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Silviculture

Edited by Ana Cristina Gonçalves

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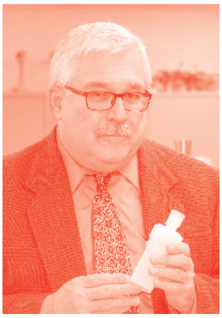



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Preface

Silviculture refers to the sustainable management of forest ecosystems, taking into consideration the product and service needs of society. It also focuses on forest stands, using several methods and techniques to ensure their conservation and productivity. Due to stand structure diversity, from even-aged monospecific to uneven-aged multi-specific stands, and to the wide assortment of forest products and services, the characterization and modeling of forest stands are made with a large suite of approaches related to the objectives, productions and services, and yields of forests.

Among other determinant practices, afforestation and reforestation are of primordial importance to forest sustainability. They are the primary drivers for the perpetuity and sustainability of forest stands as well as their related products and services.

This book focuses on silviculture as well as forestation. It contains the following seven chapters:

Chapter 1 "Silvicultural Practices in Venezuelan Natural Forests: An Historical Perspective and Prospects of Sustainable Forest Management"

Chapter 2 "Mixed Forest Plantations with Native Species for Ecological Restoration in Cloud Forests of the Venezuelan Andes"

Chapter 3 "Thinning: An Overview"

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Chapter 5 "Basic Theory and Methods of Afforestation"

Chapter 6 "Afforestation in Karst Area"

Chapter 7 "Legal and Administrative Aspects of Forest Pest and Disease Control in Japan"

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Chapter 1

Silvicultural Practices in Venezuelan Natural Forests: An Historical Perspective and Prospects of Sustainable Forest Management

Emilio Vilanova

Abstract

More than four decades of cumulative silvicultural experience in Venezuelan forests represents a significant progress towards sustainable forest management in the tropics. Here, based on an extensive literature review, expert opinions and discussions with forestry stakeholders in the country, we offer a broad overview of the history and current state of silvicultural practices in Venezuela's natural production forests. Despite important research advances, several factors including institutional and policy limitations, along with the lack of sound technical guidelines have hampered a more positive influence of silvicultural research for sustainable forest management across the country's managed forests. On an industrial scale, after an often poorly planned selective logging, and despite increasing evidences against for, a strong prominence of assisted natural regeneration (i.e., enrichment planting) characterized the post-logging management compared to other approaches. With very few exceptions, using artificial regeneration did not produced the expected outcomes in terms of tree growth, expected timber yield and survival. Finally, amidst the current political and economic upheaval in Venezuela, a broad range of lessons and policy recommendations is proposed including the strengthening of research on silvicultural options for multiple use of forests and for climate change mitigation and adaptation.

Keywords: enrichment planting, forest policy, minimum harvest diameter, research, tropical forests, Venezuela

1. Introduction

Forests in their multiple forms and types are the dominant terrestrial ecosystems on Earth, covering about one third of the globe's land area [1]. Forests represent a fundamental component of world's carbon cycle, are the habitat for biodiversity and are important for the provision of a myriad of services from which people depends for their livelihoods. Distributed over different environmental and latitudinal gradients, forest ecosystems account for at least 75% of global terrestrial productivity (GPP) [2], with tropical forests (TFs) being disproportionally relevant compared to other forests-types in the temperate or boreal regions. For example, TFs store 200–300 Pg C, about a third as much as is held in the atmosphere [3, 4].

Globally, recent estimates indicate that the forest area under management plans, mostly for timber production, has increased since 2000 reaching close to 2.05 billion ha in 2020 [5]. However, this proportion remains largely unbalanced when compared across regions, and for example less than 20% of the total forested area of South America has some type of long-term management plan [5], while high rates of deforestation and degradation continue to threaten the stability of forests, particularly in the tropics [5–7]. In terms of forest management, more than 400 million hectares (ha) of natural tropical forests have been designated as production forests [8–10]. Moreover, nearly 40% of sawn wood traded annually in tropical regions has an origin in natural forests [11], often under a "selective logging" approach in which large trees of a relatively low number of tree species are harvested in rotation cycles of 30 years on average [9, 12, 13]. The dynamics driving how tropical forests respond, and ultimately recover to this type of intervention is a function of several ecological and socio-economic factors. Yet, the characteristics of the logging practices (i.e. intensity of harvest, conventional vs. reduced impact logging), the elapsed time before the next harvest, and the post-harvest interventions are all silvicultural decisions particularly relevant to facilitate the speed of the recovery and the features of the future forest [14, 15]. Thus, throughout this entire process, silviculture plays an important role by ideally outlining the 'best' system and a set of specific practices to facilitate long-term forest management.

