516.75 Mg ha^{-1}, followed by holm oaks, with 297.21 Mg ha^{-1}; the species with the lowest value was oak, with 75.02 Mg ha^{-1}. The forests of the *24 de* Febrero villages had the highest potential for carbon storage. Deep studies are being conducted in relation to the aboveground biomass, carbon contents in trees stem, branches and leaves, and the relation to biomass dynamics and carbon stocks and other ecological aspects of village-forests.

Keywords: total biomass, carbon sequestration, allometric relationships, tropical forest

1. Introduction

The role and importance of forests in environmental issues like carbon sequestration has been evaluated by many researchers, e.g., Cook et al. [4]; de Jong et al. [10] and Dixon et al. [9]. These analyses suggest that forest conservation and their sustainable management can contribute to global carbon sequestration and conservation while providing goods and services for rural communities of many countries. De Jong et al. [10] stated that forestry and agroforestry can compensate for greenhouse gas emissions in two ways: (i) by the creation of new sinks for carbon dioxide by increasing the mass of woody material within growing trees; and (ii) through the safeguard of endangered natural forests and soils which are carbon stores.

In Mexico, conifer and broadleaf forests occupy 15.4% of the national territory; managed forests cover 7.3 million hectares, while protected forests cover 7.1 million hectares [32]. On the other hand, nearly 80% of the forest areas are communal property, and 95% of the forest exploitations originate mainly from the native temperate forests [22].

In the state of Chiapas, these forests cover 1,117,248 ha, and the rainforests occupy 2,175,948 ha [32]; therefore, the entity has the second place nationally for forest surface area and timber extraction from pine, cypress, sweetgum, holm oak, Rosy trumpet, amate ficus, cedar, and mahogany trees. In addition, its vast forest cover confers it a great potential for CO_2 sequestration.

Forest ecosystems can capture significant amounts of greenhouse gases (GHG), particularly CO_2. For this reason, in the recent decades, there is considerable interest in increasing the carbon content of the vegetation through the preservation of forests, reforestation, the creation of forest farms, and other land management methods. A great number of studies have demonstrated the ability of forest species to store carbon in their biomass [2, 26, 29, 31].

Each year, these forest areas store significant amounts of biomass that contribute to reduce carbon levels in the atmosphere [8, 23], which acquire greater significance if we consider that the CO_2 content in the atmosphere has augmented since the industrial revolution, and estimations indicate that in the twenty-first century this tendency will increase further [6, 28, 35]. All this indicates that anthropic activities have caused disturbances that contribute to the deterioration of the ecosystems [7].

Various studies performed in Mexico have proven the potential of forests to capture atmospheric carbon. Masera et al. [22] created a model simulating carbon capture during the 2000–2030 period, which divided land use into forests, rain forests, arid zones, and nonforest uses. These authors used two scenarios: the first, referred to as "baseline," and the second, as "policies." If the proposed options derived from their results were to be adopted, Mexico would be able to capture approximately 46 million tons of carbon during the 2000–2030 period. Part of this mitigation would be achieved as follows: (a) by preventing deforestation, (b) through sustainable management of the natural forests, and (c) by restoring the degraded forest areas.

CO_2 is one of the main components of GHGs and is produced by human activities when fossil fuels are utilized to generate energy and to meet other demands of the society. Deforestation processes, land-use changes, and methane concentrations resulting from agricultural and stockbreeding activities also promote climate change [35].

The increase in GHG concentrations in the atmosphere has caused the "greenhouse effect" phenomenon, which has resulted in changes in the climatic scales of the Earth [21]. The increase of CO_2 in the atmosphere produces extreme climatic events such as floods caused by hurricanes, which result in regrettable losses of human lives as well as economic losses [20].

According to Alberto and Elvir [2], carbon sequestration has been the object of study of forest research in various countries. Montero and Kaninnen [25] point out that, in southern Costa Rica, the production of aboveground biomass and carbon sequestration in managed *Terminalia amazonia* (J. F. Gmel.) plantations after 10 years was of 97.03 and 45.30 Mg ha^{-1}, respectively. In central Honduras, the production of aboveground biomass was of 80.53 Mg ha^{-1} in natural *Pinus oocarpa* forests. Carbon storage in the aboveground biomass of the pine forests of the *La Majada* village in Michoacán, Mexico, amounted to 28.85 Mg ha^{-1}, while in the conifer forests of *Tancítaro*, Michoacán, Mexico, the annual capture and storage of carbon in the aboveground biomass added up to 19.00 and 1.65 Mg ha^{-1}, respectively [13]. Based on the lack of forests studies and their potential contribution to the mitigation of climate change, the purposes of this case study were (a) the generation of basic and preliminary information about the main forest trees species and (b) the identification of rural villages with best potential regarding biomass production and carbon storage. Thus, to estimate the pine, holm oak, Mexican weeping pine, and oak forests in the forest areas from rural villages of the Frailesca region, as well as to determine the relationship between the age of trees and biomass production, allometric relationships were used.

2. Methodology

2.1. Location

The Frailesca region is made up of valleys and plains dominated by monoculture maize fields with Green Revolution technologies, surrounded by vast mountainous zones. The region is composed of six municipalities: *Villaflores, El Parral, la Concordia, Angel Albino Corzo, Montecristo de Guerrero*, and *Villa Corzo*, where traditional farming systems are practiced in small villages located within the natural forests from two natural protected areas, *La Frailescana* and *La Sepultura* reserves.

Climate is warm and semiwarm; the most predominant subclassifications are warm subhumid with occasional summer rain, followed by semiwarm humid with abundant summer rain. From May to October, the minimum average temperature ranges from 12 to 21°C (18 to 21°C in 54.9% of the region and 15 to 18°C in 37.8% of the region), while the maximum average temperature ranges from 21 to 34.5°C (30 to 33°C in 35.2% of the region and 27 to 30°C in

29.34% of the region). Average annual precipitation from May to October varies from 1000 to 2600 mm. From November to April, minimum average temperature ranges from 9 to 15°C, with averages of 12 to 15°C in 92.96% of the region, while the maximum ranges from 21 to 33°C, with averages of 27 to 30°C in 49.3% of the region and 33°C in 27.2% of the region.

The research was carried out in Villa Corzo municipality, in forest areas of 11 rural villages of La Frailesca region (**Figure 1**, **Table 1**), located between the coordinates 16°11′05″ N and 93°16′03″ W, at a mean altitude of 584 m above the sea level [3]. This municipality has a subhumid warm climate with abundant summer rains that prevails in the study area. The minimum annual precipitation is 1200 mm, and the maximum is 3000 mm, distributed among 100 and 200 days a year. The soils are affected by silt erosion precipitated by the action of the wind and by river floods; its fertility is variable, with its agricultural use conditioned by its depth and stoniness [3]. The localities have a vegetal cover that consists mainly of conifer forest secondary vegetation; montane cloud and holm oak forests; and deciduous, subdeciduous, and evergreen rainforests.

2.2. Plot selection and estimation of total biomass production and carbon storage

Dasometric data were recorded for the twenty-two 500 m^2 sampling plots (two in each village). This information was entered into a database which included data of 358 individuals of the following species: *Pinus maximinoi* H. E. Moore (pine), *Quercus robur* L. (oak), *Quercus rugosa Neé* (holm oak), and *Pinus patula* Schiede ex Schltdl. & Cham. (Mexican weeping pine) (**Table 1**).

The age of the trees was determined using a Pressler drill, as well as based on information provided by the common land holders; the normalized diameter formerly known as diameter at breast height (ND) was measured with a diameter measuring tape and the height (H) of the main stem with a clinometer [15].

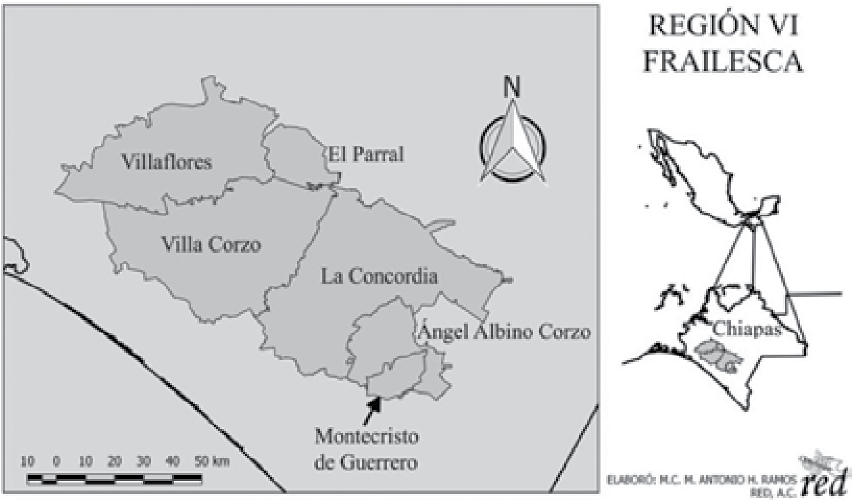

Figure 1. Location of Villa Corzo municipality in La Frailesca region (Chiapas, Mexico). Source: RED AC [36].

Villages	**500 m² plots**			
	Pine	**Holm oak**	**Oak**	**Mexican weeping pine**
24 de Febrero	+	+		
Monterrey			+	
Patria Chica			+	
La Frailesca				+
Juan Sabines Gutiérrez	+			
Bonanza		+		
La Libertad			+	
Tierra Santa	+			
La Unión	+			
Nuevo Refugio				+
Unión del Carmen			+	

Table 1. Distribution of plots in the rural villages and by the identified main forest species.

The stem volume (V; m^3) was calculated using the following equation:

$$V = \frac{\pi}{4} \cdot H \cdot ND \cdot SC \tag{1}$$

where H = tree height (m); SC = shape coefficient (**Table 2**); ND = normalized diameter (m).

In order to estimate the stem biomass, the total volume was multiplied by the density of each of the species (**Table 2**) [16]. The carbon sequestration was estimated by multiplying the biomass by 0.50, a value that represents the mean concentration of carbon for conifers [18] and a value cited in the Green House Gas Inventories of the forestry sector for Mexico [20].

The value of the biomass production in the various parts of the tree was determined based on the biomass expansion factors (BEF) and shape coefficients (SC) published by González [16] (**Table 2**), using the following expressions:

Stem biomass (SBM)

$$SBM = V \cdot r \tag{2}$$

where SBM = stem biomass (kg); V = stem volume (m^3); r = wood density (kg m^{-3}) (**Table 2**).

Tree aboveground biomass ($AGBM$)

$$AGBM = SBM \cdot BEF \tag{3}$$

Species	**SC**	**r (kg m^{-3})**	**BEF**
Pinus maximinoi H. E. Moore	0.52	507	1.25
Quercus rugosa Neé	0.39	650	1.27
Quercus robur L.	0.39	650	1.27
Pinus patula Schiede ex Schltdl. & Cham	0.25	507	1.25

Table 2. Biomass expansion factors (BEF), shape coefficients (SC), and density (r) of the studied species.

where $AGBM$ = tree aboveground biomass (Mg); SBM = stem biomass (Mg); BEF = biomass expansion factor.

Root biomass (RBM)

$$RBM = AGBM \cdot 0.30 \tag{4}$$

where RBM = root biomass (Mg); $AGBM$ = tree aboveground biomass (Mg).

2.3. Statistical analyses

The differences in the total biomass produced among species and between tissues were determined by processing the data through an ANOVA. Frequency histograms were created for the age and the normalized diameter (ND), and nonlinear regression analyses were carried out between the tree height and normalized diameter (ND) variables, as well as between the total biomass (TBM) and the tree's age by species. The mean comparison was carried out using Tukey's test for $p \leq 0.05$ [33]. Previously to all the statistical analyses, the assumptions of normality and of variance homogeneity were also verified using the STATISTICA®, version 8.0 software [33].

3. Results and discussion

The histogram of tree age by species shows that the pine plots reflected a rather heterogeneous distribution, with 12% of the trees aged 40–50 years (**Figure 2**), followed by 10% of trees aged 90–100 years. The age of holm oaks ranged between 20 and 70 years, with 10% of the individuals in an age interval of 30–50 years. The oak forests had 95 trees aged 20–40 years, while 18% of the trees of Mexican weeping pine forests aged 30–60 years (**Figure 2**). The largest number of trees with ND values of 0.1–0.4 m were oaks, followed by pines and Mexican weeping pines (**Figure 3**). In every case, like for the tree age, the highest degree of heterogeneity was found in the pine plots, with ND values ranging between 0.13 and 1.02 m (**Figure 3**). The interval between maximums, minimums, and ND (**Table 3**) made it possible to determine that the species with the highest variability were pine, holm oak, oak, and Mexican weeping pine, in this order (**Table 3**).

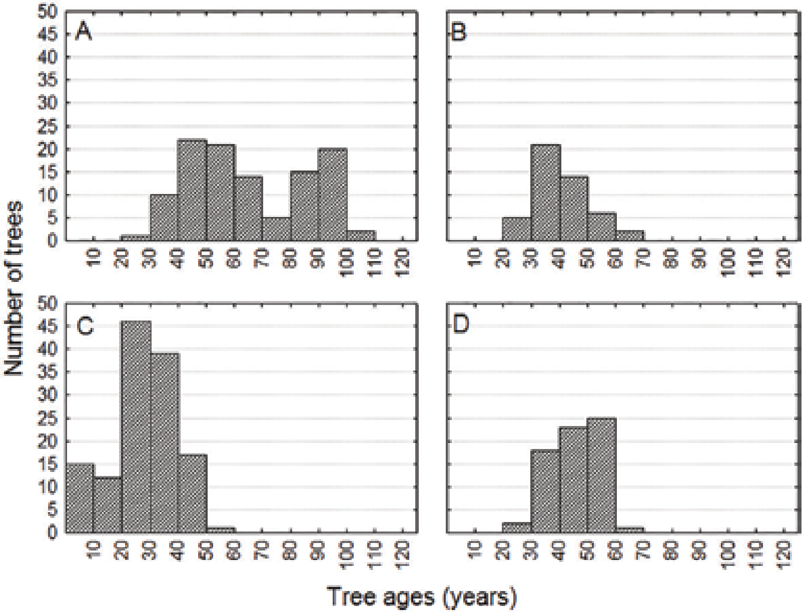

Figure 2. Frequency histograms of tree ages sampled by species in La Frailesca region (Chiapas, Mexico). A: Pine; B: Holm oak; C: Oak; D: Mexican weeping pine.

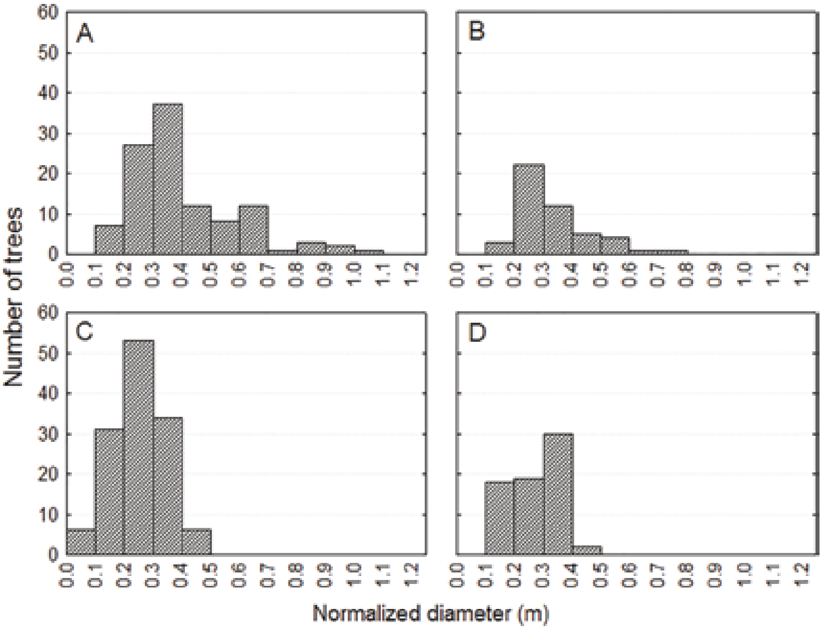

Figure 3. Frequency histograms of the ND of the sampled trees by species in La Frailesca region (Chiapas, Mexico). A: Pine; B: Holm oak; C: Oak; D: Mexican weeping pine.

Species	Mean	Maximum	Minimum	Range
	m			
Pinus maximinoi H. E. Moore	0.40	1.02	0.13	0.89
Quercus rugosa Neé	0.33	0.75	0.15	0.60
Quercus robur L.	0.26	0.46	0.05	0.41
Pinus patula Schiede ex Schltdl. & Cham	0.27	0.40	0.10	0.30

Table 3. Intervals between the maximums, the minimums, and the normalized diameters (NDs) of the studied species.

Given the close relationship observed between the ND, the age of the trees, and biomass production and carbon sequestration in various forest species [11, 12, 15], it is possible to understand the capacity of these village-forests to capture carbon from the atmosphere, particularly if it can be proven that the total biomass produced and the captured carbon increase with the age of the trees. The relationship between the height and the ND of the species was adjusted to an exponential model (**Figure 4**) called allometric, previously cited by Acosta et al. [1] and Gómez-Castro et al. [15].

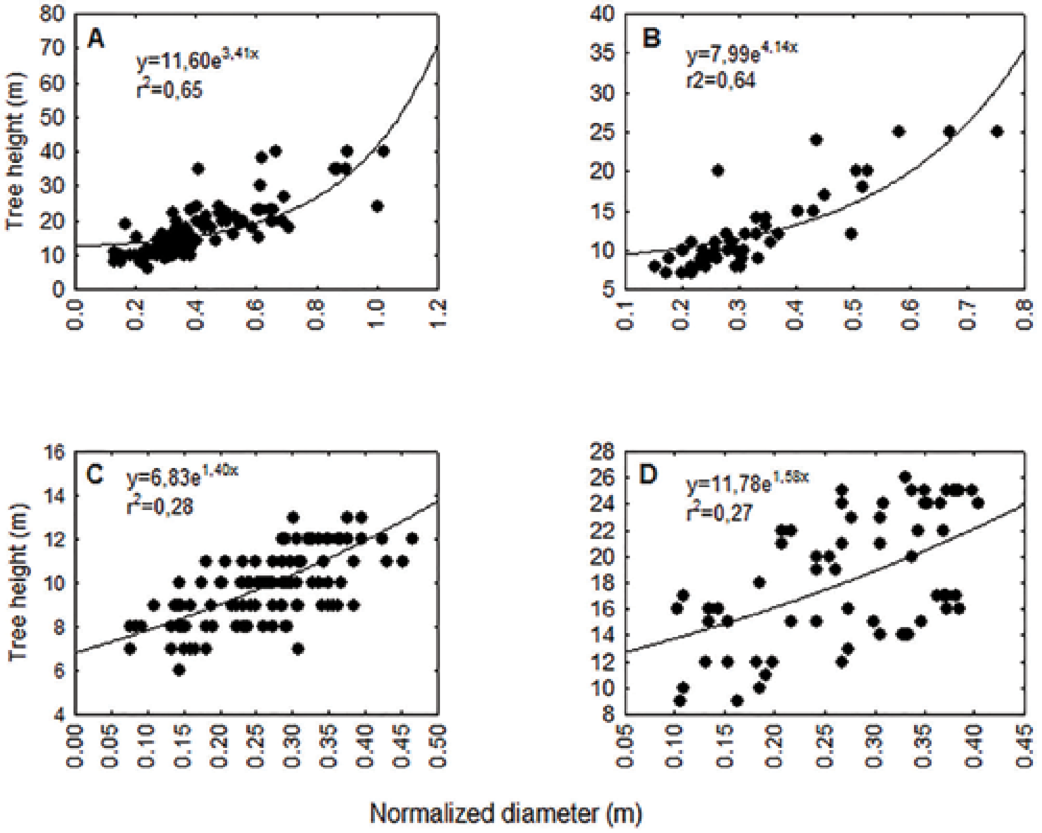

Figure 4. Exponential regression models of adjustment between the height and the ND of the studied species of La Frailesca region (Chiapas, Mexico). A: Pine; B: Holm oak; C: Oak; D: Mexican weeping pine.

The highest determination coefficients (r^2) were observed in pines and holm oaks, which also had the broadest interval between tree ages and NDs. For all the species, the mathematical adjustment produced a statistical significance of the model parameters, according to the Student's t-test (**Table 4**), a fact that corroborates the selection of the exponential model to estimate tree heights through the normalized diameter, although it suggests the need to delve into the effect of the normalized diameter interval, for which the best estimates were obtained.

For all species, the ratio of the total biomass of the tree (BMt) and the age was significantly fitted according to an exponential model (**Figure 5**, **Table 5**), with determination coefficients ranging between 0.56 and 0.85. Similar results were obtained by Rodríguez et al. [30] for the allometric relationships between the biomass production and the age of the pine trees. This result corroborates the biomass and carbon estimation studies based on the ND and ratifies the relationship between tree age and the photosynthetic processes that trigger biomass and carbon storage [27].

According to Fonseca et al. [12], both the aboveground biomass and the root biomass increase with the age in secondary forests and forest plantations. Hughes et al. [19] register an average biomass of 272.1 Mg ha^{-1} at 16 years of age. Corrales [5] registers a biomass of 162.1 Mg ha^{-1} in secondary forests aged 15 years and of 324.1 Mg ha^{-1} in primary forests in humid and very humid climates in Costa Rica.

The trees that produced the largest amount of biomass in the stem and other aboveground parts and in the roots were pines, with values above 506.92, 882.05, and 264.61 Mg ha^{-1}, respectively (**Figure 6**). Oak trees produced the least biomass. Monroy and Navar [24] cite similar results for *Hevea brasiliensis* (Willd. ex A. Juss.) Müll. Arg., with values of 73.9% stem biomass and 27.1% branch biomass, both of which increased with the age of trees.

As for the relationship between the carbon content and the total biomass production in the various components of the plant, the results suggest that the high rates registered in pine forests for both growth and aboveground carbon fixation, may be due, as Pacheco et al. [27]

Species	**Model parameters**	**Estimate**	**Standard error**	**t**	**p**	**Confidence limits 95%**	
						Min.	**Max**
Pinus maximinoi H. E. Moore	a	8.71	0.524	16.701	0.01	7.707	9.792
	b	1.51	0.099	15.193	0.01	1.313	1.706
Quercus rugosa Neé	a	6.64	0.314	21.146	0.01	6.018	7.261
	b	4.27	0.270	15.779	0.01	3.732	4.802
Quercus robur L.	a	6.83	0.414	16.475	0.01	6.008	7.647
	b	1.40	0.206	6.785	0.01	0.992	1.808
Pinus patula Schiede ex Schltdl. & Cham	a	11.78	1.251	9.417	0.01	9.283	14.277
	b	1.58	0.344	4.598	0.01	0.894	2.265

t = Student's t; p = probability of error.

Table 4. Parameters and statistical significance of the model of mathematical fit between the height and the normalized diameter.

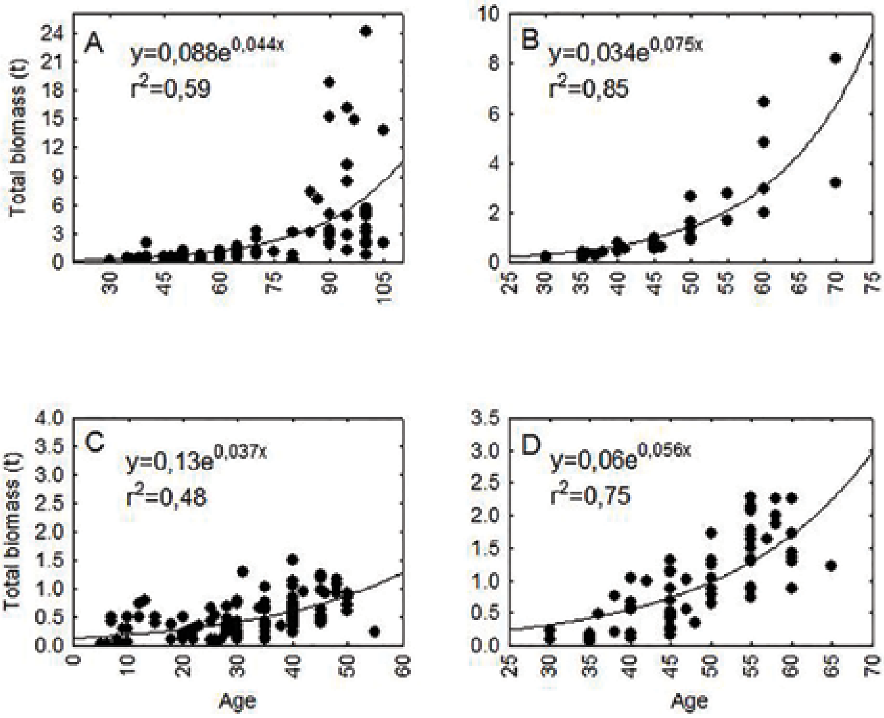

Figure 5. Regression curves of fit between the total biomass (BMt) and age of trees of (A) pine, (B) holm oak, (C) oak, and (D) Mexican weeping pine.

Species	Model parameters	Estimate	Standard error	t	p	Confidence limits 95%	
						Min.	Max
Pinus maximinoi H. E. Moore	a	0.09	0.07	1.22	0.23	−0.055	0.23
	b	0.04	0.009	5.06	0.01	0.027	0.06
Quercus rugosa Neé	a	0.034	0.016	2.18	0.05	0.002	0.07
	b	0.075	0.007	10.28	0.01	0.059	0.09
Quercus robur L.	a	0.13	0.036	3.72	0.01	0.063	0.21
	b	0.4	0.007	5.55	0.01	0.024	0.05
Pinus patula Schiede ex Schltdl. & Cham	a	0.006	0.024	2.48	0.01	0.012	0.1
	b	0.06	0.007	7.59	0.01	0.041	0.07

t = Student's t; p = probability of error.

Table 5. Parameters and statistical significance of the model of mathematical fit between the biomass and the age of the trees.

proved, to a good combination between the production of wood and cellulose compared to other species; this is helpful for the implementation of reforestation and CO_2 sequestration projects [14]. However, other factors, such as the site and the tree mass, also determine biomass

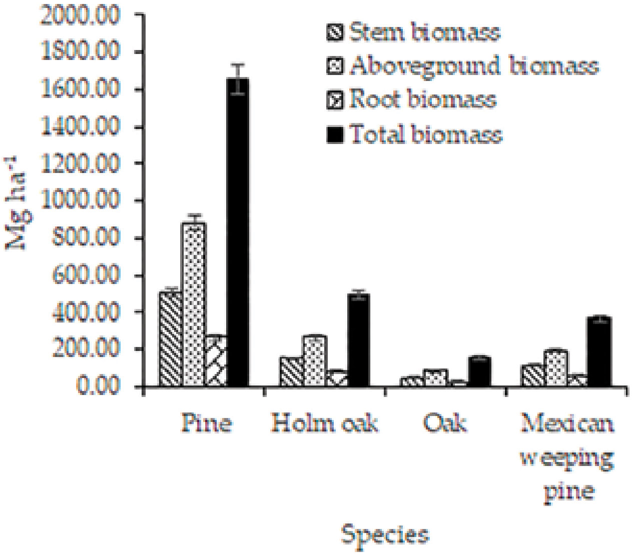

Figure 6. Total biomass production in the stem, branches, and roots of pine, oak, holm oak, and Mexican weeping pine trees in villages of La Frailesca region (Chiapas, Mexico). Note: Vertical lines in the columns represent the mean standard error.

production and carbon storage. Furthermore, this author cites a direct relationship between the sequestered carbon and growth, in both the normal diameter (ND) and the total height of the trees [34]. Analysis of the information from the various villages proved that the total biomass produced in pine trees was highest at *24 de Febrero* and *Juan Sabines* villages (**Figure 7**), with values above 2606.67 and 1045.61 Mg ha^{-1}, respectively. Therefore, these villages also had the highest values for carbon capture.

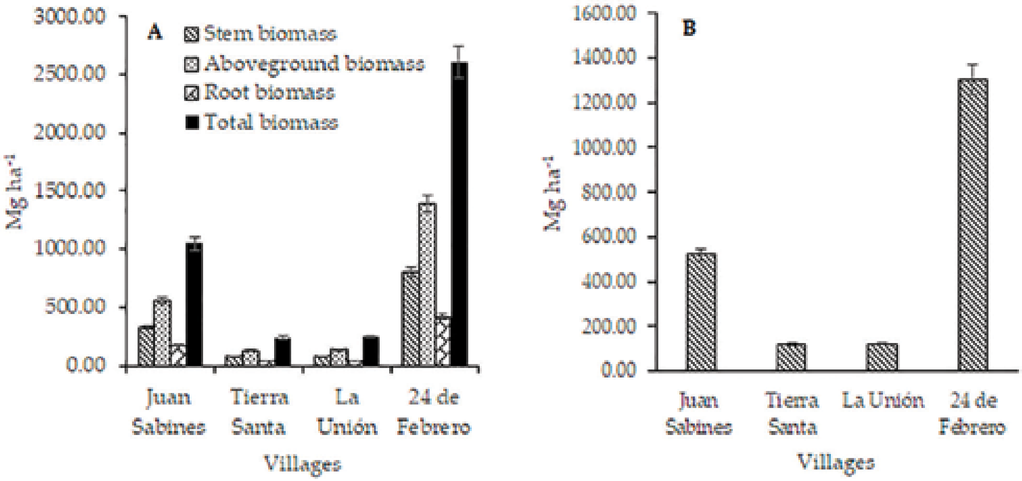

Figure 7. Total biomass (A) and carbon (B) storage in *Pinus maximinoi* H. E. Moore forests of the *Juan Sabines, Tierra Santa, La Unión* and *24 de Febrero* villages of La Frailesca region (Chiapas, México). Note: Vertical lines in the columns represent the mean standard error.

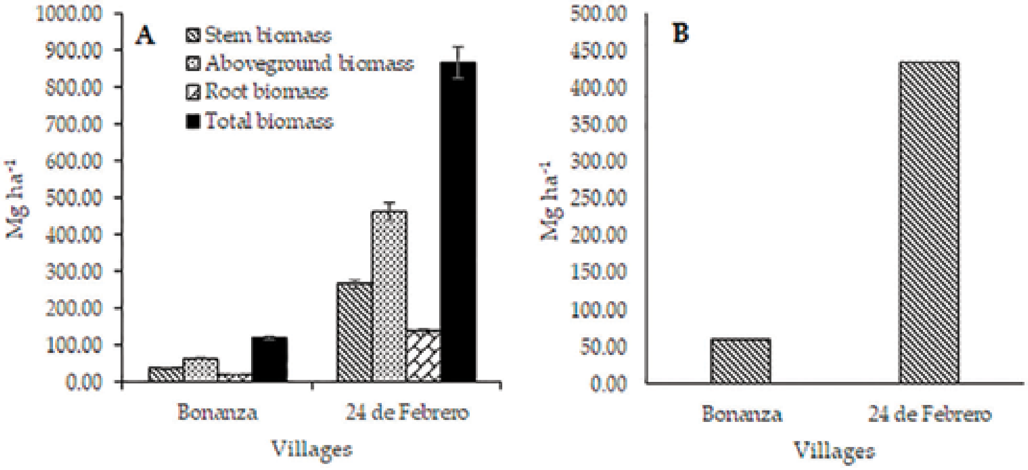

Figure 8. Total biomass (A) and carbon (B) storage in *Quercus rugosa* Neé forests in the *Juan Sabines, Tierra Santa, La Unión*, and *24 de Febrero* villages of La Frailesca region (Chiapas, Mexico). Note: Vertical lines in the columns represent the mean standard error.

Holm oak forests, which are prevalent in the *Bonanza* and *24 de Febrero* villages, produced up to 120.87 and 988.82 ha^{-1} Mg of total biomass, respectively (**Figure 8**), whereas the carbon sequestered was higher at *24 de Febrero*, with 433.98 Mg of C ha^{-1}, which proves the potential of this village for CO_2 capture and its contribution to the abatement of GHGs.

The total biomass production in oak trees was 154.03–176.21 Mg ha^{-1}, the highest record in the forest areas of the *La Libertad* village, followed by *Monterrey* (**Figure 9**). The lowest value was found at *Patria Chica*. As for carbon storage, in all the villages, it was above 50 Mg ha^{-1}; the highest was obtained in the forests of the *La Libertad* village, with 88.11 Mg of C ha^{-1} (**Figure 9**).

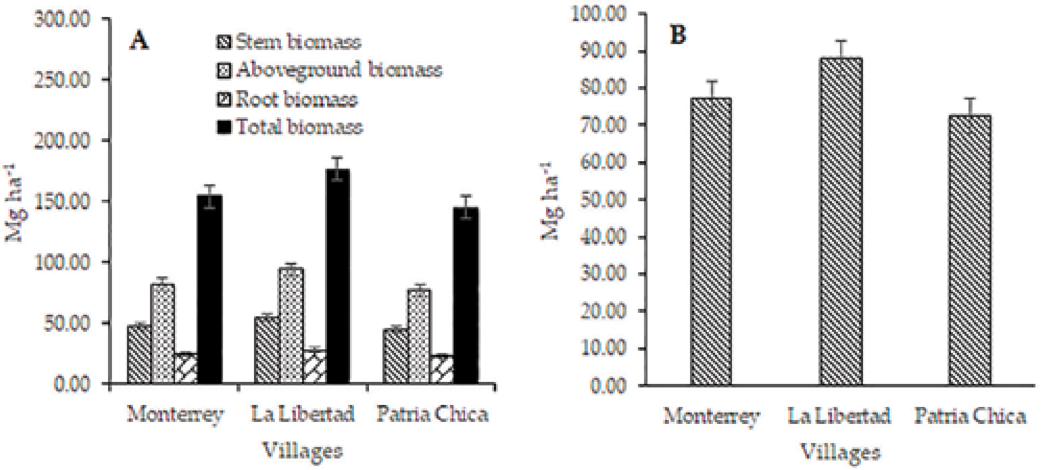

Figure 9. Total biomass (A) and carbon (B) storage in *Quercus robur* L. forests of the *Juan Sabines, Tierra Santa, La Unión*, and *24 de Febrero* villages of La Frailesca region (Chiapas, Mexico). Note: Vertical lines in the columns represent the mean standard error.

The total biomass produced by Mexican weeping pines (*Pinus oocarpa* Schiede) ranged between 281.78 and 450.52 Mg ha^{-1} (**Figure 10**). This species occurred only in the *La Frailesca* and *Nuevo Refugio* villages; the forests of the latter contributed a larger amount to this production. The highest value for carbon storage (225.26 Mg of C ha^{-1}) was found in the forests of *Nuevo Refugio*, while the *La Frailesca* village stored only 140.89 Mg ha^{-1}. As for the production of biomass per tree components (**Figures 6–10**), in all the taxa and localities, the highest accumulation occurred in the aboveground biomass, as indicated by Gower et al. [17], who proved that approximately 75% of the biomass of a tree is produced in the aboveground parts, while only 25% is accumulated in the roots.

Comparison between biomass production and carbon storage by species showed that *Pinus* spp. and *Quercus* spp. had the highest values, while *Quercus robur* had the lowest (**Table 6**). This agrees with the findings of González [16], who, after comparing three tree species, determined that *P. maximinoi* had the highest biomass accumulation, followed by *Quercus* spp. In regard to carbon, pine and holm oak had a storage potential of 516.75–247.02 Mg of C ha^{-1} (**Table 6**).

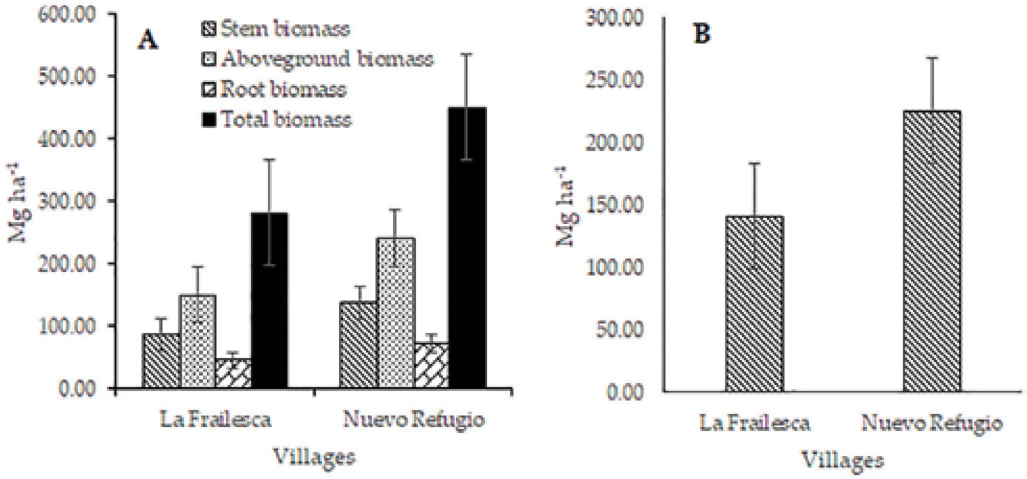

Figure 10. Biomass (A) and carbon (B) storage in *Pinus oocarpa* Schiede forests of the *La Frailesca* and *Nuevo Refugio* villages in La Frailesca region (Chiapas, Mexico). Note: Vertical lines in the columns represent the mean standard error.

Species	Total biomass (Mg ha^{-1})		Carbon storage (Mg ha^{-1})	
	Mean	Standard error	Mean	Standard error
Pinus maximinoi H. E. Moore	1033.49	557.65	516.75	278.82
Quercus rugosa Neé	494.41	373.54	247.21	186.77
Quercus robur L.	150.03	10.60	75.02	5.30
Pinus patula Schiede ex Schltdl. & Cham	366.15	84.37	183.08	42.19
General	537.94	207.32	268.97	103.66

Table 6. Total biomass and carbon storage in pine, holm oak, oak, and Mexican weeping pine forests of La Frailesca region, Chiapas.

4. Conclusions

The forests of the *24 de Febrero* villages registered the highest growth, based on their height and stem diameter, as well as on the total biomass, rendering this village one of the localities with the highest potential for carbon storage of all the studied sites. Pine, holm oak, oak, and Mexican weeping pine trees in villages of *Frailesca* region in *Chiapas*, Mexico, accumulate between 150.03 and 1033.49 Mg ha^{-1} of vegetal biomass; therefore, they are considered to have a high potential for carbon sequestration. Pine trees reached a value of 516.75 Mg ha^{-1} of C; due to its high degree of development, this species has the highest capture potential of all.

Since the present study was an initial effort to begin with data collection for preliminary insights and understandings about the village-forests in la Frailesca region, it is important to keep in the research track with deeper and longer term investigations on forest dynamics in this geographical area, especially due to its importance for biodiversity conservation, sustainable management of natural resources, and the environmental services provision for the rural and urban areas of the Chiapas state.

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Chapter 3

Forest Soil C: Stock and Stability under Global Change

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Additional information is available at the end of the chapter

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Abstract

Both plant biomass and soil C are potential to C sequestration by offsetting atmospheric CO_2. In forest, aboveground and root biomass are the major sources of soil C. Estimation of biomass is challenging due to methodological uncertainty at different spatial scale. The role of root-mycorrhizal symbiosis on long-term C storage in soil is yet to be established. Microbial biomass and community structure are crucial for C regulating processes in soil, and the genomic approach is promising for insights of soil microbial processes in relation to C stock and stability. The dynamics of dissolved organic C (DOC) in soil is extremely complex as many biotic and abiotic factors are involved in the release and flux of DOC in soil solution. Typically, the stability of soil organic C refers to recalcitrant, humified substances and physically protected materials, having the residence time of decadal to millennium scale; however, recent studies showed that microbial inaccessibility and physical protection are the dominant mechanisms. Priming is a potential process of decomposing old C and may have an antagonistic impact on soil C stock, particularly stable C in deep soil. Coupling the C pools with the stability processes is necessary for assessment of forest C balance.

Keywords: soil C stock, biomass, soil microbes, C stability, C fractionation, priming effect

1. Introduction

Carbon dioxide (CO_2) is one of the major greenhouse gases (approximately 72% of the total anthropogenic greenhouse gases) and considered as a primary agent of global warming [1]. It has been estimated that CO_2 is responsible for about 9–26% of the global greenhouse effects [2]. The concentration of carbon dioxide in the atmosphere has increased from 280 ppm of the pre-industrial era (1750) to 408.84 ppm in July 2017 with increasing rate of 2.11 ppm per year [3]. The dramatic rise of CO_2 concentration is attributed largely to human activities, and since soil is the second largest reservoir of C in the terrestrial ecosystems, there is a strong link between soil and atmospheric C through C cycle. Globally, the soil C pool is about four times larger than the atmospheric pool, and consequently, any change in the flux of CO_2 from soil to atmosphere has paramount importance in the balance of atmospheric CO_2 [4]. Among different terrestrial ecosystems, forest soil contains more than two third of the global soil organic C reserve, although forest occupies only 30% land of the earth surface, creating the highest carbon-rich domain among different land use-based ecosystems. Atmospheric C, once fixed into plant tissues through photosynthesis, is transferred to the soil as plant litter. Part of this C is stored in soils, and the major portion is released to the atmosphere through soil respiration. Some of the stored C in soil can be sequestrated as soil organic matter and/or humus for as long as a million years [5]. As such, the potentiality of forest soil for long-term C sequestration is instrumental to many research efforts worldwide. In forest, soil C stock mostly derives from decaying above and belowground plant tissues and root exudates; however, the relative contribution of fine root and accompanying mycorrhizal turnover on soil C storage are considering more vital than the C in aboveground litter [6]. Microbial biomass and the community structure of bacteria, archaea and mycorrhizal fungi contribute to soil organic C stock through biomass production at one hand and releasing stored C through decomposition and respiration processes on the other. Dissolved organic carbon (DOC) is an impotrant C pool in forest soil ecosystems, which considered as a labile and more easily degradable substrates that influence the storage of C in forest soil.

There are several mechanisms of C stability in soil such as formation of humus and other recalcitrant C, formation of organo-mineral complexes, accumulation of C in deep soil layers, etc. Many studies revealed that the considerable amounts of C are in deep soil layers. These C are considered more resistant to microbial degradation due to physical protection, chemical recalcitrance

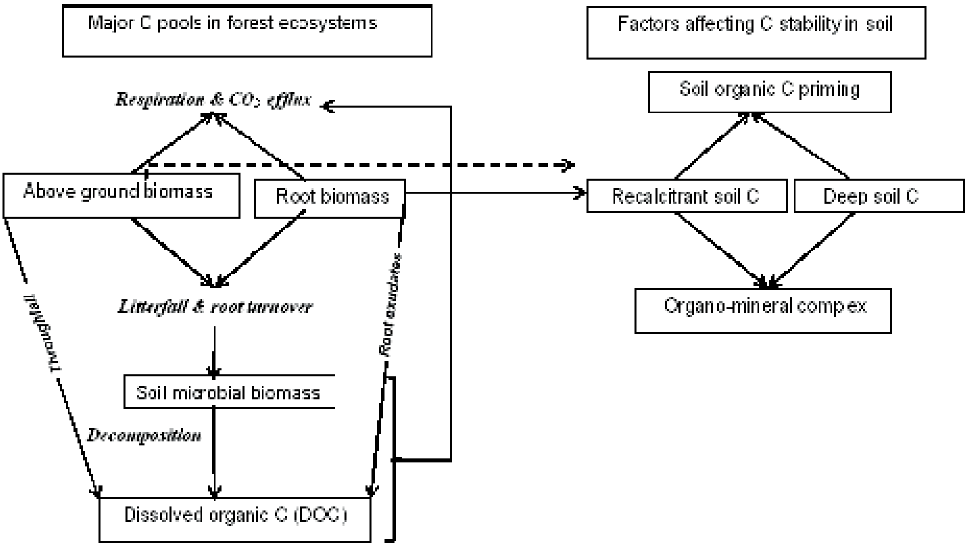

Figure 1. Link between various C pools and stability processes in forest ecosystems. Aboveground biomass and root biomass are the primary source of recalcitrant C; however, root biomass C is considered as more potential than the aboveground biomass (shown as broken line) from soil C stability perspective.
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and limited microbial activity [7]. On the other hand, priming is a C unstabling process in which old soil C is decomposed in the presence of new fresh C. The process has been receiving much attention as a potential mechanism of degradation of stable soil organic matter. Our mechanistic understanding of these processes is still unclear, because the heterogenity of soil and climatic conditions influences the processes considerably. This chapter includes the in-depth discussions on the conceptual basis and functional significance of these major aspects regarding storage and stability of soil C. The C stock part of this chapter includes the following topics: (1) contribution of aboveground plant biomass on storage of soil C, (2) fine root and mycorrhizal biomass production and turnover, (3) soil microbial biomass and community structure and (4) dissolved organic C in forest soil. The stability-related issues are as follows: (1) formation of recalcitrant C pool and C fractionation, (2) organo-mineral complexes and physical protection to soil C, (3) protected C in deep soil and (4) priming—a potential mechanism of destabilizing soil C. The relationships between C pools and stability processes are shown in **Figure 1**.

2. Storage of C in forest soil

In the soil ecosystems, C generally exists in two major forms depending on the soil type: a relatively dynamic organic pool as soil organic matter (SOM) and the inorganic forms mainly as carbonates, both are linked to atmospheric CO_2 through the processes of C cycle. Among different land use-based ecosystems, forest soil contains about one third of soil organic carbon [8]. The accumulation of organic C in forest soils depends on a number of factors such as plant biomass (above and belowground), litter quality and soil microbial activities, management practices and climatic factors such as temperature, precipitation and fire. Plant biomass C and soil C are the two C pools that constitute major part of the ecosystem C reserve [9]. Other pools are forest litter, decomposed detritus, microbial biomass, dissolved organic materials and humus. Among different processes, photosynthesis, canopy respiration, litter flux, litter decomposition, soil respiration, SOC sequestration etc. play important roles in forest C cycle. Thus, the storage of soil C is continuously influenced by dynamics of forest ecosystems and also by many biotic and abiotic factors including climate changes [10]. Because of the interdependency of these carbon pools, it is imperative to evaluate ecosystem C in an integrated approach rather than assessing any pool individually. It is worthwhile to study the controlling elements of C storage mechanisms in addition to stock size for in-depth knowledge of the processes. The factors that affecting the nature and properties of forest soil are quality and quantity of litter, soil nutrient status, root activities, interception of atmospheric deposition, canopy interactions, leaching and microbial activities [11]. C stock in soils of the three major forest biomes follows the order tropical < boreal < temperate and collectively constitutes the largest C sink in terrestrial ecosystems [9].The quantities of C stock in major global forest biomes are shown in **Table 1**. These huge organic C pools are however not constant over time. The size of the C-stock is related to the quality of soil organic matter in relation to biodegradation and subsequent inorganic C efflux from soil and supply of fresh organic material to the soil. Therefore, the balance between input of organic matter mainly from vegetation in the forest ecosystem, and losses mainly due to decomposition-respiration and leaching, determines the net C storage potential in the soil [12].

Components	**Major forest biomes of the world**			**Total**	**Proportion**
	Boreal	**Temperate**	**Tropical**		
Forest area (million hectare)	1135	767	1949	3851	31% of global ice-free land
Forest C stock (including living and dead biomass and soil) (*Pg C)	271.5	118.6	471.0	861.1	46% of terrestrial biosphere C stock
Forest biomass density (Mg h^{-1} dry mass)	95	121.5	268.9	188.3	–
Total plant biomass forest (Pg C)	53.9	46.6	262.1	362.6	42% of global forest C reserve
Aboveground biomass (Pg C)	43.8	38.8	206.4	289	80% of forest biomass C
Root biomass (Pg C)	10.1	7.8	55.7	73.6	20% of forest biomass C
Litter + dead wood (Pg)	43.1	15.4	57.6	116.1	13% of global forest C reserve
Soil organic C stock (Pg)	174.5	56.7	151.3	382.5	44% of global forest C reserve

*Pg (Peta gram) = 10^{15} g

Biomass C was estimated as 50% of dry mass. Aboveground and root biomass fractions were calculated by using root: shoot ratio of 0.23, 0.20 and 0.27 for boreal, temperate and tropical biomes, respectively [119].

Table 1. C stock in various components of forest ecosystems under different forest biomes [24].

Both natural and human induced factors can influence the concentration and stock of soil C in the forest ecosystems. The natural factors are climate, vegetation, soil quality, soil microbial populations, and forest fire, and anthropogenic factors are forest management, afforestation, deforestation, etc. [9]. Tree species (or species composition) is the single most important factor that influences SOC storage in the forest. Most of the forest ecosystem processes are generally controlled by the plant's functional traits in a species-specific way. Litter quality from broadleaved trees is generally higher than that from coniferous tree [13], but differences between broadleaved species and even between genotypes were also observed. Species-specific root turnover rate contributes to soil C dynamics in association with ecto- and arbuscular mycorrhizal fungi in temperate broadleaved forests [14, 15]. The broadleaved trees are particularly important for its higher contribution to carbon sequestration than the narrow leaved coniferous plants. The efficiency of CO_2 uptake is higher in broadleaved species than coniferous because of the low leaf area of coniferous species [16]. Forest management activities that affects SOC stocks includes thinning, harvesting, site preparation and maintaining continuous canopy cover [17], fertilization and liming [18]. Thinning practice can affect soil C storage negatively in several ways: thinning causes changing the stand microclimate by reducing evapotranspiration and increasing soil temperature and can stimulate the decomposition of the forest floor resulting a decrease in soil C pool. In addition, litterfall can be lowered in heavily thinned stands and thus decrease SOC stocks [19]. In contrast, enhanced growth of the understory vegetation due to thinning measures was observed in an experimental site in Finland which ultimately compensated for the reduction of C from tree biomass [20]. Harvesting can affect soil C storage positively or negatively. Removal of whole trees reduces seasonal litter inputs and disturbance affects forest floor and mineral soils leading to soil C loss. In addition, harvesting causes decrease in photosynthesis and can turn the forest into a C source condition [21]. The consequences of harvesting can persist for variable time frames. For example, measurement of net ecosystem C exchange showed that the increased rates of soil respiration due to harvesting continued for at least 14 years after logging [22, 23]. However, other research has shown that harvest residues left on the forest floor can also increase C stocks on mineral soils [19]. As clear cut harvesting decreases SOC stocks, continuous cover forestry may be an effective option for reduction of soil C losses following selective harvesting and thinning operations. Fire is another major disturbance that can impact soil C stocks in forest ecosystems and may have a long-term impact on C stock in soils of the boreal regions. The impact of fire on SOC stock depends on fire temperature and duration, existing SOC stock and its distribution in the soil profile and the changes in the decomposition rate of SOC following the fire event. It is obvious that most of the forest management interventions have negative impacts on the storage of soil C; however, minimizing the disturbance to soil and stand structure can reduce possibility of C loss [19]. From the above discussions, it is obvious that the nature and properties of soil organic C are highly dynamic and depend on many abiotic and biotic factors within ecosystem. In the following sections, we discuss the significance and mechanistic link of four major C pools with soil C storage in relation to climate change.

2.1. Aboveground biomass and soil C

Two major compounds that are involved in the immediate release of CO_2 to atmosphere and that are boosted by human activities are simple carbohydrates and hydrocarbons (fossil fuel). Carbohydrates in plant biomass are synthesized photosynthetically by fixing atmospheric CO_2; therefore, forest biomass has a potential for reduction of atmospheric CO_2 to some extent, and thus the higher production of biomass through afforestation/reforestation has been recognized as an effective, low cost option for mitigation of climate change impacts [1].Globally forests possess 86 ± 66 Pg of C and the biomass of living trees contributes 42% of this stock [24]. Expanding tree biomass may also increase the carbon stock in soil as 70% of soil organic C derives from plant biomass [25, 26]. Estimation of biomass is important for many purposes. At a national or regional level, when biomass is considered as a raw materials or energy source, it is necessary for planners and policy makers to know how much timber or fuel wood is available for national consumption. From an environmental management point of view, biomass quantification is important to assess the productivity and sustainability of the forest [27]. Biomass is also an important indicator of carbon sequestration, as forest biomass absorbs C from the atmosphere and stores it in the plant tissue [28].

Generally, the biomass of forest stands can be defined as the quantity of dry materials or sometimes expressed as the amount of carbon contained in woody plants (trees and shrubs) and understory vegetation per unit area (gm^{-2}). According to FAO [29], biomass is "organic material both aboveground and belowground, and both living and dead, e.g., trees, crops, grasses, stem, stump, branches, bark, seeds, and foliage." Aboveground biomass is the total amount of biological material present above the soil surface over a specified area [30]. Tree biomass is generally divided into different components such as foliage, branches, stem, stump, etc. on the basis of physiological functions. As atmospheric CO_2 sequestrates in the plant biomass through photosynthesis processes, the quantification of the vegetative biomass is essential in forest ecosystem studies in order to estimate carbon pools at multiple scales [31].

2.1.1. Forest biomass and climate change

Net primary production (NPP) is the annual plant biomass that remains in the woodland ecosystem after release of CO_2 as autotrophic respiration. Part of this NPP is subjected to another two processes, viz., decomposition and heterotrophic respiration, when biomass transfers to the forest floor as litter. These ecosystem processes are generally controlled by the plant's functional traits in a species-specific pattern [32]. Tree biomass is analogous to primary production as biomass accumulates atmospheric carbon through photosynthesis. Therefore, the net primary production (NPP) is generally estimated by measuring plant biomass and thus considered as a basic parameter in ecosystem research [33]. However, estimation of forests biomass has received much attention in recent years because of firstly, anthropogenic emissions of CO_2 are thought to be partially offset by increasing forest biomass [34] and secondly, a change in biomass regionally is associated with important components of climate change [35]. Therefore, accurate estimation of biomass is necessary for better understanding of deforestation impacts on global warming and environmental degradation at one hand and ecosystem C sequestration and storage on the other. Woody biomass is particularly important for long-term C sequestration. Generally, as a rule of thumb, 1 m^3 wood stores ~0.92 t CO_2 and the woody biomass fixes C depending on its maturity and post-harvesting use [34]. Although the use of wood product as biofuels results in the release of stored C immediately, it provides sustainable C benefits as a substitute of fossil fuel. Alternatively C may be fixed for hundreds of years if it is used for houses or furniture [34].

Deforestation is the human-induced conversion of forest to non-forest land use and causes immediate emission of huge forest carbon stocks through land clearing [1]. Due to deforestation, C is released from both plant biomass and emission of soil C due to disturbance. Forest degradation, especially nonsustainable harvesting, anthropogenic disturbance and collection of fuel wood, causes substantial reduction in forest C stock. Thus, the destruction of forest biomass has raised concerns over global warming and climate changes at a global scale. Conversely, sustainable forest management measures and preventing deforestation can play key role in mitigation of climate change [1]. As the most widely distributed terrestrial ecosystems on earth, forest contains huge C stock in living biomass (in 2005, it was equivalent to more than 1 million metric tons of CO_2 [34]) resulting the uptake of atmospheric carbon and the conversion of greenhouse gases to biomass. Therefore, forests play a significant role in the global climate change, through both CO_2 absorption and emission.

2.1.2. Estimation of forest biomass

To fulfill the requirements of the Kyoto protocol, it is necessary to estimate the removal and accumulation of C in forest biomass. The field measurement is considered to be accurate but proves to be very costly and time consuming [35]. The conventional method of biomass estimation is based on field measurements. However, this approach is time consuming, labor intensive and difficult to implement in remote areas [35]. For small-scale studies, the conventional method may be appropriate; but for studying the area of wider spatial scale or the issue of studying carbon sequestration, the use of the field measurement approach is much more challenging. The most common approach to estimate the aboveground biomass includes harvesting and measuring the dry mass of sample trees and use of allometric regression functions [36]. Allometric functions established in one area are often expected to be applicable to an areas with a similar climate and other conditions, e.g. site conditions, silvicultural measures, etc.; therefore, species-specific equations are getting wide acceptance globally. The role of remote sensing technologies for forest biomass assessment has also been recognized [37], and many studies had been conducted for this purpose worldwide. Forest information, including species, crown closure, age, height and volume, can usually be acquired through aerial photo interpretation [38]. In this technique, multiple regression models are developed based on integration of satellite images and vegetation inventory data and thus provide a method for biomass estimation. Geographic information systems (GIS) is recognized as a powerful tool for ecological studies and combination of GIS data and modeling techniques can improve the model performance. Lidar (light detection and ranging) technology is an active remote sensing tool that provides three-dimensional vertical measurement of ground target and thus can be quantified certain forest attributes such as mean stand height, horizontal and vertical crown dimensions, etc. Using such attribute data, forest characteristics like stem diameter, basal area and aboveground biomass can be calculated from allometric relationships [39]. Although remote sensing techniques provides information on stand-related parameters, Franklin and McDermid [40] pointed out that most of the orbital sensors are inadequate to fully capture forest stand parameters with high level of confidence.

2.2. Fine root and mycorrhiza

Most studies on the role of species traits in C cycling have focused on aboveground biomass, while only recently the importance of root litter for soil C cycling in forest has become apparent [41, 42]. Belowground tree biomass includes all structural coarse roots, fine roots, mycorrhizal fine roots and the mycorrhizal hyphal mycelium [43]. The contribution of coarse roots is mainly as support organs and as long-distance transport pathways, and the fine roots in association with mycorrhizal fungi facilitates nutrient and water uptake, and the uptake of nutrients often involves secretion of root exudates [44]. Although aboveground forest biomass accounts for the majority of the total accumulated biomass in the forest ecosystem, recently research studies on the functions and ecological role of root biomass have been receiving more attention, realizing the fact that root production contributes about 33% of the global annual net primary production [45]. In forest ecosystems, fine root production and turnover represent a considerable proportion of C flux, which transfers to soil organic matter pool through root decomposition and rhizodeposition [42, 46]. In general, root derived C is recognized as more recalcitrant than leaf C. In a decomposition experiment of fresh leaf and root litter from Norway spruce (*Picea abies*), Hansson et al. [47] found that roots decompose more slowly than needles due to litter quality, especially higher lignin content in spruce root (35–37%) compared with needles (15%), suggesting significant contribution of root-derived C to soil organic C storage.

A symbiotic association between mycorrhizal fungi and plant root systems is extremely important in nutrient acquisition and soil C storing and more than 90% of plant species form this mutualistic association. Fungi colonize in the root systems of plants and develops a network of filaments to absorb the soil nutrients which are not available to plants and make it accessible to plants; in return, fungus obtains C sources (energy) from the host plant. Godbold et al. [48] reported the mycorrhizal external mycelium as dominant pathway through which plant photosynthetic C compounds are transferred to soil organic matter. A study with different tree species showed that up to 60% of plant photosynthates C are transferred to soil by mycorrhizal hyphae [48]. Thus, mycorrhiza plays an important role in plant growth and root development. The biomass and products of mycorrhiza make a significant contribution to soil C and N pool. Rillig et al. [49] found that the quantity of mycorrhiza can be more than free living soil microbial biomass in some tropical forest (0.08–0.2% of total C in old forest). In addition, ectomycorrhizal and ericoid mycorrhizal fungi can produce N degrading enzymes and thus allowing them greater access to organic N sources than arbuscular mycorrhizal fungi. Therefore, soil ecosystems, dominated by ectomycorrhizal and ericoid fungi might have higher soil organic C stock than the arbuscular mycorrhiza due to greater N availability [50].

2.2.1. Measurement of fine root

The relationship between global climate change and plant growth and the role of forests as C sequester have encouraged the refinement of the estimates of root biomass and production. In comparison to aboveground biomass, the estimation of belowground biomass is more complicated and laborious. As a result, fewer case studies have been conducted to investigate tree root biomass at a stand level, and more uncertainties exist in belowground biomass estimation on a large scale. Different procedures and techniques have been followed to study the fine root biomass and turnover in the field. However, so far no one technique has been recognized as the best universally. The direct approaches are sequential soil coring, ingrowth cores, minirhizotrons and root mesh, while the indirect methods include carbon fluxes or nitrogen budget approaches and correlations of root biomass or production to pools or fluxes of limiting abiotic resources [43, 51].

Sequential soil coring: The sequential coring method is the most common approach to estimate fine root biomass and turnover in the field and thus the estimation of belowground NPP. The coring depth depends on the age and type of forest species but typically 0–30 cm depth is employed for estimating root biomass, as coring to this depth has been shown to capture a high percentage of the total fine biomass [52]. The difference between biomass estimates at each sampling date is used to estimate the fine root production. Among the different approaches of data analysis, estimating fine root NPP is the most commonly used approach where the differences in biomass between the maximum and minimum fine root biomass measured during a year [51]. A second approach, introduced by Santantonio and Grace [53], includes the Compartmental Flow Model or Decision Matrix method, which estimates changes in live and dead root biomass and losses from dead roots due to decomposition. The third approach was introduced by Persson [54] where all positive differences in root biomass between each sequence of sampling dates were summed up. If the intervals between the root sampling are too long, the intervening variation can be lost [55]. Vogt et al. [51] suggested that since a mean fine root biomass value is obtained by integrating all sampling intervals during the year, the error is less in this method compared to other methods of measuring production.

Ingrowth cores: The ingrowth core technique involves the replacement of a mesh bag filled with root-free soil into a cored or augured hole, and after a period of time when new roots grow into the core, the whole mesh bag is removed from the hole by complete excavation. The ingrowth core method has been used alone or in association with the sequential core method to estimate fine root production [55]. An over estimation of root productivity due to high proliferation of new root growth into the competition-free spaces is the major disadvantage of ingrowth core method. The artificial repacking of the soil may alter bulk density. However, it allows the direct calculation of fine root production and is thus especially suitable for comparison of fine root production between sites and treatments.

Minirhizotrons: The minirhizotron uses a clear transparent tube with a miniature camera, which is inserted into the ground. The camera fitted inside the tube can capture photographic images of fine root growth at different depths outside of the tube surface. The minirhizotron technique allows spatial sampling by the placement of multiple observation tubes in the ground. Within the last decade, the use of minirhizotrons has become a favorite method of many researchers [56, 57]. The minirhizotron technique can be used to obtain (1) quantitative information on root length, rooting density, root dynamics, lateral root spread and the depth of rooting and separation of roots into structural/functional diameters classes and (2) qualitative information on root color, branching characteristics, patterns of senescence and observations of parasitism and symbiosis [58].

Root mesh method: The root mesh method has been proposed as an alternative technique that overcomes the problems associated with the conventional methods for estimating root production [59]. Using this method, root production is estimated by placing a mesh vertically into forest soil for a specific period of time and then measuring the number and weight of root that grow through the mesh. The procedure that is much easier than other methods requires only simple equipment and causes minimal soil disturbance [59]. Recently, ground penetrating radar (GPR) is becoming popular to estimate root biomass because of less laborious and nondestructive nature of the approach. In this method, 3D images of the root system can be acquired by close range remote sensing techniques. Although a convenient and promising approach, the accuracy of the method is low for estimating entire root system [60, 61].

2.3. Soil microbial biomass and community structure

Soil microorganisms are main players of most of the biogeochemical processes in forest soils such as decomposition, respiration, fixation of nutrient elements and soil formation and can be quantified as soil microbial biomass—a broad index to express the relative amount of microbes in soil. As soil microbes themselves constitute a very active C pool, they are considered as labile C in soil. Typically, microbial biomass comprises about 1–5% of total organic C in soil [62]. The quantity of soil microbial biomass is influenced by the quantity of organic matter and also availability of nutrients from these organic sources. Because microbes are entirely depend on these substrates for their energy requirements. Due to their key roles in cycling and availability of soil nutrients, formation of soil structure and sensitivity to management and land use, microbial biomass is recognized as an indicator of soil quality. On the other hand, soil microorganisms and enzymes are the most important factors for decomposition and subsequent respiration processes and thus directly link to the storage and stability of soil organic C.

Among the microbial community in soil, bacteria and fungi are the key decomposers. They breakdown and mineralize litter and decompose most of the plant biomass produced in the terrestrial ecosystem. Fungi are generally considered to play a dominant role in decay processes, and its degrading mechanisms and enzyme systems have been well documented [63, 64]. In contrast, although recognized as important decomposer, bacteria received less attention and its role in leaf litter decomposition remains unclear [63]. In general, the mechanism of heterotrophic respiration includes the mineralization of C that is incorporated to soils through plant litter and rhizodeposition and conversion to CO_2 by the soil microbes. Therefore, soil microbial community plays a crucial role in the respiration process. In this process, fungi play the dominant role than bacteria through their higher enzyme capabilities and affect soil C cycling processes efficiently. However, the role of soil microbial community structure on soil respiration in forest ecosystems is still unclear, particularly in tropical regions [65].

2.3.1. Factors affecting microbial biomass

Many soil biotic, abiotic and environmental factors can affect the respiration process. Climatic factors, particularly temperature and precipitation, play major rules on distribution of soil microbial community from local to regional scales. As global climate change predictions indicated significant changes in temperature and rainfall patterns, it is uncertain how climate change may reshape the soil microbial community and the functioning. Temperature sensitivity of soil microorganisms can affect their relative abundance and function. It has been shown that in temperate forest increasing temperature by 5°C can change the relative abundance of bacteria and bacterial: fungal ratio [66]. Similarly, soil moisture is also an important determinant of microorganisms and change in precipitation can alter soil microbial biomass through changing structure of microbial community. However, the response of different microbial community to soil moisture may not be similar. For example, Kaisermann et al. [67] reported the shifting of one dominant fungal group to another due to small change in soil moisture availability, while soil bacteria remained constant.

Soil properties such as organic matter content, soil pH and soil texture are strongly correlated with soil microbial biomass and community structure at local level. Soils with higher organic matter content are generally high in soil microbial biomass, most probably because of higher nutrient availability. Decomposable soil organic matter can provide sufficient C and N for growth and activities of soil microorganisms and consequently higher microbial biomass in soil. Soil pH is one of the most influential factors that affect microbial community in soil. In fact soil pH regulates multiple soil parameters that are closely related to microorganisms, such as C and nutrients availability, solubility of metals, crop growth, etc. Rousk et al. [68] reported higher fungal growth and lower bacterial growth at low pH (~4.5) and below pH 4.5 inhibited all microbial growth due to release of free aluminum. Clay minerals in soil can influence the distribution of soil microbial biomass through absorbing organic compounds and influencing substrate availability. A positive correlation between clay content and soil microbial biomass C in soil was reported by some investigators [69]. Tree species could influence the composition and function of soil microbial community structure by changing forest microclimates, quality and quantity of above and belowground litter production, production of root exudates and symbiotic association with mycorrhiza and other fungi [70]. Thus, in general, it is presumed that tree species can influence soil microbial biomass C and N. However, Liu et al. [71] reported that the influence of individual species identity on microbial communities of soil and rhizosphere could be pronounced when the trees were grown in monoculture. For example, the concentration of total phenols and condensed tannin in leaf litter can suppress soil microbial biomass depending on the concentration of these compounds. Qu and Wang

[72] found that low concentration of phenolic enhance soil microbial biomass by providing energy source for microbes; however, the allelopathic effects of high concentration phenolic can suppress soil microbial biomass C.

2.3.2. Methodological aspects

Microbial biomass pool size in soil can be measured by various direct and indirect approaches. The basic principal of these methods is based on the estimation of the quantity of C present in the biomass of soil microbes. The direct methods include straight microscopic counting and counting of density-based most probable number (MPN) of microbes in soil dilution series. However, MPN method is suitable for bacterial cell and fungal spores [73]. The indirect methods include fumigation and extraction method, extraction of phospholipid fatty acids (PLFAs) and substrate-induced respiration (SIR) methods. The classical method of measuring soil microbial biomass includes the extractions of C in soil samples with and without chloroform fumigation. Fumigation process lyses the microbes in soil producing higher CO_2 than nonfumigated soils, and C in microbial biomass can be calculated from the difference between chloroform treated and untreated samples. The method yields a good estimation of soil microbial biomass although some microbes may be registrant to chloroform fumigation. Another common indirect approach is extraction of phospholipid fatty acids (PLFAs), which is the integral component of microbe's cell membrane and accurately reflects the amount of microbial biomass in soil. An important advantage of this method is the analysis of soil microbial biomass according to different taxonomic groups. Recently, DNA- and RNA-based analyses are receiving more attention, because the genetic materials are powerful tools to discriminate various microbes more accurately. DNA can be extracted and analyzed following polymer chain reactions (PCR). To analyze soil microbial diversity, 16 s ribosomal RNA (rRNA) sequencing technologies are also considered very useful approach [74]. Although there are some limitations with analyzing soil samples, the genetic approaches are seemed to be promising for the studies of soil microbial ecology in future.

2.4. Dissolved organic C in forest soil

Dissolved organic C (DOC) is a heterogeneous mixture of various organic compounds in soil solution, derived from both biotic and abiotic origins and recognized as a bioavailable and mobile C pool in soil. DOC consists of a wide range of substances ranging from low molecular weight compounds such as aliphatic carboxylic acids to large polymers of fulvic and humic acids. The major part of the DOC comprises macromolecules of fulvic and humic acids and the dominant low molecular weight compound is monocarboxylic acid [75]. Other constituents of DOC include sugars, organic acids, dissolved nutrients (C, N, P, and S) and low molecular phenolic [76].

2.4.1. Ecological significance

DOC plays important roles in many processes of C cycles in lithosphere and hydrosphere including C flux to deep mineral soil layers and aquatic bodies. Tree biomass C is transferred to soil systems through litter fall, throughfall and subsequent decay processes. Release of

DOC from vegetation and soil organic matter is recognized as an important pathway of C flux during these processes. Studies on DOC production and flux are indispensable for evaluating the impacts of environmental changes on C cycles in terrestrial ecosystems. In deep mineral soil, most of the DOC is subjected to adsorb on mineral surfaces or formation of particulate organic materials (POM), which are highly resistant to decomposition and thus potential for long-term C storage [77]. The movement of DOC from the litter layer to lower mineral soil plays an important role in the activities of belowground autotrophic and heterotrophic soil microorganisms and leaching loss of C and N. DOC is the major form of C that can be sequestrated by clay particles in the deep soil layers or hydrologically transported from the forest floor to underground and surface water resulting eutrophication and thus can damage water quality. The negative impact of DOC on C storage was also reported by some investigators [78]. DOC in dissolved organic matter provides readily available C as a source of energy for soil microbes to stimulate the degradation of old organic matter and thus can have negative impacts on soil C storage. Recently, Jones et al. [79] proposed to use DOC as soil quality indicator in large-scale soil and land use survey, because DOC contains different chemical compounds that are specific to particular functional soil and plant types.

2.4.2. Production

In forest soil, dissolved C compounds are released from both fresh and decomposed organic materials. Other sources of DOC are throughfall, microbial biomass and roots exudates [76]. Between these two primary sources, humus has been recognized as the more dominant sources of DOC than the recent fresh litter. In forest floor, incompletely decomposed litter by fungi might be the most important source of dissolved organic materials, and the microbial metabolites contribute significantly to the amount of DOC released during the degradation processes. It is obvious that DOC production is closely related to litter decomposition on forest floor and a positive correlation between carbon decomposition rate and DOC leaching in forest ecosystems has been reported by some investigators [80]. McClaugherty [81] reported that 33% of the soluble compounds in sugar maple litter gradually leached to DOC pools.

2.4.3. Factors controlling DOC production

Both field and laboratory experiments showed that the production rate of DOC was controlled largely by substrate (litter and soil organic matter) quality, activities of soil microorganisms and temperature [82]. At the field level in northern Germany, annual carbon transport from litter layers of alder and beech forest was estimated as 0.8–1.4% of annual gross carbon production [83]. In temperate forest ecosystems, it has been estimated that annual DOC flux from forest floor to mineral subsoil horizons was 115 to 500 kg C ha^{-1} [84]. But below the mineral soil layers, the concentration of DOC is very low, most probably because of fixation of organic molecules on the mineral particles or mineralization of organic substances to CO_2. The quantity of annual DOC production depends on numerous biotic and abiotic factors. In forest ecosystems, tree species identity can influence the production of DOC, because of species-specific quality and quantity of litter which is primary source of DOC. In general, coniferous trees accumulate higher amount of organic C in the forest floor than the deciduous broadleaved species and consequently affect quantity of DOC in soils. However, in a study with coniferous (*Abies grandis* and *Picea abies*) and broadleaved species (*Fagus sylvatica* and *Quercus robur*), Strobel et al. [85] found that although quantity differs significantly between tree species, the composition, chemical properties and reactivity of DOC were not influenced by tree species identity. Abiotic factors such as temperature and soil pH found to have impact on the amount of DOC accumulation in forest floor and peat soils [82].

2.4.4. Measurement

Determination of dissolved organic nutrients in soil received less attention than in fresh water and marine ecosystems. In general, dissolved organic compounds in soil can be extracted directly from soil solution by centrifugation except when soils are in dry condition [86]. A more suitable approach is chemical extraction with various extracting solvents. The most common method consists of extraction with potassium sulfate (0.5 M) at 20°C, shaking and centrifuging. The concentration of DOC in the extract is measured by total organic C (TOC) analyzer. Shaking time, temperature and the ratio between soil weight and volume of extraction solution (w/v) are the most critical factors during the analysis. Overall, the efficiency of DOC recovery depends on the sample preparation and extraction procedures [86].

3. Stability of storage C in soil

Soil organic C stock is sensitive to climatic, ecological and management changes, and thus, the stability of soil C is one of the major sources of uncertainty to forecast the impacts of forest ecosystems on future climate change. In general, the stability is thought to be an inherent property of organic matter and mainly depends on the quality, decomposability and turn over time of organic substances. However, some researchers argued that stability is not a molecular property of soil organic matter and therefore it cannot be intrinsically stable, rather the surrounding environment that developed by soil physicochemical and microbial properties is the main determinant of soil C stability [7]. For example, although lignin and lipids have long molecular structure and recognized as recalcitrant in nature, the isotopic analysis revealed that under appropriate conditions these compounds can be decomposed faster than the other organic compounds [7]. As the components of soil organic matter are complex and heterogeneous in terms of origin, chemical structure and biodegradability, the existing time of these components in soil varies widely and eventually affects the long-term storage in soil. Therefore, simply estimating soil carbon content is not sufficient to study soil carbon storage in relation to ecosystem carbon balance. It is necessary to characterize the quality of soil organic C in relation to biodegradability and subsequent residence time in soil. In the following section, we will discuss the basic concepts of C fractionation in relation to stability along with two contrasting mechanisms related to soil C stability and finally how these processes influence the deep soil C stock. A summarized description of various soil organic C stability processes and their mechanistic interpretations are presented in **Table 2**.

3.1. Fractionation of C in soil

The dynamic nature of SOM causes release of C and other elements through the decomposition processes. But the decomposition rate and turnover time of different organic compounds vary considerably. Although a wide range of physicochemical to biogeochemical and environmental factor affects the process, the chemical nature (quality) of organic compounds is the first regulator of decomposition dynamics [87]. The chemical properties of soil organic substances that reflect the ability of microbial decay referred as biochemical quality [88]. Soil organic matter is the heterogeneous mixture of substances with variety of biochemical quality. Therefore, fractionation of soil organic carbon refers to repartition of SOM into several discrete pools on the basis biodegradability. This concept of fractionation is simple and suitable for ecological research. Other approaches of carbon fraction include the physical fractionation to quantify free and physically protected organic fractions and the chemical fractionation of humus into fulvic acid, humic acid and humin [89]. Carbon in SOM has been divided into several pools on the basis of decomposition rate and turnover time. The most common approach is the two pools system, in which the C fractions having rapid turnover time is termed as labile and the slow one as recalcitrant. The labile pool is further divided into two types, labile-I and labile-II, based on the chemical composition of organic compounds [87, 90].

3.1.1. Labile C pool in soils

Labile soil carbon refers to the organic C fraction having a turnover time of hours to less than a few years in contrast with recalcitrant C having turn over time of several thousand years [91]. Chemically labile C is largely composed of carbohydrate, polysaccharides of plants (hemicellulose, starch residues) and microbial origin (chitin) and cellulose. Polysaccharides of both plant and microbial origin (hemicellulose and starch) termed as labile-I, which is hydrolysable with acids of medium concentration. On the other hand, labile-II fraction is largely cellulose and rather resistant to decomposition. This fraction is hydrolysable with very high concentration of acid [87]. However, transformation of labile-II to labile-I can be occurred during decomposition of litter. For example, cellulose can be quickly cleaved by exoenzymes into simple sugars, which are readily metabolized by soil microorganisms. Labile C compounds are generally easily biodegradable; however, some labile C cannot be subjected to microbial attack due to protection by clay particle or coated by recalcitrant materials, hence labile C must be chemically degradable and physically accessible to microbes [92].

3.1.2. The recalcitrant C pool in soils

The recalcitrant C pool consists of more stable (chemically humified and physically protected) C compounds in soil organic matter with a slow turnover time. It includes humic substances, lignin and related compounds along with fats, waxes, resins and suberins [93]. These compounds consist of large polymers which cannot pass through cell membrane; in addition, the irregular chemical structure and complex bonding cause these substances to be resistant to enzymatic attack. Using an acid hydrolysis technique, Collins et al. [94] isolated recalcitrant C and found it comprised 30–50% of total soil organic carbon in US Corn Belt soils and estimated the mean residence time (MRS) of 2600 years for this fraction.

Process	Major causes	Long staying C compounds	Residence time	Mechanisms of stability
Recalcitrance/ selective preservation	Aromatic polymer structure	Lignin, phenolics, tannin, cutin, suberin, chitin, melanin, phospholipids, ceratin	Years	Polymethylenic structure, hydrophobicity, recalcitrant C-C bonding and aromatic ring prevent initial decomposition and progressive alteration of residues leads to relatively resistant substances
	Atomic bonding	n-Alkanes, alkanoic acid	Decade	
	Complex macromolecules/poly aromatic	Humic polymers	Decade	The residues of decaying lignin mix up with fats, waxes, etc. resulting complex humus which are resistant to common microbes
	Fused aromatic structure	Black C	Century	Exact mechanism is unknown. Complex and highly condensed aromatic chemicals might prevent microbial decay.
Physical protection by aggregates	Inaccessibility of microbes, enzyme and oxygen	OC (organic C) in macro and micro aggregates (>20μm & 20–250μm)	Years-Decade	Cell secretions, root exudates and mucus act as cementing agents along with fungal hyphae to form aggregates. Pore size distribution control the accessibility (pore size limit for bacteria is < 2μm).
		OC in clay microstructure including intercalated phyllosilicates	Century	
		Trapped inside organic macro-molecules	Century	
Organo-mineral associations	Microbes unable to utilize organic molecules sorbed on the mineral surfaces	OC on the layer silicates, sesquioxides and amorphous clays	Century	Strong boding with OH- on the edges and sorption sites of phyllosilicates clays, strong Fe-O-C bonds by ligand exchange and cation bridge are the major mechanisms
Organo-metal complex	Direct interaction with Fe^{3+}, Al^{3+} and Ca^{3+} ions and through microbes	OC in podzols, metal-rich soils and in dissolved organic matter (DOM)	Century	Al toxicity inhibits functions of microbes and extracellular enzymes, alters the size of organic molecules and thus reduces microbial decay. Precipitation of dissolved organic C with metal ions enhances the stability of C compounds
Suboptimal biogeochemical environment	Unknown. Probable causes: scarcity of decomposer, nutrient and energy	OC in deep soil layers	Millennium	Exact mechanism is unknown

Data sources: von Lutzow et al. [120] and Schmidt et al. [7].

Table 2. Various processes of C stability in soil organic matter: residence time and stability mechanisms.

3.1.3. Ecological significance of soil carbon fractionation

The dynamics of soil organic matter influences different types of ecosystem services such as release and sequestration of C, energy and nutrients, affect structure and functions of soil the microbial community etc. SOM comprises heterogeneous mixture of organic compounds with different degradability depending mainly on the susceptibility to microbial attack, which influence overall residence time of organic C in soils. Therefore, the fractionation of SOM into discrete degradable pools provides information about the following ecosystem processes.

Release and sequestration of C: Increasing emissions of carbon dioxide from soil is related to rapid biological decomposition of soil organic matter and thus enhance global warming [95]. The labile fraction of soil organic matter plays a dominant role in the CO_2 efflux process due to its rapid turnover rate. Typically, the turnover rate of labile C in organic compounds such as soluble sugars, starch and carbohydrates is very rapid, as fast as a few days to a few years. In contrast, the contribution of recalcitrant C pool in long-term C storage is enormous. Lignin and some physically protected labile SOM can stay in soils for several thousand years [95]. Therefore, C fractionation provides information about both short- and long-term soil C responses to changes in the soil environment.

Provide nutrients and energy to plants and microbes: The labile C pool has potential to provide nutrients to other primary users and thus is associated with ecosystem productivity in the short term [96]. During initial stage of SOM decomposition, readily degradable C components, especially carbohydrates, are the major source of energy for microorganisms to synthesize new cells [96]. Plant nutrient elements such as nitrogen, phosphorus, potassium, etc. are released from labile portion of SOM within a short time and become readily available to other plants. As forest soils are generally deficient in plant nutrients due to high demands of major nutrients, the labile fraction of SOM plays a vital role in plant nutrient supply.

Structure and functions of microbial community: Fractionation of C into labile and recalcitrant pools is based on microbial degradability of the SOM. These two broad pools further consist of different organic compounds, and many microorganisms are involved in the degradation of these compounds. Soil microbial biomass itself is a component of labile SOC pool because of its availability to other decomposers within a short period [96]. Other labile compounds of SOM are generally decomposed by rapidly growing opportunistic microorganisms [63]. Specific groups of fungi and bacteria are involved in enzymatic degradation of more recalcitrant parts of SOM. Cellulose is degraded by hydrolytic and cellulolytic organisms that produce enzymes to breakdown polymers, and degradation of lignin is dominated by different types of fungi, especially white rot, soft rot and brown rot [63]. Labile and recalcitrant C pools in soil thus influence the size, composition and function of soil microbial community engaged in decomposition of a particular pool.

3.2. Organo-mineral complexes and physical protection to soil C

The association between soil organic matter and mineral particles is recognized as a fundamental mechanism for long-term storage of C in soil ecosystems by preventing immediate microbial attack. The process depends on the physical and chemical and morphological/structural properties of the two components, involved in the interaction, i.e. organic materials and mineral particles [92]. Different types of aluminosilicates, primary minerals such as quartz and feldspars, Fe and Al oxides and hydroxides can interact with various forms of organic C such as aromatic C, polysaccharide and carboxyl C; however, metal oxides are more active than aluminosilicates in forming association, because of large-specific surface area with high density of surface charges [97]. A simple approach to estimate the organic C fixation capacity of a soil is to study the correlation between soil organic C and specific surface area of clay particles in soil [98]. However, an approach combining mineralogical, chemical and microbial research approaches could be more effective to explain the complex associations. Soil microorganisms especially Gram staining bacteria can accumulate the adhesive substances like extracellular polysaccharides, proteins and other polymeric substances on the surface of minerals, and thus the variability in microbial community composition can affect organo-mineral associations [99].

3.2.1. Mechanistic considerations

Different types of soil minerals showed mechanistic variations in fixing organic molecules on their surface area. In phyllosilicate clays, organic C generally adsorbed as oxidized C compounds. The hydroxyl group at the edges of the clay particles forms a strong bonding with carboxyl and phenolics groups of organic matter [100]. Some expanding minerals such as smectites can hold oxidized organic matter through weak H-bonding [101]. In addition, polyvalent cation bridges between dislocated charges on the surface of phyllosilicates and negatively charged organic functional groups have been proposed as a potential mechanism for organo-mineral associations [101]. Primary minerals such as quartz and feldspars also play important role in storing soil organic C, although these minerals generally low in specific surface area and reactive sites. Jones et al. [79] reported the aliphatic C and protonated N as dominant forms of organic compounds on the surface of these minerals. Similarly, significantly higher proportion mineral-associated aliphatic C was found in soil with high sand content by some investigators [102], implying that the H-bonding of non-ionic lipids, polysaccharides and waxes might be played a major role in bonding between organic matter and quartz and feldspars. Among different types of minerals, Fe and Al oxides and hydroxides play the dominant role in sorption of soil organic compounds. Both crystalline and amorphous minerals of Fe and Al can bound organic C; however, amorphous metal oxides are more efficient in stabilizing this soil organic C [103]. Metal oxides carry a net positive charge promoting a strong exchange reaction with negatively charged organic domains such as carboxylic acid and phenolics [79].

3.2.2. Role of soil aggregates

Soil aggregates provide protection of soil organic C against microbial decay, which is attributed by inaccessibility of decomposers and enzymes due to mechanical barrier provided by aggregates. During the formation of soil aggregates, soil organic matter mainly decomposed plant debris, bacterial and fungal biomass acts as a cementing agent between mineral particles and forms microaggregates (<250 μm). These microaggregates are further combine together to form a larger aggregate, termed as macroaggregate (> 250 μm) using fine roots and fungal hyphae as binding agents [104]. However, roots and hyphae can decomposed rapidly; therefore, the macroaggregates are less stable. Some mycorrhizal hyphae contain glycoprotein, glomalin, which can form more stable macroaggregates in soil [105]. Soil textural and mineralogical composition particularly clay particles affect the formation of aggregates and hence the protection for soil C [106]. The size and distribution of intra-aggregate pore space play an important role in effective protection to organic C, because the larger the pore space the greater the oxygen and air supply resulting a faster decomposition and release of CO_2. The quality of protected C can affect the stability of aggregates depending on the pore size. If the protected C is the non-particulate labile, the microbial decomposer can have easy access to the labile C through the pore space, causing the instability of C.

3.2.3. Role of clay minerals

Isotopic and elemental analysis indicated that interactions between organic C and amorphous clays of Fe and Al oxides are the dominant mechanisms to protect soil C from the decomposers and enzymes [107]. Kögel Knabner et al. [101] reported that the stabilization through the formation of organo-mineral complexes, particularly sorption of organic molecules on Fe oxides and aluminosilicate clay minerals, is more common in deep soil than the surface. Selective dissolution extractions provide useful information about the types of stabilization present in these soils, both qualitative and quantitative. Fe and Al hydroxides and poorly crystalline phases are involved in the stabilization or organic matter. Changes in the relative amounts of these substances across the chronosequence may have important implications for the potential stability of SOM. Close interaction of the surface of fine roots to the surface of soil minerals can enhance the stability of root-derived exudates, which are generally readily decomposable, by formation of organo-clay complexes. The process may be particularly important in forest ecosystems where the absence of tillage or other mechanical operations prevents the admixture of aboveground litter with the mineral soil. Therefore, physical protection of soluble organic matter by soil mineral may be more likely in case of root exudates.

3.2.4. Methods of evaluation

Various physical, chemical and mineralogical techniques are generally applied to study the quality, quantity and turnover of organic matter adsorbed on the mineral matrix by isolating the organic and mineral parts. The physical separation of organic and mineral compositions can be accomplished by combination of ultrasonic disruption, particle size and density fractionation [108]. In the first step of physical approach, soil aggregates are disrupted by sonification and agitation in water, and secondly, fractionation of mineral particles is based on their sizes by sedimentation or dry and wet sieving methods. Density fractionation includes separation of light and heavy organic fractions by floating and ultrasonic techniques. In chemical methods, adsorbed organic substances can be extracted using hydrolyzing, oxidizing and desorbing reagents. Chemical fractionation of C in soil organic matter could be performed following acid hydrolysis methods [88]. Oxidation of unprotected organic matter by oxidative reagents is a common to isolate the labile C, leaving behind the residual C on the clay surfaces [108]. Despite the development of numerous separation techniques, the direct investigations of organo-mineral association for explicit information regarding link between mineralogy and organic compounds are still lacking. Recent studies focused on spatial distribution of organic molecules on clay surfaces with structural features of the association by scanning transmission X-ray microscopy (STXM) and near edge X-ray absorption fine structure (NEXAFS) [97, 109]. These techniques might be a powerful technique to study the organo-mineral assembles at nanoscale [106].

3.3. Priming: a potential mechanism of destabilizing soil C

In studies of soil organic C turnover, priming effect can be defined as an extra decomposition of organic C after addition of easily decomposable organic substances to the soil [110]. This process has received much attention as a potential mechanism of degradation soil organic matter [107, 111] and thus important for modeling the fate of ecosystem C dynamics under future climate change impacts. The concern over the huge impacts of elevated atmospheric CO_2 on earth's climate is leading many management efforts to enhance C storage and stability in soil; however, the protection of already stored soil C is equally important. Priming of soil organic C is a potential threat to relatively stable soil C that could be considered as a valuable ecosystem resource. Some authors argued that the higher plant productivity under elevated CO_2 supplies a large amount of labile organic C to soil continuously and thus can compensate for the loss of C by priming [111]. However, the recalcitrant and protected C (from microbial attack) that lost by priming are generally buried under top soil and obviously more stable than the fresh C [7]. Although considerable research activities are being carried out globally, our mechanistic understanding regarding the priming process is still poor.

3.3.1. Mechanisms

The mechanism of priming effect can be illustrated by 'Co-metabolism' theory [112]: addition of labile organic substance provides energy for microbes promoting the synthesis of enzymes that are capable of degrading recalcitrant C in soil and thus facilitating SOM mineralization. Labile organic substances are leached out from the fresh substrates and transported as dissolved organic C (DOC). The sources of labile C could be the root and root exudates, decomposed litter and applied fertilizers. As a part of ecosystem processes, plant litter in forest provides soil microbes with different organic compounds containing essential sources of energy and nutrients. These include readily available sugars and amino acids and easily degradable cellulose and protein, which can stimulate different groups of microbial decomposer to growth and degrade complex organic substances [107]. These mechanisms are particularly destructive for recalcitrant soil organic matter pools, which were so far protected from microbial attack due to energy and N limitation [113]. Negative priming was reported by some authors when fresh labile C suppresses soil organic matter decomposition in the presences inhibitory substances, resulting no loss of C due to the presence of labile substrates [114].

3.3.2. Factors influencing the priming process

As priming is a biogeochemical phenomenon, many biotic and abiotic variables can influence the magnitude of the process. Supply of dissolved organic C (DOC) from fresh sources is an important factor at initial stage of priming process. Release of low molecular weight compounds such as sugar, amino acids, other organic acids, etc. through hydrolysis and leaching processes from fresh organic matter is the first step of priming process. In this connection, soil hydraulic properties, especially drainage conditions and hydraulic conductivity of soil, might be important factors for DOC transportation. Favorable conditions for water movement lead to easy DOC redistributed throughout the soil profile to facilitate priming effects in buried soil organic C [113]. Secondly, soil microbial community composition and structure play a vital role in priming by promoting the growth of new microbes, producing necessary enzymes and attacking the old organic substrates. The dynamics of microbial biomass particularly the role of microbial necromass in priming mechanisms is still unknown. Future climate change projection indicated increase in temperature and atmospheric CO_2 concentration resulting increase in NPP and supply of fresh litter in forest ecosystems. Therefore, the priming effects will likely be more intensive; however, it depends on the responses of the associated factors like quality and quantity of DOC, precipitation, etc. Significant impacts of climate change on vegetation dynamics have been predicted in climate models which could alter not only DOC quality but also the community composition of soil microorganisms and thus can directly affect the priming process.

3.4. Protected C in deep soil

The major part of soil C occurs at the surface layer of soil profile because of accumulation of organic matter from above and belowground plant litter and also due to maximum soil microbial and root activities. But due to natural and anthropogenic soil, organic matter can be distributed deeper soil horizon as well. Although top 30 cm sampling depth has been recommended by IPCC for soil C inventories [115], the typical sampling depth is 1 meter to estimate soil C stocks. Most of the previous studies on soil C were limited to the upper 15 to 30 cm of soil because of the difficulties associated with sampling. However, recent studies confirm that considerable amounts of C can accumulate in subsoil layers (below 40 cm), and in some ecosystems, more than half of stabilized C is below 30 cm [116]. The deeper the soil profile the older the C and less susceptible to decay processes. The origin, nature and properties and most importantly the dynamics of deep soil C (below 1 m depth) under changing global environment are largely unknown. Fontaine et al. [107] estimated mean residence time of soil organic matter 2000–2010,000 years at subsoil layers below 20 cm. Although microbial activity is the primary active agent for soil organic matter stabilization and mineralization, the mechanisms underlying the processes at deep soil layers are still unclear. It is hypothesized that the restricted biotic activities due to oxygen limitation could be the major cause of slow C turnover in deep soil layers. In addition, subsoil C is considered more resistant to microbial degradation due to physical protection and chemical recalcitrance of substrates.

3.4.1. Controlling factors

Vertical distribution of SOC storage is controlled by many biogeochemical and climatic factors, yet at identical soil and climatic conditions, vegetation and soil texture are likely to be more active factors [26]. In forest ecosystems, plant-induced processes such as litter input and root activities might be the dominant factors in regulating the vertical C distribution. Physical processes regarding landscape formation such as erosion, landslides and flood may cause burial surface soil resulting accumulation of C in deep soil layers. One of the major sources of deep soil C stock is the root systems including the production of root necromass, release of exudates and turnover of the mycorrhizal hyphae network [117]. The contribution of these processes to SOC storage is uncertain except the contribution of root necromass because of rapid turnover of labile C in these substrates. However, C release from root exudates and mycorrhizal turnover may be potential route to soil C storage through the formation of soil micro aggregates or clay coatings [43]. Another important process leading to the distribution of C storage at deep soils is bioturbation by earthworms some of which can reach to 1–2 m depth [116] and transport of fresh SOM into the burrows and mixed with mineral soils.

3.4.2. Dynamics of deep soil C

The dynamics of deep soil is difficult to evaluate. In general, the mineralization of C in soil organic materials is largely mediated by extracellular enzymes produced by soil microorganisms. The processes vary significantly depending on the quality and quantity of substrate, available nutrients and soil properties [118]. These factors differ greatly between top soil and deep soil in forest ecosystems [116]. Little is known about catalytic properties of soil enzymes and their functional link to the microbial community structure in deep soil horizons, where abiotic environmental conditions are quite different compared with top soils. Microbial priming has been recognized as a major process that involve in mineralization of recalcitrant deep soil C [107]. Two possible situations may occur in case of microbial degradation of organic C in this condition. First, the abiotic environment at deep soil, particularly scarcity of oxygen, can create adverse conditions for microbial functioning; secondly, microbes cannot acquire sufficient energy from recalcitrant compounds to sustain their activities, resulting the slowdown of microbial decay. The exact mechanisms are not well understood; therefore, more research efforts are needed for explicit understanding of deep soil C dynamics and hence to protect this valuable ecosystem resource.

4. Conclusions

Organic C sequestration in forest is considered as a potential mitigation option for climate change by storing atmospheric CO_2 in the tree biomass and soil organic matter. Ecosystem C inventory is essential for C accounting, control of greenhouse gas emission, forest conservation and land development programs. In addition, due to establishment of the Kyoto Protocol, the interest of scientists and policy makers has been increased dramatically about the studies of C storage and its stability in soil. Among different C pools in forest ecosystems, aboveground tree biomass is the first sink of atmospheric C and plays a crucial role in ecosystem C balance. Tree roots provide many ecosystem services, of which resource acquisition from the soil and contribution to C sequestration are the most crucial. Fine roots are recognized as the most dynamic part of the root systems and regulate belowground C flux and net primary production. The typical methods of biomass estimation are extremely laborious and time consuming. Recently, remote sensing, GIS data and 3D canopy structure images are used for estimation of aboveground biomass, and software-based image analysis is receiving much attention in root studying. In spite of these methodological advances, a robust method for estimating tree biomass is still far reaching. Soil microbial biomass and community structure are extremely crucial for regulating dynamics of soil organic C and subsequent emission and storage in soil. Dissolved organic C is also important active C pool in soil that stimulates microbial activities and thus might have link with soil respiration.

Stability of soil organic C is coupled with various biogeochemical processes in the soil and therefore regulated by multiple biotic and abiotic factors. Thus, the link between different pools and processes are crucial for understanding the soil C storage and stability. Soil organic matter in the deep soil layers is potentially stable due to long residence time and surrounding soil properties. The accumulation of these highly processed C is influenced by the translocation of dissolved organic carbon (DOC) through the soil profile. Plant biomass can also be a major determinant of the vertical distribution of C in soil, through above and belowground C allocation patterns. Similarly, root activities such as incorporation of structural coarse roots, mycorrhizal fine roots and hyphal mycelium can influence the vertical distribution of organic C. The effect of clay content on soil C stability is well established, particularly in deeper layers with higher proportion of protected organic molecules. Although a large portion of fine root C returned to atmosphere through root and rhizomicrobial respiration, root residues can stay in soils for long time compared to aboveground litter and thus contributing significantly to the SOC stock. Priming process can impact soil C stock negatively. Fresh litter and root may also stimulate the microbial activities that leading to rapid decomposition of old C in soil, creating an antagonistic effect on the storage of soil organic C. Advanced analytical techniques can be instrumental for explicit understanding of these complex interactions at ecosystem level. Overall, the interdependency of various soil C pools and processes is a fundamental determinant of storage and stability of forest soil C that need to be addressed by applying advanced technologies, such as molecular, isotopic and genomic analytical toots.

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Section 2

Forest Management and Restoration

Chapter 4

A Review of Selected Methods to Determine the Economic Value of Forest: Polish Research

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Additional information is available at the end of the chapter

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Abstract

Value is a multifaceted term, and for this reason, forest economics uses a set of various categories of forest value. Two basic categories of value, with which we deal in economics, are connected with market value and non-market value. Market value is specified by the market as a result of interactions of supply and demand. Occasionally, market value is referred to in Polish literature as exchangeable value and value in exchange. Non-market value is a value ascribed by consumers to a good or service, for which there is no real market. In such cases, many methods have been used over the years to appraise the value of forests. This study presents a synthetic review of scientific thought connected with forest valuation. Concepts of static and empirical schools are discussed, indicating the role of Polish scientists in the development of forest economics in terms of forest value appraisal.

Keywords: forestry economics, forest value, statics, forest rent, empirical methods

1. Introduction

Analysis of historical interactions of man and forest indicates that forestry is a form of land use that supplies a vast array of diverse benefits. A comprehensive set of these benefits is defined as forest functions. Their range and level depend both on forest character and the adopted forest management system [1]. Identification of individual forest functions is the task of both the fields in forestry sciences and economic practice, resulting in the practically unlimited diversity of these functions and multitude of criteria for their classifications. However, the dominant aspect is connected with needs and expectations of individual groups and entire societies [2]. From the point of view of spatial management, forest combines four types of space: natural space—comprising nature elements creating conditions to sustain life of various species and having an ecological value; social space—i.e., occupied by individual communities, which strive to satisfy their needs, as a result of which it acquires social value; cultural space—as an area, in which heritage of material culture is embedded, thanks to which space is assigned cultural value; and economic space—in which economic activity is conducted, thanks to which space acquires specific usability, utility and economic value [3].

Efforts to give forestry a rational economic basis are almost as old as regulated forestry itself. During the last two centuries, many well-known forestry scientists and practitioners have worked on this topic [4]. Valuation of assets is a significant element in contemporary politics and in economy. Literature on the subject presents several definitions of value. When talking about "value", the term should either be specified or be understood as an "umbrella concept", comprising several incommensurable types of values [5]. Within the broader sense of "value" in relation to environmental aspects, it was analysed, e.g., by [6] and [7].

We need to stress the fact that the concept of economic value itself is ambiguous. It may be assumed that in forestry we deal with at least two basic categories of economic values, i.e., the current forest value (consumer value) and expected value. The former value is the commercial value of the entire forest to be felled and sold. It is forest value understood as capital allocated to provide periodical income, with the income already existing. In the latter case when we determine the expected value of a forest, it does not provide any income yet or this income is only partial. The complete income from that forest may only be expected in the future and its capitalised amount at present makes it possible to estimate the current expected value of forest. The above approach to forest value may be classified to the area of material value, treating forest as an economic object. Within this approach, we may distinguish directions, which base value directly on manufacturing or production costs (the concept by Smith from 1776), costs of reproduction (the concept by Carey from 1859) or the price of labour (the concept proposed by Marx in 1867).

Apart from the object-based school in the history of forest valuation, we may also distinguish a subject-oriented trend. It was represented by such researchers as [8–12], Wiëser [13] and primarily [14]. The subject value theory is inseparably connected with liberalism. It was initiated by the first representatives of social economics, fathers of the Austrian, French and German schools based on the so-called marginal utility theory. Proponents of the subject-based forest value indicated the need to establish value in view of individual economic goods in relation to human needs. For this reason, this school of thought considered the aspect of utility as the capacity to satisfy needs of a single individual (consumer).

Representatives of post-classical economics have formally proclaimed ideas of classical economics, but with their attempts to order and develop its theses, they have also simplified its essence. The primary representative of post-classical economics J. B. Say, when investigating economic phenomena disregarding social relations of production, treated economic processes as a specific form of barter, i.e., an exchange of services between production factors participating in economic activity. Say was of an opinion that utility, i.e., the upper limit of exchange value, is the source of value of a good. The lower limit of exchange value is connected with production costs, which following A. Smith he considered to be equivalent to the cost of wages of production factors. In the case of local competition, the market mechanism approximates the market price to production costs. In turn, the share of individual production factors in the realisation of the newly produced value is equivalent to their contribution in the generation of this value. It needs to be stressed that in this concept all production factors have the capacity to generate value. Say linked the process of generating the utility value with the general process of value formation [15].

A Polish scientist, Stefan Studniarski [16] adopted the economic theory of value proposed by Liefmann in 1917 to the needs of forest economics and distinguished the objective and subjective value in forestry. He claimed that the objective value is a technical term, since it defines technical usability of a good or product and provides essential information from the economic point of view. In turn, subjective value defines the relative attitude of a given subject to a specific good. As such, it is developed in the emotional sphere, and outlays incurred to produce this good are not the primary foundation for its valuation. At present, in forest economics, we may distinguish the economic value and the non-economic value of forest, which directly corresponds to the division proposed by Studniarski.

As we can see, the concept of value has been changing with time. As a result, it has contributed to methodological changes in the foundation for forest valuation, particularly since the concept of valuation is directly connected with value. Valuation is the process of value attribution; every valuation is based on a specific ethics determining the value system applied and uses its own 'language of valuation' [17, 18]. The most typical valuation is connected with the monetary representation of value. Monetary valuation of ecosystem services is a widely used approach to quantify benefits supplied by the natural environment to the society [19].

The greatest interest in forest valuation was observed around the 1850s, when capital was acknowledged as the dominant production factor and forest land was considered to be the only capital engaged in forest production. At the same time, the adopted sustainable and continuous utilisation of timber resources limits forest market turnover. This results in the disturbance of its marker prices. In such a situation, valuation methods were developed for forest land and stand based on income generated by forest income from timber sales.

Also in Poland, forest economics has focused on the problem of forest valuation, particularly after WWII in relation with the transformation of the socio-economic system and the adopted different approach to valuation of forest resources. Within the last 30 years, the concept of forest management has again been changed. The former raw material model has been replaced with multi-functional economy. This also determines the development of general concepts in contemporary forest management and thus—also methods to valuate forest resources. These methods aim at the establishment of a comprehensive appraisal of natural resources, i.e., material and non-material components, in order to improve their condition, to ensure their protection and stewardship in accordance with the principles of sustainable and multifunctional forest management. Apart from the productive function (as a source of raw materials), forests also serve diverse non-productive functions (not related with raw materials): protective, recreational and health-promoting. As it was stated by [20], in Poland, these functions are not marketable goods. For this reason, in Poland as well as many other countries, purchase and sale transactions are very rarely conducted in practice. So far, there are no universal solutions facilitating appraisal of the economic value of forests in such a situation. Over the years, attempts have been made to create solutions connected with valuation of forests. Generally proposed methods of forest valuation may be divided into two basic groups, i.e., based on a static notion of forest, and empirical, within which methods based on tables of stand value coefficients have been developed in Poland. Thus, Polish researchers have contributed to the still on-going search for methodological solutions connected with forest valuation.

2. Static methods of forest valuation

The problem of a static approach to forests has been discussed for years, e.g., [21–25], Mohring (2001) and [26]. In the history of forest economics, the period of intensive development of the so-called forest statics may be described as the classical period, since it was influenced by the liberal free market economics of Adam Smith in the form of the science on appraisal of forest value and profitability of forest holdings based on a static notion of forest. It was the period in the development of economics concerning forest management starting in the first half of the nineteenth century. It has provided theoretical foundations for the valuation of forest value and profitability in forestry [27].

The period of the so-called static approach to forest valuation was connected with changes in the concept of the objective of forest production. At that time, the material function of forest production was predominantly an economic function. The idea of sustainable timber harvesting was replaced by the objective of ensuring maximum income on capital invested in a forest holding, a concept consistent with the spirit of free enterprise. This was manifested in the idea that "the task for foresters is to obtain the possibly high financial income from the forest rather than the greatest timber volume" [Heeg [28]]. This objective was predominant in forest management systems in many countries for the next 150 years. It was manifested in the mathematical form in the well-known formula of economic equilibrium by M. Faustmann used to determine such age of stand (u), at which income from timber harvesting (Du) would be equal to the cost of compensation (payment) of land capital (B), represented by interest on that capital calculated for the period, over which it was frozen in forest production [29]. Faustmann's basic economic equilibrium formula, published in 1849, influenced the economic model of forestintermediate management for many years:

$$Au + \sum D_n \times 1.0p^{u-i} = \mathrm{B} \times (1.0p^u - 1) + \mathrm{C} \times 1.0p^u \qquad (1)$$

where Au is the netto income (cash unit/hectare), Dn is income (revenues - costs) from using the forest before the harvest period, B is the value of land capital (cash unit/hectare), C is the cost of forest regeneration (cash unit/hectare), p is the forest interest rate (%) (e.g., for 4%, p = 4), u is the rotation age, and i is the current age of stand.

As it was reported by [26], a common notion in forest and natural resource economics is that the celebrated 'Faustmann formula' was discovered by a young Hessian forester Martin Faustmann in 1849 and that the 'Faustmann rule' or the Faustmann-Pressler solution to the optimal rotation age was derived from it a decade later by a distinguished professor of forest mathematics Max Robert Pressler (e.g., [30, 31]). The Faustmann model is well accepted in economics [32, 33], and it has a "myriad" of applications [23] with high practical relevance worldwide [34]; however, in a wide range of communities of forest and environmental scientists, forestry managers and other forestry-related stakeholders, there is still a very sceptical view towards this model.

Deegen et al. [35] claim that the misunderstanding is threefold: first, overexploitation and deforestation are interpreted as a consequence of unrestrained competition and uncontrolled markets. Second, the effectiveness of decentralised coordination of the self-interest actions of individuals will be totally underestimated. Calls for limitation of competition and market controls, i.e., replacing decentralised with centralised coordination, become louder. Third, how can the prices that reflect today's needs know the concerns of future generations? As a result, they later proposed empirical methods, which are discussed in the later part of the paper.

The scientific foundation for the static approach to forests was provided by the model of a normal forest, i.e., a forest holding composed of pure (single-species) forest management units. In the forest static approach, a normal forest is a forest with an adequate model structure, i.e., normal (appropriate) proportions of age class areas, and thus the same normal reserve, growth increment, annual cut, etc. A forest holding with the abovementioned normalcy characteristics in theory guaranteed continuity and uniformity of forest management both in the temporal and in the spatial terms. The static model of a forest holding determined the organisation and management method; more importantly, it determined the methods of forest valuation and assessment of profitability for that forest holding. It defined the forest holding as a planned system of concurrent activities, which over a certain period are to supply staple goods required to satisfy human needs. In the static forest approach, a forest holding is not an actual entity, but rather purely abstract (the so-called normal forest), not present-day, but based on expected effects in the distant future and not constituting a finite whole, but a mechanical set of individual stands. Forest itself was treated as capital, which had a primary task to generate income for its owner. In view of the above, methodological discussions on the appraisal of forest value based on the static approach were founded on the assumption that forest needs to be treated as capital, as if it was deposited in a bank and from which a certain amount of interest is due.

Finally, representatives of the static approach calculated land value mainly using Faustmann's formula [29]:

$$B = \frac{A_u + \sum D_n \times 1.0\,p^{u-i} - C \times 1.0\,p^u}{1.0\,p^u - 1} - \frac{V}{0.0p} \tag{2}$$

where V is the annual cost of forest administration; the other denotations as previously.

This formula became the paradigm of the approach to forest valuation, assuming forest to be a static entity as well as the foundation for the theory of the highest ground rent, understood mathematically as the interest on forest soil capital. We need to stress here that this formula was based on the so-called static assumption that forest economics is based not on an already existing stand created by nature, but unforested land. This led to a further assumption that interest on capital, such as land (B) and interest on administrative capital (involved labour) (V), was treated as production costs, which should be covered by net income. In turn, net income was determined from the difference in gross income and expenditure deferred to stand rotation. In the estimation practice, representatives of the static valuation school most frequently used approximation formulas for income forest value (formulas by Höninger, Riebel, Glaser).

It was also attempted to determine land value based on incurred costs for the establishment of tree culture and costs of land purchase. For example, Heyer, on the basis of Faustmann's formulas (Eq. (2)), proposed determination of land value in relation to the land purchase price and costs of establishment of a forest plantation on that land. This assumption was expressed in the following equation:

$$B = K \times 1.0p^{i} + \frac{C \times \left(1.0p^{i} - 1\right)}{0.0p} \tag{3}$$

where K is the capital (land purchase price); the other symbols as previously.

Considerable contribution to the further development of this scientific school was provided by a Polish scientist, Stefan Studniarski, who pointed to the fact that in the case of the income-based method of forest land valuation, this value is established by discounting the value of the product (such as timber in forestry) reduced by forest regeneration costs deferred over the entire rotation period. As he stated, in this situation, the following assumption has to be true: the value of future utility (timber harvested over the entire stand cycle) may be calculated at the time of stand regeneration based on these costs and the assumed forest interest rate. However, this is not the case, since the value of this product is modified not by the incurred costs and the assumed forest interest rate, but rather by the sale prices of individual dimensional timber grades. The abstract character of the concept for the income land value completely disregards actual processes of exchange of goods and services.

In the case of stands, their value was calculated by determining the present value of expected pure income from one rotation cycle. In this case, investigations within the static approach were based on a mathematical formula proposed in 1854 by Oezel:

$$HE = \frac{Au + \sum D_{n} \times 1.0p^{i} - (B + V) \times \left(1.0p^{u-i}\right)}{1.0p^{u-i}} \tag{4}$$

where HE is the stand value; the other symbols as denoted previously.

The above formula was modified by Heyer, who proposed calculation of stand value based on the sum of present values of all costs (the value of land and human capital) less the value of potential income.

$$HE = (B + V) \times \left(1.0p^{i} - 1\right) + C \times 1.0p^{i} - E \times 1.0p^{i} \tag{5}$$

where E is the present value of potential income obtained from the forest; the other symbols as denoted previously.

Also in this case, the same Polish scientist, Stefan Studniarski, indicated that from the economic point of view all applied cost items (soil (B), management costs (V) and forest regeneration (C)) are abstract and as such they are of little use in accurate valuation.

Representatives of the static approach to forest valuation were also familiar with the concept of sale value of the stand, which later lost in importance. The sale value of land was considered to be such a value, which a given land plot has in relation to sale of its land and analogously, it was proposed to determine the sale value of stands by comparing it with other such transactions. However, as it was observed earlier, it was problematic since such comparisons were sometimes impossible. For this reason, capital solutions connected with economic forest valuation should be considered when developing theoretical foundations for forest valuation. This is particularly important, since-as it was shown by Paua Anthon Samuelson, awarded the Alfred Nobel Bank of Sweden prize in economics-the Faustmann concept concerning forest valuation may be applied also in new areas of economic activity (Plotkowski [36]). Stand value may be estimated by capitalisation of cash flows and updating the future (expected) revenue and net costs. Applying the theory of money value, we may determine the value of a stand at any specific age (NPV) as a sum of updated (to that age) net cash flow (NCF) in individual periods (years) of its life (Zajac and Swietojanski [37]). Net cash flows constitute the difference between future revenues and costs of their generation and discounted interest:

$$\mathrm{NPV}_{\mathrm{i}} = \mathrm{K0} + \sum_{\mathrm{i=0}}^{\mathrm{u}} \frac{\mathrm{NCF}_{\mathrm{i}}}{(1.0\mathrm{p})^{\mathrm{i}}} \tag{6}$$

The Faustmann theory of discounted cash flow analysis was presented by [25]. They described three valuation approaches: compounding costs, compounding discounting annuities and discounting future cash flows [25]. At present, the method of forest value estimation is not a problem. It is rather the determination of an adequate interest rate.

Apart from the concept to treat forest as bank capital within the framework of the static forest valuation approach, another school was founded indicating that forest is a form of capital, which had the primary task to generate income to its owner in the form of forest rent. In this case, forest value is directly related to the amount of income, which may be obtained assuming sustainable forest management. The volume of this net income corresponded to capital for a specific rent:

$$r = A_u + \sum D_n - (C + V) \tag{7}$$

where r is the value of net income (rent); the other symbols as denoted previously.

In view of the above assumption, it was proposed to calculate forest value assuming capitalisation of net annual income (rent) according to the formula:

$$W = \frac{r}{0.0p} \tag{8}$$

where W is the forest value calculated based on capitalisation of net annual income; the other symbols as denoted previously.

In forest economics, the above formula is referred to as the capitalisation equation, while the term 1/0.0p is called the capitalisation index or valuation index.

The method of determining the value (price) of forest through capitalisation of net income was identical to the method of determining land price, which according to Marx in capitalist economies is simply capitalised income of land lease. The theoretical foundation for this formula was provided by the proportion:

$$\frac{r}{Ko} = \frac{p}{100} \tag{9}$$

symbols as denoted previously.

It results from the equation that the ratio of the value of rent (net income) to forest value (forest capital) is identical to the ratio of the part (percentage) to one (to 100%). The above proportion may be used not only to calculate forest value, but also the value of the rent:

$$r = Ko \times 0.0p \tag{10}$$

as well as the value of interest rate:

$$0.0p = \frac{r}{Ko} \tag{11}$$

symbols as denoted previously.

It needs to be stressed that the valuation multiplier is inversely proportional to the adopted interest rate. Thus the higher the interest rate, the higher the rent, but the lower the value of forest (capital). As a result, we should be cautious when establishing interest rates. The determination of forest value by capitalisation of net income resulted in very low or even negative values for forests currently having no stands. In the mid-1800s, this led to the development of the so-called component methods of forest value estimation. In terms of the approach to forest as a form of capital both in theory and in practice, two schools of rent accounting were developed, i.e., forest rent and ground rent. In the theory of the forest rent, the entire forest, i.e., land together with stands and other property located on that land, e.g., buildings, structures and roads, was assumed to constitute the initial capital for a forest holding. Forest value was calculated by capitalisation of the forest rent, typically annual and perpetual. This method is also called the indirect method, since forest value is calculated from the income generated by the forest. In the theory of the ground rent, both forest land and fixed assets located on that land, i.e., buildings, structures and roads, were assumed to be the initial capital of a forest holding, while stands were treated as products with a long maturation (turnover) period and as such they were elements of working capital. Thus the value of forest land and the value of stands were calculated depending on age. As a result, forest value was composed of two elements, i.e., the value of land and the value of stands. This method is sometimes called the direct or component method.

A significant contribution to forest value estimation using the static approach to forest valuation is connected with the studies by of a Polish scientist, Ostrowski. In 1976, he proposed the determination of the economic value of forests from the quotient of net income and forest interest rate, increased by adding the value of fixed assets involved in production. The result also needed to be corrected by the quality index of the forest holding(s). The example forest holding quality indexes for Poland proposed by Ostrowski ranged from 0.8 to 1.2. The proposed formula took the following mathematical form:

$$Wg = \left(\frac{r}{0.0p} + Wst\right) \times s \tag{12}$$

where Wg is the economic value, Wst is the value of fixed assets engaged in production, s is the forest holding quality index; the other symbols as denoted previously.

In turn, Ostrowski proposed to calculate the level of rent based on information concerning net income, the forest management plan and actual forest management as well as the level of commercial afforestation:

$$r = Dc + (E - U) \times Sp + pz \times w \tag{13}$$

where Dc is the annual net income based on accounting data (net income), E is the annual cut (merchantable timber), U is the volume of harvested dimensional timber grades (merchantable timber), Sp is the mean stumpage price per 1m^3 timber, pz is the area of commercial afforestation (apart from regeneration of current logging sites) and w is the mean costs of reforestation of 1 ha.

The formula proposed by Ostrowski [38], appropriate from the point of view of both accounting and forest valuation principles, could be used in practice, provided an economic equilibrium was found between the value of forest production and costs incurred on the operations of the forest holding. This formula assumed that net income is generated from forest economic activity. As it is generally known, in forestry, this condition is not always fulfilled due to the effect of the differential rent. For this reason, another Polish scientist Podgórski proposed a modification of Ostrowski's formula and replaced forest rent (r) with the value of the annual cut calculated from the product of volume of timber to be harvested in a given year (E) and the stumpage price of a cubic meter of timber (Cnp). As a result, Ostrowski's formula modified by Podgórski took the following form:

$$Wg = \left(\frac{We}{0.0p} + Wst\right) \times s \tag{14}$$

where We is the value of the annual cut; the other symbols as denoted previously.

As it was previously mentioned, the static approach to forest value estimation is not problematic; it is rather the determination of an adequate interest rate that may be a problem. In the case of methods based on the static approach and the percentage or rent calculations, the interest rate is of paramount importance. The manner of its determination has always been considered dubious, as it has not been definitely established whether it should be identical in the percentage and rent calculations. Nevertheless, it was assumed that it should be constant throughout the entire rotation during the stand life. Attempts were made to apply the ordinary interest rate and then compound interest, and subsequently, it was the arithmetic mean and the geometric mean of the two. Finally, it was decided to refer to compound interest, but problems with the adoption of the level of interest rate could not be resolved.

Some literature sources present an opinion that interest rate is a comparative measure, which may be used to determine value, and as a result of such an economic character, it has to be uniform in all valuation cases. A discussion on this subject was presented by [39], who indicated that some researchers, e.g., [40], assumed a constant value of interest rate at 3%. In turn, [41] claimed that the conventional (risk adjusted) discount rate to be applied on farmland investments was 3% and on pure money capital 5%. For this reason, he felt it reasonable to be something in between for forest, thus as an integer, 4%. Lehr and Borggreve [42] were strong opponents of a low interest rate, and for a permanent forest culture, they applied rates of 4–6%, while for less permanent cultures, they applied rates of up to 10%. Some researchers express an opinion that in the case of capital with an extensive productive life interest rate (p) is 1–3% [43]. In turn, Piekutin and Skreta [44] claimed that the level of forest interest rate decreased from 5% in the early 1800s to approx. 1% at present.

In the twentieth century at the turn of the 1980s and 1990s, Polish scientists Podgórski and Kikayi [45] investigated the empirical determination of the volume of forest interest rate (p). They stated that a reliable method to determine the forest interest rate is provided by the annual cutting budget (Eu) in relation to the structure of standing timber resources (Znp):

$$0.0\mathrm{p} = \frac{\mathrm{E}}{\mathrm{Znp}} \tag{15}$$

In financial terms, they proposed to calculate the volume of forest interest rate (p) from the ratio of the value of the assumed yield and the value of standing timber resources.

Podgórski and Kikayi [45] were of an opinion that replacing the category of net income from a forest holding with the volume or value of the allowable annual cut is more consistent with the contemporary forest holding practice, while modified forest interest rates may be used for monetary valuation of both forest land and standing timber resources. In Poland, the forest interest rate calculated in this manner is approx. 2%.

More recently, [46] modified that opinion and proposed calculation of forest interest rate from the ratio of the volume of harvested timber (U), which is currently the source of income, and standing timber resources (Znp), or from the ratio of current increment of forest (Pb), which will be the source of income in the future, and standing timber resources:

$$0.0p = \frac{U}{Znp} \; or \; \frac{Pb}{Znp} \tag{16}$$

where Pb is the current increment; the other symbols as denoted previously.

For example, in Poland, the increase of forest (Pb) is 60 million m^3 and standing timber resources is 2 billion. It follows that the forest interest rate is 3%.

$$0.0p = \frac{Pb}{Znp} = \frac{60\,000\,000}{2\,000\,000\,000} = 0.03; p = 3\% \tag{17}$$

From the natural point of view, the above formulas for the determination of forest interest rate may be considered rational. Current increment in stand volume is the actual natural effect of forest management, which may generate income in the future. For this reason, it is proposed to call the calculated forest interest rate the natural interest rate. In turn, the degree of forest use, resulting from the allowable annual cut, is the technical effect of forest management, since its volume is determined by the principles of calculation and establishment of allowable annual cuts.

Investigations conducted at the Department of Forest Economics, the Poznań University of Life Sciences (Poland), confirm that from the technical and natural point of view, the forest interest rate ranges from 2 to 3%.

3. Empirical methods to estimate forest value

Empirical methods to estimate forest value were developed as a result of criticism of the static methods voiced, e.g., by Glaser, Hölinger and Köstler. Critics of methods based on the static approach (particularly the percentage methods and forest rents) were of an opinion that using one formula it is not possible to estimate forest value regardless of stand age. This is because when there are no mature stands there is no income and no rent and there is no simple dependence throughout the entire stand life cycle between income (rent) and costs of running forestry operations and forest value. Thus, with years, representatives of the static approach to forest valuation replaced static formulas for the calculation of stand value with simplified empirical formulas. In the appraisal of stand value, two developmental phases were distinguished, i.e., merchantable mature stands (mainly at rotation age) and non-merchantable immature stands. Later from the group of immature stands, second-growth forests and sapling stands were distinguished as a separate group of stands. Such a division into three groups of stands, i.e., second-growth forest and sapling stands (stands of the youngest age classes), stands of medium age classes as well as mature and overmature stands, is still used in empirical methods. The value of second-growth forest and sapling stands was determined using the outlays method, i.e., based on the sum of all costs incurred on the establishment, tending, protection and other costs paid from the moment of establishment to the moment of forest valuation. Any income from commercial thinning was deducted from this value. This method has not been disputed. However, it is considered objectionable to add up costs over longer periods, i.e., 20–30 years. It is acceptable to estimate the value of mature stands (at rotation age and older). The value of these stands is estimated using the realisable value method, i.e., based on income, which may be attained at cutting the entire stand. The greatest problems are posed by estimation of value in the case of immature stands in medium age classes. For such stands, the so-called expected value is determined. A certain (acceptable) value is the value of mature stands, and for this reason, it has been attempted to reduce (discount) this value using respective indexes to the value at the estimation age. In this respect, Glaser and his followers were most successful. At the initial stage of his research, Glaser assumed that the value of stands changes in proportion to their age. As it was reported by [39] and Podgórski [47], age as the reducing factor was adopted for the first time by Martin. Thus assuming that the age of stand is of key importance for its value, the following proportion was developed:

$$\frac{\mathrm{Ai}}{\mathrm{Au}} = \frac{\mathrm{i}}{\mathrm{u}}, \text{ thus } \mathrm{Ai} = \frac{\mathrm{Au} \times \mathrm{i}}{\mathrm{u.}} \tag{18}$$

where Ai is the expected value of estimated stand and Au is the realisable value of stand at rotation; the other symbols as denoted previously.

This formula, while mathematically correct, provided overestimated results, particularly in the case of younger stands. In further research, it was assumed that stand value changes in proportion to stand age squared. Thus a proportion was constructed:

$$\frac{\mathrm{Ai}}{\mathrm{Au}} = \frac{\mathrm{i}^2}{\mathrm{u}^2} \text{ therefore } \mathrm{Ai} = \frac{\mathrm{Au} \times \mathrm{i}^2}{\mathrm{u}^2} \tag{19}$$

symbols as denoted previously.

In turn, this formula gave overestimated results. Today, we know that there is no simple dependence between the value of a stand and its age or age squared. The third version of Glaser's formula is referred to as the corrected Glaser's formula. In order to eliminate inaccuracies of the previous formulas, Glaser proposed a further correction introducing to the formula regeneration costs for 1 ha forested area (C):

$$Ai = \frac{(Au - C) \times i^2}{u^2} + C \tag{20}$$

symbols as denoted previously.

Under Polish economic conditions also this formula did not yield satisfactory results. For this reason, Glaser's formulas were adapted to the situation of Polish forestry by Prof. Jan [48]. A significant contribution of Świąder to forest economics in the area of forest valuation is connected first of all with the determination of age class periods, within which respective methods of forest value estimation should be applied. He stated that the reproduction cost method, which replaced the method of generated costs, should be applied to 20-year periods in the case of coniferous species and to 30-year periods for broad-leaved species except for oak, for which it is 40 years. Świąder proposed to calculate the value of stands of that age using the following formula:

$$Wr = (C + Kk \times n + Ko \times i) \times z \times p \times BWP \tag{21}$$

where Wr is the reproduction value of second-growth forest or sapling stands, C is the one-off cost of establishment of second-growth forest, Kk is the recurring costs connected with forest tending measures, n is the number of tending operations, Ko is the protection and administrative costs, i is the age of second-growth forest or sampling stand, z is the stocking index, p is the area and BWP is the site index.

The value of costs included in formula 19 needs to be understood as mean costs for the last three years calculated on a regional scale, rather than a single forest holding. We have to stress here the proposal of [48] to apply the site index (BWP). That researcher rightfully believed that the mean reproduction cost is an equivalent of stand value only in the poorest forest sites. The proposed use of the quality conversion coefficient aimed at the adjustment for the higher value of second-growth forests and sapling stands growing on quality sites. This conversion coefficient is calculated from the product of the growing stock of a mature stand on a site of a given quality class to the growing stock of a mature stand in the site of the lowest quality class for a given species. Data on the growing stock are collected from respective stand yield tables. It needs to be stressed here that in the above formula we do not deal with the cost price (expenditure), but rather the value of forest reconstruction in a given year under specific conditions based on current costs. The use of current costs is a certain form of deferral of costs incurred 10 or 20 years ago, at the time when the actual expenditure was incurred.

Ref. [48] modified also Glaser's formula concerning the expected value and established for that method specific age boundaries. He reported that this method needs to be applied above the limits established for the reproduction cost method, i.e., for coniferous species starting from 21 years to the assumed rotation and for broad-leaved species starting from 31 years to the assumed rotation, while for oak stands from 41 years to the assumed rotation, respectively. Valuation of such aged stands should be performed according to the following formula:

$$Ai = (Au - C) \times \frac{i^2}{u^2} \times z \times p \tag{22}$$

symbols as denoted previously.

Other Polish researchers also proposed their own modifications to Glaser's formula for the calculation of expected value. For example, Trampler proposed to calculate the expected stand value from the following formula:

$$Ai = C + (Au + \sum Du - C) \times \frac{i}{u} - \sum Di \tag{23}$$

where Di is the value of pole stand to the age of stand at valuation (i) and Du is the value of intermediate stands to rotation age; the other symbols as denoted previously.

Another Polish researcher [49] was of an opinion that the conversion factor resulting from the ratio of estimation age (i) and rotation age (u) of stands may not be a key element in the estimation of stand value. He argued that labour time is the only inherent condition for forest production operations. For this reason, he believed that an appropriate approach to the valuation of medium-aged forests is provided by the method based on outlays of required labour, which needs to be performed in the process of timber production. A logical consequence of this statement was to limit the entire calculation, using cost ratios, to the potential value by reducing the actual value of a mature stand. For this purpose rather than the coefficient resulting from the ratio of stand estimation age and stand rotation age, he used the ratio of stand production costs to stand estimation age and total stand production costs to rotation age:

$$Ai = (Au + \sum Du - \sum Di) \times \frac{ki}{K} \times z \tag{24}$$

where ki is the incurred costs of timber production to age of stand value estimation (i) and K is the total costs of timber production incurred until rotation age; the other symbols as denoted previously.

However, in this method, it is the reduction factor (ki/ *K*) that causes problems, since in practice we may use solely calculated costs, as actual silviculture costs incurred over a period of several decades are difficult to assess. Moreover, it was assumed in that method that stand value changes in proportion to costs incurred on silvicultural measures; as we know—also from studies outside the field of forestry—there is no simple correlation between incurred costs and the value of produced goods (apart from specific cases).

Another method applied to valuate mature and overmature stands was based on realisable value. This method provides the actual market value of stands, since it is established based on actual, i.e., market timber prices. For this reason, in practice, it has been considered least controversial. It was assumed that for mature and older stands the value of these stands is equivalent to the value of timber dimensional grades found in the estimated stand less costs of timber harvesting:

$$Au = \sum (M \times P) - Kp \tag{25}$$

where M1, M2 … M is the volume of individual dimensional grades; P is the prices of individual dimensional grades; and Kp is the costs of timber harvesting and extraction.

We face problems with the practical application of this method, since in some cases stands of the same species and in the same age are separated only by a compartment boundary or a road, but they differ in their rotation age. For example, 90-year-old pines may sometimes be estimated based on the expected value, since their rotation age was established at 100 years, while in another case, on the basis of realisable value, as their rotation age was determined at 80 years. We know from experience that stand value fluctuates stepwise (it is not a continuous function) with changes in the calculation formula. This is the case when the reproduction cost method is replaced with the expected value method or the transition from the method based on expected value to that of realisable value. The abovementioned methods of estimating stand value, such as proposed by Prof. J. Świąder, were implemented in Poland in 1963 on the power of the directive of the Ministry of Forestry and Wood Industry (the Journal of Laws Dziennik Urzędowy Ministerstwa Leśnictwa i Przemysłu Drzewnego, no. 4, 1963) [Podgórski [50]].

4. Yield table methods in estimation of forest value

An original contribution of Polish science to research on stand value estimation is connected with the development of stand yield tables. The three methods presented above, i.e., reproduction costs, expected value and utility (realisable) value, were applied at the Forest Research Institute when developing stand value tables for individual types of forest trees depending on their age and site quality classes. These tables were amended and updated several times. They may be extensively applied in practice, particularly in the determination of losses due to premature stand cutting.

As it was reported by [20], for the purpose of yield tables, stand value based on the incurred costs was determined using the following formula:

$$Wki = C + Kz \times n + Ko \times i \tag{26}$$

where Wki is the stand value calculated based on incurred costs; C is the one-off costs connected with establishment of second-growth forest; Kz is the costs recurring several times during stand growth; Ko is the fixed costs, recurring annually; n is the number of cost recurrences Kz to age "i"; i is the age of valuated stand.

The expected value of stands was established using the formula:

$$Wi = \frac{C + (Wu + Du) - C}{u} \times \frac{i}{u} - Di \tag{27}$$

where C is the cost of establishment, fill-in planting and tending of second-growth forest; Du is the realisable stumpage value of pole stand to rotation age u; Di is the realisable stumpage value of pole stand to age i; Wu is the realisable stumpage value of stand at rotation age; u is the rotation age; and i is the age of valuated stand.

Realisable value of the stand was determined using formula 23 given above.

Calculated values of stands were determined using the above mathematical functions and they are given in the form of tables depending on age and stand quality class in terms of 5-year intervals. Tables contained values in monetary units for a fully-stocked stand of 1 hectare. For this reason, these data were becoming obsolete rather fast and needed to be updated. An essential change in the manner of stand appraisal was introduced in the 4th edition of the tables published in 1985, in which it was decided not to present stand value in monetary units; instead, an arbitrary unit factor (value index) was introduced in the form of "1 m^3 2nd grade pine lumber". This made it possible to determine stand value with no frequent table updates required to compensate for changes in prices and costs. In the next 5th version of the tables published in 1991, considerable changes were introduced to the method of stand value calculation. These changes were connected with the method to determine value based on incurred costs and expected value. Value based on incurred costs (Wki) was calculated using the following formula:

$$Wki = C + Kp \times n + (Ko + r) \times i \tag{28}$$

where C is the one-off costs connected with the establishment of second-growth forest, Kp is the costs recurring several times during stand growth, Ko is the fixed costs, n is the number of recurrences Kp, r is the forest rent and i is the stand age.

In this version of the tables, the expected value (Wi) was determined using the following formula:

$$Wi = (Kz + ri) + \frac{(Wr + Wp - Kz - ri) \times i}{u} - Wpi \tag{29}$$

where Kz is the cost of establishment of second-growth forest, ri is the forest rent at age i, i is the current stand age, u is the rotation age, Wr is the value of timber from mature stand, Wpu is the total value of pole stand to rotation age and Wpi is the total value of pole timber harvested to age i.

In 1997, due to the change in timber classification introduced in Poland in 1993 and its transformation from use-based to grade and dimension-based, the 6th version of the stand value tables was updated. The previously adopted arbitrary conversion unit of "1 m^3 2nd grade pine lumber" was replaced by a new unit, i.e., "1 m^3 softwood lumber". Moreover, the tables were adapted to amended legal regulations.

In 2002, the next 7th version of the tables was prepared, in which new stand value indexes were established, taking into consideration current timber prices and costs of its production, as well as the new conversion unit, i.e., "1 m^3 total timber". In this version, the respective value indexes are given in terms of 1-year intervals. These tables (the 7th version) constitute an attachment to the currently binding Regulation of the Minister of the Environment of 20 June 2002 and they are the legal foundation for the estimation of losses due to single indemnity for premature stand cutting (the Journal of Laws Dziennik Ustaw no. 99, item 905).

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Planting and Harvesting Decisions: A Review and Extension to the Case of Two Alternative Tree Species

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Additional information is available at the end of the chapter

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Abstract

When to optimally harvest even-aged trees is a dominant concern in forest economics. In the literature, it was considered when the land is available for just one harvest (Wicksell setting) or multiple harvests (Faustmann setting). In this chapter, we will review the rotation lengths under both settings and focus on the impact of timber price variations on planting and harvesting decisions when one or two tree species are available for planting. When timber prices are rising but a species must be replaced by a more attractive species, rotation lengths before the switch are either constant and equal to the Faustmann's rotation, increasingly higher than the Faustmann's rotation, or decreasingly lower than the Faustmann's rotation. If timber prices are stochastic, forest managers prefer longer rotation lengths when the switch to the alternative tree species is about to occur as a means of postponing decision-making and waiting for more information related to timber prices. The possibility of replacing the planted species with an alternative species increases the land value, especially when timber prices are uncertain.

Keywords: forestry rotation, land value, alternative species, alternative land use

1. Introduction

A dominant research topic and practical concern in forest economics have been the age at which even-aged trees should be optimally harvested and how this age, known as the rotation length, depends on timber price levels or price change rates. In the literature, it was considered under two settings: when the land can be planted just one time, referred to as the Wicksell's rotation, and when the land can be planted an unlimited number of times, referred to as the Faustmann's rotation.

The rotation problem was first resolved using some simplifying assumptions, such as constant timber prices and the availability of just one species for planting. The rotation equation that allows for the calculation of the optimal tree rotation length requires comparing the net marginal benefits of letting trees grow to the opportunity cost of the current harvest augmented by the land opportunity cost. The land's value stems from all future harvests or other alternative agricultural or non-agricultural uses.

Gradually, certain assumptions were relaxed. For instance, a generalized version of the Faustmann formula applies when stumpage prices and costs are known time functions in [1]. Rising timber prices were considered in [2] and changes in stumpage prices and costs in [3]. Others considered stochastic timber prices as in [4–9].

The presence of another alternative tree species was considered, for instance, in [10], in which an alternative species is available for afforestation. Two mutually exclusive tree species exist in [11, 12], but choosing one species leads to permanently losing the other. More recently, forest rotation was studied in situations in which two tree species existed with rising timber prices [13] or stochastic timber prices [14].

The rest of this chapter is organized as follows. In Section 2, we will first review the basic equations that allow for determination of the rotation lengths under Wicksell and Faustmann settings. We will then focus on the impact of timber price variations on rotation length and planting decisions.

In Section 3, we will present and discuss the model of two tree species as in [13, 14], in which two alternative tree species are available. Both species have two different stochastic timber prices that are assumed to follow a geometric Brownian motion and two deterministic age-dependent volume growth functions. We, therefore, assume the existence of two alternative species and the availability of bare land to be planted with a single species once (Wicksell setting) or an unlimited number of times (Faustmann setting). When the number of plantings is greater than one, it is possible for forest managers to replant the same species or repeatedly switch to the other species. The switch to the alternative species can also be considered a switch to an alternative agricultural or non-agricultural use.

In Section 4, we will study the problem of choosing between two alternative tree species with non-stochastic rising timber prices. This is an extension of Faustmann's problem with the presence of two alternative tree species. When the switch to the alternative species occurs, planting and harvesting the current species take place with rotations that are either increasingly higher, decreasingly lower, or equal to the Faustmann rotation.

In Section 5, we assume that timber prices are uncertain. More precisely, we discuss the problem of choosing between two alternative tree species with stochastic timber prices when the land is available for one rotation only. This is an extension of Wicksell's problem. This section helps us understand the value of waiting for more information before making decisions with irreversible consequences, a situation that will appear in Section 6 under the Faustmann's setting.

In Section 6, we will discuss the problem of choosing between two alternative tree species with stochastic timber prices when the land is available for an unlimited number of rotations. This is an extension of Faustmann's problem when two species are available. When the switch is about to take place, the rotations are increasingly longer and higher than the Fautsmann's rotation. Switching later allows forest managers to postpone decision-making and to wait for more information while trees continue growing.

Section 7 offers a conclusion and proposes future research topics.

Recall that the focus of this chapter is the economics of harvesting and planting, which are related given that harvesting opens up the option of planting, and so on. The planting decision has been overlooked in the literature despite its meaningful economic consideration, especially in a context of price uncertainty. In the remaining sections, the planting decision will be considered by assuming the land is initially bare.

2. Basic Wicksell's and Faustmann's rotation lengths

Timber prices have long-term, positive trends that are higher than the rate of increase of other goods prices (see for instance [15]). In [16], real softwood stumpage price trends are estimated at 1.8 to 3.8% (cited in [3]). At a disaggregated level, different tree species may experience differentiated timber price trends, given their distinct abilities to sequester carbon (see for instance [17]) or resist diseases and parasites. Under these conditions, forest managers may find it profitable to switch from one species to another or more generally to switch from timber to other agricultural or non-agricultural uses (see for instance [3]).

The Wicksellian rotation refers to the age at which a stand of even-aged trees will be harvested one time only. When timber price and regeneration costs are constant, the Wicksellian rotation length a_w is constant, determined implicitly as the solution to the following equation, as in [9]:

$$\frac{V_a(a_w)}{V(a_w)} = r, \tag{1}$$

where a_w is the Wicksellian rotation length, r is the discount rate, and $V(a)$ and $V_a(a)$ are, respectively, the growth volume function and its first derivative.

Eq. (1) shows that the harvesting decision is further delayed as long as the marginal benefit $V_a(a)$ from continuing to let trees grow is higher than the opportunity cost of the current harvest, which is $rV(a)$.

When timber prices evolve at a constant rate μ, the rotation length remains constant and equal to the implicit solution of the equation:

$$\frac{V_a(a_w)}{V(a_w)} = \delta, \tag{2}$$

which is the same as Eq. (1), where r is replaced by $\delta = r - \mu$ to account for a price increase resulting in higher rotation lengths.

When timber prices p are constant and it is possible to plant the land and harvest the stand an unlimited number of times according to the Faustmann's setting, the rotation length a_F, referred to as the static Faustmann's rotation length, is constant from one rotation to the next. This is determined implicitly by the following Faustmann's rule as, for instance, in [18]:

$$\frac{V_a(a_F)}{V(a_F)-\frac{D}{p}}=\frac{r}{1-e^{-ra_F}}, \tag{3}$$

Where, D is the constant regeneration cost.

The static Faustmann's rotation depends on the timber price level as long as the regeneration cost D is positive. In this situation, one known implication is that a one-time rise in the timber price level implies a decrease in the Faustmann's rotation length. An increase in timber revenues, therefore, makes longer rotations less attractive. Note that when the regeneration cost is equal to zero or accounted for in the timber price, the Faustmann's rotation length is independent of the price level as it is determined implicitly as the solution to:

$$\frac{V_a(a_F)}{V(a_F)}=\frac{r}{1-e^{-ra_F}}, \tag{4}$$

When timber prices evolve at a constant rate μ and regeneration costs are absent or accounted for in timber prices, the rotation length remains constant and equal to a_f, the implicit solution of the equation:

$$\frac{V_a\left(a_f\right)}{V\left(a_f\right)}=\frac{\delta}{1-e^{-\delta a_f}}, \tag{5}$$

where $\delta = r - \mu$. It can be shown that $\frac{\delta}{1-e^{-\delta a}}$ is decreasing in δ for a given age a. Therefore, Eq. (5) admits a higher solution for the rotation length when δ decreases (or μ increases). Hence, the rotation length increases when the price rate of change μ increases. This result holds when regeneration costs are positive, as shown numerically in [2]. This result also holds when timber prices follow geometric Brownian motion and regeneration costs are absent or accounted for in the price timber as shown in [14] and discussed in Section 5.

As pointed out in [2], some confusion exists in the literature with respect to timber price impact on forest rotation length. To clear up this confusion, it is important to distinguish between a one-time static increase in the price change rate and its impact on the rotation length on the one hand, and the continuous increase of timber prices in time and their impact on successive rotation lengths on the other. For instance, when timber prices increase exponentially at a constant rate in the presence of constant regeneration costs, succeeding rotation lengths continuously decrease over time, as shown in [2]. Under these conditions, successive rotation lengths converge after some rotations to a certain length that can be higher or lower than the static Faustmann's rotation length a_F. However, when timber prices are rising or stochastic and a conversion of the site to an alternative use is possible, changes over time in the optimal rotation lengths remain uncertain, as in [3, 13, 14], for instance.

Nevertheless, when timber prices increase exponentially at a constant rate in the presence of constant regeneration costs, if the rate of change increases statically (a one-time rise in terms of comparative analysis), then the rotation length also increases, as in [2]. This result is also verified in the case of rising or stochastic timber prices and the possibility of changing land use, as in [13, 14].

To isolate alternative land use impact on the planting decision as well as on the rotation length, regeneration costs are assumed to be nil or are accounted for in the timber prices in the following sections. When applicable, uncertainty over timber prices is considered by assuming that timber prices follow geometric Brownian processes. Along with the absence of regeneration costs, it can be shown that under these conditions, the land and stand value functions are homogeneous with respect to the timber prices as shown in [13, 14]. They can, therefore, be expressed as functions of time and the ratio of the two species' timber prices where the numeraire is one of the timber prices. This allows for the representation of the forest rotation as a function of the timber price ratio only. The rotation problem becomes autonomous in the sense that rotations do not depend on time explicitly, but rather implicitly through their dependence on timber prices. For more information, especially concerning the algebra, readers may refer to [13, 14].

3. Two-tree harvesting model

In the following sections, unless specified otherwise we assume that forest managers can plant one species among two available species, P and $P^{'}$. The timber growth function for species P (respectively $P^{'}$) is $V(a)$ (respectively $V^{'}(a)$). We assume that the timber price of species P (respectively $P^{'}$) follows a geometric Brownian motion with drift μ (resp. $\mu^{'}$) and volatility σ (resp. $\sigma^{'}$):

$$dp = \mu p dt + \sigma p dz \tag{6a}$$

$$dp^{'} = \mu^{'} p^{'} dt + \sigma^{'} p^{'} dz \tag{6b}$$

where time indices have been omitted, $dz = \varepsilon\sqrt{dt}$ and $dz^{'} = \varepsilon^{'}\sqrt{dt}$ are the increments of Wiener processes, and ε and $\varepsilon^{'}$ are standardized Gaussian white noises whose correlation is ρ. For notation simplification, variables that depend on time are not indexed unless it is necessary.

The relative price $\theta = \frac{p^{'}}{p}$ is time variable while $\delta = r - \mu > 0$ and $\delta^{'} = r - \mu^{'} > 0$ are constant parameters, where r is the discount rate. We assume that $\delta > 0$ and $\delta^{'} > 0$; otherwise it would be optimal to delay the investment forever. Each tree species is characterized by a timber volume growth function with the following properties:

Assumption 1 *There exists* $\underline{a} > 0$ *and* $\underline{a}^{'} > 0$, *such that the timber volume functions* $V(a)$ *and* $V^{'}(a)$ *are continuous over* $[0, +\infty[$, $V(a) = 0$ *over* $[0, \underline{a}]$, $V^{'}(a) = 0$ *over* $\left[0, \underline{a}^{'}\right]$; $V(a)$ *and* $V^{'}(a)$ *are positive, continuous, differentiable and concave over* $[\underline{a}, +\infty[$ *and* $\left[\underline{a}^{'}, +\infty\right[$ *respectively. In addition,* $\lim_{a\to+\infty} V_a(a) = 0$ *and* $\lim_{a\to+\infty} V_a^{'}(a) = 0$.

A volume growth function is usually convex and then concave. Assumption 1 allows for avoidance of the convex part that would result in complex but economically uninteresting considerations. At the same time, this assumption reflects that trees need time to provide commercial timber volume.

4. Choosing between two alternative tree species with rising timber prices: An extension of Faustmann's problem

In [13], timber prices rise according to the two processes in Eq. (7a) and Eq. (7b), corresponding to two alternative tree species with deterministic growth functions according to Assumption 1.

$$dp = \mu p dt \tag{7a}$$

$$dp' = \mu' p' dt \tag{7b}$$

In particular, when the planted species has a low rate of price change, a switch to the alternative species will certainly occur in the future. The same model can be applied in situations where timber harvesting must be terminated to affect the land to other agricultural or non-agricultural use. For the sake of facilitating discussion in the remaining sections, the alternative land use is assumed to be planting the alternative tree species.

Assume in the remaining sections that the price of the planted species P has a trend μ, that alternative species $P^{'}$ has a trend $\mu^{'}$, and that μ is lower than $\mu^{'}$. It is shown in [13] that, since the land and stand value are homogenous in timber prices, planting and harvesting decisions can be illustrated in the plan (a, θ) where a is the stand age and $\theta = \frac{p^{'}}{p}$. The age or forest rotation of the planted species can be expressed as a function $a(\theta)$.

Figure 1 illustrates a numerical example of the planting and harvesting decisions where $\mu = 1\%$, $\mu^{'} = 3\%$, and $r = 4\%$, as in [2]. The rotation is expressed as the harvesting age $a(\theta)$, a function of $\theta = \frac{p^{'}}{p}$, represented by the dashed line. To focus on the role of timber prices, in [13] the same volume growth function is assumed for both species: $V(a) = V^{'}(a) = V_{\infty}\left(1 - e^{-\alpha\left(a-\underline{a}\right)}\right)$ where $V_{\infty} = 100$ is the timber volume when the age tends to infinity, $\alpha = 0.01$, and $\underline{a} = 5$ is the minimum age for positive growth. The Faustmann's rotation for species P is $a_f = 18.05$ years.

The most crucial findings are the following. There exists a threshold θ_0 ($\theta_0 = 0.68$) such that on bare land, species P must be planted if the timber price ratio θ is lower than θ_0, and species P′ must be planted if the timber price ratio θ is higher than θ_0. When θ is sufficiently low, that is, when p is much higher than p′, species P must be planted and harvested successively for some time. After a sufficiently long period of time, species P′ will become sufficiently attractive to justify a switch from species P to species $P^{'}$, as $\mu^{'}$ will be higher than μ. As prices are assumed to be certain, a permanent switch to species $P^{'}$ will take place when species P is harvested at $\theta > \theta_0$. When the permanent switch to species $P^{'}$ occurs, its forest rotation $a_f^{'}$ is constant, determined implicitly by Eq. (8) as follows:

$$\frac{V_a\left(a_f'\right)}{V\left(a_f'\right)} = \frac{\delta'}{1-e^{-\delta' a_f'}}, \tag{8}$$

Now reconsider the situation where species P is still attractive, as the price p is still low with respect to p' or, equivalently, θ is sufficiently below the switching threshold θ_0. In the plan (t, θ_t) in **Figure 1**, when the stand P is growing, point (t, θ_t) moves obliquely according to:

$$\theta_t = \theta_{t=0} e^{(\mu' - \mu)t}. \tag{9}$$

As illustrated in **Figure 1**, the harvesting age $a(\theta)$ is composed of two boundaries (dashed line), an upper boundary and a lower boundary. The upper boundary is a set of downward-sloping and upward-sloping segments. The lower boundary is one upward-sloping segment starting from $(\theta_0, 0)$.

When the trees are growing, trajectory (θ_t, t) evolves obliquely according to Eq. (9) and hits the upper boundary on one of the downward-sloping segments. When the trajectory hits the upper boundary at a given downward-sloping segment, trees are harvested and the same species is replanted a given number of times. For instance, when the trajectory hits the upper boundary on the last downward-sloping segment (at θ_t higher than θ_0), trees are harvested and the alternative species are planted for the first time. The alternative species continues to be replanted and harvested forever, with rotation lengths constant and equal to a_f' given by Eq. (8). When the trajectory hits the upper boundary on the downward-sloping segment at θ higher than θ_1 but lower than θ_0, trees are harvested and the same species P is replanted one more time before the switch to the alternative species P' occurs. The upward-sloping segments

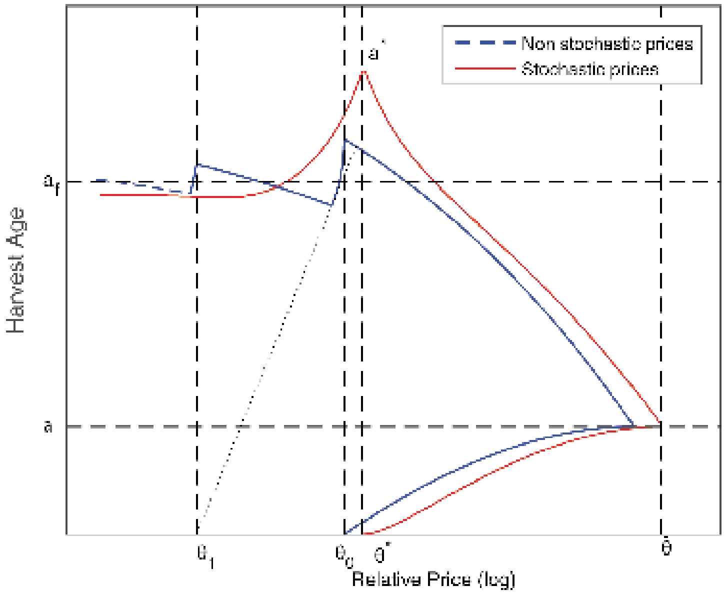

Figure 1. Harvesting boundaries when timber prices are certain and stochastic ([13, 14]).

are never optimally reached, but they signal the change in the remaining number of harvests of the same species. From left to right, the last downward-sloping segment of the upper boundary can be called a switching boundary, whereas all the remaining segments of the upper boundary constitute the replanting boundary.

Similar to the upward-sloping segments of the upper boundary, the lower boundary cannot be reached after a sequence of optimal harvests of species P. Assume that a forest manager has, for whatever reason, a stand of species P at age a and θ higher than θ_0. Note that this situation cannot be the result of a sequence of optimal harvests. Under these conditions, the forest manager must either harvest the stand immediately if point (θ, a) is below the lower boundary or wait to let the trees grow if point (θ, a) is above the lower boundary and then harvest the stand when the upper boundary is reached. At that time, stand P must be replaced by the alternative species P′.

Most importantly, successive optimal harvests of stand P take place according to the following mutually exclusive situations in which the trajectory (θ_t, t) hits the upper boundary:

1. Successive rotations lengths are all equal to the static Faustmann rotation length $a_f = 18.05$ (y) of the planted species given by the Eq. 5. It is the case for $\theta = 0.39$, 0.56, and 0.80 at which the switch occurs.
2. Successive rotations lengths continuously increase and remain higher than the static Faustmann rotation a_f. For instance, successive rotations are 18.45, 18.74, and 19.27 (y) when relative prices are equal respectively to 0.36, 0.52, and 0.76 at which the switch occurs.
3. Successive rotations lengths continuously decrease and remain lower than the static Faustmann rotation a_f. For instance, successive rotations are 17.67, 17.40, and 16.96 (y) when relative prices are equal respectively to 0.42, 0.60, and 0.84 at which the switch occurs.

A rise in the increase rate μ of the planted species results in a higher rotation length and a later switch to the alternative species. However, a static one-time increase in the price of the planted species (θ decreases) that is sufficiently small to not alter the number of remaining harvests of the same species before the switch results in an increase in all successive rotations lengths. Depending on the price level, a price increase sufficiently high to affect the number of remaining harvests before the switch may result in a decrease in successive rotations lengths.

5. Choosing between two alternative tree species with stochastic timber prices: An extension of Wicksell's problem

Reconsider Wicksell's problem in the presence of two alternative species P and $P^{'}$ and assume that the land is still bare. Timber prices are stochastic and evolve according to Eq. (6a) and Eq. (6b). Assume that the forest manager must plant a single species once at no cost. There then exists an interval (see **Figure 2**) in the timber price ratio $\theta = \frac{p^{'}}{p}$ during which it is optimal to keep the land bare (see [13]). This situation remains until the timber prices differentiate enough for the forest manager to select the species with the highest stand value to plant. This waiting period applies despite the absence of any regeneration costs. The reason is that the availability

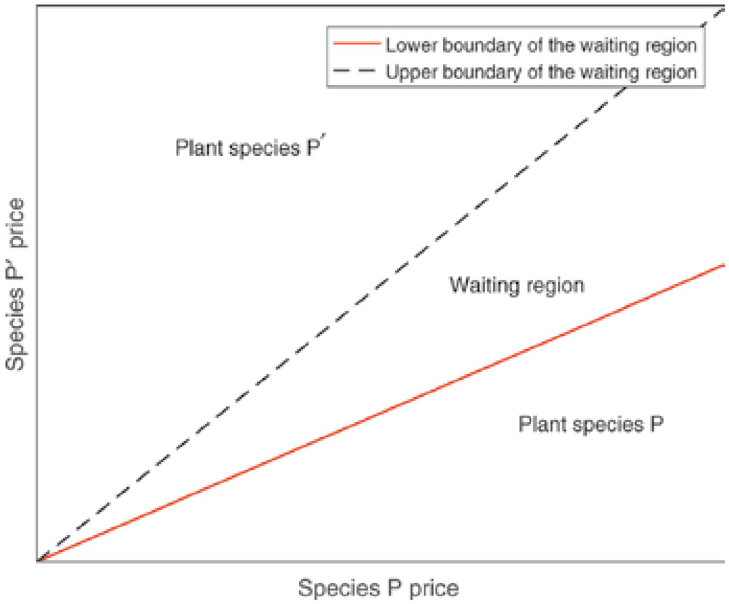

Figure 2. The one-rotation planting decision when two species are available.

of the land for a single rotation represents an opportunity cost for the forest manager if he or she commits the mistake of choosing the wrong species. The land value is proportional to the timber price once planted but follows a non-linear function of timber prices during the waiting period. This situation illustrates the value of the option to decide later once more information becomes available.

Figure 2 illustrates the decision-making process. If both timber prices (p, p') are represented by a point below the continuous line, then species P must be planted. If both timber prices (p, p') are represented by a point above the dashed line, then species P' has to be planted. Finally, as long as both timber prices (p, p') are represented by a point between the continuous and dashed lines, that is in the waiting region, the planting decision must be delayed. One way to avoid making the wrong decision is to postpone making the decision as shown in the literature related to investing under uncertainty using real options analysis (see for instance [19]). If the uncertainty increases (σ or σ' increase), then the waiting region becomes larger. If the timber price trend of one of the species increases, then the corresponding planting region becomes larger. For more information, readers may refer to [20].

6. Choosing between two alternative tree species with stochastic timber prices: An extension of Faustmann's problems

In this section, we will consider Faustmann's problem when two tree species P and P' are available with timber prices following the stochastic processes in Eq. (6a) and Eq. (6b). The price of one species may be lower or higher than the price of the other species at any time, and the converse may occur later. Under these conditions, forest managers must choose the species to be planted, decide on rotation lengths, and decide whether to replant the same species or to plant the alternative species and whether it is optimal to keep the land bare for some time as in Section 5. It is shown in [14] that it is never optimal to keep the land bare after a harvest as in the Wicksellian setting. The reason is that instead of waiting to let timber prices differentiate clearly, forest managers are better off planting the most promising species and switching to the alternative species if it turns out that they have planted the wrong species.

Figure 1 illustrates the rotation length as a function of timber price ratio $\theta = \frac{p'}{p}$ (continuous line) when species P with the lowest trend μ is planted (recall that μ is lower than μ').

As in Section 4, when timber prices are rising and non-stochastic, if the land is bare, species P must be planted if θ is lower than a certain threshold θ^*, and species P' must be planted if θ is higher than θ^*. The harvesting boundary encompasses an upper boundary and a lower boundary. The upper boundary is composed of a replanting boundary corresponding to θ varying from zero to θ^* and a switching boundary corresponding to θ varying from θ^* to $\overline{\theta}$. When the replanting boundary is reached, species P is harvested and immediately replanted. When the switching boundary is reached, species P is harvested and immediately replaced by species P′ and so on. The maximum rotation is reached at θ^* when the replanting and switching boundaries meet.

Although the converse may occur, θ increases in average with time as μ' higher than μ. Successive rotations lengths may increase or decrease similarly to the non-stochastic situation studied in Section 4, as the replanting boundary is non-monotonic (**Figure 1**). In general, the impact of a static increase in the timber price of the planted species (θ decreases) on its rotation length remains ambiguous except when the species is harvested for the last time. At that time, when timber price p increases, θ decreases and the rotation length increases. The longest rotation lengths are obtained when the planted species is about to be harvested for the last time and the switch to the alternative species is about to occur. The rotation length is higher when the switch occurs at θ closer to θ^*. The reason is that forest managers prefer delaying the decision to switch when timber prices are uncertain by choosing longer rotation lengths; it is better to delay the decision to switch while trees are still growing rather than harvest earlier and wait for timber prices to distinguish them sufficiently as in Section 5. It is also shown numerically in [14] that the rotation length increases when the price rate of change increases, as in [2] and [13].

Figure 3 represents the land value in terms of the numeraire p when only species P is available and when both species P and P' are available, with certain or stochastic timber prices. Obviously, the land has a higher value when both species are available than when only one species is available. Furthermore, when both species are available, the land has a higher value when timber prices are uncertain than when timber prices are certain. This result is well known for real options analysis. When timber prices are uncertain, the availability of two species allows forest managers to profit from an increase in timber prices but

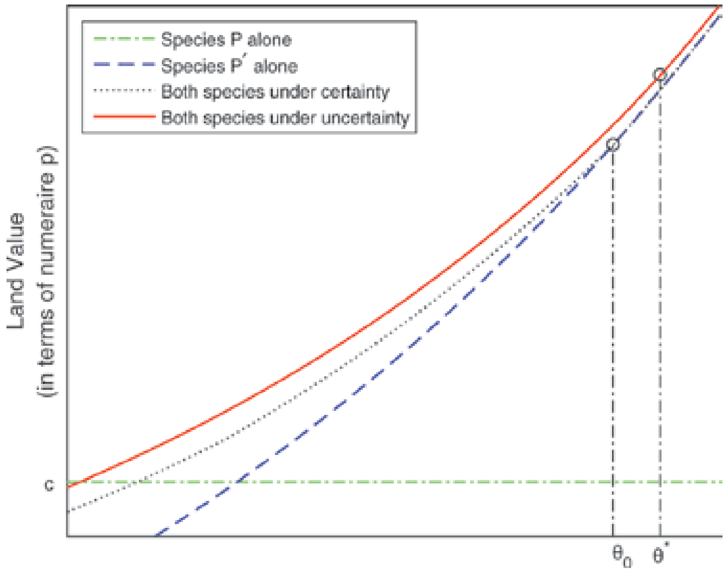

Figure 3. Land value with certain and stochastic timber prices ([14]).

protects them at least partially from price decreases, especially when the two species do not have highly correlated timber prices.

7. Conclusion

Under the Faustmann's setting when only one species is available, regeneration costs are constant, and timber prices are rising, then successive rotations lengths decrease continuously over time. However, this result does not hold when it is possible to switch to another alternative use of the land. When timber prices are rising and it is desirable to switch to an alternative tree species, successive rotations lengths before the switch are either equal to Faustmann's rotation length, increasingly higher, or decreasingly lower. When timber prices are uncertain, successive rotations lengths before the switch may increase or decrease. When the switch is imminent, the rotation length increases. This is an optimal way for forest managers to postpone deciding when prices are uncertain and allow trees to continue growing.

Planting and harvesting decisions are prototypes of investment decisions as decision makers face uncertain future costs and revenues as well as irreversible action, especially with respect to harvesting old and mature trees or planting with positive cost, as is the case in practice. These decisions can be managed proactively using some sources of flexibility, such as delaying irreversible decisions. Actions with irreversible consequences should be delayed until sufficient information becomes available to justify committing resources. In addition to delaying decisions, other sources of flexibility exist, such as investing sequentially, which may be applied by investors facing high levels of uncertainty.

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To the memory of Pierre Lasserre.

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Linking Science and Management in a Geospatial, Multi-Criteria Decision Support Tool

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Additional information is available at the end of the chapter

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Abstract

Land managers are often faced with balancing management activities to accomplish a diversity of objectives in complex, dynamic ecosystems. In this chapter, we present a multi-criteria decision support tool (the Future Forests Geo-Visualization Decision Support (**FForGeoVDS**)) designed to inform management decisions by capturing information on how climate change may impact the structure and function of forested ecosystems and how that impact varies across the landscape. This interactive tool integrates spatial outputs from various empirical models in a structured decision framework that allows users to customize weights for multiple management objectives and visualize suitability outcomes across the landscape. As a proof of concept, we demonstrate customized objective weightings designed to: (1) identify key parcels for sugarbush (*Acer saccharum*) conservation, (2) target state lands that may serve as hemlock (*Tsuga canadensis*) refugia, and (3) examine how climate change may impact forests under current and future climate scenarios. These case studies exemplify the value of considering multiple objectives in a flexible structure to best match stakeholder needs and demonstrate an important step toward using science to inform management and policy decisions.

Keywords: spatial modeling, landscape modeling, integrated modeling, GIS, structured decision-making, forest ecology, forest health, climate change, multiple criteria decision support, geo-visualization, Future Forests Geo-Visualization and Decision Support tool

1. Introduction

Over the past several decades, climate patterns across the globe have been shifting, accompanied by a wide range of biological changes. In the Northeastern US, it is predicted that climate change will continue to restructure forests and alter the services they provide over the coming century. To refine our understanding of how climate may impact forested ecosystems, and how this information can be used to inform decision-making, our research group developed a spatial, multi-criteria decision support tool (Future Forests Geo-Visualization Decision Support (**FForGeoVDS**)) that integrates empirical models of current and future forest structure and function in an interactive weighting algorithm. Combined with climate projections, the resulting decision support framework allows stakeholders to compare outcomes for a variety of management objectives under various climate conditions on a pixel-by-pixel basis, thus reflecting the complexity of the Northeast's heterogeneous landscape. This merging of scientific knowledge with management needs is an important step toward improving our understanding of the relationships between forests and climate, but more importantly affecting change on the ground by informing management and policy decisions.

What makes this **FForGeoVDS** tool unique is several-fold:

1. The integration of disparate empirical ecosystem models allows us to incorporate and aggregate information on a variety of forest structural characteristics, processes, and ecosystem services provided. This set of ecosystem model inputs will be modified and updated over time as new models are developed and vetted by the scientific community.
2. The incorporation of several climate projection scenarios allows users to examine potential changes in modeled forest structure and function under alternative climate conditions.
3. A dynamic structured decision framework allows users to interactively select and weigh multiple management objectives for an integrated assessment of their overall management goals.
4. The structured decision framework is applied independently to each 30 m pixel across the landscape, providing spatial detail not typically included in decision analyses. This allows land managers to visualize outcomes across the landscape and identify various locations ideally suited for specific outcomes or those best suited to accomplish multiple outcomes.
5. The ability to continually expand and update the tool via the integration of new ecosystem models as they become available.

The overall goal of this chapter is to provide a test case for how the **FForGeoVDS** tool can integrate disparate research products in a structured decision support tool to provide spatially detailed quantification of the relative likelihood of meeting multiple objectives under both current and future climate scenarios. Here we give an overview of the **FForGeoVDS** tool's spatial modeling framework and demonstrate the general structure based on the integration of five empirical models developed to quantify: sugar maple distribution and health, forest fragmentation risk and hemlock distribution, and vulnerability to hemlock wooly adelgid (HWA). Using the **FForGeoVDS** tool, we present three case studies to demonstrate how these models can be integrated to visualize and prioritize locations across the landscape that meet a variety of management objectives. These case studies correspond to three hypothetical users with a decision to make:

- A Land Trust Director who wants to identify private properties for possible acquisition to both conserve sugar maple habitat and minimize forest fragmentation on habitat corridors;
- A State Forest Planner who wants to identify a state forest for hemlock monitoring and possible stand improvement where hemlock is likely to tolerate hemlock wooly adelgid infestation in order to maintain important wildlife habitat; and
- A Climate Action Advocate who wants to demonstrate how our ability to maintain both sugar maple and hemlock across the state is likely to change with the twin pressures of forest conversion and projected climate change.

This demonstration exemplifies a structured algorithm for decision support and an optimized spatial output to visualize and assess relative objective success across the study area. However, it is important to note that this approach is not designed to serve as an optimization exercise that differentiates among management alternatives. Instead, it serves as a spatial assessment and visualization tool to allow users to identify where across the landscape priority management objectives are most likely to be successful under status quo management acitivities, and how this might change under climate change scenarios. It also does not permit assessment of management objectives or alternate scenarios that are not already captured in modeled inputs or spatial data layers. As such, it is limited by the ecological modeling products available. However, we anticipate that over time, additional ecological models can be incorporated into the decision tool based on feedback on stakeholder needs and continued progress in scientific modeling for the region.

This development of the **FForGeoVDS** tool has been aided by a broad audience of stakeholders who have been continually involved in the design process. The research team has held multiple participatory design sessions with professionals from multiple state and federal agencies, nonprofit land management organizations, and advocacy groups. These stakeholders have provided invaluable input into the necessary features and functionality of the decision support tool, as well as how this tool will be situated in the larger ecosystem of tools available to support decision-making about Vermont's forests. While many tools exist from many different agencies, there is a notable gap in the ability of managers to combine projections of climate change with forest management objectives in a flexible way pertinent to their scale and extent of interest. This puts the **FForGeoVDS** tool in a particularly good position for adoption and use, and these stakeholder engagement sessions have helped to shape both the research team's activities and the expectations of the eventual users of the tool.

2. Background on decision support for forest management

The use of decision support systems is gathering increased attention in natural resource management because of the demand for increasing transparency, potential for economic, legal, and political conflicts, the increasing volume of data available to help inform decisions, and improved technologies to integrate, analyze, and summarize this information [1]. This is particularly true for forest ecosystem management, where growth and yield models can be used to directly predict forest response to management alternatives [2]. Such tools are also proving useful when applied to adaptive management decisions, particularly as it relates to the impacts of climate change [3, 4].

There are many methodological approaches to decision support, including artificial neural networks, knowledge-based systems, and multi-criteria decision models [2]. While the aggregation algorithms differ, decision support systems are built on a basic framework that includes structuring the decision problem, identifying various alternatives or scenarios to compare, determining the preferences of decision-makers, and evaluating and comparing the decision alternatives [5]. Choosing the appropriate decision support methodology is dependent upon the actors that are involved in the process, desired goals, and available information [6].

Here we developed an interactive, multiple criteria, structured decision framework to help land managers visualize how climate change may impact forest structure, conditions, and processes across the landscape. Multiple criteria decision systems (MCDSs [7, 8]) provide flexibility in the selection and weighting of management objectives using a structured framework for evaluating outcomes for multiple, potentially conflicting objectives in quantitative units or qualitative rankings. This framework involves three main concepts: value scaling (or standardization) of various objectives, criteria weighting by the user, and a combination (decision) rule to aggregate across criteria [9] to evaluate the relative value of each alternative scenario in achieving the desired weighted objectives [6]. A key component of this approach is allowing decision-makers to determine which objectives are of interest, and what relative weights each of those objectives should carry in the analysis.

This approach is becoming more common in natural resource management applications to evaluate and compare broad alternatives for specific parcels or management units. More recently, it has become common to integrate temporal and spatial results to produce spatial decision support systems by coupling with GIS tools [9, 10]. For example, the integrated grid based ecological and economic (INGRID) landscape model, was developed by [11] to support landscape management decisions. INGRID was designed to simulate the ecological effects of management alternatives and costs for dry grasslands using empirical models that predict the risk of plant and animal extinction under alternative conditions, management scenarios, and disturbance regimes. While input rasters were spatially explicit, the resulting decision assessment was based on overall proportion of cells with desired biodiversity outcomes, without examination of spatial patterns in those outcomes across the landscape.

However, this same approach could be applied in a geospatial model where each pixel across the landscape is processed through the structured decision framework and evaluated independently. The name "Geovisual Analytics for Spatial Decision Support" [12] suggested for this approach emphasizes the importance of interactive, visual interfaces, as an effective way to synthesize and quantify information to support problem-solving. This requires an approach that first articulates decision objectives, integrates information from a multitude of data, provides a flexible, interactive interface for customization, and then provides data analysis and visualization, across multiple scenarios in order to select the right course of action and where across the landscape that can be achieved [12]. Because this approach is typically disseminated through online visualization tools, it has the potential to provide access to a wider audience of potential stakeholders, enabling many actors to work together and providing an interface adaptable to the needs of different actors [12].

This highlights one of the key components of what can be a generally "techno-centric" approach to decision support: stakeholder engagement. The human decision-makers are an integral component to decision-making, providing information that often dominates the process [13]. Solutions based on complex algorithms, frameworks, and models can be integrated with approaches, such as collaborative learning, expert models, and continuous stakeholder participation during development in order to design systems better able to cope with the complexities of managing ecosystems with the potential for many conflicting interests [14].

Looking ahead, the future of decision support systems for forest management will likely move toward more comprehensive forest models and the need to consider climate change as a major driver [15]. This will require careful parameterization and empirical validation before implementation in decision support tools and at the same time will increase the required expertise to actually run the models and analyze the resulting output [16]. The Future Forests Geo-Visualization Decision Support (**FForGeoVDS**) tool presented here exemplifies this move toward more comprehensive, integrated forest models with parameterization for various climate-change scenarios. Other promising developments include the use of the Internet to enable easy access to data and public engagement with these tools and the resulting outputs. This presents an opportunity for new decision support applications, such as the one presented here, to bring decision support to a broader audience of public, commercial, nonprofit, and private stakeholder groups.

3. Approach

Here we demonstrate the structure and utility of the new multi-criteria Future Forests Geo-Visualization and Decision Support (**FForGeoVDS**) tool, designed to integrate empirical models of forest structure and function under various climate conditions. This pilot is based on several current empirical models (forest fragmentation risk, sugar maple abundance, sugar maple canopy condition, eastern hemlock abundance, eastern hemlock HWA risk), integrated in three user case studies. Here we walk through various components of the **FForGeoVDS** approach, including a summary of the empirical models that provide the foundation for the decision support tool (Section 3.1), the geospatial interface used to integrate across models (Section 3.2), customized weighting scenarios (Section 3.3), and the structured decision calculations (Section 3.4) used to aggregate decision outcomes. The final outputs include maps that quantify the weighted suitability (Section 3.5) on a pixel-by-pixel basis, highlighting where management objectives are most likely to be met and providing a quantitative basis for parcel comparison.

Because of the flexible design of the **FForGeoVDS** tool, we are able to demonstrate how the mean "weighted suitability" decision outcome can be captured across the landscape and compared across towns, parcels, and climate scenarios to inform three different customized management objectives (Section 4).

3.1. Input empirical models

The **FForGeoVDS** tool relies on empirical models designed to capture specific ecosystem structure and function that can be used as a proxy for management outcomes. Here we incorporate five disparate models: tree species distribution (sugar maple and hemlock percent basal area maps [17]), forest fragmentation risk [18], sugar maple canopy condition [19], and eastern hemlock susceptibility to hemlock wooly adelgid [20]. For all input models included in any decision support tool, it is critical to provide easily accessible information about model development and validation to add credibility to the tool and allow stakeholders to assess the reliability of the resulting products. Here we present a brief summary of each model included in this pilot, including links to publications for additional information.

3.1.1. Tree species distribution model

Traditional forest classifications are often limited to coarse forest type classes, but advancements in remote techniques allow for more detailed mapping of species abundance and composition. This is particularly useful in the Northeast where species composition is often mixed and information beyond “dominant species” is required.

For the models included here, [17] employed a spectral unmixing technique based on multi-temporal Landsat imagery to capture the proportion of each species present in each pixel. Algorithms based on US Forest Service Forest Inventory and Analysis plots were used to convert spectral unmixing scores to basal area composition. Combining basal area maps for all key species, fractional basal area was calculated to quantify relative abundance of each species and to develop a thematic species map for comparison to more traditional species mapping products (**Figure 1**). Validation with 50 field inventory plots showed the resulting thematic forest classification mapped 15 forest classes with overall accuracy = 42%, KHAT = 33%, fuzzy accuracy = 86% at the pixel level, providing more detail and higher accuracy than existing forest-type mapping efforts.

Model description [17]:

Units: Percent basal area (0–1).

Spatial extent: Northern New York, Vermont, and New Hampshire; Western Maine.

Resolution: 30 m.

Time interval: Historic and current.

Climate: No significant climate influence.

3.1.2. Forest conversion model

To assess potential future changes in forest cover, [18] measured historical changes in forest cover in the northern forest region and used correlations between these historical changes and

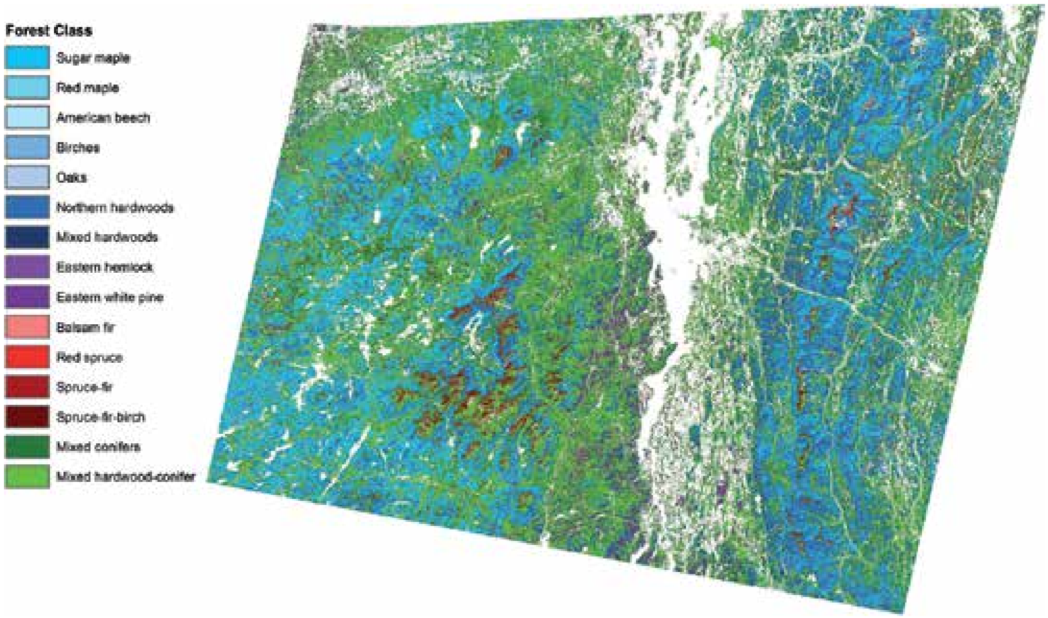

Figure 1. Forest cover map spanning northern New York and Vermont produced by integrating spectral unmixing of multi-temporal Landsat imagery and a rule-based, OBIA classification scheme.

other spatial variables to estimate future transitions into and out of forest at three time steps across the landscape.

This forest conversion model can be used to prioritize conservation in critical areas where deforestation is likely, or where projected deforestation will cause significant fragmentation of the landscape. It can also be used to avoid areas with high fragmentation risk if the management activity will not be able to control that process.

Resulting forest conversion probability maps (**Figure 2**) provide projections of places likely to experience deforestation or forest regrowth, but only insofar as historical drivers remain constant; this assumption may be weaker for later time frames.

Model description [18]:

Units: Probability of change (0–1).

Spatial extent: Northern New York, Vermont, and New Hampshire; Western Maine.

Resolution: 30 m.

Time interval: Historical, current and probability of transition maps available for 2030, 2060, and 2075.

Climate: No significant climate influence.

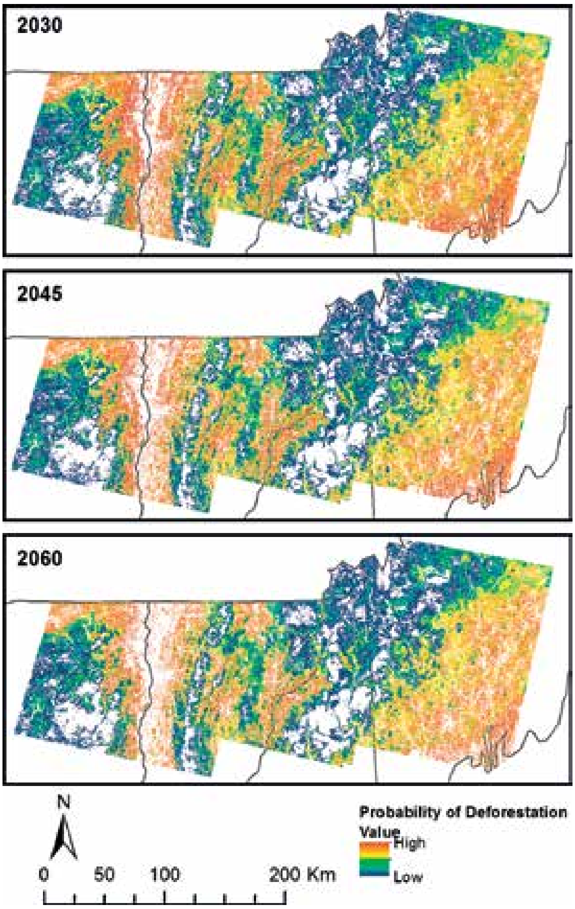

Figure 2. Probability maps show the likelihood of transition from forest to nonforest for each of three time steps. Higher values represent a higher probability (0–1 scale) of deforestation. Lower values indicate locations where deforestation is unlikely.

3.1.3. Sugar maple stress index model

This study compared a suite of climate metrics to field assessments of sugar maple canopy condition across Vermont [19]. Five climate metrics were significantly related to sugar maple decline. The influence of climate is comparable to that of insect outbreaks and other disturbance events, indicating that in Vermont, sugar maple is a climate-sensitive species.

The model identified areas across the state where climate has historically been better or worse for sugar maple (**Figure 3**). Substituting climate projections in the model indicates that climatic conditions are likely to become less favorable for sugar maple over time, with up to 84% of current sugar maple stands negatively impacted by climate change. However, locations of "climate refugia" should also be available to maintain sugar maple in spite of changing climatic conditions. Considering the role of sugar maple in Vermont's economy and culture, managing this resource into the future as climate changes is of great concern and could pose a considerable challenge.

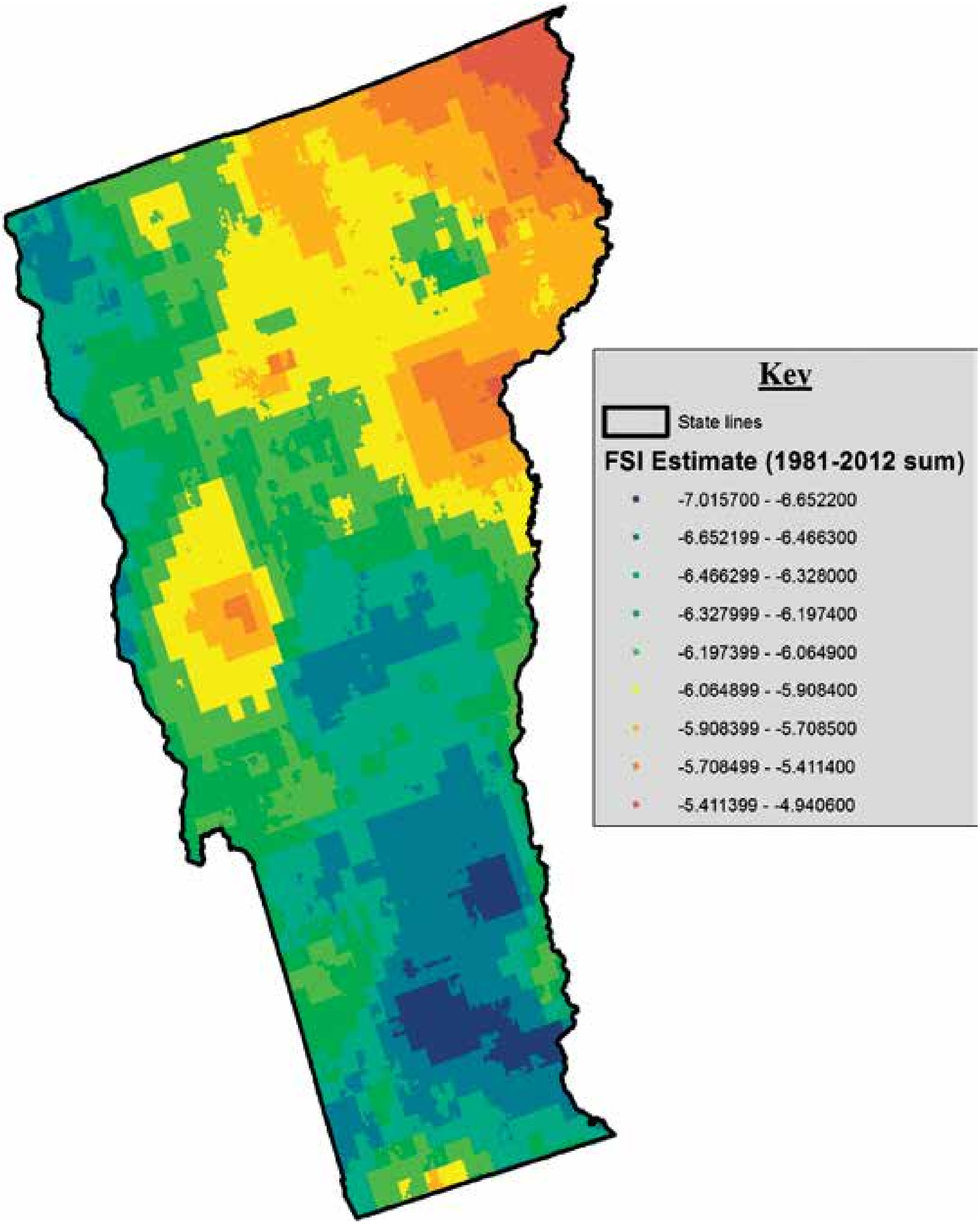

Figure 3. Climate impacts on sugar maple health from 1981 to 2012. Higher forest stress index (FSI) values (warmer colors) represent locations with more severe climate induced sugar maple decline over the historical record. Lower FSI values indicate locations with minimal climate induced sugar maple decline.

This study was designed to isolate the impacts of climate on sugar maple condition and does not capture additional stressors that may result from other biotic or abiotic agents. As such, it should be used only to differentiate relative regions of climate favorability for sugar maple.

Model description [19]:

> *Units:* Forest Stress Index (FSI) near 0 represents average canopy condition, Lower values = healthier than normal, Higher values = higher decline than normal.
>
> *Spatial extent*: Vermont.
>
> *Resolution*: 800 m.
>
> *Time interval*: Historic (1971–2000 norms), current (1981–2010 norms), and projected (2050 and 2100 under two climate scenarios).
>
> *Climate:* Significant climate influence captured for A2 (high) and B1 (low) emission scenarios.

3.1.4. Hemlock wooly adelgid susceptibility model

To measure the relative risk of hemlock wooly adelgid (HWA)-induced decline in eastern hemlock stands, [20] quantified changes in hemlock basal area increment (BAI) to quantify decline on 41 hemlock stands across New England. They determined that slope, GIS calculated Hillshade (capturing heat load), and January minimum temperatures were significant predictors of hemlock decline rates following infestation. The authors built a model to differentiate stands likely to decline rapidly (susceptible) from those likely to decline over longer time periods (tolerant) (**Figure 4**). The model correctly classified 80% of the 41 original sites with 73% accuracy on 15 independent validation sites.

To apply this model across the study region, GIS data layers for key input variables were used to produce a spatially explicit model that predicts the likelihood of hemlock growth declines following HWA infestation. This was applied using both historical temperature norms and a projected 2°C increase in minimum January temperatures.

Model description [20]:

> *Units:* Probability of rapid HWA-induced growth decline (0–1 scale).
>
> *Spatial extent*: New England.
>
> *Resolution:* 30 m.
>
> *Time interval:* Historic (1971–2000 norms), current (1981–2010 norms), and projected (2050 and 2100 under two climate scenarios).
>
> *Climate:* Significant climate influence captured for A2 (high) and B1 (low) emission scenarios.

3.2. Geospatial integration

Integration of many spatial data layers requires careful preprocessing. These inputs include empirical model inputs (raster layers) as well as other ancillary data necessary in

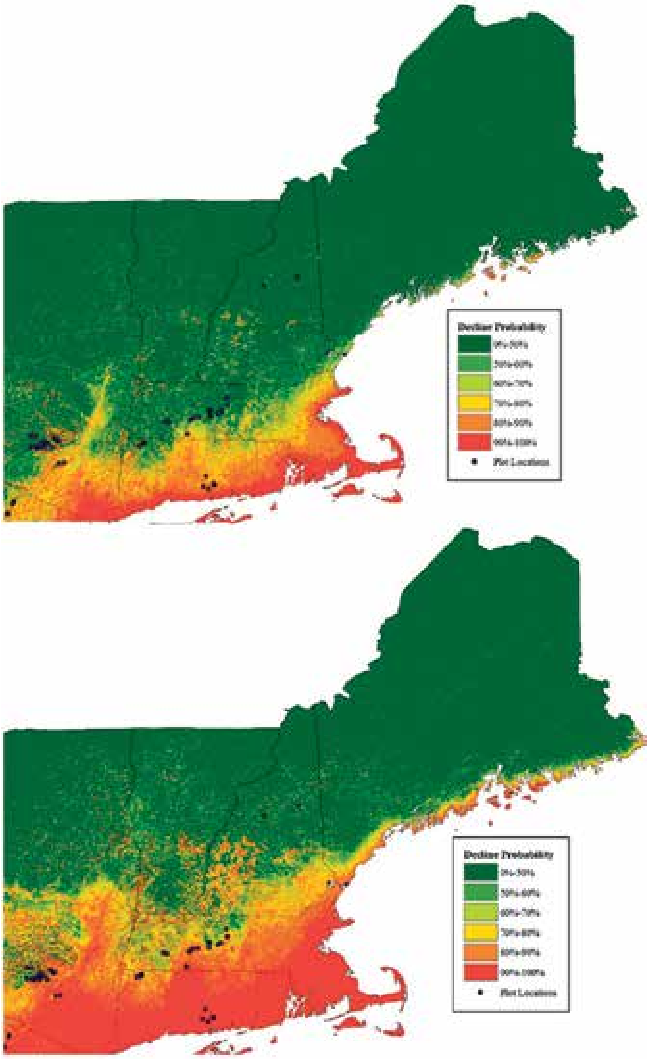

Figure 4. Hemlock risk map shows probable locations of declining sites for current (top) climate and projected (bottom) climate conditions. Map units capture the probability quartile (0–50, 50–60, 60–70, 70–80, 80–90, 90–100%) that a given location has site conditions favoring rapid hemlock decline. High values indicate locations where mortality will occur quickly following HWA infestation. Low values indicate locations where hemlock is likely to tolerate HWA infestation.

customizing the decision framework (e.g. land ownership class, political boundaries, etc). The first step was to develop a common raster grid framework to which all data were transformed. The 30-m USGS DEM for Vermont was used as the base frame for standardizing the various input data, providing the common projection (USA Contiguous Albers Equal Area Conic USGS), cell size (30 m), extent and snap raster for aligning all inputs to a common grid. Each input raster was projected, clipped to the extent of the Vermont study area, and then resampled to the common grid cell size and alignment. Continuous variables were interpolated using bilinear resampling, while categorical variables were interpolated using the nearest neighbor resampling.

In order to integrate these outputs, each was first normalized to the same relative zero to one scale. When necessary, a percentile-based linear transformation was used to convert values for each pixel to a scale where 0 represents the lowest values for that metric and 1 represents the highest value for that metric. Units indicate the percentile value for a given value within the larger population of values contained in the input raster for the full study area raster. Species percent basal area for hemlock and sugar maple as well as the hemlock wooly adelgid probability risk model did not require any transformation as they generate data on a zero to one scale. The Forest Stress Index data ranged from −0.176002 (historical norms) to 2.166101 (A2 scenario). The probability of forest conversion was available for three time steps (2030, 2060 and 2075); to flatten these into a single raster, the maximum probability of conversion was calculated for each cell and used as the final input to the decision support tool. This processing was accomplished with a combination of Python scripts and ArcGIS models and run in the ArcGIS 10.4 software package.

3.3. User customization

The **FForGeoVDS** online tool requires that users first select their desired management objectives from the list of empirical models available. Here we include empirical models designed to capture forest structure or condition attributes described above (Section 3.1). Users select their desired management objectives, define an area of interest, weight each objective based on their overall management priorities, and define the directionality for each objective. The area of interest is used to calculate summary statistics and generate final maps. The weightings of each objective must sum to 100. The directionality of each objective determines whether higher values from the empirical model correspond to higher desirability for the user. As an example, higher probability of conversion may be desirable from someone looking for areas to purchase development rights, while lower probabilities of conversion may be desirable for someone looking to establish long-term monitoring plots. Once these customizations are set, the user can generate suitability maps of their chosen landscape.

3.4. Suitability calculations

FForGeoVDS calculates a suitability score for each pixel in the defined area of interest from the user-defined weights and directionality for each objective (**Figure 5**). Using the normalized input rasters, it is relatively easy to apply user-defined weights to calculate an overall "weighted mean suitability" score. Normalized values for each pixel are used as inputs to a weighted mean calculation based on user-defined weights. The desirability settings are incorporated by inverting the zero-to-one scores as needed such that positive values capture desirable outcomes and negative values capture undesirable values. The result is an integrated, weighted mean where higher value pixels identify where across the landscape the stated combination of management objectives is most likely to be met (**Figure 5**).

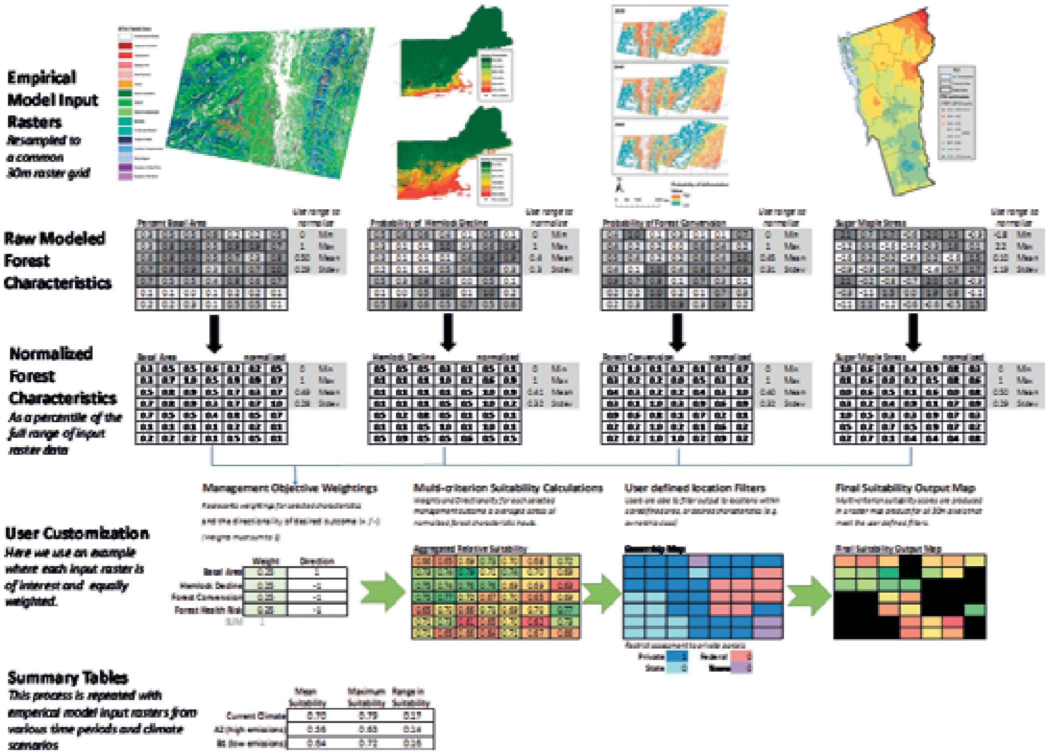

Figure 5. The GIS model used to calculate final suitability scores includes processes to normalize each selected input raster, rescale to match desired outcomes, weighted based on user customization and averaging for each pixel across the input study area. The resulting output suitability rasters are then summarized for the user-defined area of interest.

3.5. Suitability outputs

Outputs from the customization **FForGeoVDS** tool include maps of **weighted average suitability** outcomes where 0 represents the least suitable locations and 1 represents the most suitable locations for the specified combination of management outcomes. Users can manipulate a threshold to filter out pixels that do not meet a desired level of suitability. In addition, weighted average suitability data are summarized for user-defined areas of interest, including mean-weighted average suitability and the proportion of pixels above any user-defined thresholds of suitability. In this way, users can quantitatively compare both the extent and magnitude of suitability in addition to visualizing patterns across the landscape. With the addition of ancillary data layers as filters or for zonal statistics, users can use suitability values to compare properties for targeted efforts, relative success of various management objectives for a given property or the potential changes in suitability under various climate change scenarios. With this summary statistical information, users have critical information on which to base decisions and to more effectively communicate how management success is likely to change under various scenarios.

It is important to point out that the suitability score produced using the **FForGeoVDS** tool provides a 0–1 suitability scale based on the *full population* of conditions captured in the database across the full study area. Thus, if the user selects only their parcel for analysis, the scores that are returned do not represent the best and worst locations for that specific parcel only, but instead returns scores that are relative to the best and worst locations for the entire area captured in the database (in this example, across the state of Vermont). As such, it is possible that a given property may contain no highly suitable locations relative to what is available across the state. If at the end of this analysis no "best" location to achieve management goals has been identified, this does not mean that the analysis was worthless. Often the insights gained may suggest other management options, rather than seeking outcomes that may not be likely given the conditions within a selected area. Because this analysis is flexible, the decision-maker is always free to explore other management objectives available in the tool to identify what that given parcel may be best suited for.

Another approach to decision support for clearly defined locations involves limiting normalization and subsequent suitability calculations to the range of values contained within the user-defined area rather than the full population of values across input rasters. One widely adopted approach to this location-specific analysis is the Simple Multi-attribute Rating Technique (SMART [21]). SMART analyses create an interval scale for each metric, and then base aggregation algorithms on the range of those scores within the specific decision problem area [22], not the range of all scores across the broader landscape. This is particularly useful when the range of possible scores is limited within a specific problem, which can limit the influence for that particular objective relative to other objectives in the multi-criteria problem. SMART weighted aggregation methods then allow the user to obtain a measure of the overall benefits of various locations relative to other locations only within the decision problem area. However, this approach is not suitable for all circumstances, for example when scores relative to other locations outside of the specified area are also of interest, or when there is a possible interaction between the some of the criteria [22].

While this tool has been purposefully designed to examine relative suitability across the entire study area, and considering the full range of information available within our input models, it can easily be modified to use the SMART approach described above. Future iterations of the **FForGeoVDS** tool will include user preference options to base normalization on the full range of population values (*population*) or to limit normalization to the range found only within the user-defined area (*SMART*). We also anticipate creating an option for users to aggregate using the *weighted average* or *additive weights* option. This further flexibility will allow for users to customize the type of information that output maps and tables are able to provide.

4. Use case studies

Here we present three hypothetical case studies to demonstrate different ways that the tool can be used to inform management. This includes customized objective selection and weights defining the relative importance of each objective, as well as a directionality statement defining whether high or low values are desired as ideal outcomes for each management objective.

These customized settings were then linked to our input empirical models to examine how suitability varies across the landscape, identify target management locations, and, where

Land Trust Director (Sugar Maple Focus)		
Objectives	**Weight**	**Desirability setting**
Sugar maple basal area	40	High desirable (1)
Hemlock basal area	0	
Sugar maple stress	40	Low desirable (−1)
Hemlock susceptibility	0	
Forest conversion risk	20	High desirable (1)
Sum	*100*	

Table 1. Customized objective weights and desirability settings to identify high sugar maple abundance, high conversion risk, and low climate-induced stress for sugar maple across the area of interest.

climate was identified as a potential driver, compare how suitability outcomes may change under different climate conditions. Note that not all management objectives will have climate scenario products. Some of the empirical models included here have found significant climate effects, resulting in products for multiple climate scenarios (e.g., sugar maple condition and hemlock risk). Other empirical models (e.g., fragmentation, species distributions) found no relationship between climate variables and thus do not include different climate scenarios for comparison in the **FForGeoVDS** tool.

4.1. Case study 1: land trust director

Context: A hypothetical land trust organization can secure funds to acquire new properties for conservation. The land trust has a focus on promoting habitat connectivity, and the donor is interested in the iconic sugar maple. Combining these, the land trust's goal is to identify private properties that are (1) dominated by sugar maple, (2) at higher risk of development, and (3) at a lower risk of climate change-induced maple decline, weighted as described in **Table 1**. They will use output information to identify parcels best able to provide sugar maple refugia under climate change, while also maintaining forest connectivity.

Area of interest: Private, nonconserved lands in Vermont.

Input empirical models: Sugar Maple Stress Index Model, Percent Sugar Maple Basal Area, Forest Conversion Model.

Ancillary data layers:

Town boundaries and parcel boundaries – Vermont Centre for Geographic Information.

Habitat connectivity: Vermont Habitat Blocks and Wildlife Corridors: An analysis using Geographic Information Systems, Vermont Fish & Wildlife Department, 2011, by Eric Sorenson - Vermont Fish & Wildlife Department and Jon Osborne - Vermont Land Trust.

Conserved Lands, Vermont Protected Lands Database, Vermont Centre for Geographic Information. Last updated 2016.

Summarize by: Town and parcel.

Climate scenarios: Sugar Maple Stress Index Model with inputs for Historic norms, A2 (High emissions) and B1 (low emissions) scenarios.

Output maps (**Figure 6**) of this weighted prioritization of high sugar maple abundance and low sugar maple climate induced stress, with high risk of conversion, show that management suitability on private lands across the study area varies geographically, with ideal conditions concentrated in northern and central Vermont. It is also clear that climate change has the potential to decrease suitability considerably across the landscape, although with effects that differ geographically.

The user has defined the area of interest as private lands overlapping habitat with high connectivity value and set a suitability threshold of 60 to summarize output data. Within this area of interest, summary statistics were generated for each town (**Table 2**), allowing the hypothetical user to identify potential regions to target for land acquisition. These summary statistics identify several towns where the prioritized management objectives are likely to be met (highest mean suitability values for pixels above the 60 threshold). Zonal statistics, output in table format, summarize the mean suitability score above the threshold, area of the town above a threshold, and the percent of the total town area that meets the suitability threshold. This allows of hypothetical land trust to prioritize locations to search for potential conservation properties. Under historical norm climate conditions, St. Albans Town comes out as a top location, with the highest mean suitability score and over 1.2 million hectares that meet at least a 60th percentile for suitability.

However, considering future climate conditions, a new ranking emerges due to differential projections of climate change across the study area (**Table 3**). Under the low and high

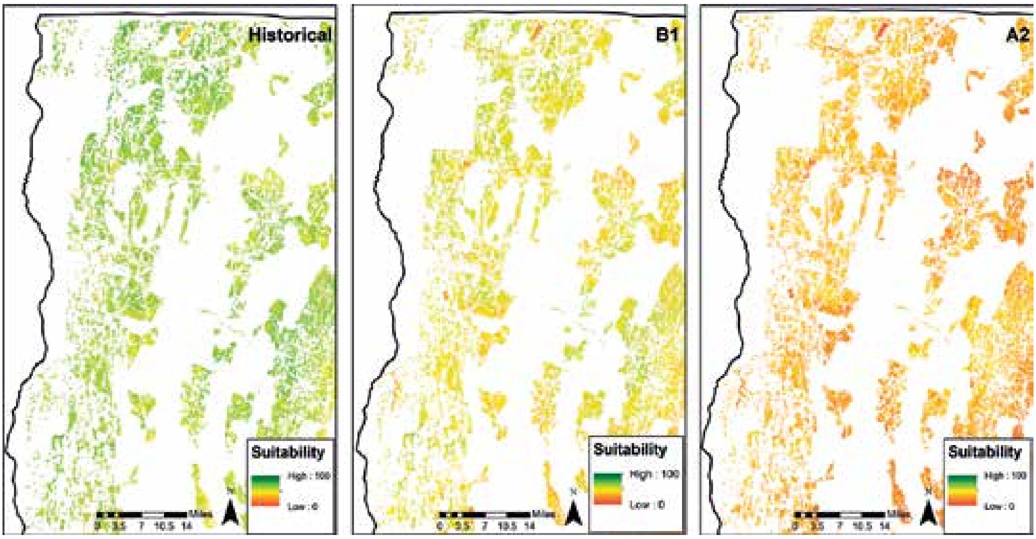

Figure 6. Output maps of weighted suitability for every 30 m pixel where sugar maple is present on privately owned parcels across the selected study area demonstrate how relative suitability differs geographically and under various climate change scenarios.

Summary Statistics by Town—Current Climate			
Town	**Mean suitability above threshold**	**Area above threshold (ha)**	**% Town area**
St. Albans	70.1	1,215,486	9
Montpelier	69.9	729,567	31
Highgate	69.6	2,532,465	18
Swanton	69.4	1,430,865	10
Berlin	69.3	1,458,972	17
Franklin	68.7	2,008,881	21
Waterbury	68.5	807,084	7
Mooretown	68.5	820,449	9
Fairfield	68.4	3,899,907	24
Georgia	68.2	2,588,922	25

Table 2. Summary statistics on suitability outcomes under current climate conditions are summarized by the towns and sorted by mean suitability above a 60 suitability threshold.

Town	Mean suitability above threshold	Area above threshold (ha)	% Town area
Summary Statistics by Town—B1 Low Emissions Scenario			
Orange	63.6	2,106	0
Addison	63.5	120,528	1
Troy	63.2	209,223	2
Highgate	63.1	628,641	4
Newport	63.1	127,494	1
Irasberg	63.0	216,594	2
Barton	62.9	70,551	1
Montpelier	62.8	228,420	10
Johnson	62.7	57,834	1
Weybridge	62.7	95,013	2
Milton	62.7	476,928	3
Summary Statistics by Town—A2 High Emissions Scenario			
Troy	60.6	648	0
Franklin	60.3	81	0
Highgate	60.0	81	0

Note, only those towns with area above the 60 suitability threshold are included.

Table 3. Summary statistics on suitability outcomes under the B1 (low emissions, upper) and A2 (high emissions, lower table) climate scenarios are summarized by town and sorted by mean suitability above a 60 suitability threshold.

emission scenarios, mean suitability and area above the 60th percentile threshold drop for all towns. But this drop is more significant for some, with St. Albans Town dropping off of the top 10 list for both low and high emission scenarios entirely. This indicates that if the Land Trust wants to consider sugar maple resilience under future climate conditions, the town of Orange (low emission) or Troy (low and high emissions) is likely to have more suitable locations. However, the area above the 60th percentile threshold is low for both towns, indicating that it may be difficult to find available properties for conservation efforts. If the small proportion of parcels that are most suitable for purchase are not for sale, the Land Trust could target a town such as Troy, VT that has a similar mean suitability but over a much larger area.

Parcel maps can then be used to identify key properties to target within the town of Troy for conservation purchase. The suitability output maps (**Figure 7**) show us that there are several properties with a high suitability score for potential conservation efforts.

This exercise allows the Land Trust to identify and compare potential properties for sugar maple conservation in locations where forest conversion risk is relatively high, but sugar maple abundance and resilience in the face of climate change are also high. This represents a tangible, decision product that can be used to guide and justify actions taken by the Land Trust to preserve potential sugar maple refugia and attract potential funding.

4.2. Case study 2: state forest planner

Current models suggest that hemlock wooly adelgid will disperse through the majority of eastern hemlock's native range, but with impacts that vary widely depending on site conditions. This hypothetical state forest planner wants to identify hemlock stands within state forests that contain high-density eastern hemlock stands that are most likely to tolerate HWA infestation, weighted as described in **Table 4**. The goal is to manage for hemlock in areas that are likely to serve as long-term seed source for this species.

Area of interest: Vermont State Forests.

Input empirical models: Percent Hemlock Basal Area, Forest Conversion Model, Hemlock Risk Model.

Ancillary data layers: Conserved lands, Vermont Protected Lands Database, Vermont Centre for Geographic Information. Last updated 2016.

Summarize by: Forest management unit.

Climate scenarios: Hemlock Risk Model with inputs for Historic norms, A2 (high emissions), and B1 (low emissions) scenarios.

Output maps of this weighted prioritization of high hemlock abundance and low hemlock risk on state lands indicate that much of the state's forest land is suitable for hemlock management under current climate conditions (**Figure 8**). However, some state forests, particularly those in central and eastern VT, are particularly at risk from the invasive HWA under future climate scenarios. In particular, we begin to see the impact of heat and water stress on the relatively drought intolerant hemlock manifest in lower suitability scores on southern facing slopes.

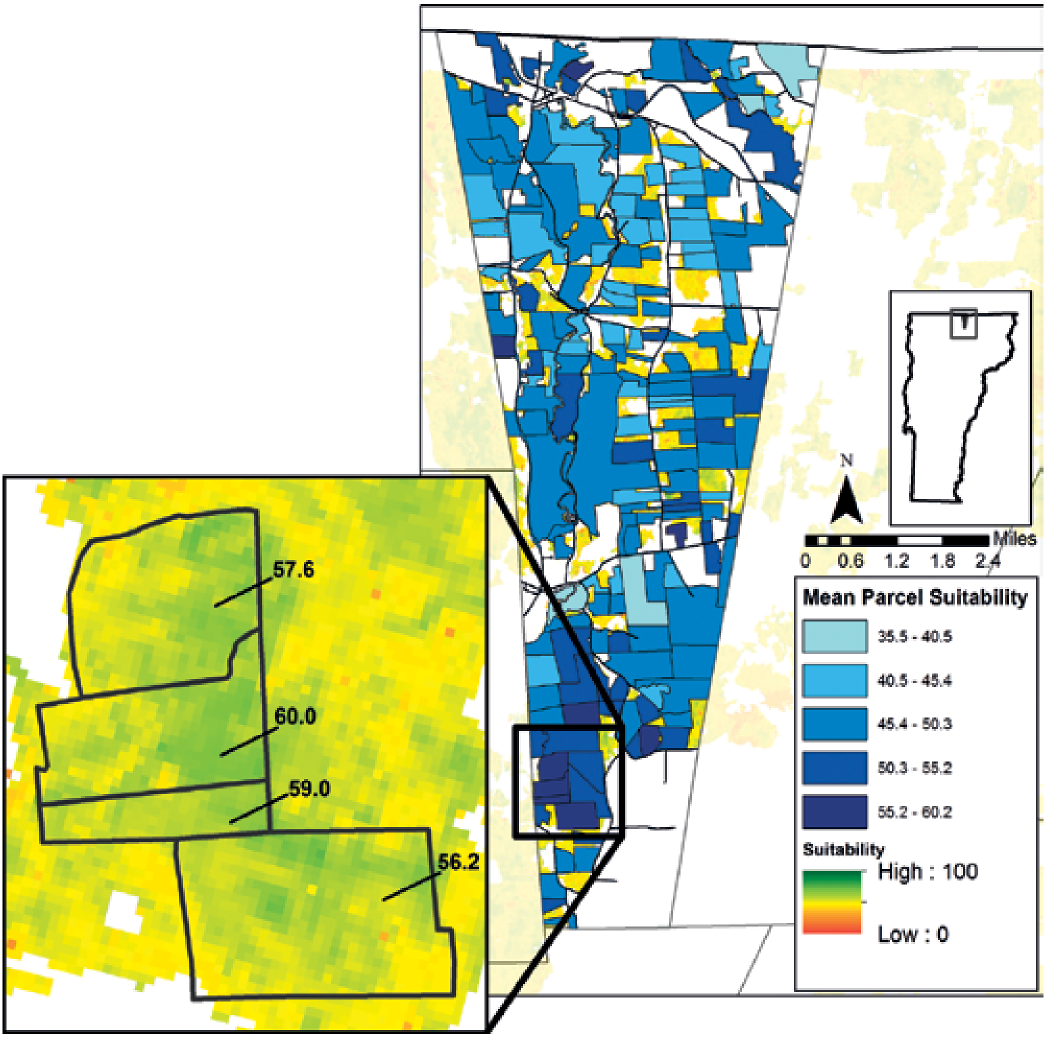

Figure 7. Mean suitability for parcels >25 acres (10.1 ha) for the town of Troy, Vermont, and a close-up look at suitability rankings on four parcels with high means (inset).

State Forest Planner (Hemlock Focus)		
Objectives	**Weight**	**Desirability setting**
Sugar maple basal area	0	
Hemlock basal area	50	High desirable (1)
Sugar maple stress	0	
Hemlock susceptibility	50	Low desirable (−1)
Forest conversion risk	0	
Sum	*100*	

Table 4. Customized objective weights and desirability settings to identify high hemlock abundance and low HWA risk across the area of interest.

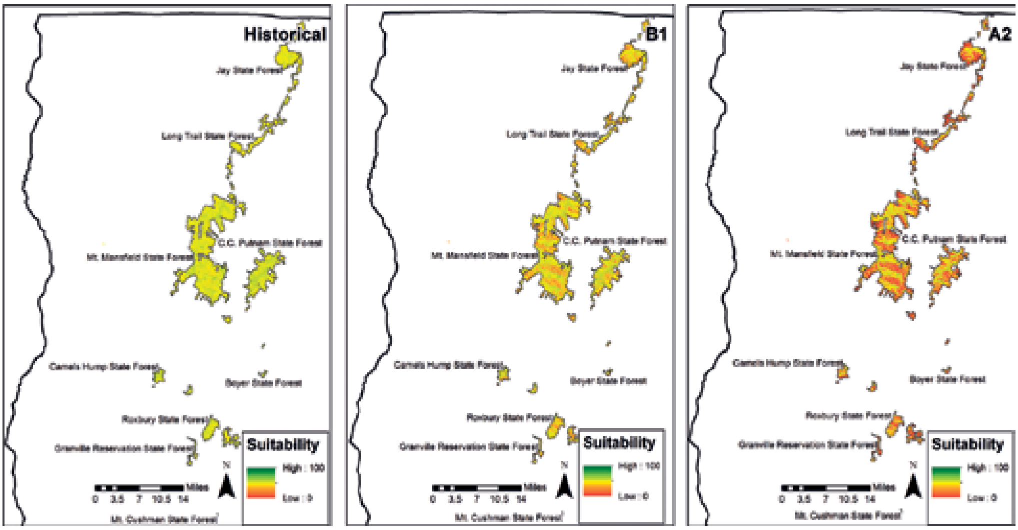

Figure 8. Output maps of weighted suitability for every 30 m pixel on state-owned parcels across the selected study area demonstrate how relative suitability for hemlock refugia differs geographically and under various climate change scenarios.

The State Forest Planner has used Vermont's state forest boundaries as the area of interest and as the unit for generating summary statistics and has set a suitability threshold of 60. Opting for summary data in chart rather than table form, the State Forest Planner can see which of their forests are ideally suited for hemlock management, and quantify which are most likely to decrease in suitability under a changing climate (**Figure 9**).

In this example, Camels Hump State Forest has the highest proportion of its area above the 60th percentile suitability threshold under current climate conditions, while Mt. Mansfield has the highest total area. Considering the proportion of suitable land, two smaller forests rank highest under the climate change scenarios, with 12.9% of Boyer State Forest above the threshold under the low emission scenario and 4.7% of Mt. Carmel State Forest above the threshold under the high emission scenario. But in terms of absolute area, Mt. Mansfield and CC Putnam State Forests provide the largest base of suitable landscape. Comparison of the climate scenarios also helps to identify how vulnerable hemlock stands in the different forests may be to climate change. For example, the proportion of suitable hemlock stands in the West Rutland State Forest changes modestly between the climate norms and B1 scenarios. In contrast, the Camels Hump State forest loses almost half of its suitable hemlock under the B1 scenario.

These maps and data give the Forest Planner a set of criteria that they can use to evaluate against other factors, such as funding, staff capacity, and institutional opportunities for experimentation. The forest planner can take these results and analyze trade-offs to target a smaller forest as a demonstration project or spreading management efforts across a larger land area while considering the long-term effects of climate on their hemlock management

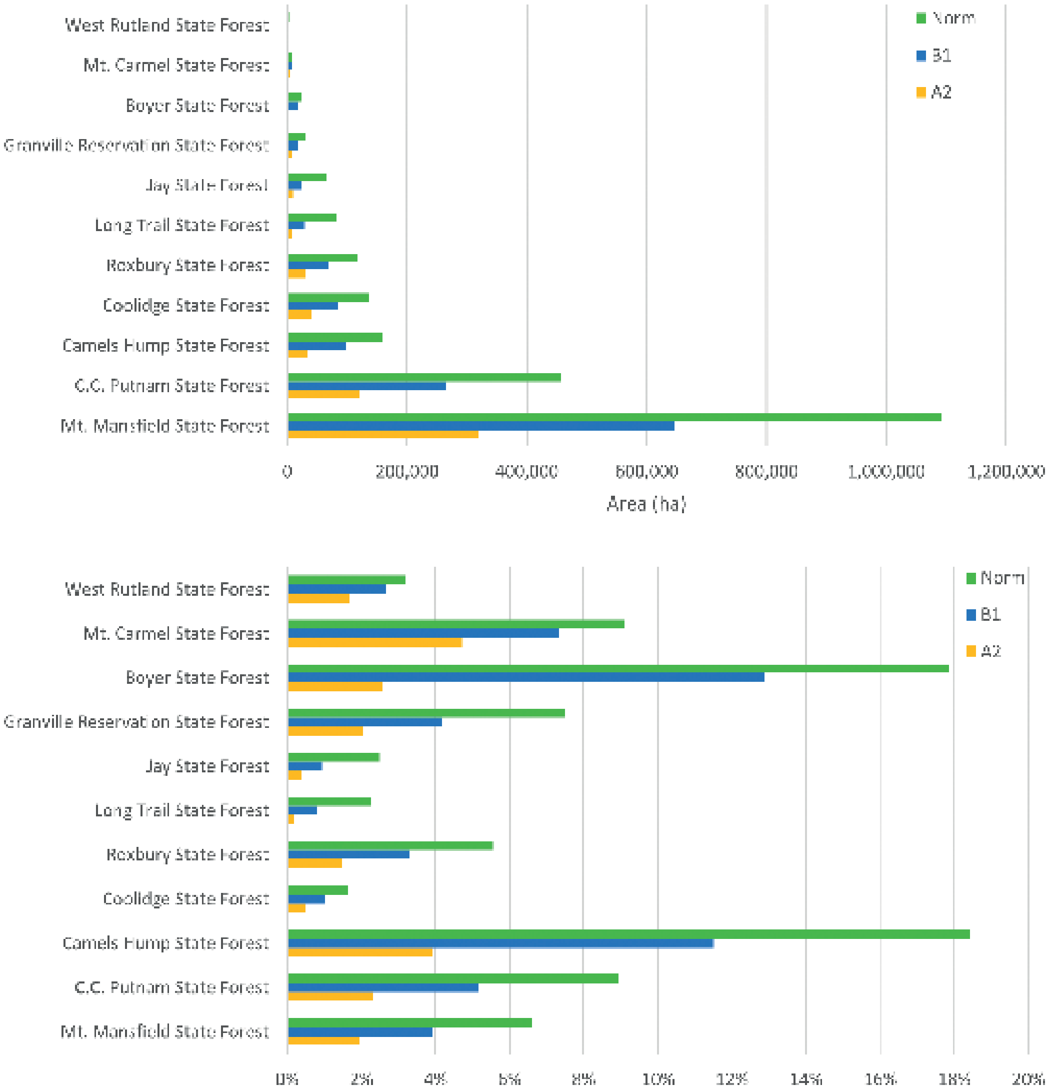

Figure 9. Graphical output summarizes the total area (top) above the specified suitability threshold for the areas of interest (VT State forest lands) under three different climate scenarios. This information can also be summarized as percent of total area (bottom), which helps to emphasize the importance of smaller parcels.

goals. This information provides necessary information to target management activities on the ground and justification for management decisions.

4.3. Case study 3: climate action advocacy group

Presenting before the State Legislature, this organization is looking to demonstrate the potential severity of climate change impacts on the state's largest natural resource: its forests.

Specifically, they want to quantify how our ability to maintain sugar maple and eastern hemlock stands across the state is likely to change under various climate projections, with all management objectives equally weighted as described in **Table 5**. They are particularly interested in private lands because Vermont's forests are 80% privately owned, and they want legislators to provide new incentives for landowners to implement climate-resilient management strategies for these important species.

> *Area of interest*: Vermont private lands.
>
> *Input empirical models:* Percent Hemlock Basal Area, Percent Sugar Maple Basal Area, Sugar Maple Stress Index Model, Forest Conversion Model, Hemlock Risk Model.
>
> *Ancillary data layers*: Conserved lands, Vermont Protected Lands Database, Vermont Centre for Geographic Information. Last updated 2016.
>
> *Summarize by*: State of Vermont.
>
> *Climate scenarios*: Sugar Maple Stress and Hemlock Risk Models with inputs for Historic norms, A2 (high emissions), and B1 (low emissions) scenarios.

Output maps of this objective prioritization that equally weight all available management objectives demonstrate the potential impacts of climate change on the region's sugar maple and hemlock resource (**Figure 10**). The user has chosen nonpublic, nonconserved lands as their area of interest to capture the private ownership that dominates in VT, and selected to generate summary statistics for the entire region.

First, it is interesting to note how the geographic patterns of suitability scores vary significantly for this combined outcome weighting (emphasizing both sugar maple and hemlock) compared to an emphasis only on sugar maple in Case Study 1 and only on hemlock in Case Study 2. This highlights the importance of targeting specific management objectives, including allowing for a complex combination of multiple management objectives. The decisions that result from these differential weightings can vary widely.

Advocacy Group (Climate Change Focus)

Objectives	**Weight**	**Desirability setting**
Sugar maple basal area	20	High desirable (1)
Hemlock basal area	20	High desirable (1)
Sugar maple stress	20	Low desirable (−1)
Hemlock susceptibility	20	Low desirable (−1)
Forest conversion risk	20	Low desirable (−1)
Sum	*100*	

Note that the key information desired from this case is a comparison of how climate impacts our ability to maintain suitable habitat for all listed management objectives.

Table 5. Customized objective weights and desirability settings to integrate across all management objectives.

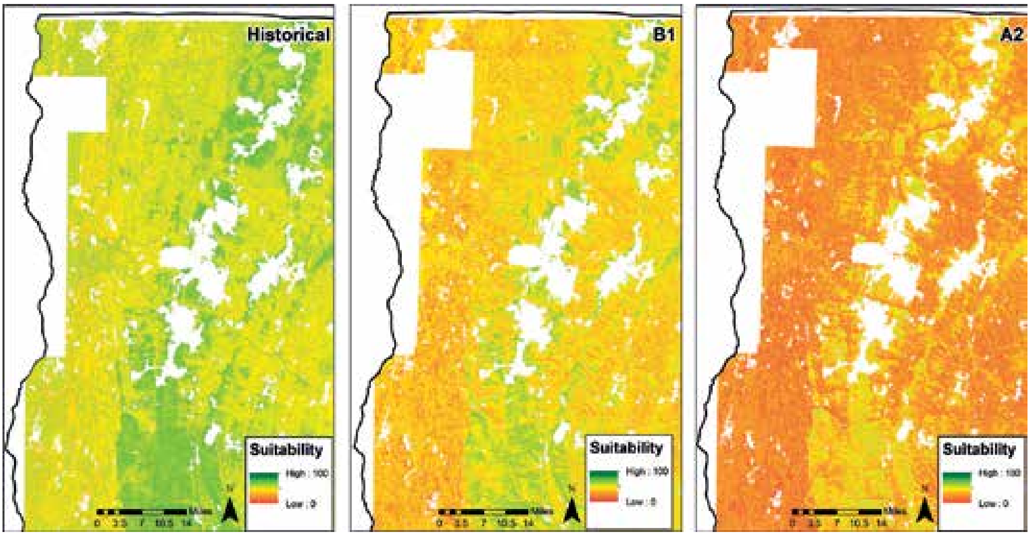

Figure 10. Output maps of weighted suitability for every 30 m pixel on private owned parcels with either hemlock or sugar maple present demonstrate how relative suitability differs geographically and under various climate change scenarios for this customized analysis equally weighting all available management objectives.

Second, it is striking how drastically overall suitability declines under both low and high emission scenarios (**Table 6**). Across the study area, suitability drops considerably, with a study-wide mean of 56.3 under historical norms dropping to 44.3 under low emissions and 28.8 under high emissions. This drop is particularly pronounced along the Lake Champlain Valley where a large portion of Vermont's population is located.

Using the current suitability values as a baseline representing the current "standard" of forest health, our hypothetical climate advocate stakeholder group could quantify the proportion of the region's forest lands that fall more than a standard deviation below current "mean/baseline" suitability. This provides a tangible value to present to policy makers they hope to influence to promote climate policy. In this management prioritization scenario, the user can say that 65.6% of the region's forest will become less suitable to sustaining a healthy sugar maple and hemlock forests under a low emission scenario. They can further demonstrate that 95.2% of the region's forest will become less suitable under a high emission scenario.

Climate scenario	Mean suitability	Maximum suitability	Range in suitability
Historical norms	56.3	79.9	55.6
B1 – low emissions	44.3	73.3	58.5
A2 – high emissions	28.8	65.6	59.7

Table 6. Summary statistics of suitability scores across the entire study area demonstrate how suitability can be expected to change under various climate scenarios.

With these substantial drops in suitability, the advocate can make an argument that Vermont will face significant challenges in maintaining hemlock, and maple species on private lands under even generous climate change scenarios, arguing for proactive efforts by the state to encourage landowners to take action. This may include implementing management efforts on private lands directed at managing forests for resilience and resistance. Or it could be used to justify policy changes to limit or mitigate greenhouse gas emissions. The maps, summary statistics, and underlying science provide the advocate with a set of materials to take to a lobbying session or public hearing where they can advocate for better protections and management of important forest resources across the state.

5. Conclusions

One challenge that many natural resource managers face is the uncertainty of climate change and the effects that large-scale changes will have on resources at a parcel- to landscape scales. With the ability to explore different climate change outcomes for a particular set of objectives, managers can find bounds to their expectations, lowering the barrier to incorporating climate change into management planning.

While many forest decision support tools exist, there is a notable gap in the ability to combine projections of climate change with current ecosystem models, linked to discrete forest management objectives in a flexible way pertinent to their scale and extent of interest. This puts the Future Forests Geo-Visualization Decision Support (**FForGeoVDS**) tool in a particularly good position for adoption and use to a broad audience of stakeholders. Because outputs include quantitative metrics to identify locations across a property or landscape where various management objectives are most likely to be met, the possibility of selecting different locations is highly dependent on objective weightings. This may prove particularly useful for conservation efforts, where resources to conserve targeted properties are limited, but conservation goals are complex.

The case studies we have presented here outline three potential uses for this tool. The ability to refine and target areas meeting multiple objectives, as in the case of the land trust director, or the ability to compare the suitability of a short list of state-owned properties, as in the case of the state forest planner, provides powerful new ways of combining scientific models, GIS data layers, and climate projections to generate detailed maps, useful summary statistics, and motivating graphics. However, we expect to see a more diverse range of users as the decision tool is refined and additional empirical models are included (e.g., wildlife occupancy, recreational patterns, carbon storage, etc.). This increases the number of management objectives one can consider and thus the robustness of the tool.

The **FForGeoVDS** research team has actively engaged with land managers and decision makers from nonprofits, state and federal natural resource departments and agencies, and advocates to understand what questions they face and what outputs and supporting material a decision support system focused on climate change and forests should provide. These working sessions have proven invaluable in the development and design of the tool. Because of the diverse nature of these stakeholder groups, initial feedback has ranged from a desire for more detail, flexibility and complexity, to a desire for more simplistic approaches. This exemplifies the importance of continuous interaction with stakeholders to find the best fit for the most users and the need for continual updates, additions and increased functionality to the tool itself.

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Chapter 7

Alternative Forest Restoration Techniques

Sebastião Venâncio Martins

Additional information is available at the end of the chapter

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Abstract

This chapter seeks to describe the need to adapt the forest restoration to the different regional environmental filters and the different ecological opportunities of the landscapes, through the adoption of techniques not conventional or alternative restoration. When starting this text, it should be made clear that all restoration models and techniques have their environmental and socioeconomic importance, since they contribute to the return of forest ecosystems to a non-degraded state, with direct or indirect impacts on the recovery and conservation of hydrological and nutrient cycles, biodiversity, agricultural production, and the minimization of climate change. Therefore, there is no pretension here to present a set of models and techniques that are “superior and unique” and that should be standardized and followed throughout the country. To be clear in this text, there are innumerable possibilities and alternatives for forest restoration in Brazil, given its continental dimensions, with remarkable climatic, edaphic, cultural, and socioeconomic diversity. Therefore, there is no single restoration model or technique that can be applied widely and on a large scale; what is important is to take advantage of the remaining potential for ecosystem regeneration by adapting more appropriate techniques for each situation.

Keywords: ecological restoration, natural regeneration, nucleation, direct seeding, topsoil transposition

1. Introduction

As we started this text about what is being called “alternative forest restoration techniques,” it should be made clear that all restoration models and techniques have their environmental and socioeconomic importance since they contribute to the return of forest ecosystems to a non-degraded state, with direct or indirect impacts on the recovery and conservation of hydrological and nutrient cycles, biodiversity, agricultural production, minimization of climate change, and the well-being and quality of life of human populations.

Therefore, it is not our claim to present a set of models and techniques that are "superior and unique" and that should be standardized and followed all over the country. Indeed, standardization and the attempt to create rigid norms for restoration projects are strong features of the more conservative and traditional forest restoration, as it is well illustrated in the text by [1].

Forest restoration's record in Brazil and in all the world, well documented by [2–6], reveals that innumerous initiatives of restoration have begun in a more empirical way, based on practical knowledge and without following preestablished norms, even because the "Ecology of Restoration" as the Science that guides the foundations of restoration (see [7]) did not even exist at that time. However, it cannot be denied that these first initiatives of ecological restoration of the forests, although without much scientific criteria, have resulted, in many cases, in forests with good quality of restoration, within the context of total or partial return of biodiversity and environmental or ecosystem services.

Over this trajectory of forest restoration in the world, one can see from the last two decades a great advance in the number of projects and area in restoration, with important initiatives of large-scale restoration and mainly focused on the riparian forests (areas of permanent preservation), which have been intensified with the demands arising from the 21st Session of the Conference of the Parties to the United Nations Framework Convention on Climate Change (COP 21, Paris, France), reinforced in the COP 22 (Marrakech, Morocco) and COP 23 (Bonn, Germany). In this new trend of restoration, an attempt is also made to standardize or dictate very strict rules for forest restoration, which often fails to consider the differences and specificities of each region, which presents a set of filters that rule the structural and functional organization and biodiversity of natural ecosystems and, of course, of ecosystems to be restored as well.

Thus, the objective of this chapter is to explain the need to adapt forest restoration to the different regional environmental filters and to the different ecological opportunities of the landscapes, translated here as the potential for self-recovery—or resilience—of ecosystems through the adoption of unconventional or alternative techniques of restoration, either alone or combined with traditional techniques.

2. The traditional forest restoration

Although it is sometimes difficult to establish a strict dividing line between what is traditional or conventional and what is alternative in the forest restoration, it is possible to consider as traditional the reforestation in a total area with fixed spacing between the planted seedlings and, as an alternative, all initiatives that escape this pattern of restoration.

It is important to emphasize that the ecological succession responsible for the regeneration process of forest ecosystems is initiated by the seeds present in the soil bank or from the seed rain and by the growth of stalks and root buds of various forms of life, usually forming regeneration nuclei; then why do we seek to restore forests only by planting tree seedlings, following alignments and predefined spacings?

The answer is not simple; it has a bias in the use of silvicultural and agronomic techniques, in the state of local degradation and landscape, in the differences between seed and seedling offerings, and in the costs of restoration in the different regions [6, 8].

The heterogeneous reforestation with seedlings of native species has its origin and mirror, the forestry, that is, this type of traditional forest restoration follows exactly the silvicultural techniques applied in the reforestation with *Eucalyptus* and *Pinus* to produce wood for cellulose, charcoal, etc., exchanging species, only.

To make it clear, this approach does not seek to criticize or diminish the socioeconomic and even environmental importance of *Eucalyptus* or *Pinus* forestry for timber production, which generates thousands of jobs, strengthens the country's economy, and helps to reduce pressure on forests and to protect soils from erosive processes, among many other benefits [9].

When, for example, a forest restoration project is implemented with seedling planting following an alignment and spacing of 3 × 3 m between seedlings, with fertilization of 200 g of NPK 6-30-6 per pit, nothing else is being done than to follow exactly the spacing and fertilization used in many reforestations with *Eucalyptus* in soils with low levels of phosphorus. Although the defenders of this restoration model may refute by claiming that restoration uses a high diversity of native species, in some cases from 80 to 120, following functional groups with pioneer and non-pioneer lines or fill and diversity lines.

Once more, with the artificial organization that tries to impose the forest to be restored, the classic paradigm of ecology and succession [10] is present as never when defining functional groups, spacing, number of species, and, worse, strict parameters and criteria for monitoring such forests in restoration, based on control of mature forests. It is important to highlight that a diversity similar to that of the original ecosystem can be achieved over time, even with the planting of a single or few species, provided that the landscape is resilient, as verified, for example, in the Forest Garden of Campos do Jordão, State of São Paulo, Brazil, where the planting of *Araucaria angustifolia* alone served as a catalytic regenerating forest of the other regional native species [11].

It must be made clear that native forests have the diversity and arrangement of species controlled by ecological processes such as dispersal, predation, competition, nutrient cycling, and physiographic factors such as altitude and terrain slopes, among others, and that besides tree species, various other life forms compose the forest ecosystem, such as epiphytes, creepers, understory plants, and associated fauna. So, it is not uncommon to find forests considered as "restored" that even after two or more decades of traditional reforestation still maintains planting alignment and a virtually empty understory with extremely low density of regenerant shrub-trees.

Nevertheless, it cannot be disregarded that in certain situations, it is necessary to plant native tree species, for example, in the case of highly anthropic landscapes where resilience has been lost or is very low.

In addition, in the context of wood production of native species ("e.g., see [12]"), which is more like a rehabilitation of degraded areas, and not actually restoration, reforestation is more recommended, actually, since it allows the combination of groups of species of wood of different uses and characteristics in the planting lines, since a production control is necessary. Moreover, in relation to implantation of agroforestry or agroforestry systems—rehabilitation models—the planting of predefined lines of native trees and agricultural crop lines seems to be the most appropriate. It is worth noting that these rehabilitation models for the production of timber and non-timber products, when temporary, can be converted into restoration models, although they maintain the characteristics of the productive system for a long time [13].

In the context of forest restoration, interpreted as an ecological restoration of forest ecosystems, what is also essential to make clear is that planting of seedlings does not always need to be in the total area, nor does it always have to have a predefined, narrowed spacing, regardless of region, landscape, climate, etc. Therefore, it is not possible to accept for forest restoration a generalization of reforestations with predefined spacings, usually 3 × 3 m or even 2 × 2 m, and many species, which neglect the potential for forest regeneration, which could be stimulated through alternative techniques, with sensible reductions in restoration costs and the formation of forests through a more natural and sustainable process in the long term.

It is not possible to adopt in a generalized way for the different Brazilian regions, with their different ecological peculiarities, public policies of forest restoration based only on the reforestation through the planting of seedlings in total area, which ends up being more unfeasible than contributing to the advance of the restoration.

A good example of this paradigm shift has been occurring in the State of São Paulo, Brazil, where a set of legal norms, through resolutions issued by the Secretariat for the Environment (SMA), was initially launched to improve project quality and to speed restoration. These first resolutions, although well intentioned, unfortunately complicated and made it difficult rather than facilitating or stimulating restoration in the State and, after a long debate between the scientific community of São Paulo and of other states, which resulted in at least three publications of great impact [1, 14, 15], ultimately have become appropriate and have accepted different restoration models and techniques, provided that the final result is successful. In this context, the following are some alternatives for forest restoration, which, in general terms, pursue the formation of restored forests in a more ecological and sustainable way and with lower costs than traditional reforestation.

3. Revamping traditional reforesting

It is possible and feasible to make a reforestation with less traditional and less conservative. To do so, it is necessary to innovate and to create new alternatives of planting arrangements in the field, and this is configured as an open field for avant-garde studies.

Changing planting spacings according to the reality of each landscape or region means allowing the intensification of planting of seedlings to the regions or farms that really need this type of intervention. Thus, changing spacing may represent a reduction of 1100 seedlings per hectare (at 3 × 3 m spacing) to half or even a third of that, which at first may seem a reduction in the demand for seedlings, but when it is considered that using fewer seedlings per hectare can restore more hectares, all in all, the production and commercialization of seedlings may be little impacted or not impacted at all.

It is very common in reforestation to clean the area to be planted first. Such cleaning often eliminates everything; nothing is left after cutting followed by application of herbicide in a dirty pasture. In this cleaning of the area, it is common for many seedlings that were starting a process of succession to be eliminated, to give place to the planted seedlings, some that do not always occur in the region.

One of the first assumptions to make reforestation more ecologic and alternative is the use of regenerants, for example, if spots or nuclei of natural regeneration occur in an area of a few hectares, they must be maintained and the planting spacing altered. As a rule, in the vicinity of forest fragments, natural regeneration is more abundant, and it is reduced as it moves away from the remnant; this is a pattern of forest succession [16] in abandoned pastures [17–19], for example, and for these cases, the planting of seedling in broader spacings, such as 4 × 4 to 6 × 6 m in the parts with more regeneration, or even no planting of seedlings if regeneration is already very intense (**Figures 1** and **2**) is an alternative.

Another important aspect to be considered in traditional reforestation in total area is the uniformity, represented by the more or less standardized height of the seedlings. Although certain pioneer species show very rapid growth, for example, *Schizolobium parahyba*, as a rule, most native species grow slowly in height, and thus in the first years of planting, the area becomes a uniform carpet of seedlings of short height. Such an environment is unattractive for seed dispersal birds to move from isolated forest fragments in the landscape.

An alternative to make the planted area more attractive to birds is the installation of artificial perches made of bamboo, *Eucalyptus* sticks, and other materials [3, 20]. From these perches,

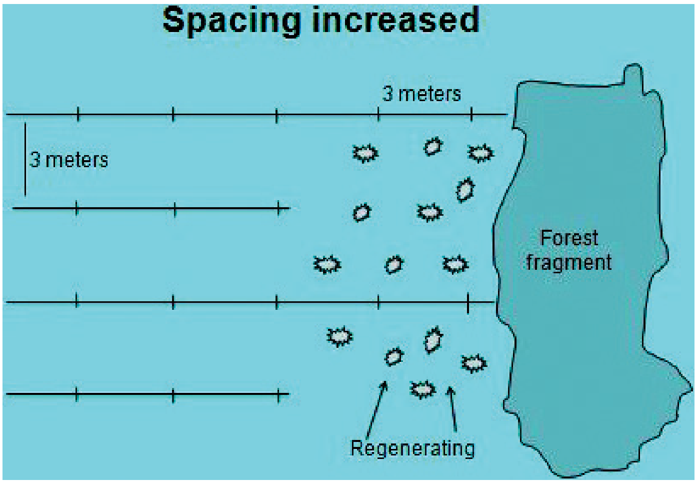

Figure 1. Schematic representation of the increase in spacing between seedlings according to regeneration potential.

Figure 2. Extracts with different regeneration potentials; in the upper part of the slope (A), only the enrichment with planting of seedlings in nuclei in the regeneration faults was indicated; in the lower part of the slope (B), the planting in smaller spacing, always maintaining the regenerants, was pointed. Picture: Sebastião V. Martins.

installed at intervals of 20–50 m, birds can move and disperse fruit seeds that they used in forest fragments still existing in the landscape. With the installation of perches in the reforestation, larger spacings can be adopted inasmuch as nuclei of natural regeneration tend to be formed around the perches.

There may be also natural perches by planting fast-growing species, including exotic species, since some have much higher growth rates than most native species, provided they are not invasive. In this sense, the planting of isolated trees of fast-growing species, at 50-m intervals within the wide space of the reforestation with native ones, can be a good alternative. When these isolated trees reach a height of about 10 m, they can be killed with herbicide application on the stem and become dry perches for some years.

Another way to make reforestation less conventional and more ecological is by neither adopting planting alignments nor spacing between seedlings, that is, to carry out planting at random. Thus, in the development of the planted forest, the planting lines will not be maintained, as in traditional reforestation, making the restored forest more similar to a native forest. As the planting of the seedlings becomes more flexible, it is possible to increase the seedlings in worse stretches, for example, in exposed and compacted or eroded soils, and to increase the distance between seedlings in stretches that already present arboreal regenerants or colonization by ruderal shrubs.

4. Natural regeneration

Natural regeneration, understood as the process by which an ecosystem impacted by natural or anthropic disturbances recovers its total or partial biodiversity, its structure and functioning, through the successional sequence over time, is undoubtedly the most ecological and cheapest manner of restoring forests.

Several studies have been published and proved the viability of natural regeneration as an alternative to forest restoration in abandoned pastures [17–20], in mined areas [16, 21] and in large-scale restoration [8, 22].

The study of [19] in Puerto Rico demonstrated that one restoration strategy for tropical forest in abandoned pastures is simply to protect the areas from fire and allow natural regeneration to produce secondary forest. In accord with the authors, this strategy will be most effective if remnant forest (i.e., seed sources) still exists in the landscape and soils have not been highly degraded.

Probably the study of [8] about "Natural regeneration potential of native forests in the different regions of the State of Espírito Santo, Brazil," is the strongest evidence of the role of natural regeneration as an alternative to enable large-scale forest restoration in Brazil. This study, the result of a major project that analyzed natural regeneration on farms throughout the state of Espírito Santo, revealed that in a 33-year period, 18,979 forest fragments were naturally regenerated in lands in Espírito Santo, occupying an area of 106,554.87 ha. The study also showed that in the State of Espírito Santo, 60.88% of its total area, equivalent to 2,804,431 ha, has a high potential for natural regeneration of forests and in most of such areas, planting of seedlings for forest restoration was not necessary.

Despite all this potential of natural regeneration detected in Espírito Santo, which is probably repeated in other States, and it should be even greater in the region covered by the Amazon Rainforest, the study by [8] also revealed that there are areas of that state with low regeneration potential, notably the far North, where only the abandonment of the areas does not guarantee the regeneration of the forest and restoration interventions are necessary.

The findings of this study in Espírito Santo are very relevant because they indicate that large-scale forest restoration cannot follow a single model or a standardization. The five defining elements of ecological restoration—ecological, economic, social, cultural, and political—described in [23, 24] also move toward it, from this alternative approach.

By considering only the economic element, it is necessary to reflect on the main bottleneck to enable forest restoration through reforestation in a total area, which is its high cost, which varies greatly from one region to another, but which is hardly less than R$ 7000.00 per hectare, not being rare examples exceeding R$ 20,000.00 per ha, and being suggested an average of R$ 10,000.00 per ha [2].

Through natural regeneration, this cost is substantially reduced to one-third or even zero, since it basically comes down to the cost of fencing and to construct fire breaks of areas under regeneration, for the isolation of livestock and fire, costs that also occur in traditional reforestation. **Figure 3** shows a ciliary forest restored through the natural regeneration process, where the only intervention was the enclosure of the APP range, to prevent entry of cattle.

But as it was evident in that study in Espírito Santo, where most of the state's territory has a high potential for natural regeneration, there are areas that the forest may not regenerate naturally, and thus even natural regeneration cannot be indicated as exclusive restoration technique for all regions, landscapes and situations of environmental degradation.

Figure 3. Restored ciliary forest by means of natural regeneration. Forest restoration project LARF-UFV. Picture: Sebastião V. Martins.

As regeneration depends on three basic ecological mechanisms—seed rain, soil seed bank, and regrowth of vines and roots—in very anthropic landscapes, with a long history of intensive use only for agriculture or livestock and with forest remnants absent or very isolated and degraded, this process may not occur or be extremely slow.

In conclusion, before recommending natural regeneration as an exclusive restoration technique for a particular area, it is essential to make a diagnosis of its regeneration potential. The diagnosis should consider the landscape in which a particular area to be restored is inserted, the distance of forest fragments remaining in the landscape, the presence of regenerants and their abundance, and, if possible, the soil seed bank [25, 26]. In this aspect, [27] point that the natural regeneration techniques, however, are not innately superior or always more appropriate for restoring forest ecosystems than artificial techniques and the intervention must be based on management objectives, informed by evaluation and interpretation of site conditions, and incorporate silvicultural knowledge and skills. Where management objectives are best served by controlling the timing of restoration and the species composition of the restored forest, active intervention at the regeneration stage is critical [27].

In regenerated forests dominated by one or a few species, enrichment management can be recommended, but costs still tend to be much lower than total area planting. A study developed by [28] about polydominant spruce-broadleaved forest long-term economic use in Moscow, Russia, found good results with two management techniques: group-selective cutting aimed at imitating the treefall gaps' natural mosaic structure of uneven-aged forests and combining gap falling with planting of species that occupied dominant position in preagricultural forest.

5. Nucleation techniques

For situations in which only the abandonment of a particular area has not resulted in progress of the natural regeneration process, by the factors already related, it is possible to stimulate and accelerate the process through the adoption of alternative techniques, such as nucleation, exclusively or combined with the reforestation.

The development and successful application of nucleation techniques as alternatives to improve—in terms of cost reduction and sustainability of restored ecosystems—has been widely documented [3, 29–37].

In a review of nucleation researches in several countries, [38] concluded that the results of experimental tests of applied nucleation indicate that the density and diversity of colonists is higher in planted nuclei than in areas where no planting takes place (e.g., passive restoration) and that these studies suggest that the applied nucleation strategy has the potential to restore deforested habitats into heterogeneous canopies with a diverse community composition while being cheaper than projects that rely on plantation designs.

5.1. Seedling nuclei

In abandoned areas, largely isolated in agricultural and pasture landscapes, the emergence over time of sparse, or even dense, coverage of ruderal herbs and shrub nuclei and even pioneer tree species, often forming monodominant communities, where one or a few species are established is common, but succession does not advance in terms of species diversity and biomass. As the main factor hindering the progress of forest regeneration in these situations is the lack of seedlings input, since the sources are very distant and the soil is degraded in terms of seed richness, the output is to potentiate regeneration through artificial introduction of propagules, either seedlings or seeds or both.

The planting of seedling nucleus in areas in slow regeneration process is an excellent alternative to make forest restoration possible in a more ecological way and at a low cost in comparison with the reforestation in a total area.

The ecological principle of seedling planting in nuclei is the realization that in many situations of degradation of terrestrial ecosystems, succession does not begin simultaneously, covering all the abandoned area, a pasture, for example, but generally small nuclei of pioneer facilitating species of succession are formed, which expand over time (**Figure 4**). In their study [39],

Figure 4. Natural regeneration nuclei in degraded pasture. Picture: Sebastião V. Martins.

on primary succession in the Canadian Rockies, pointed out that in looking for plants that might serve such nuclei for colonization during primary succession, plants that fix nitrogen should be considered.

There are several types of nuclei, in terms of number of species and distances between seedlings, but certainly Anderson's Groups are still the most used. In this model, five cross-shaped seedlings with four peripheral seedlings and one central seedling were planted at 0.5 m between the central and peripheral seedlings [35].

Alternatively, seedling nuclei can be formed by planting four seedlings at the ends of a 1-m-side square, with a central seedling, preferably the four peripheral seedlings of pioneer species (each of a species) and the central non-pioneer species (late secondary or climax) that requires greater shading.

Among the advantages of this technique, in ecological terms, is the fact that the pioneer seedlings (**Figure 5**), when growing faster, provide the necessary shading for the late species planted in the center of the nuclei, in addition, formation of nuclei, with seedlings closer than in the conventional plantations, form clusters more resistant to the weather and to the attack of herbivores and pests. The nuclei must be distanced according to the potential of natural regeneration of the area, which allows a reduction in the total of seedlings per hectare, from 1100 to 1666 seedlings from traditional reforestation to about 200–400 seedlings with the nuclei, which, ultimately, dramatically reduces the costs of restoration.

The planting in nuclei also allows a single crowning for the nucleus as a whole, that is, for the five seedlings, which represents a strong reduction in the costs of implanting and maintaining restoration projects, since the crowning around the seedlings in many regions where herbicide application is not accepted by environmental agencies is the only way to avoid competition with aggressive exotic grasses.

Figure 5. The nucleus of pits for planting and *Euterpe edulis* Mart. Seedlings planted in an enrichment nucleus. Forest restoration project LARF-UFV. Pictures: Sebastião Venâncio Martins.

Experimental reforestations were carried by [40] in the Sierra Nevada Protected Area (southeast Spain) with the aim of comparing the survival and growth of seedlings planted in open areas (reforestation) with seedlings planted under the canopy of preexisting shrub species. Their results showed that nurse shrubs had a stronger facilitative effect on seedling survival and growth at low altitudes and sunny, drier slopes and that pioneer shrubs facilitate the establishment of woody, late-successional Mediterranean species and thus can positively affect reforestation success in many different ecological settings.

The study of [41] in the Trevenque area, Sierra Nevada, Spain, tested the hypothesis that the use of shrubs as nurse plants is an alternative technique of reforestation with greater success than traditional techniques, in which preexisting vegetation generally considered a source of competition. They compared the traditional planting of *Quercus pyrenaica* seedlings in open areas with the planting of seedling nuclei in the canopy of a pioneer shrub *Salvia lavandulifolia*. The results showed that *Quercus* survival was 6.3× higher when planted under individuals of the pioneer shrub than open areas and, therefore, the use of shrubs as nurse plants for *Q. pyrenaica* reforestation is a viable technique to increase establishment success [41].

5.2. Plant residue and soil transposition

In areas with soil degradation, such as compaction and erosion, common in degraded pastures and mined areas, planting of seedlings may not be sufficient to provide forest restoration. In these situations, planted seedlings may present good initial growth, whether they find nutrients, organic matter, and free soil in the pits, but this growth can be slowed down as the root system depletes the resources of the planting pits and starts to explore the compacted or very poor soils around them. The final result ends up being the formation of an environment with small isolated trees, type bonsais, with soil exposed between them and without understory formation. Obviously, a vegetation with such characteristics tends to perish.

To enhance forest restoration in these highly degraded environments, the transposition of soil/litter and vegetal residues such as antlers and bark is an ecological and low-cost alternative. The surface soil, or topsoil, and the litter layer that covers it, when taken from native forests, contain, in addition to nutrients and organic matter, a rich seed bank, formed not only by pioneer tree species but also by herbaceous, shrub, epiphyte species. Therefore, the top soil/litter set is a rich and diversified component of the forests, and its transposition in nuclei or islands to degraded areas produces the necessary stimulus for the triggering of natural regeneration [29–31, 33–35]. However, it should be remembered that topsoil and native forest litter must be removed from areas where environmental licensing has authorized the suppression of vegetation for mining activities, impoundment of watercourses, among others.

In this sense, a study carried out by [42] in a secondary forest on the campus of the Federal University of Viçosa, in Viçosa, State of Minas Gerais, showed that the removal of layers of soil only 5 m in depth and litter, in ranges of 1 × 2 to 1 × 8 m, spaced at about 5 m, did not have a significant impact on forest regeneration, with retreatment sites naturally recovered after 1 year. Despite the need for new tests to evaluate the impact of topsoil and litter removal in other forests, these first results indicate the possibility of applying this technique from forests in legal reserve areas, through a project to be analyzed by the qualified environmental agency.

The plant residues that have been most used in forest restoration in degraded soils are the semi-composite bark of *Eucalyptus*, a very abundant material in pulp and charcoal production companies, and the branches resulting from legal suppression of vegetation or tree pruning. These residues provide nutrients and organic matter, which are essential for enhancing the chemical and physical properties of degraded soils, as well as to provide shelter and food source for insects, rodents, and reptiles, stimulating food chains [3, 30].

Although the application of these nucleation techniques is more common in areas with soil degradation, nothing prevents soil/litter nuclei and residues from being used also in areas with traditional reforestation. As with the installations of perches, the deposition of residues and top soil in the narrow bands or islands within the reforestation is a way to increase biodiversity and stimulate ecological processes, as well as to enable a reduction in the number of seedlings planted.

5.3. Direct seeding

One of the main factors that prevent or hinder the progress of the natural regeneration process in highly anthropized landscapes is the lack of seed rain with species diversity. Some ruderal and pioneer species with wide anemochory dispersion can reach in some situations great distances and colonize certain areas forming monodominant vegetation, since other species with more restricted zoocoric dispersion do not achieve the same success.

As the regeneration process does not advance in these areas due to the distance from the seed sources and/or the low supply of dispersers, the artificial dispersion through direct seeding becomes a promising alternative.

The seeding direct was recommended by [43] as technical for restoration of degraded Norway spruce forest in the Forstamt Weissenhorn, Bavarian region of southern Germany. The authors appoint that the direct seeding seemed a promising and cost-effective alternative to planting.

Despite being an alternative technique of restoration that shows good results, it also presents some limitations so it should not be recommended for every situation [37, 44–46]. In very degraded soils, such as mined areas and old pastures with exposed soils, the simple sowing of native species may not be sufficient to trigger the regeneration process. The previous prepare of soil by scarification in the case of compacted soils and the provision of a source of nutrients and organic matter is often necessary.

Also, one cannot expect much from direct seeding in brachiaria (*Urochloa decumbens* Stapf) pastures, where the main factor of inhibition of regeneration tends to be precisely the aggressiveness of the exotic grass and its previous control through herbicide application and cutting, among others, becomes necessary.

However, direct seeding has great potential for use in the enrichment of swiddens and pastures, where a regeneration process has already begun but at a very slow pace and with very low diversity. Thus, through sowing, species that have regional occurrence can be introduced into areas in regeneration, but cannot reach via dispersion.

Figure 6. Forest restoration of mined areas with random planting of seedlings and direct sowing of *Cajanus cajan* (L.) Millsp. Forest restoration project LARF-UFV. Pictures: Sebastião Venâncio Martins.

A detailed study on the viability of direct seeding as a forest restoration technique is found in [46]. In it, the authors discuss very satisfactory results of the application of a new mechanized direct seeding methodology adopted in a large scale in the state of Mato Grosso, Brazil, in which seeding agricultural machines were adapted for seed sowing of native species. In addition, they present a review with excellent results of experiments of direct seeding of different species in the Brazilian biomes.

An option to reduce the costs and the use of agrochemicals in restoration areas is the direct seeding of green manures described by [47] and implemented in restoration projects coordinated by the Forest Restoration Laboratory of the Federal University of Viçosa (www.larf.ufv.br). In this technique, seeds of herbaceous and shrub species of the family Fabaceae (formerly Leguminosae), with symbiotic association with *Rhizobium* and fixation of atmospheric nitrogen, present rapid growth in poor and degraded soils and, therefore, can be sown together with seeds of arboreal native species or together with the seedlings planted in larger spaces or in nuclei. Several species of green manure have been used in direct seeding to cover the soil fast and reduce competition with grasses such as brachiaria, and it is worth mentioning *Cajanus cajan* (L.) Millsp. that in addition to enhancing soil, it rapidly leaves the system and releases space for growing seedlings (**Figure 6**).

6. Final considerations

The progress of forest restoration in the world is undeniable, which has gained strength in recent years from the 21st Session of the Conference of the Parties to the United Nations Framework Convention on Climate Change (COP21) and the recent water crisis that has increased the perception of society about the importance of preservation and restoration of riparian forests.

In this scenario, all forest restoration initiatives already carried out or under way are very important, although some have not achieved the expected results and others have presented very high costs. All over the world, there had been a great demand for studies and forest restoration services, mainly by mining, power generation, and pulp production companies, which are being increasingly encouraged to restore their Permanent Preservation Areas and Legal Reserves, not only in compliance with environmental legislation but also to certify their activities and export their products.

As it was pointed out in this text, there are innumerable possibilities and alternatives for forest restoration, with remarkable climatic, edaphic, cultural, and socioeconomic diversity between the countries and the different types of impacts to which its ecosystems are subjected. Therefore, there is no ready-made recipe, a single restoration model or technique that can be applied widely and on a large scale. The important thing is to take advantage of the remaining potential for ecosystem regeneration by adapting more appropriate techniques for each situation.

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Section 3

Forest Resources and Biodiversity

Review, Rethink, and Reconsider Human-Nature Interaction in the Urban Context

Foo Chee Hung

Additional information is available at the end of the chapter

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Abstract

Human and nature are intimately linked with each other. However, such relationship is changing as a result of the drastic changes brought by rapid urbanization in the last few decades. This leads to the necessity in redefining and re-exploring the human-nature interaction, particularly in the urban context. By taking the lowland dipterocarp forest in a highly urbanized region in Malaysia—Klang Valley, as the topic of discussion, this chapter examines and discusses how human-nature interaction can vary with regard to different degree of naturalness. Having an understanding on such relationship is important as it can help in formulating a more compelling conservation goal that keeps both human and nature intact while at the same time appealing to and accepted by a wider range of audience.

Keywords: forest, naturalness, human-nature interaction, urban, quality of life

1. Introduction

Ecosystem is intimately linked to human well-being throughout human history [3]. Human depend on the services provided by ecosystem for their sustenance, livelihoods, and survival. Evidence shows that more than 1.35 billion people, in which many of them living in the indigenous communities or inhabit some of the biologically richest and most threatened areas on Earth, are depending on continued access to basic necessities provided by those ecosystems such as clean air and water, food, fuel, clothing, medicine, and shelter for fulfilling their physical, economic, and spiritual well-being [6]. Nevertheless, there is a sign of disengagement of humans from the natural environment in the last few hundred years, due to the enormous shift of people away from rural areas into cities [9]. According to the United Nations Population Division [19], more than 48% of people worldwide live in urban areas

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currently, and it is projected to exceed 60% by 2030. Some developed countries have exceeded this figure. For example, in the USA, half of the population live in suburbs and an additional 30% live in urban centers (U.S. [21]); while in the UK, about 90% of the citizens live in cities [20]. A rapidly growing country like Malaysia is projected to reach 70% of the total population that live in the urban area by the year 2020 [7].

As this shift took place, human beings started to redefine their relationship with the natural world, where too often the nature is considered as expendable and the ecological processes that sustain us are hidden from view [11]. In a highly urbanized region, for example, the ways that urban dwellers have contact with the natural environment are through viewing the nature, spending leisure-time in the nature, or involving themselves in environmental conservation issues (**Figure 1**).

Their daily life, in terms of economic and social-cultural, is more likely influenced by the built environment than nature. Besides, in contrast to the rural or indigenous communities that depend largely on the provisioning services (food, jobs, raw materials, shelter, etc.) supplied by the natural environment, urban dwellers have much freedom of choice in finding substitutes for those services. Moreover, there are trade-offs between ecosystem services, in which different priorities are given to different ecosystem services in order to maximize the well-being of the urban dwellers. Therefore, disconnection may probably occur between certain ecosystem services and urban dwellers' well-being, and it is not surprising to find out whether the natural environment is not being considered as a main contributor to the urban dwellers' well-being.

The direct consequence of such disconnection is the intensified estrangement from the nature [11] and the gradual loss of people's appreciation to the nature regarding their reliance on the

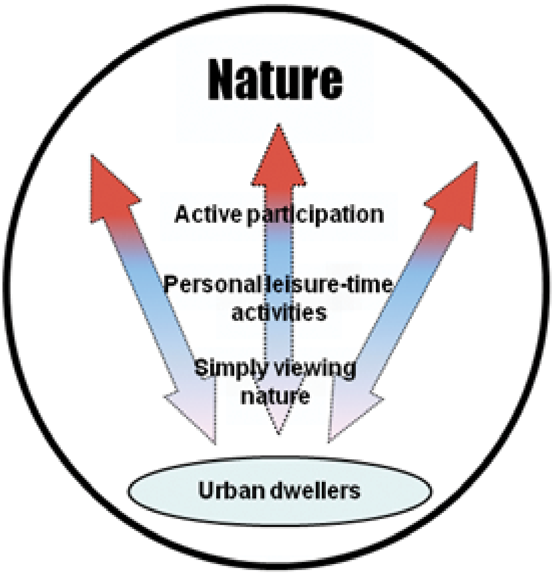


Figure 1. Ways of contact with nature from an urban community's perspective. Source: Adapted from [17].

natural environment [15]. This phenomenon was described by Pyle [14] as the "extinction of experience" syndrome, which is the formation of apathy towards environmental concerns as a result of people's alienation from the nature due to the homogenization and reduction of local flora and fauna in their living place, following the quickening pace of artificial surface replacement and the permanent loss of a bunch of valuable natural habitat. However, as a species that has spent most of its existence hunting, gathering, and/or growing food, it is doubtful that human have been able psychologically to remove themselves from the nature [12].

It is believed that there is universal inborn human desire to re-establish the tenuous link with nature, which is referred to as biophilia by Wilson [23], a term to describe both negative and positive responses to elements of the nature that had survival value to our evolutionary predecessors. Although recent human history is characterized as a gradual shift from living in rural to living in cities; whenever there is a chance, most people would prefer to live close to the nature (i.e. waterfront, green park) because through human evolutionary past, the nature had survival value for obtaining food and water. Moreover, with the increasing knowledge and information about natural science and environmental issues, particularly the adverse impacts caused by human technology to the living system, people become more aware of their reliance on the biosphere. Consequently, this leads to the shift in values and perceptions of human-nature relationship. As the values shift to a more holistic perception of people's relationship to nature, so will the people's expectations shift to a greater desire to complement their urban lifestyle with elements of nature [12].

In this chapter, human-nature interaction in the urban context is explored by taking the lowland dipterocarp forest in a highly urbanized region in Malaysia—Klang Valley, as a topic of discussion. The chronology of the lowland forest degradation in the Klang Valley is reviewed first. Then, the challenges with regard to forest conservation from the national, state, and local level are discussed. Finally, the chapter is devoted to understand how different natural attributes of the urban nature may have influences on the human-nature interaction, thereby affecting people's responses and expectations they placed on these remnants. The data used in the discussion include both primary and secondary sources, ranging from local community questionnaire survey to regional demographic, real estate, and land use statistics.

2. Lowland forest ecosystem and regional development in Malaysia

The lowland forest, or more precisely, the lowland dipterocarp forest (since it is mostly dominated by trees from the *Dipterocarpaceae* family) is one of the most complex, dense, and species-rich forests. These forests form a green girdle right around the Earth's equator, representing the epitome of plant development below 300 m above sea level (**Figure 2**). On one hand, it has great value for wildlife conservation and scientific research; on the other, it is the type of forest that is under enormous threat due to its value for commercial timber extraction.

The contribution of lowland forest ecosystem to the regional development was mostly through providing land and forest resources (mainly timber). According to Vincent and Hadi [22] on the relationship between deforestation and agricultural expansion in Peninsular Malaysia, the

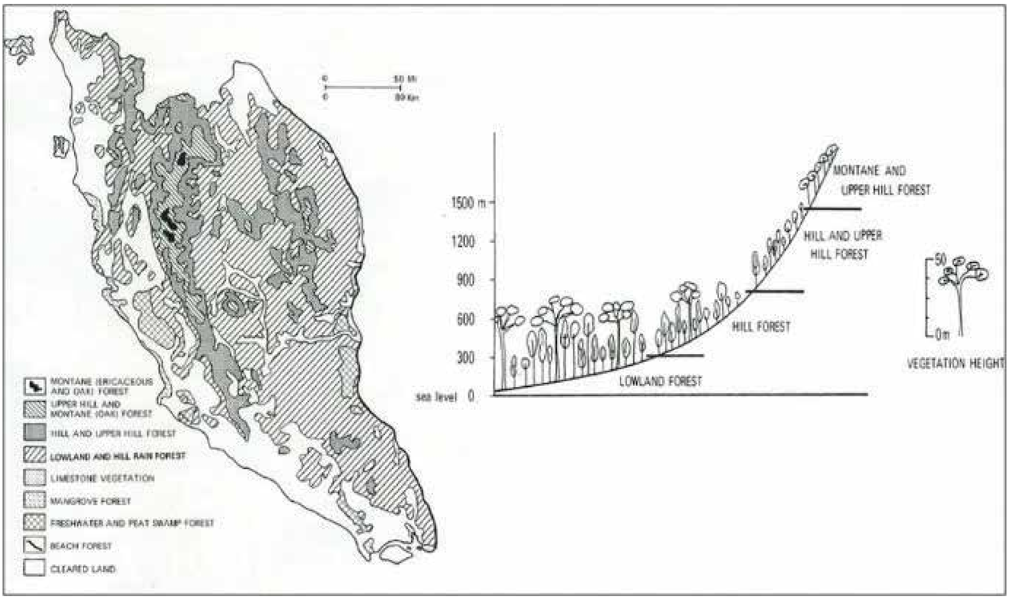


Figure 2. Varying forest structure with elevation in Peninsular Malaysia. Source: Adapted from [1].

agricultural expansion in the peninsula has gone through three distinct phases. The first phase (1904–1932) depicted the boom in rubber planting, where an annual average of 49,000 ha was opened up during this period for rubber estate. The second phase, between 1932 and 1966, saw the agricultural expansion slow down to an annual average of 24,000 ha, whereas the third phase (1966–1984) illustrated a renewed rapid agricultural expansion, at an average of some 57,000 ha a year, where the extensive tracts of natural forests have been cleared especially for the oil palm planting, which increased to some 1.12 million ha during the same period. As a whole, a total of 1.36 million ha of forests were lost to other uses from 1966 to 1984.

The conversion of forest into agricultural land use was intensified with the existence of the Federal Land Development Authority (FELDA). Established in 1956, FELDA has been the principal land development agency in the peninsula, where it has played a key role in transforming virgin forest land into plantations and settling landless families. By providing a package of physical, social, and technical infrastructure aimed at bringing the technological strengths of the plantation sector to smallholder agriculture, FELDA has successfully addressed rural poverty in the peninsula. Since its establishment, FELDA has developed over a million hectares of agricultural land, with over 70% planted with oil palm. FELDA's strategy was to develop forested land into viable plantations to settle landless families. Clearly, FELDA's impact on land use has been considerable, and the figures are somewhat impressive.

Apart from providing land resource for further development, natural forest was also an important source of log input for the domestic primary timber processing industry. The contribution of timber industry to the nation's economy had gained importance since decades

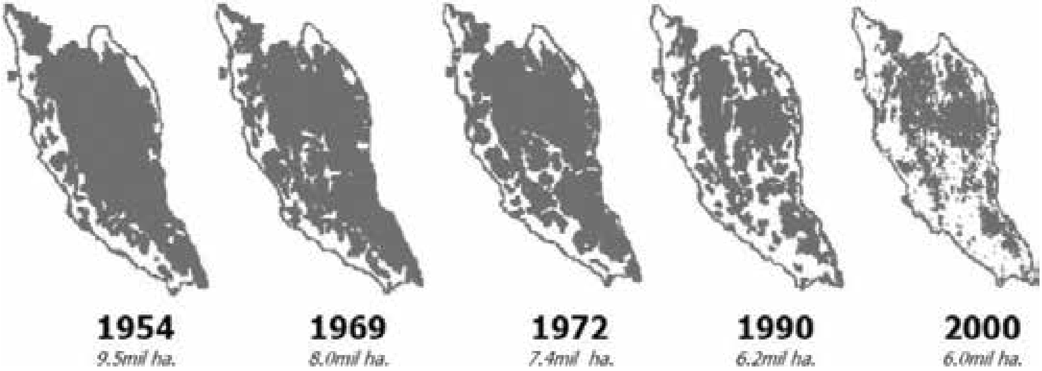


Figure 3. Fragmentation of natural forest cover in Peninsular Malaysia, 1954–2000. Source: National Physical Plan 2, Federal Department of Town and Country Planning.

ago. For example, in 1985, the export earnings from the timber sector were US$ 1.9 billion and further increased to US$ 5.4 billion in 1994, which accounted for about 9% of the nation's export earnings. In 2005, the total export of timber and timber products was valued at RM 21.51 billion or 4% of the country's total export earnings of RM 533.79 billion. In fact, in the same year, the forestry sector in Malaysia accounted for about 8.2% of the country's GDP of RM 262.17 billion and provided direct employment to 270,670 persons, representing 2.6% of the country's workforce [18].

As like other developing countries, the degradation of forest ecosystem in Peninsular Malaysia indicates a typical example of the excessive demand for ecosystem services stemming from economic growth, demographic changes, and individual choices. In another word, it is the trade-off being made from one service (provision of natural resources) to another service (ecological function). Under the impact of modernization, large area of forests has been whittled away in order to provide food (as forests turned into agricultural land), raw materials (such as timber), and land for the rapidly urbanizing population (**Figure 3**). Such fast disappearing of forest cover in Peninsular Malaysia has once worried Marshall [10], who expressed his concern by stating that "Nothing may remain of the lowland-dipterocarp forests by 1990 if deforestation continues in such ever-increasing rate." To date, the densely populated areas in Peninsular Malaysia today have less natural forest. The so-called undisturbed forests can only be found on the mountain or hilly areas, while most of the remaining lowland forests are either disturbed or secondary forests [13].

3. Development pressure and degradation of Forest ecosystem

Being one of the highly urbanized regions in Peninsular Malaysia, Klang Valley (**Figure 4**) has been undergoing dramatic transition in both demography and economics. One of the most outstanding features of urbanization in Klang Valley is the dramatic growth and dominance of the Kuala Lumpur city, and the strengthening of its influence within the settlement system of the Peninsular Malaysia. As the country's capital, Kuala Lumpur first emerged in the late 1850s as a small trading port serving the tin mines in the Klang River Basin. With only a population about 2000 in 1870s, Kuala Lumpur further grew into a city as a result of the mining boom in the river basin and the successful development of rubber plantation in its surrounding areas. Continuing economic growth and modernization have made the city becoming more vital of its role as a political, administrative, and cultural significance in the country; and eventually, in 1974, Kuala Lumpur was proclaimed as a Federal Territory, together with the extension of its boundary from the original 93 to 243 km^2.

Klang Valley is formed as a result of spill over development of Kuala Lumpur to its surrounding areas. Originally, "Klang Valley" refers to the conurbation that embraces the Federal Territory of Kuala Lumpur, the dormitory-industrial city of Petaling Jaya, the Selangor state capital—Shah Alam, and the Klang city where Port Klang is located. But, throughout years of urbanization and the emergence of other new towns adjacent to the Kuala Lumpur-Port Klang development axis, the term "Klang Valley" eventually represents the combination of Kuala Lumpur and the other four surrounding districts of Selangor state – Petaling, Klang, Gombak, and Hulu Langat, with a total area of 424 km^2 and a population of nearly 6 million in 2010. Extensive land use change associated with the increasing urbanity has systematically seized the limited green area. Much of the original forest cover has been replaced by urban land use, and development has even encroached into the foothills [24]. Nearly 60% of the land in the region has undergone at least one change in land use from its original natural forest state as early as 1986, in which the major conversion has gone to the commercial plantation, followed by built-up area and mining.

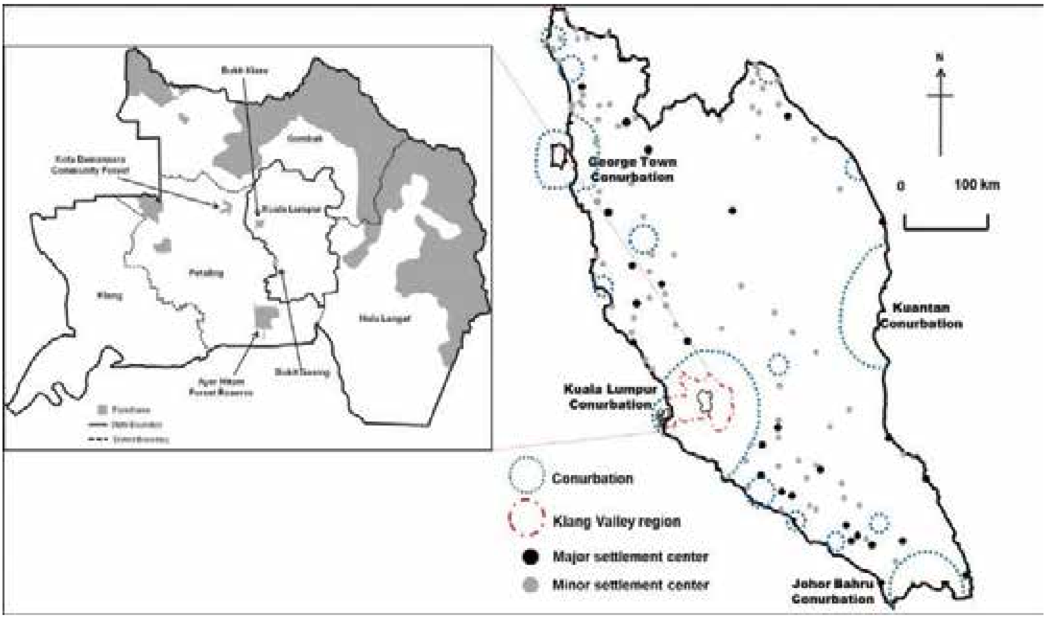


Figure 4. Location of Klang Valley and other conurbations in peninsular Malaysia.

The lowland forest in Klang Valley region was fragmented and further downsizing, and eventually sandwiched by the residential and commercial area. Only a few sites of small parcel lowland forest left in this region (<2% of the total land area), and all of them are located within the highest population density part of the region - Petaling district, facing the threat from housing development in their surrounding areas. For example, the Kota Damansara Community Forest (gazetted as forest reserve in 1898) located in the midst of Petaling Jaya, a satellite city of Kuala Lumpur, once sprawling over 6590 ha, has shrunk steadily over the years, where now only 321.7 ha is left. Parcel by parcel of forest land had been first converted into rubber plantations, and then into oil palm estates. While in the last 15 years, those parcels are mainly for the development of residential houses. Another typical example is the Ayer Hitam Forest Reserve (gazetted as forest reserve in 1906) located in Subang Jaya—a residential town in the Petaling district, where hectares of land from the forest reserve has been gradually converted into different uses, from originally 4270.7 ha in 1906–1176 ha in the present.

4. Challenges for lowland forest conservation in Klang Valley

The conservation value of the lowland forest is undeniable, and perhaps, invaluable to some extent. There have been a number of studies on the uniqueness of lowland forest in Peninsular Malaysia, such as those conducted by Forest Research Institute Malaysia (FRIM) at Pasoh Forest Reserve and at Ayer Hitam Forest Reserve by University Putra Malaysia. However, knowing the value of the forest is not enough as the intensity of the value is subject to change with different scales.

From the national point of view, 55.2% of Malaysia is still covered by forest in 2008, where 44.4, 57.4, and 65.5% of the Peninsular Malaysia, Sabah, and Sarawak are forested, respectively. These figures are arguable as they only give a general view of forest hectarage, regardless of the types of forest found in Malaysia such as lowland dipterocarp, hill dipterocarp, upper hill dipterocarp, oak-laurel, montane ericaceous, peat swamp, and mangrove forest; which lead to the conclusion that the country is still "green." Furthermore, Malaysia has been recognized as one of the 12 megadiversity countries in the world by the 2001 Global Diversity Outlook as the country contains at least 60% of the world's known species. States that have been listed as the main areas for endemism are from the East Malaysia (Sabah and Sarawak), which accounted for 67.9% of the total forested area in the country. Meanwhile, the Selangor state, where Klang Valley region is located, only contributes 1% of the total forested area (**Figure 5**). Thus, logically, more conservation efforts will be given to maintaining forest resources from those major contributors, particularly the two states from the East Malaysia that are playing the main role as the "reservoir" of the country's species diversity, uniqueness, and abundance; while trade-offs is made with regard to small-scale remnant forests situated in urban area.

Apart from that, the existing forest policy and legislation are also conducive to lowland forest conversion. Under the Article 74 (2) of the Federal Constitution and Section 11 of the National

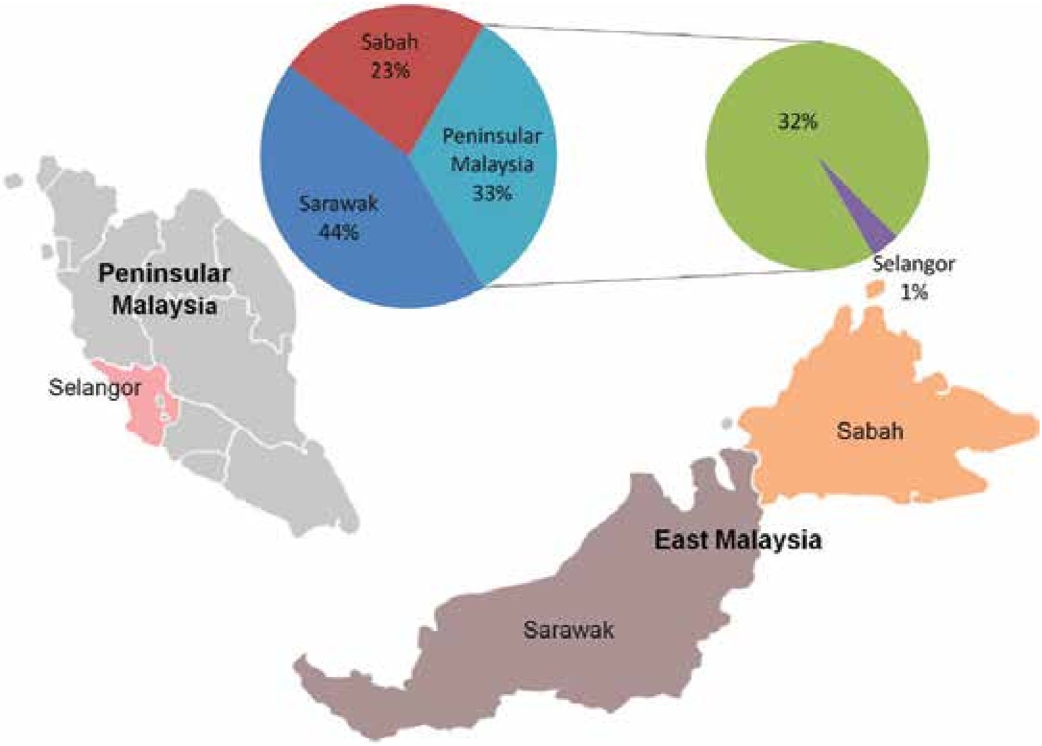


Figure 5. Share of forested area by major states in Malaysia.

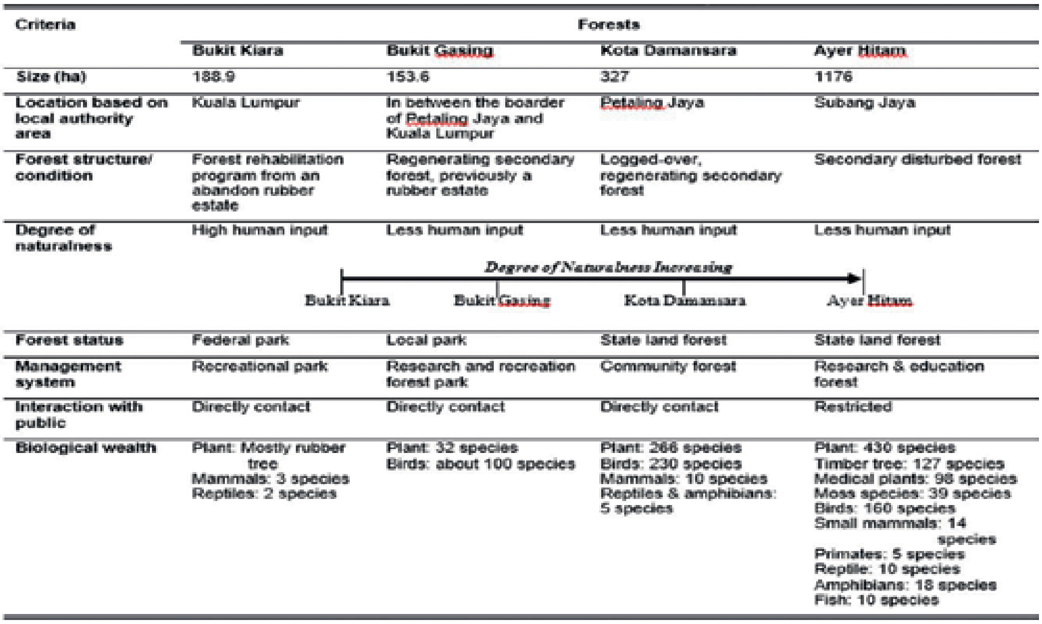
Forestry Act (1993), the individual states are given exclusive jurisdiction over their respective forest land, in which reserved forests are considered as an alternative land use, and the individual state authority if satisfied that any land in a permanent reserved forest is no longer required for the purpose as classified under Section 10 of the National Forestry Act (1993) (such as timber production forest, soil protection forest, soil reclamation forest, flood control forest, water catchment forest, forest sanctuary for wildlife, virgin jungle forest, amenity forest, education forest research forest, and forest for federal purposes), and if the requirement for economic use of the reserved forest is higher than that it is being utilized, the state authority may excise that forest land from the permanent reserved forest. Since there are difficulties from the practical point of view in conserving small-sized remnant forests, coupled with the fact that conversion of forested land into other land uses, especially to housing development, is more promising in terms of higher rate of return on investment, these forested lands are always preferred to be whittled away for further development.

5. Human-nature interaction in Klang Valley

Despite its relatively lesser biodiversity as compared to those big-scale rural reserves, small-scale remnant forest in urban area is capable of providing a wide range of environmental benefits that contribute to the liveability of modern city and the well-being of the urban dwellers, which are used to be overlooked in the past. Following the improved life standard, educational level, and the stressful of the urban environment, people's awareness, understanding, as well as appreciation to these urban natural environments, particularly those in their neighborhood, are becoming higher.

This is evidenced through a study conducted by the author in the surrounding residential areas of four remnant forests in Klang Valley region: (i) Bukit Kiara Forest Park (BKFP), (ii) Bukit Gasing Forest (BGF), (iii) Kota Damansara Community Forest (KDCF), and (iv) Ayer Hitam Forest (AHF) (**Table 1**), with regard to the connection and attachment between urban dwellers and remnant lowland forests in Klang Valley.

These forests are selected as study site because they are inner urban forests, surrounded by the residential/commercial areas, and subject to different local development pressure from the vicinity. Besides, the present forest condition and forest status also affects the selection of the study sites as well. Since the author wishes to examine the extent to which different forest naturalness may affect people's perception on the forest, each selected site was ensured to be unique and representative to its own structure/condition and management approach. For example, BKF is a botanical garden originated from the forest rehabilitation program from an abandon rubber estate; while AHF is a secondary disturbed forest, contributing to logged-and-treated activity between 1930 and 1965, which is then gazetted as a forest reserve and



Criteria	Forests			
	Bukit Kiara	Bukit Gasing	Kota Damansara	Ayer Hitam
Size (ha)	188.9	153.6	327	1176
Location based on local authority area	Kuala Lumpur	In between the boarder of Petaling Jaya and Kuala Lumpur	Petaling Jaya	Subang Jaya
Forest structure/ condition	Forest rehabilitation program from an abandon rubber estate	Regenerating secondary forest, previously a rubber estate	Logged-over, regenerating secondary forest	Secondary disturbed forest
Degree of naturalness	High human input	Less human input	Less human input	Less human input
Forest status	Federal park	Local park	State land forest	State land forest
Management system	Recreational park	Research and recreation forest park	Community forest	Research & education forest
Interaction with public	Directly contact	Directly contact	Directly contact	Restricted
Biological wealth	Plant: Mostly rubber tree Mammals: 3 species Reptiles: 2 species	Plant: 32 species Birds: about 100 species	Plant: 266 species Birds: 230 species Mammals: 10 species Reptiles & amphibians: 5 species	Plant: 430 species Timber tree: 127 species Medical plants: 98 species Moss species: 39 species Birds: 160 species Small mammals: 14 species Primates: 5 species Reptile: 10 species Amphibians: 18 species Fish: 10 species

Source: A compilation by the author from various unpublished local environmental report.

Table 1. Characteristics of the four selected study areas.

is leased to a university for the purposes of education, research, and extension. These two forests represent the extreme of man-made and natural continuum along the degree of naturalness. Both KDCF and BGF are placed in between such continuum, showing their tendency towards more "man-made" and "natural," respectively.

Through questionnaire survey conducted among residents living in the surrounding area of the forests, the author found that size and the degree of naturalness of a forest does have influence on the way human interact with the nature. As high as 43% of the survey respondents living in the surrounding area of AHF did not realize the existence of AHF, as compared to 8, 10, and 13% of the respondents from BKFP, BGF, and KDF, respectively (**Figure 6**).

To note, the awareness of the forest existence among residents decreased with the increasing size of the forest as well as the higher degree of naturalness of the forest. When the respondents were asked whether the forests are in anyway contributing to their quality of life, similar trends of result were observed, in which 92% of the respondents living in the surrounding area of AHF perceived that their quality of life is not influenced by the forest, as compared to 8, 11, and 30% of the respondents from BKFP, BGF, and KDF, respectively (**Figure 7**).

These findings indicated that people tended not to consider the forest as a contributor to their life quality without direct contact with the forest. This is the case of AHF, where being gazetted as a research and educational forest, it is restricted to the public accessibility. While for other forests where people can have direct contact with, the tendency to take into consideration the forest as a determinant of their quality of life is increasing. This is because by having a direct contact with these forests, in fact, enables residents in the surrounding areas to develop a higher attachment to these remnants, thereby becoming more aware of the benefits provided by these remnants to their quality of life. However, due to the increase of human intrusion, the degree of naturalness of these forests was decreasing as well as their richness of biodiversity.

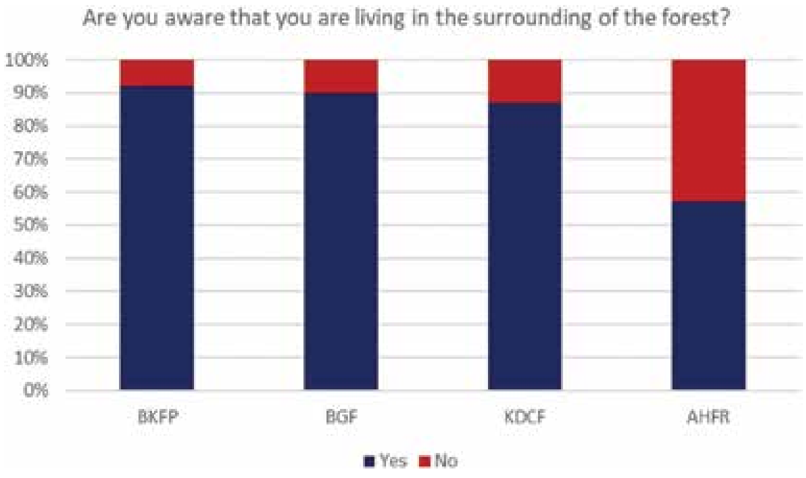


Figure 6. Awareness of the forest existence.

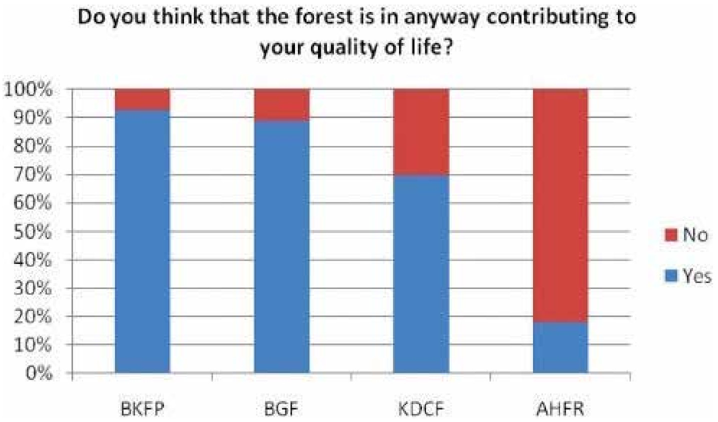


Figure 7. Perception on forest in benefiting quality of life.

The attachment of urban dwellers with urban natures can also be explored from the concept of "sense of place" as suggested by Shamai [16]. A sense of place is a piece of the whole environment which has been claimed by feelings. It is an umbrella concept that includes all the other concepts such as attachment to place, national identity, and regional awareness. Different levels of sense of place that imply an increasing intensity of feeling about a particular place at a given time can be measured and ranked on an ordinal scale [8]. According to Shamai [16], having a sense of place consists of three phases, where the first phase is belonging to a place, the middle phase is attachment to a place, and the highest phase is commitment to a place.

The author does not intend to explore in detail the concept of sense of place in the present study, but to use the "sense of place" concept as a guidance in assessing the level of "willingness to contribute" of the survey respondents towards the issue of conserving a small-scale urban nature. This is because the author opined that both conservation and attachment to a place share a lot of similarities. First, both of them required the commitment to a place, and this commitment can be depicted in a continuum, ranging from not having any feeling to a deep commitment towards a place. Second, both of them are dealing with the relationship of human to a place. While sense of place is a measure of how deep a person is attached to a place, it is also feasible to be used in measuring to what extent a person is willing to contribute in conserving a place. It is assumed that the higher or deeper a person is attached to a place, the more he/she is willing to contribute in the place conservation. Third, the essence of sense of place lies in the beholder's senses and mind. This is also true for conservation, where it involves a person's feelings, attitudes, and behavior towards a place which varies from person to person. As like Aiken and Leigh [2] said, "From the perspective of experience, conservation is a point of view, a state of mind, perhaps an emotive call to action; it implies a particular view of land and life."

A scale of five level of willingness to contribute was used and it is shown in **Figure 8**. Respondents were asked to rate five statements which representing different level of willingness to contribute with a scale from 1 to 5, where 1 is 'strongly disagree' while 5 is 'strongly agree'. Each statement

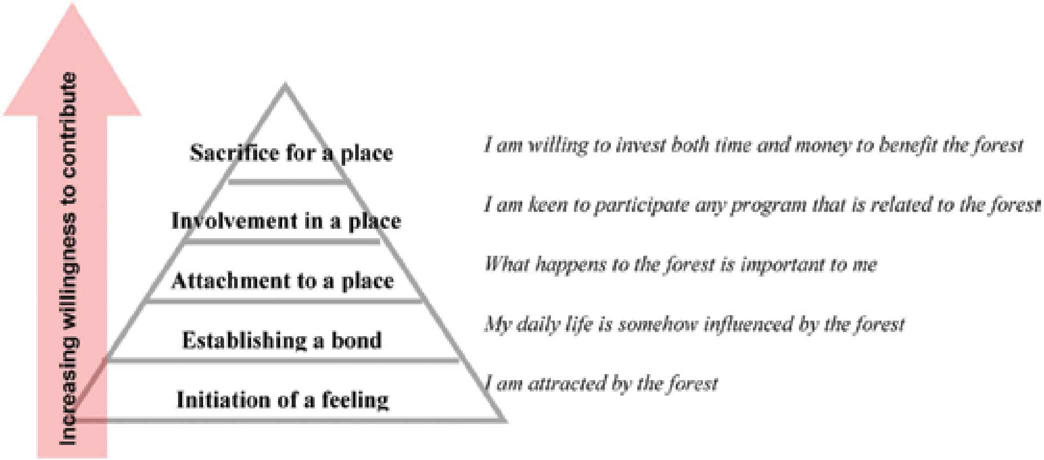


Figure 8. Level of willingness to contribute.

was assigned different weight that is compatible to its level of willingness to contribute. The rating given by the respondents for each statement was then multiplied with the assigned weighting and sum up to obtain an aggregate score of level of willingness to contribute. These aggregate scores were then divided into four groups; thus, four segments of respondents were obtained representing four levels of willingness to contribute: (i) high (>60), (ii) mid-high (46–60), (iii) mid-low (34–45), and low (<31).

As shown in **Figure 9**, the willingness to contribute was increasing along the natural and man-made gradient, where the highest level of willingness to contribute (with a score > 60) was leveling up, while the lowest level of willingness to contribute (with a score ≤ 30) was decreasing, as the forest becoming more and more "man-made" or "human-control." This is reasonable because willingness to contribute tends to increase with the increase of a person's attachment to a place. Through direct experience and engagement with the activities related to the forest, such as shaping the forest landscape and giving suggestion on forest management, a person will likely to imbue the forest with meaning. This meaning, in turn, may become a motivation for him/her to further contribute to benefit the forest which was deemed to be part of his/her life. Typical examples can be observed in both BGF and KDCF, where through direct engage in forest landscape design (either by gardening or by providing feedback to the management authority), the people's attachment to the forest becomes stronger and have higher desire to voluntarily contribute their effort, time, or even money to conserve the forest.

The main point to be highlighted here is that naturalness does play an important role in influencing people's responses and attachment they placed on the forest. The author successfully explores another aspect on how human interact with nature in the urban context, in addition to studies conducted by Foo [5], Foo and Kidokoro [6] and Farahwaheeda et al. [4]. Understanding people's views and expectations on forests is the first and foremost step

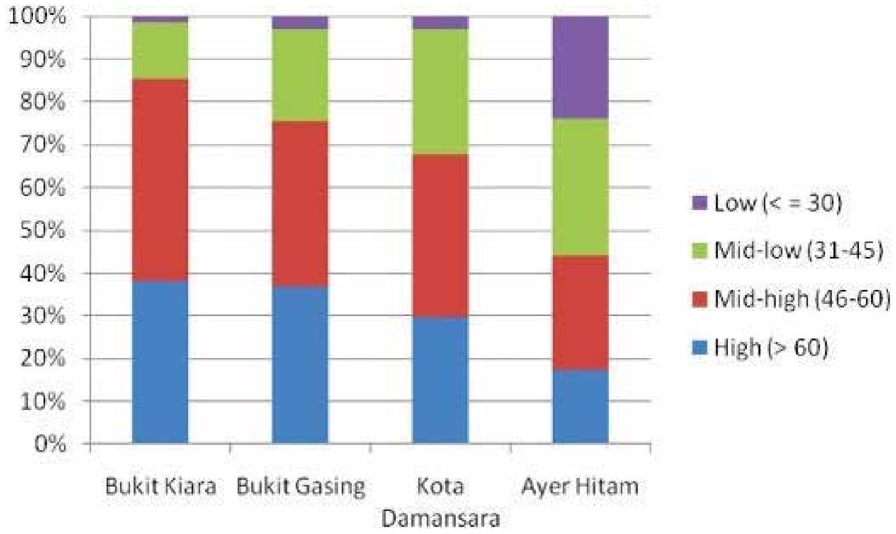


Figure 9. Distribution of attachment level on each study site.

in ensuring the success of forest conservation. Even though the scientific understandings on forestry have achieved a great improvement throughout these decades, the importance of people's responses to what forests are and how they should be managed is still playing an undeniable important role, particularly in conserving the declining (in size) and degrading (in biodiversity) urban natures.

With the increased environmental awareness and the realization of the adverse impacts posed by urbanization on them, urban dwellers become more appreciated with these remnants than before. Improving quality of life is not merely a matter of physical development. According to the World Health Organization (WHO, 1993), quality of life is defined as "an individual's perception of their position in life in the context of the culture and value systems in which they live and in relation to their goals, expectations, standards, and concerns." If this is the case, the authorities must consider that people highly regard urban natures as a contributor to their quality of life, when making any decision upon converting these remnant forests into other uses.

6. Conclusion

Lowland forests in Malaysia have been fragmented and greatly reduced by conversion to plantations and urbanization. Being located in the country's highly urbanized region, the survival of the lowland forest is often subject to profound anthropogenic stresses and invasions. The understanding of the interaction between human and the natural environment can help in formulating a more compelling conservation goal that keeps both human and nature remain intact while at the same time appealing to and accepted by a wider range of audience. To date, so far, efforts to natural environment conservation are mainly focused on large areas with less human presence, bio-diverse, and relatively untouched. As a result, much more challenges are expected to be encountered in conserving a natural environment within a highly urbanized area where lands have been a scarce resource for absorbing further development pressure. Besides, once the land was found to have economic value; pressure would promptly mount to have it excised from the protected status so that development could proceed. As the aim of development is to improve and enhance human well-being through land conversion, unless land conservation has a relatively stronger reason to convince the decision makers, otherwise decision is always made in favor to the former.

Therefore, the author suggests that an effective environmental conservation program must not only be based on the assessment of conservation value that most often derived from the ecological point of view, such as the number of threatened species, endemic plant species, etc. Instead, the understanding of the changes happened in the surrounding areas of the preserved site, such as land use and demographics change, economic and social activities, people's perception and attitudes towards the role and function of the preserved site, etc., is also necessary, in order for proper policy implication to be drawn and to be further used for strengthening the existing conservation value.

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Advances in the Assessment of Climate Change Impact on the Forest Landscape

Melih Öztürk, Şahin Palta and Ercan Gökyer

Additional information is available at the end of the chapter

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Abstract

Changing climates threaten the habitats and ecosystems at variable extents throughout the world. Forests are unique habitats and ecosystems that are vulnerable by the consequences of climate change. The climate change causes disturbances, alterations. and shifting on the forests that can be diagnosed at the tree and stand scales, as well as can be monitored and analyzed at the landscape scale. Furthermore, some recent researches concentrate on conveying the forest tree and stand-level shifting and disturbances to the forest landscape level by upscaling. In this study, the climate change impacts on the forest landscapes; principally, the disturbances including the drought-induced mortality, growth and productivity failures, and insect outbreaks are evaluated. Secondarily, climate change-induced alterations of the forest species distributions and forest landscape compositions, dynamics of the forest biodiversity, and tree migrations are discussed by focusing particularly on the relatively recent advances involving the modeling procedures. Ultimately, monitoring the climate change-driven shifting phenology of the forest landscape through the remote sensing techniques is referred in this study. Moreover, the study examples dependent upon the climate-ecological modeling and satellite data assessment of the forest landscapes throughout the world are also referenced. The landscape-scale assessment of the climate change impacts on the forest ecosystems provides integrated and comprehensive approach toward the proposal of sustainable mitigations and solutions to the phenomenon.

Keywords: landscape ecology, ecological models, leaf area index (LAI), biodiversity, tree migration, remote sensing techniques

1. Introduction

Earth's climate had experienced natural alterations due to changes in the solar radiation and atmospheric, oceanic, and terrestrial forces, drivers, and components over the past centuries [1].

However, the anthropogenic effects have triggered the recent climate change since particularly the twentieth century. The emissions of the greenhouse gases, primarily the carbon dioxide that exceeded their average concentrations in the atmosphere, have led to the gradual warming of the earth's surface. Therefore, the biogeochemical processes have been influenced by this change in the climate. The biogeochemical processes have in turn affected the climate contributing its change [2]. Hence, terrestrial ecosystems play an important role in the climate change processes as these ecosystems are significantly vulnerable from the consequences of climate change. Among those terrestrial ecosystems, forest landscapes constitute the major part that is both influencing the climate change and being influenced by the consequences of climate change [3].

Forest landscape consists of the total forest ecosystem within the field of vision that is either comprehended by the naked eyes and/or can be observed through technological instruments and remote sensing techniques. Autumn and winter sceneries from the forest landscapes in Western Black Sea Region of Turkey can be seen in **Figure 1**. The interactions between the forest landscapes and climate are rather intensive and complex principally by means of biogeochemical and hydrological cycles. Basically, climate is effective in the process of exchanging minerals among the plant, soil, and water due to their interactions. Indeed, the forest landscapes are unique ecosystems that are also vulnerable to the climate-induced natural and anthropogenic disturbances. For instance, in particular for the energy-limited environments, increasing temperatures stimulate the photosynthetic and metabolic activities of the forest trees urging their primary productivity and leading them to deposit more carbon that is sequestrated from their environment [4].

Figure 1. Autumn and winter sceneries from forest landscapes in Western Black Sea Region of Turkey (Photos by Melih Öztürk).

This situation makes the consequences of the forest fires more devastating, primarily from the point of releasing huge amounts of carbon dioxide to the atmosphere. On the other hand, the leaf carbon uptake reduces suddenly when the maximum temperature limit occurs [4].

The leaves of the trees are also included into the primary products of the forest landscapes. Hence, increasing temperatures stimulate the growth and amount of the leaves, which are also indicated by some vegetation parameters [5]. Leaf area index (LAI) is one of those vegetation parameters, defined as the one-sided area of the canopy leaves over the projected crown area of that canopy [1]. The LAI plays a key role on determining the level of the growth and productivity as well as on the hydrology of the forest ecosystems [6]. Indeed, the increasing LAI leads to the increment in the evapotranspiration rate to some extent [7]. Therefore, LAI is used as a key parameter in order to predict climate change impact on the growth, productivity, and hydrology of the forest landscape. On the other hand, the temperature-induced increasing metabolic activity of the forest trees triggers their roots to absorb more nutrients and water from the soil [8]. The water absorbed from the soil is eventually lost to the atmosphere through evapotranspiration [9]. As long as this water deficiency could not be met by the precipitation, drought stress occurs for the forest trees under the warming temperatures.

The climate change-induced forest landscape disturbances do primarily involve biome destruction, habitat isolation, and insect invasion. Furthermore, the climate change is able to transform landscape ecology, enhance or reduce photosynthetic activity, interrupt carbon balance, disrupt hydrology and biodiversity, and shift the phenology of the forest ecosystems. Furthermore, land use conversions may be triggered within the forest landscapes as consequences of climate change. On the other hand, the developments and new techniques are introduced in order to monitor and assess the direct and indirect impacts of climate change on the forest landscape. Hence, improvements toward those assessments are discussed together with the new modeling procedures within the scope of this study.

2. Methodology and overall studies

The scientific literature that concentrates and reviews the integrated climate change and forest landscape studies and researches is evaluated and discussed within the concept of this chapter. The world's climate has been changing particularly for the recent decades and is projected to change in the future. Climate change models especially for the forest landscapes and basins are proposed by the recent researchers. Spatial and temporal explicit landscape ecology models that consider patch-corridor-matrix interactions are produced [10]. Hydrological models and associated nutrient cycle models compatible with these climate change models are adapted also for the forest landscapes. Besides, ecological and biodiversity models that project lateral and vertical distribution of the forest species are developed. Moreover, spatially and temporally dynamic land use models are generated regarding the possible land use conversions and reorganization of the forest landscapes. In addition, innovative remote sensing techniques determine the leaf area index (LAI) of the forest landscapes to acquire their phenological and ecophysiological dynamics. These dynamics can be obtained through spatial and temporal analyses of the satellite data under the recent climate change phenomenon and under the prospective climate change scenarios. The LAI can also be indirectly observed and analyzed beneath and just above the canopy through some technological instruments. There are studies conducted for the forest landscapes within the continental scale such as done for Europe [11] and within watershed scale in Turkey [12]. There are also landscape-scale modeling attempts for simulating the dynamics of Oregon (USA) and Alpine (Austria) forest ecosystems [13]. The usage of the LAI parameter in order to define the impact of climate change is gaining importance particularly during the silviculture applications [14].

3. Warming climate

Climate warming indicates the increase in the average annual, seasonal, or monthly temperatures. The warming trends in the climate can be encountered at the local, regional, landscape, and global scales. According to the IPCC reports, global temperature is predicted to increase between 2 and 4°C especially for the second half of this century and is projected to increase toward the next century [15]. Future climate scenarios indicate the heat increases and associated droughts at the regional and global scales. The forest landscapes suffer the warming effects of the climate generally in the form of the drought stress and water deficiency. The warming climate is projected to cause the snow, constitute smaller fraction within the total precipitation which means lower snowpack water storage, and alert for possible summer water deficiency [16]. The summer water shortages particularly imply the drought stress posing threat for the forest trees, stands, and ecosystem.

The role of the forest ecosystems in order to produce microclimate and climate moderating capacity cannot be ignored for diagnosing and understanding climate change impacts on the forest landscapes. For the subalpine and temperate zones of the Swiss forest landscapes, the long-term moderating capacity of the forest canopy reached to 3.3°C being higher below the dense canopy and particularly during the spring when the vegetation period initiated [14]. Climate warming influences the density of the forest canopy and thereby affects the regeneration potential of the forests and the sustainability of the forest landscapes.

3.1. Climate warming and drought stress assessment

Climate change directly puts pressure on these forest ecosystems such as in the form of heat increase and drought stress within the landscape. Increasing temperatures stimulate the evapotranspiration leading to the water loss from the forest soil. The over loss of water from the soil results in the deficiency of soil moisture unless it is met by the precipitation. The long-term soil moisture deficiency alarms the first signals of drought within the forest ecosystem. As is known, the tree roots absorb water and uptake the nutrients dissolved in that water. The soil moisture deficiency restricts the nutrient uptake of the forest trees, which then handicaps the forest growth and productivity and disrupts the landscape health. The prolonged drought may have direct or indirect impacts on the species distribution and composition, biodiversity, net primary productivity, and faunal wildlife of the forest ecosystems [17].

The long-term climate warming stimulates the photosynthetic and metabolic activity of the forest trees extending their vegetation periods. The extended vegetation periods allow the forest trees to sequestrate more carbon and nutrients within their plant organs. Although the concentration of the carbon allocated by the forest tree leaves can be negligible compared to the concentration in the stem [18], outlasting leaves lose more water by transpiration leading to the drier soils. The sustained dry soils cause physiological stress particularly for the water-demanding forest tree species.

The studies emphasize the severity of drought in particular at the Western [19] and Southwestern [20] United States forest landscapes. According to a synthesis of the studies about the drought impacts on the forest dynamics, structure, and biodiversity, the hydrothermal surplus and deficit values are generally referred in order to anticipate the severity of the droughts [19]. They also mentioned about the developed models that would project the forest responses to those drought impacts. Forest landscape models focus on the drought impacts as well as the land use impacts on the forest ecosystems [19]. Additionally, ecological modeling procedures for projecting the land use dynamics and associated hydrological responses under the climate scenarios are proposed [21]. Recently, one of these modeling applications involve simulation of the forest stand dynamics and land use conversions in order to predict watershed water budget [21].

3.2. Assessment of climate change impacts on forest growth, yield, and productivity

The productivity of the forest landscape is particularly dependent upon the basic environmental factors including the light, water, temperature, and site nutrients [6]. These are the main drivers that influence ecophysiology of the forest trees and determine the forest landscape health. Since the temperature and radiation stimulate the forest growth, climate warming especially favors the net primary production (NPP). Most of the studies that focus on the interactions between the climate change and forest productivity have indicated the positive trend of the forest productivity with the climate change [22]. The carbon dioxide is the main determinant factor for such trends. Generally, forests tend to occupy the warmer climates [22]. Leaf area index is used as an indicator of forest growth and productivity and can be used for determining the distribution gradients of the forest tree species [12].

The basic assessment methodology of the forest growth response to climate change at the individual tree and species level is correlating the climate data with the tree-ring width and the diameter at breast height (DBH) measurements. Thus, the effects of increasing or decreasing temperatures, water abundancy or deficit on the tree-ring width changes, and tree radial growth alterations are to some extent analyzed and predicted. The tree-ring width measurements are conducted with the integrated precise instruments and software technologies. Besides, tapes, digital calipers, compasses, and more recently the laser technology serve for the DBH measurements.

The collected individual tree radial growth data provide establishment of a measured tree network at the forest species and stand levels. These forest species and stand-level data together with the climate data can be represented within the regional, landscape, or global scales using definite geo-statistical techniques of geographical information system (GIS) applications. Thereby, the spatial impacts of climate change on the growth and productivity of the forest landscapes are explicitly displayed thanks to the digital maps. Consequently, the validity of all these forest landscape data is able to be tested and analyzed dependent upon the comparison with the satellite data using the remote sensing techniques.

There are many studies in the literature that focus on the climate change impacts on the forest growth and productivity. Based on the tree-ring width calculation of variable forest species, a dendroecological study concluded that the northern limestone Alpine forest landscapes had overall been robust to the climate change [23]. The study insisted that the fir had relatively been drought-tolerant species compatible with climate change, whereas spruce had been the most sensitive species to the drought [23].

3.3. Climate change impacts on forest species distribution, biodiversity, and tree migration

Climate is a significant driving factor for the distribution of the forest species throughout the landscapes. Therefore, biodiversity of the forest landscapes differ based on the alterations among the regional climates [24]. Moreover, dependent upon their climatological requirements; primarily upon the temperature and precipitation tolerability limits, forest trees tend to migrate toward either warmer or colder altitudes or regions. Climate change impacts on the distribution and biodiversity of the forest species are evaluated and analyzed at different spatial scales ranging from individual trees to communities and biomes [25]. The biodiversity of the forest landscapes is affected by the climate change and in turn feedback the climate, encouraging the possible changes [25]. Climate change and land use lead to large shifts in biodiversity within the forest landscapes [25]. The rapid and extreme large shifts will be experienced especially at the ecotones of the semiarid landscapes [26]. One of those landscape-scale large shifts occurred in the 1950s at the Northern New Mexico where forest patches had become fragmented due to drought-induced mortality of ponderosa pines [26]. Possible global warming induces the treeline shift to the upper altitudes and toward the higher latitudes [27]. Such as the treeline shift, climate change has pronounced impacts on the forest tree migration, which is not only related with the genetic characteristics of the species but also considerably associated with the environmental conditions of the forest landscape.

4. Modeling and simulation procedures

Conceptual or numerical modeling and simulation procedures provide integrative comprehension of the forest ecosystems and natural landscape environment. In the recent years, climate change models are the basis for simulating and projecting the climate change impacts on the forest landscapes. Climate change models are particularly based on the atmospheric carbon dioxide concentration change scenarios [28]. The early modeling studies that concentrated on the assessment of the climate change impacts on the forest landscapes date back to almost 30 years ago. However, up to date, then, functions of the models had broadly been dependent upon solely to the climate, almost ignoring the tree physiology and other ecological and environmental factors such as soil characteristics, pests, pathogens, etc. [28]. Indeed, climate may not be only determinant, for instance, to simulate the tree species distribution throughout the forest landscapes.

The developments in the assessment of climate change impacts on the forest landscapes thank to two major procedures; implementation of modeling-simulation methods and application of remote sensing techniques. Modeling procedures principally involve the simulation of drought-induced stresses and phenological and physiological responses to these stresses. Consequently, the forest tree growth and productivity, disturbances, mortality and diebacks, tree migration, redistribution of the species composition, biodiversity alterations, pest and pathogen invasions, and harms are all modeled under the changing climates and related scenarios.

4.1. Drought stress, wind disturbance, growth, productivity, and tree mortality models

Climate change-induced drought and heat stress could shift the tree species composition, forest structure, and geographical distribution of forest landscapes in many regions [29]. The drought essentially causes tree mortality and disruption of patches within the forest landscapes. Therefore, the spatial heterogeneity within these drier forest landscapes impairs regeneration and provokes associated subsequent disturbances [30].

The wind damage to the forest trees is one of the severe disturbances within the forest patches [31]. In order to simulate the impacts of wind disturbances on the forest ecosystems, a process-based model was developed and coupled with the iLand (landscape simulation model), where the level of disturbance was accounted on the individual trees and on forest structure dynamically [32]. The model was tested for simulating windstorm damage on a forest landscape of Sweden, showing that the predicted results were highly compatible with the satellite-driven data [32]. The model was then improved in order to analyze its response to the bark beetle disturbances in a forest landscape under the +4°C climate warming, which is predicted to increase the disturbed area almost threefold [33]. One of the fundamental models that intend to assess the impacts of global changes on the growth and productivity of the forest landscapes is the forest growth model (3-PG) [34]. Referring to the satellite data, the physiologically based process model is used to simulate the response of maximum periodic annual increment (PAI) of forests to the climate change scenarios [34].

Drought stress has direct impacts on the forest landscape dynamics. Out of the models, LANDIS-II, the forest landscape disturbance and succession model, come into prominence that it is used for the analysis of the landscape dynamics [35]. Other empirical models can serve to this forest landscape disturbance and succession model [35]. One of the studies with the LANDIS-II revealed that the drought stress had affected species composition and total biomass in a forest landscape of Wisconsin (USA) [35]. They indicated that the forest tree species had responded to the length of the drought rather than the severity of that drought. The results of the model emphasized the significance of drought on the forest dynamics simulation and carbon storage [35].

The tree mortality may emerge based upon the climate change-induced drought-heat stress and associated ecophysiological failures, pest and pathogen outbreaks, and fires [29]. The realistic assessment of tree mortality within the forest landscapes should not be solely dependent upon the basic climate-tree mortality interaction models but also be supported with field observations. A review of these models can be seen in **Table 1**.

Model name	Focal task	References
3-PG (physiologically based process model)	Forest growth and productivity modeling	Coops and Waring [34]
MAPSS (Mapped Atmosphere-Plant-Soil System)	Vegetation (including forest) density and distribution modeling	Neilson [36] and Bachelet et al. [37]
MCI (combination of MAPSS and CENTURY)	Dynamic global biogeography projection	Bachelet et al. [37]
DISTRIB (empirical model)	Forest community distribution simulation	Iverson and Prasad [38]
DISTRIB and SHIFT (spatially explicit cell-based model)	Predicting colonization potential across fragmented landscape and tree migration	Prasad et al. [39]
FVS (forest vegetation simulator)	Landscape forest dynamics: stand dynamics, species composition, growth, and yield	Crookston et al. [41]
iLand (individual-based biodiversity model using empirical response functions)	Forest landscape biodiversity modeling: tree, ground cover, insect, and beetle biodiversity projection	Seidl et al. [13] and Thom et al. [42]
TreeMig (spatially explicit-dynamic forest landscape model)	Tree migration prediction, considering landscape patterns, reproduction, growth, competition, and mortality	Lischke et al. [43] and Nabel et al. [44]
LANDIS-II (spatially interactive forest landscape model)	Tree species migration, growth, mortality, succession, disturbance, and drought stress modeling	Scheller and Mladenoff [45] and Gustafson and Sturtevant [35]
LANDIS PRO (regional-scale forest landscape model)	Forest composition change due to population dynamics, dispersal, and harvest	Wang et al. [46]

Table 1. Models for assessing climate change impacts on forest landscapes.

4.2. Models of climate change impacts on species distribution, biodiversity, and tree migration

According to a study, the species distribution and biodiversity models that respond to the climate change impacts are summarized until then [25]. The summary indicates that the initiation of these models dates back to the 1990s. The global climate models that indicate the climatic drivers of the forest species distribution and biodiversity assisted the biodiversity models at the stand or landscape scales. Out of these models, the Mapped Atmosphere-Plant-Soil System (MAPSS) that simulates interactions between the biosphere and atmosphere regarding the climate change impacts dates back to 1995 [36]. The model capable of projecting forest distribution considers vegetation parameters such as leaf area index (LAI) and stomatal conductance in order to compute the hydrological balance within the landscape [36]. The equilibrium MAPSS model together with the dynamic MCI model that is an integrated version of MAPSS and CENTURY models were used in a study to simulate the biogeographical distribution of the vegetation across the US landscapes [37]. According to the findings of the study, a moderate rising of the temperature led to the denser vegetation and more carbon sequestration, whereas a large temperature rising resulted in large vegetation shifts and carbon losses [37].

The modeling approaches at the tree species and forest community levels have also become prevalent. An empirical model DISTRIB that relies on regression tree analysis was introduced in order to propose potential future distributions and suitable habitats for 80 forest tree species at the Eastern US landscapes under the climate change scenarios [38]. They predicted that the climatic change would have led to the slight increment in the tree species richness across the landscapes while maples, beeches, and birches would have lost most of their areas under all the climate change scenarios [38]. More than a decade later, the DISTRIB model was applied again for the Eastern United Sates forest landscapes under the future climate change scenarios [39]. They incorporated a spatially explicit cell-based SHIFT model to anticipate the colonization potential of the multiple tree species under the current fragmented habitats and notably the dominance potential of the oaks within the forest landscapes in order to inquire suitable habitats under the influence of the climate change [39]. The integrated model was successful in estimating the colonization potential, proposing that a narrow area of the suitable habitats for oaks would likely be occupied within 100 years [39].

Modeling the treeline shifts assists the scientific knowledge around the comprehension of the impact of climate change at the landscape scale and serves to develop forest landscape management proposals under the possible impacts. According to a study that produced spatiotemporal model for the treeline dominated by mountain pine at the Northern Calcareous Alps in Austria, the spread of the trees was slow due to the low growth rates and long generation terms [27]. On the other hand, in order to assess the uncertainty about the models of climate change effects on tree range distributions, eight models involving the ones of niche-based, process-based, growth index, and dynamic global vegetation were compared [40]. The study concluded that the model conflicts particularly originated from the rising carbon dioxide assumptions, whereas the models reconcile for the range contraction of Scots pine and for the range expansion of some Mediterranean tree species [40]. In a study, forest vegetation simulator (FVS) was adapted to respond to the effects of possible climate change accounting the regeneration, growth, mortality, and climate-induced genetic reactions [41]. According to the projections of the FVS model, the tree mortality tended to influence the stand dynamics within the landscape of the Western USA proposing the need for introducing new model as Climate-FVS [41].

Some of the recent models that intend to simulate the climate change impacts on forest landscape biodiversity not only project the disturbance of tree species but also predict the diversity of ground cover vegetation and insects. The iLand model is one of those complicated forest landscape biodiversity models that simulated the climate change impacts on the tree cover and composition change together with the responses of other organisms including the ground vegetation, spiders, beetles, and insects [42]. The model investigated the effects of the disturbance regime in a national park of Austria, indicating that the increasing frequency and severity of the disturbance had overall been beneficial for the mountainous landscape biodiversity [42].

Difficulty in modeling the forest tree migration under the climate change scenarios arises dependent principally upon the complexity of the vegetation and site parameters. One of the tree migration models at the landscape level is the TreeMig which is a dynamic and spatially explicit forest landscape model considering the reproduction, growth, competition, and mortality of the forest trees [43]. The model was tested for the Alpine landscape of Valais, Switzerland, where sudden and severe temperature declines led to rapid forest diebacks and rising temperatures retarded the species colonization [43]. The forest landscape model, TreeMig, was capable of producing landscape patterns as a result of endogenous and exogenous factors [43]. The TreeMig forest landscape model was then modified to simulate northward migration of the European hop hornbeams along the fragmented and climatically variable Alpine landscapes of Switzerland [44]. They concluded that the interannual climatic changes had significantly influenced the migration potential of the European hop hornbeam trees [44]. Another model that can simulate the forest tree species migration, growth, succession, and mortality under the influence of particularly the climate change, landscape fragmentation, and interspecific competition is the LANDIS-II [45]. In a study at the northern Wisconsin of the USA, the LANDIS-II was applied to inquire the aboveground biomass change and multiple forest tree species migration [45]. According to the results of their simulation, the landscape fragmentation obstructed the forest tree species migration and range expansion [45]. The regional-scale version of the previous model, LANDIS PRO, was run from 2000 to 2300 in order to account for the forest species composition changes due to multiple climate change scenarios and regarding the succession and harvest processes within the Central Hardwood Region of the USA [46]. The LANDIS PRO model predicted that the forest composition had tended toward xeric species rather than mesic species for the regional landscape [46].

5. Remote sensing techniques for assessing climate change impact on forest landscape

The assessment of the climate change impacts on the forest landscapes particularly concentrated on the two issues: modeling and remote sensing techniques. The remote sensing technique is used to monitor the forest growth, productivity, canopy height, and biomass over the past 30 years [47]. The usage of these techniques for the assessment of the impact of climate change on the forest productivity and growth has been gaining prevalence for the last decades in particular [22]. The remote sensing techniques allow the assessment of climate change impacts on the forest landscapes through monitoring the climate change-induced postfire impacts, shifting phenology of the forests trees, ecophysiological responses of the forest landscapes to the climate change, and growth and productivity measurements using the satellite data.

The forest disturbances in the landscape scale are particularly monitored through these techniques. These monitoring processes and procedures have intensively concentrated on the vegetation changes and responses to those disturbances. Forest landscape composition is also affected by the disturbances triggered by the climate change. These disturbances involve the harms caused by the insect outbreaks and windthrows. The remote sensing techniques are relatively able to predict the level of such disturbances. Therefore, these techniques are frequently referred in the researches that investigate the impacts of climate change on the forest landscapes.

5.1. Remote sensing of climate change-induced shifting phenology

The air temperature is generally estimated to increase with the ascending altitude, based on the principle of lapse rate that generally ranges between 0.4 and 1°C per 100 m.a.s.l. [48]. The phenology and ecophysiology of the forest landscape are directly influenced by this air temperature alternating along the altitudinal gradients. Hence, the phenological stages of the forest landscapes including principally the foliation processes; budburst, leaf onset, leaf expansion, and defoliation processes; senescence, discoloration, leaf fall are driven primarily by the climate. Consequently, any change in the climate is possibly to shift the timing of all these phenological processes and patterns.

The remote sensing techniques supply the spatial and temporal monitoring of these phenological stages and patterns of the forest landscape. The relatively key parameter in order to analyze these spatiotemporal changes is normalized difference vegetation index (NDVI) [47]. There have been global studies which concentrated on analyzing the climate change effects of alternating the phenology of the overall vegetation based on their NDVI data [49]. However, the satellite-based data should be validated with the ground-based data. By comparing the field data with the satellite data, a study tried to validate the land surface phenology of the mixed temperate forest at the landscape scale [50]. They used MODIS (moderate-resolution imaging spectroradiometer) and enhanced vegetation index (EVI) in order to inquire phenology along the springs of 2008 and 2009 [50]. The budburst stages were successfully derived by the satellite data which were also compatible with the ground-based data [50].

The insect injuries are relatively the major disturbances emerging as a consequence of climate change. In order to map the forest landscape composition following the post-disturbance by spruce beetle, a study used Landsat satellite data together with an image processing tool [51]. As a result of the study, they concluded that their percent canopy cover model could approximately predict the observed values [51]. The multipurpose vegetation parameter, LAI, is also used as indicator during the remote sensing studies with the purpose of assessing the climate change impacts and associated disturbance impacts on the forest landscapes [52, 53]. The tree mortality rates due to the heat-induced and greenhouse gases can be alerted for a possible climate change [54]. Drought-induced forest diebacks may occur, which can be detected by the satellite data indicating the change in the landscape color [55]. Hence, a study used Landsat imagery in order to detect drought-induced tree mortality in the forest landscape of Central Texas (USA) [55]. Furthermore, the usage of unmanned aerial vehicle (UAV) is among the novel and emerging technologies and techniques [56] that can practically be used for monitoring the phenology and ecophysiology of the forest landscapes affected by the climate change [57].

6. Conclusions and recommendations

Climate change not only influences mechanisms of the forest ecosystems but also directly or indirectly triggers extreme events such as landslides altering the physical structure of the forest landscapes [58]. Therefore, climate change impacts on the forest landscapes should be considered and handled in a broader scale involving forecast of the possible subsequent disturbances and estimate of the ecosystem's adaptation potential. Hence, the determination of the adaptive capacity of the forest landscapes is particularly significant from the point of developing sustainable management proposals for these fragile landscapes [59]. The studies referred in this study not only suggest the existing climate change-associated die-offs, insect and pathogen invasions, and outbreaks but also warn for the future vulnerability to climate-induced tree mortality, physiological stresses, land use transformations, etc.

On the other hand, climate change is a phenomenon influencing the urban forest environment as well as the natural forest resources and landscapes. Although the climate change prevails within both the urban and rural environments, the warm air originating particularly from the human-induced urban heat islands rather affects forest landscapes closer to the urban fringes. Therefore, climate change impacts on forests should not only be addressed within the stand level but also be handled and mitigated at the landscape scale. Consequently, the sustainable forest landscape management objectives could be achieved under the recent and potential climate change phenomenon.

The ecological models for the optimum management of the forest landscapes should be developed and advanced in order to account and anticipate even the latent consequences of the climate change together with the pronounced and apparent impacts. It is suggested that the future model projections about the disturbance regimes on the carbon balance would assist forest management [60]. These models should also indispensably be integrated with the continuous spatial and temporal monitoring procedures of remote sensing techniques based on satellite data and aerial photographs. However, the models that lack field observations and validations are somehow pending questions and uncertainties about thoroughly assessment of the climate change impacts on the forest landscapes. In addition, it is necessary that the independently conducted local and regional studies and researches should be conveyed and gathered for the sake of supporting the global knowledge store on climate change and forest landscapes.

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Chapter 10

Relationship Between Bird Diversity and Habitat along a Pine-Oak Successional Forest in Southern Mexico

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Additional information is available at the end of the chapter

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Abstract

Few studies have related bird species richness and abundance with vegetation structure at different successional stages in pine-oak forests of Mexico. We studied changes in the bird community across three successional stages of pine-oak forest: early, intermediate, and mature. Also, we related bird community attributes—including generalist and specialist birds—with vegetation variables. We analyzed the vegetation and estimated bird abundances in 10-min-count periods in 10 circular plots per successional stage. We recorded a total of 71 bird species: 21 were specialists and 50 were generalists. The completeness of species richness was between 79 and 88% in the sampled habitats. Diversity profiles were consistently larger in the intermediate stage, except for its species richness that was not different between this and the early stage. We found a more richness and a higher number of individuals of generalist in the early and intermediate stages. The abundance of specialists was higher in the mature forest. An ordination analysis showed that generalists were associated to different variables. This suggests that these species can adapt to different forest conditions. Some specialist birds were more abundant in sites with high dominance of trees. Our results confirm the importance of maintaining not only mature forests but also young successional stages in order to conserve the species typical of secondary pine-oak forest bird species.

Keywords: succession, temperate forests, bird community, species richness, change land use

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1. Introduction

The composition and the structure of vegetation influence the abundance and the distribution of birds in different environments [1–4]. In temperate forest ecosystems of Mexico and other parts of the world, many anthropogenic activities (i.e., agriculture, overharvesting, and fires) have increasingly modified the composition and structural characteristics of vegetation [5, 6]. As a consequence, many of these landscapes are currently characterized by secondary forest intermixed with scrublands, grasslands, and farmlands [7]. In these landscapes, vegetation in later successional stages is usually taller and has greater biomass, higher number of vertical structural components, and higher number of tree species [7, 8]. Due to their complexity, these habitats often provide a variety of food and nesting sites for specialist birds, in particular [3, 9]. Earlier stages, on the other hand, have a well-developed low layer and thicker vegetation. Thus, bird species richness and abundance vary across forest successional stages [10]. Many bird species, for example, are favored from disturbed sites that are often found in early successional stages, as these may have more habitats available for generalist species (i.e., species that use a relatively wide set of habitat resources [11–13]). On the other hand, Connell [14] suggests that species richness is greater in sites in the intermediate phase of succession because species characteristic of both early and mature forests (i.e., generalists and specialists) can coexist.

The presence of bird species across forest chronosequences (i.e., sequential changes in the structure of the community spatially and temporally reflected) can help in reverting fragmentation processes, as bird species participate in various ecological interactions (e.g., pollination, seed dispersal, and nutrient cycling). For example, in a montane oak forest in Costa Rica, Wilms and Kappelle [15] found that numerous birds that forage on the fruits of mature forest trees also forage for resources in pastures. Thus, birds may serve as key dispersal agents, transporting tree seeds from mature, closed forests to non-forested secondary scrublands or pastures.

Several studies have shown the relationship between bird communities and vegetation structure in temperate forests [10, 16, 17]. However, few studies have related bird species richness and abundance with vegetation structure at different successional stages in pine-oak forests in the Americas, and these have mainly evaluated changes in bird communities throughout post fires succession following fires in Canada and the United States [18]. In Mexico, there are some studies that have described changes in bird structure in sites of pine-oak forest with different levels of disturbance [19–21]. These studies have found a general pattern of a greater number of bird species in the undisturbed forests than in the disturbed forests, but there are no reports on the response of bird communities to plant succession in temperate ecosystem.

Our main goal was to evaluate the relationships between the richness and abundance of the bird community and the vegetation structure of three successional stages of a pine-oak forest (early, intermediate, and mature forests) in southern Mexico. In addition, we evaluated these relationships between vegetation and generalist and specialist birds. We hypothesized that bird species richness and abundance would rise in the intermediate stage (according to intermediate disturbance hypothesis [14]), due to the presence of both generalist and specialist birds. We also expected a greater presence of specialist bird in mature forest and an increase of generalist birds in the early stage. The results of this research can contribute toward improving the management policies of this ecosystem in the region, which has been continuously affected by anthropogenic disturbances.

2. Methods

2.1. Study area

This study was carried out in the locality of Xocomanatlan in the state of Guerrero between 17°33′21″ and 17°31′30″ N and 99°41′02″ and 99°38′16″ W (**Figure 1**). This area corresponds with the biotic province of the Sierra Madre del Sur which presents a high concentration of endemic species. The study area has an average elevation of 1900 m.a.s.l.

2.2. Sampling sites

Three sites of pine-oak forest in different successional stages were selected: early (last disturbance approximately 5 years ago), intermediate (last disturbance approximately 20 years ago), and mature (last disturbance approximately 40 years ago). At the mature forest site, large-scale agricultural activities have been largely impeded by the topography (inclination >45 grades) and the rocky substrate of the zone. The tenants have partially logged trees to obtain forest timber, although elements of the original flora are retained. The early and the intermediate successional sites were cleared by the traditional slash and burn method to plant corn and, to a lesser extent, peach, pear, apple, and hawthorn (*tejocote* in Spanish) orchards. The date of the last major disturbance and the type of management at each site were obtained from informal interviews with farmers.

2.3. Vegetation sampling

At each selected site, we randomly delimited 10 plots of 0.78 ha (**Figure 1**). Within each plot, we traced two perpendicular lines, oriented to the four cardinal points, with ropes. The taxonomic identity of all trees and shrubs with branches intersecting the ropes and with diameters at breast height (DBH) ≥ 10 cm was determined. The foliage cover of each of these plants was

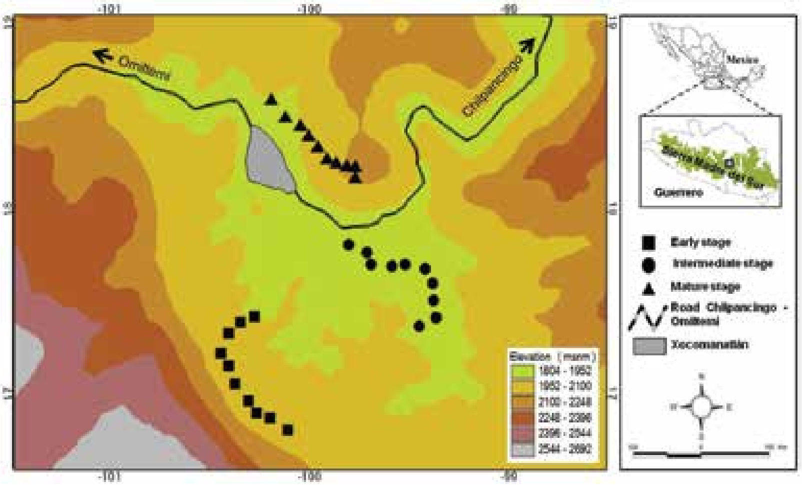

Figure 1. Location of the study area in the state of Guerrero in southern Mexico, showing the sampling plots in three successional stages of a pine-oak forest.

estimated by the ellipse formula, using the length of the maximum and the minimum diameters of the foliage [22]. Foliage stratification is a measure that reflects the foliage density and was estimated using an optical square marked with two perpendicular axes [23]. The square has three mirrors arranged so that an observer can look horizontally through the device and assess the height of objects above. In each plot, we recorded foliage height and the number of times that foliage touched the point of intersection of the two axes. This procedure was repeated every 50 cm along the two transects, resulting in a total of 50 observational data points. Foliage heights were estimated with the electronic clinometer Haglöf HEC and were grouped in 1 m intervals. With this data, we evaluated the foliage height diversity (FHD) using the Shannon-Wiener index. Detailed vegetation analyses for these sites are described in Almazán-Núñez et al. [7].

Plants were identified by consulting botanical specimen collections, and plants were deposited at the Herbarium of the Chemistry and Biological Sciences Faculty of the Autonomous University of Guerrero.

2.4. Bird survey

Birds were surveyed from October 2013 to April 2014 in the same plots where vegetation was sampled. The bird censuses were performed during the hours with the highest bird activity in the morning (07:00 to 11:00 h) and the afternoon (16:00 to 19:30 h). We used the point count method with a fixed radius of 50 m (10 point counts per site for a total of 30 points). This radius was selected because the vegetation structure of the three successional stages allowed observations to be performed at this distance. Each point count was separated by 200 m to avoid pseudoreplication. All individuals seen or heard within a 10-min period at each point were recorded. This time lapse is long enough to allow most birds present, including rare species, to be counted, and short enough to minimize the probability of counting the same bird more than once [24]. Each site was visited 30 times (for a total of 90 visits to all successional stages). The abundance of each species was the average number resulting from all visits. Raptors and vultures were excluded, as our point count sampling method was not designed to effectively sample these bird groups. Each observed bird species was categorized into two groups based on Stotz et al. [11] and personal observations: (1) specialist (i.e., birds inhabiting the forest interior that are rarely found in other habitat types) and (2) generalist (i.e., birds that use a variety of habitats including the forest interior, secondary forest, forest edges, and open areas).

2.5. Data analysis

We calculated the average and the standard error of the abundance and the richness of birds and of structural parameters (e.g., foliar cover, FHD) and of the plant community (e.g., richness and abundance of trees and shrubs) based on the vegetation data collected in the three successional stages. These variables were compared with a one-way analysis of variance (ANOVA) and multiple comparisons with Tukey's HSD. The Chao2 estimator was used to assess the comprehensiveness of the bird species survey. This nonparametric species richness estimator has been found to give reasonable approximations at small-grain-size sample units (i.e., circular plots [25]). Calculations were done with the software EstimateS version 9 [26]. To compare bird richness and diversity between seral stages, we used the number of effective species, which is the number of species of equal abundance expected in a community, and three levels of diversity (^0D, ^1D, and ^2D, respectively) were calculated. The first index corresponds to species richness and does not consider abundance (^0D). In the second index, species are weighted according to observed abundances (^1D). The third index (^2D) gives more weight to the dominant species [27, 28]. For this analysis, an estimated diversity profile was performed by using an analytical method to obtain accurate, continuous, and low-bias diversity especially for $q \leq 1$ [29]. In these comparisons, 200 replicate bootstrapping and confidence intervals at 84% were used, which are appropriates for inference equivalent to $P < 0.05$ [30]. This profile was calculated in the R package iNEXT [31].

Both the richness and abundance of the generalist and specialist bird species in each successional stage were compared and analyzed using a Kruskal-Wallis (K-W) test and an ANOVA, respectively. The significant differences were contrasted with a Mann-Whitney U test (in the case of K-W test) and multiple comparisons of Tukey's HSD (for the ANOVA). To evaluate the relationship between the abundance of specialist and generalist birds and the 10 habitat variables, a canonical correspondence analysis (CCA) was used. CCA is a multivariate ordination technique for associating species abundance data with multiple environmental factors [32]. From an initial examination of the correlation matrix of the 10 habitat variables, we eliminated several from the analysis that were redundant with other variables. Then, the CCA was executed using the uncorrelated variables. We eliminated bird species with less than three observed individuals from the CCA, as very low numbers can reflect random observations [33]. This analysis was performed in MVSP package v. 3.22 [34].

3. Results

3.1. Vegetation sampling

Tree richness and abundance were significantly highest in mature forest in comparison to the early and the intermediate stages (**Table 1**), while shrub richness and abundance did not show significant differences between the three successional stages (**Table 1**). The structural variables were significantly highest in the mature forest in comparison to the other two stages, except for plant density, which was greater in the intermediate stage, although this difference was not significant (**Table 1**).

3.2. Richness, abundance, and diversity of birds

We recorded a total of 71 species corresponding to 25 families (Appendix). Among these, 53 species were residents and 18 were migrants. Also, 21 species were classified as specialists and 50 as generalists (Appendix). In the intermediate successional stage, 47 species were recorded, in the early stage 45 species, and in the mature stage 36 species. The Chao2 richness estimator showed that the 79, 84, and 88% of the total expected richness, respectively, to the early, intermediate and mature stages were recorded.

Variables	Successional stage		
	ES	IS	MS
Tree richness	1.7 ± 0.26a	2.7 ± 0.30b	3.9 ± 0.31c
Shrub richness	2.3 ± 0.33a	2.6 ± 0.56a	1.9 ± 0.43a
Tree abundance	5.5 ± 0.82a	11 ± 1.55b	17.2 ± 1.46c
Shrub abundance	11.9 ± 1.86a	9.6 ± 2.47a	7.1 ± 1.26a
Shannon-Wiener index	1.1 ± 0.08a	1.3 ± 0.14a	1.4 ± 0.09a
Simpson dominance	0.4 ± 0.03a	0.4 ± 0.05a	0.3 ± 0.03a
Foliar cover (m^2)	249.1 ± 67.77a	539.1 ± 73.13b	659.3 ± 86.25b
Foliage height diversity (FHD)	1.7 ± 0.17a	2.1 ± 0.10b	2.3 ± 0.05b
Plant density	42.3 ± 6.90a	47.1 ± 7.93a	43.7 ± 4.38a
Total height	16.7 ± 1.19a	20.0 ± 0.70ab	22.8 ± 1.03b

Different letters denote significant differences ($P < 0.05$) according to Tukey's HSD tests. ES, early stage; IS, intermediate stage; and MS, mature stage.

Table 1. Average values (± SE) of habitat variables in three successional stages of pine-oak forest in southern Mexico.

The intermediate successional stage showed a higher diversity profile than the mature stage and was higher in the order q1 and q2 (abundance and dominance, respectively) than early stage, but not in q0 (richness), as confidence intervals overlapped. Diversity in the intermediate and early stages was not different (**Figure 2**).

The richness of specialist birds was not significantly distinct between the three successional stages (**Figure 3A**). The richness of generalist species was highest in the early and intermediate

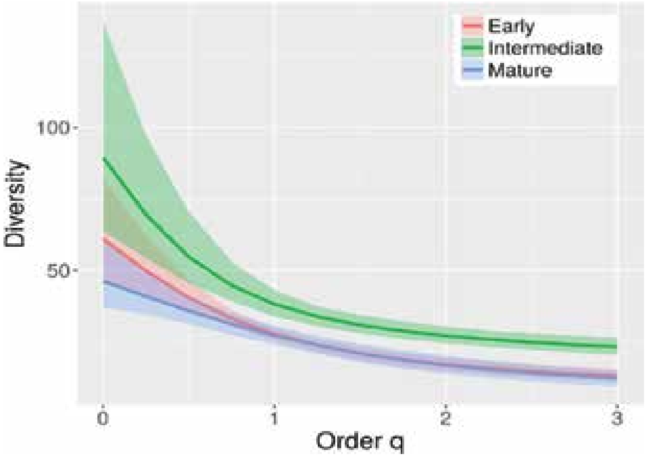

Figure 2. Estimated diversity profiles of bird assemblages across three successional stages, early, intermediate, and mature, in a secondary pine-oak forest in southern Mexico. Shaded polygons denote the 84% confidence intervals.

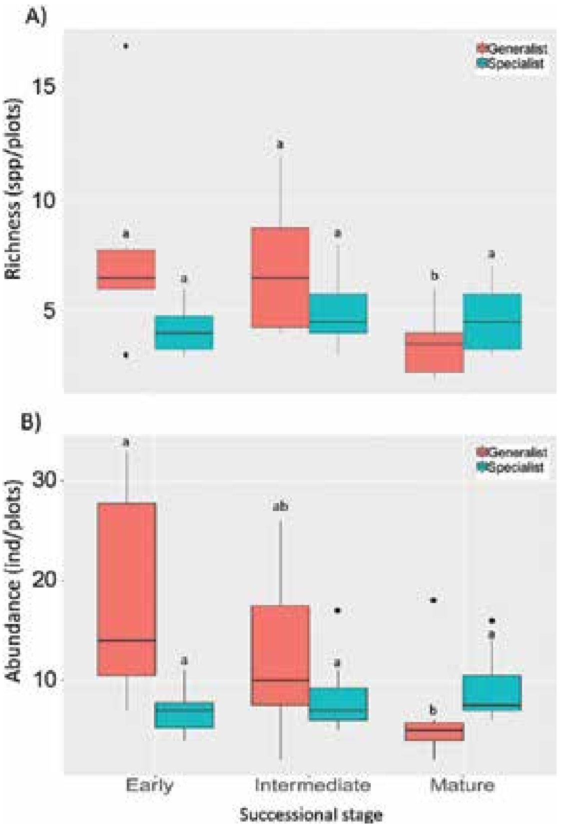

Figure 3. Boxplot of (A) richness and (B) bird abundance of generalist and specialist birds across three successional stages of a pine-oak forest in southern Mexico. Different letters indicate significant differences ($P < 0.05$) among successional stages, according to Mann-Whitney U test (richness of generalists) and Tukey's HSD (abundance of generalists). Within boxes, horizontal line indicates the median; box boundaries indicate interquartile range; whiskers indicate minimum, and maximum values and black dots are *outliers.*

stages, and the lowest species richness was found in the mature stage (**Figure 3A**). These differences were significant between successional stages ($X^2 = 7$, $df = 2$, $P = 0.030$; **Figure 3A**). The average abundance of specialist birds between the three successional stages was not significant (**Figure 3B**). Meanwhile, the abundance of generalist species was higher in the early stage and lower in the mature stage ($F = 5.32$, $P = 0.01$).

3.3. Relationship between distribution of generalist and specialist birds and habitat variables

The first two canonical axes of the CCA explained 85.5% (first axis = 56.8%; second axis = 28.7%) of the total explainable variation in the relationship between the distribution of generalist birds and habitat variables. The three main groups were revealed by this ordination analysis of generalist birds (**Figure 4A**). The first group is formed by species that forage in the foliage and that are associated with the height of plants and FHD, including warblers, such as

Setophaga ruticilla, *S. townsendii*, *Oreothlypis superciliosa*, and *Mniotilta varia*; vireos, such as *V. gilvus* and *V. solitarius*; and other migrants such as *Pheucticus melanocephalus* and *Polioptila caerulea*. The second group was formed by species associated with tree abundance and the diversity of plants (Shannon-Wiener index), including *Mithrephanes phaeocercus*, *Setophaga petechia*, *S. coronata*, *Cardellina pusilla*, and *Junco phaeonotus*. Finally, the third group was formed by *Aphelocoma coerulescens*, *Columbina inca*, *Catharus ustulatus*, *Sialia sialis*, and *Turdus migratorius* as well as the hummingbirds *Amazilia beryllina* and *Hylocharis leucotis* that were associated with shrub abundance (**Figure 4A**).

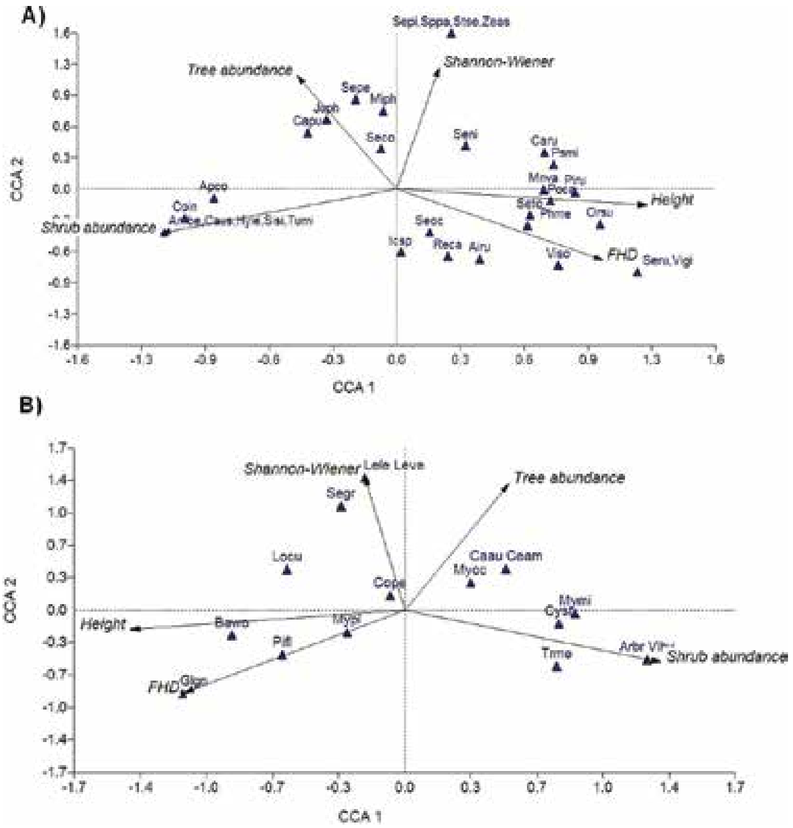

Figure 4. Canonical correspondence analysis of generalist (A) and specialist species (B) along the successional gradient of a pine-oak forest in southern Mexico. Variables: FHD, foliage height diversity; height, total height of the plants. Species: Airu, *Aimophila rufescens*; Ambe, *Amazilia beryllina*; Apco, *Aphelocoma coerulescens*; Arbr, *Arremon brunneinucha*; Bawo, *Baeolophus wollweberi*; Capu, *Cardellina pusilla*; Caru, *Cardellina rubrifrons*; Caau, *Catharus aurantiirostris*; Caus, *Catharus ustulatus*; Ceam, *Certhia americana*; Coin, *Columbina inca*; Cope, *Contopus pertinax*; Cyst, *Cyanocitta stelleri*; Glgn, *Glaucidium gnoma*; Hyle, *Hylocharis leucotis*; Icsp, *Icterus spurius*; Juph, *Junco phaeonotus*; Leaf, *Lepidocolaptes affinis*; Lele, *Lepidocolaptes leucogaster*; Leve, *Leptotila verreauxi*; Locu, *Loxia curvirostra*; Mefo, *Melanerpes formicivorus*; Miph, *Mitrephanes phaeocercus*; Mnva, *Mniotilta varia*; Myoc, *Myadestes occidentalis*; Mymi, *Myioborus miniatus*; Mypi, *Myioborus pictus*; Orsu, *Oreothlypis superciliosa*; Phme, *Pheucticus melanocephalus*; Pifl, *Piranga flava*; Piru, *Piranga rubra*; Poca, *Polioptila caerulea*; Psmi, *Psaltriparus minimus*; Reca, *Regulus calendula*; Seco, *Setophaga coronata*; Segr, *Setophaga graciae*; Seni, *Setophaga nigrescens*; Seoc, *Setophaga occidentalis*; Sepe, *Setophaga petechia*; Sepi, *Setophaga pitiayumi*; Seru, *Setophaga ruticilla*; Seto, *Setophaga townsendi*; Sisi, *Sialia sialis*; Sppa, *Spizella passerina*; Stse, *Stelgidopteryx serripennis*; Trme, *Trogon mexicanus*; Tumi, *Turdus migratorius*; Vigi, *Vireo gilvus*; Vihu, *Vireo huttoni*; Viso, *Vireo solitarius*; Zeas, *Zenaida asiatica*.

On the other hand, the CCA of the specialist birds explained 92.1% (first axis = 60.5%; second axis = 31.6%) of the total explainable variation. The distribution of some specialists, including woodcreepers, such as *Lepidocolaptes leucogaster*, the dove *Leptotila verreauxi*, and the warbler *Setophaga graciae*, were associated with the diversity of plants (Shannon-Wiener index; **Figure 4B**). Moreover, species as *Glaucidium gnoma*, *Piranga flava*, and *Baeolophus wollweberi* were associated with FHD and the height of plants. Other species such as *Arremon brunneinucha*, *Vireo huttoni*, and *Trogon mexicanus* were associated with shrub abundance, while *Myadestes occidentalis*, *Catharus aurantiirostris*, and *Certhia americana* were associated with tree abundance to a lesser extent (**Figure 4B**).

4. Discussion

Our results showed that bird diversity profile was higher in the intermediate successional stage than the other stages, except for its species richness that was not different to the early stage. These results support partially the intermediate disturbance hypothesis [14], which states that within a broad range of environmental disturbance levels, species diversity is maximized at an intermediate level of anthropogenic and natural disturbances, because competitively inferior, disturbance-tolerant species and competitively dominant, disturbance-sensitive species coexist when disturbances are neither too rare nor too frequent [35]. We also found less bird species in mature forest than in young secondary forests. This result coincides with those found by other authors along successional gradients in different types of vegetation in the Neotropics of Mexico [36, 37], Costa Rica [15, 38], and Ecuador [12, 39]. In addition, land-use history can influence the type of vegetation that regenerates after a disturbance. Gaps in the canopies of disturbed forests, for instance, promote an increase in the productivity of the lower vegetation strata [7, 38]. As a consequence, secondary successional habitats can provide a complex mosaic of microhabitats that can attract many bird species, such as *Junco phaeonotus*, *Mitrephanes phaeocercus*, and *Myadestes occidentalis*, which are favored of the open canopies to foraging. In our study area, pine tree richness was highest in the early successional stage, while pine tree abundance increased in the intermediate successional site [7]. This can explain the high richness of birds in the first successional stages that use these trees as foraging sites, such as warblers (*Cardellina pusilla*, *C. rubrifrons*, *Setophaga pitiayumi*, *S. striata*, *S. coronata*, and *S. townsendii*). In addition, recent studies have suggested that birds associated with mature forest might actually prefer early-successional habitat to mature forest during the post-fledging period to replenish energy [40]. Although this aspect was not evaluated in this study, it is probable that some birds, such as *Trogon mexicanus*, *Contopus pertinax*, *Cyanocitta stelleri*, and *Catharus aurantiirostris*, can move during the post-fledging period to the early stages in search of resources, as it has been observed in this stage.

In our study area, the higher diversity of birds at the early and intermediate stages was also likely due to the presence of peach, pear, apple, and hawthorn orchards, which may provide a diversified food supply for birds during certain periods of the year [41, 42]. On the other hand, there was a high dominance in the early and mature stages compared to the intermediate stage. Different studies showed that in the early stage this is a typical pattern, where species associated to this stage reach high abundance values in relation to other species [43], as it was the case of the resident species *Aphelocoma coerulescens* and a migratory one, *Regulus calendula*. In mature forest, the dominance is not a common pattern, but in our study area, this result can be explained by the presence of resident species that were observed actively during the nesting period (March–April), such as *Trogon mexicanus* and *Myioborus pictus*.

On the other hand, the higher richness of generalist species in the intermediate and the early stages resulted to a large extent from the presence of Neotropical migrants that can readily use such habitats in the first stages of succession [12, 13]. In fact, there was an increase in the richness and abundance of generalist birds in these stages, which differed with the mature stage, something that is not occurred with the specialist birds whose presence was the same in the three successional stages. Moreover, the ordination analysis showed that the generalist birds were associated to different variables such as height of plants, FHD, and tree and shrub abundance. These correspondences are based according to their behavior and feeding habits, demonstrating that these species can adapt to different forest conditions [4, 44]. Particularly, species such as *A. coerulescens*, *Columbina inca*, *Catharus ustulatus*, *Amazilia beryllina*, *Hylocharis leucotis*, *Sialia sialis*, and *Turdus migratorius* were more abundant in sites with predominance of shrubs. These species have been associated with open areas with scattered trees [45]. Also, other bird groups (i.e., *Oreothlypis superciliosa*, *Polioptila caerulea*, *Piranga rubra*, *Mniotilta varia*, *Setophaga townsendi*, *Setophaga ruticilla*, *Vireo gilvus*, *V. solitarius*) were more associated with total height of the plants and foliar height diversity or FHD, because their habitat uses and foraging behavior include branches and leaves [46]. Specialist birds' abundance increased in the mature forest, which coincides with previous studies that showed that specialization in bird communities generally increases in the course of forest succession [47]. Some specialist birds such as woodcreepers (*Lepidocolaptes leucogaster*) were related to the abundance of trees and Shannon-Wiener index (giving more weight to plant richness), as these birds need trees to make their nests in cavities and forage on trunks and bryophytes [48]. On the other hand, other specialist birds were associated with shrub abundance and corresponded to their ecological needs. Particularly, shrub dwellers such as *Myioborus miniatus* and *Vireo huttoni* use low strata in vegetation in search for food and even for cover, since they are ground foragers like *Arremon brunneinucha*.

5. Conclusions

Our results demonstrate the importance of maintaining not only the mature forest but also young successional stages for conservation of secondary pine-oak forest bird species. According to this, the forest policies and management planners must take into account how land and forest are used in pine-oak ecosystems of southern Mexico, in order to improve landscape heterogeneity and multiply different habitat and foraging site opportunities for bird species.

Acknowledgements

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A. Appendix

Composition of bird species in three successional stages of pine-oak forest in southern Mexico. *Seasonality*: R, resident; M, migratory. *Bird groups*: G, generalist; S, specialist. *Successional stage*: ES, early stage; IS, intermediate stage; MS, mature stage. The relative abundance of bird species per successional stage is presented

Family	**Species**	**Seasonality**	**Bird groups**	**Successional stage**		
				ES	**IS**	**MS**
Columbidae	*Columbina inca*	R	G	0.015	0.004	
	Leptotila verreauxi	R	S		0.009	
	Zenaida asiatica	R	G		0.009	
Cuculidae	*Piaya cayana*	R	G	0.008		
Trochilidae	*Lampornis clemenciae*	R	G		0.009	
	Archilochus alexandri	R	G	0.008	0.009	
	Amazilia beryllina	R	G	0.008		
	Hylocharis leucotis	R	G	0.030		
Strigidae	*Glaucidium gnoma*	R	S			0.013
Trogonidae	*Trogon mexicanus*	R	S	0.019	0.040	0.138
Picidae	*Melanerpes formicivorus*	R	G			0.031
	Picoides scalaris	R	G		0.004	
	Picoides villosus	R	S		0.004	0.006
Furnariidae	*Lepidocolaptes leucogaster*	R	S		0.022	
	Lepidocolaptes affinis	R	S		0.004	
Tyrannidae	*Mitrephanes phaeocercus*	R	G	0.049	0.084	0.006
	Contopus pertinax	R	S	0.026	0.044	0.038
	Empidonax fulvifrons	R	G	0.004		
Vireonidae	*Vireo huttoni*	R	S		0.004	
	Vireo solitaries	MI	G	0.004		0.019
	Vireo gilvus	R	G			0.013
Corvidae	*Cyanocitta stelleri*	R	S	0.053	0.018	0.006
	Aphelocoma coerulescens	R	G	0.162	0.026	
Hirundinidae	*Stelgidopteryx serripennis*	R	G		0.022	
Paridae	*Baeolophus wollweberi*	R	S		0.022	0.069
Aegithalidae	*Psaltriparus minimus*	R	G		0.048	0.038
Certhiidae	*Certhia americana*	R	S	0.011	0.013	

Family	**Species**	**Seasonality**	**Bird groups**	**Successional stage**		
				ES	**IS**	**MS**
Troglodytidae	*Thryomanes bewickii*	R	G	0.004		
	Troglodytes aedon	R	G			0.006
Polioptilidae	*Polioptila caerulea*	MI	G	0.004	0.022	0.025
Regulidae	*Regulus calendula*	MI	G	0.094	0.040	0.069
Turdidae	*Sialia sialis*	R	G	0.023		
	Myadestes occidentalis	R	S	0.056	0.066	0.019
	Catharus aurantiirostris	R	S	0.015	0.018	0.019
	Catharus ustulatus	T-MI	G	0.011		
	Turdus migratorius	R	G	0.023		
Mimidae	*Melanotis caerulescens*	R	G	0.004	0.004	
Ptiliogonatidae	*Ptiliogonys cinereus*	R	G	0.004	0.004	0.031
Peucedramidae	*Peucedramus taeniatus*	R	S			0.013
Fringillidae	*Euphonia elegantissima*	R	G	0.004		
	Loxia curvirostra	R	S		0.022	0.019
	Spinus notatus	R	G	0.008		
Passerellidae	*Arremon brunneinucha*	R	S	0.008		
	Atlapetes pileatus	R	S			0.013
	Aimophila rufescens	R	G	0.004		0.006
	Melozone albicollis	R	G	0.011		
	Spizella passerina	MI	G	0.015	0.026	
	Junco phaeonotus	R	G	0.064	0.070	
Icteridae	*Icterus spurius*	MI	G	0.004		0.025
	Icterus bullockii	MI	G		0.004	
Parulidae	*Mniotilta varia*	MI	G	0.008	0.031	0.031
	Oreothlypis superciliosa	R	G		0.004	0.013
	Setophaga ruticilla	T	G			0.006
	Setophaga pitiayumi	R	G		0.013	
	Setophaga petechia	MI	G	0.004	0.009	
	Setophaga striata	T	G		0.013	
	Setophaga coronata	MI	G	0.053	0.057	0.013
	Setophaga graciae	R	S		0.035	0.006
	Setophaga nigrescens	MI	G	0.004	0.018	0.006
	Setophaga townsendi	MI	G	0.015	0.018	0.031
	Setophaga occidentalis	MI	G	0.026	0.004	0.019
	Cardellina pusilla	MI	G	0.015	0.013	
	Cardellina rubrifrons	MI	G		0.013	0.006
	Myioborus pictus	R	S	0.060	0.066	0.156
	Myioborus miniatus	R	S	0.011	0.004	

Family	Species	Seasonality	Bird groups	Successional stage		
				ES	IS	MS
Cardinalidae	*Piranga flava*	R	S	0.011	0.013	0.069
	Piranga rubra	MI	G		0.004	0.006
	Piranga bidentata	R	G	0.004		
	Pheucticus melanocephalus	R	G	0.004	0.004	0.013
Thraupidae	*Diglossa baritula*	R	G	0.004	0.004	0.013
	Sporophila torqueola	R	G	0.045		

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