As an applied discipline, silviculture traditionally has aimed at controlling the establishment, composition, structure, growth, and the role of trees within a forest to create a more predictable production system [16, 17]. While objectives of forest management have changed globally in the last two decades with an increasing relevance for conservation of biological diversity, carbon storage, and other ecosystem services, the design, planning and application of silvicultural practices is still very much oriented towards timber production that often reduce structural and biological complexity, and has become a prevalent driver of change in many tropical managed forests [18–21]. From the outset, tropical silviculture faces the challenge of reconciling timber production as a primary goal with long-term conservation of forest ecosystems, so thresholds of extraction intensity coupled with silvicultural treatments needs to be compatible with the maintenance of biodiversity and other ecosystem services, as well as the financial viability for all the actors involved [9]. Accomplishing this goal, while difficult, seems more feasible than a few decades ago, given that the levels of ecological knowledge of tropical forests has increased enormously in the last 20 years, which means that there has never been a sounder scientific basis from which to guide forest management in the tropics [22].

This chapter discusses the history of silvicultural practices in natural forests of Venezuela, a country with one of the longest history of forest management in the tropics [23, 24] (**Figure 1**). First, an outline of the context of Venezuelan forestry is presented, including an overview of forest extension, forest types and the characteristics of forest management in the country. A review of the main silvicultural systems historically applied with considerations of their effectiveness and impacts is shown to finalize the analysis with a general proposal of recommendations to improve how forests in Venezuela could be sustainably managed in the twenty-first century.

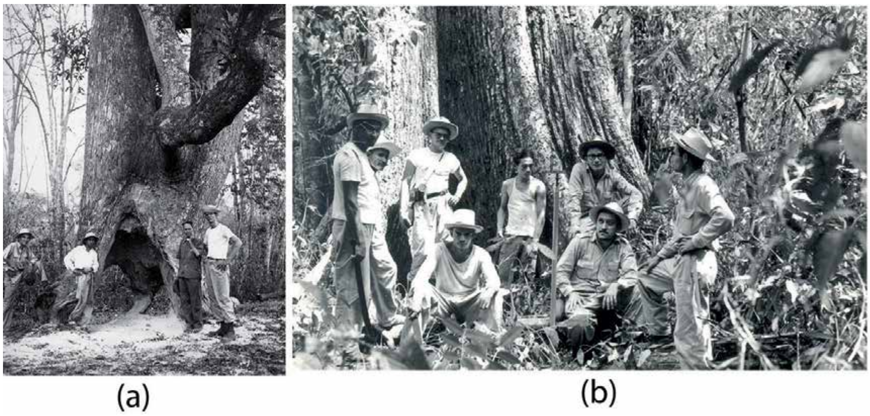

Figure 1.
(a) Early forest exploration in Turén Forest Reserve, Portuguesa state, Venezuela, circa 1965. Photo: Courtesy of Giorgio Tonella; (b) forestry students in early 1970s doing forest inventory in Caparo Forest Reserve, Barinas State. Photo: Lawrence Vincent.

2. A general description of Venezuela's forest cover

Venezuela has a total land area of about 916,445 km^2 with 45–50% of this area covered by different types of forests [25, 26]. The variation in forest cover in Venezuela in the last three decades has followed a similar trend as in many tropical countries with a notorious peak in forest loss during the early 1990s and a slightly declining trend in deforestation rates towards the end of the twentieth century [6]. Over a longer time period, Pacheco et al. [25] found that between 1920 and 2008 Venezuela had, on average, an annual rate of forest loss of 0.30% per year, with a net decrease of 26.4% in the national forest cover for the entire period. It is around the beginning of the 1950s that a sharp increase in deforestation especially in the Western Plains ecoregion occurred, an area that remained as one of the national hotspots of deforestation for a long time [27]. With the somewhat historical unbalanced distribution of Venezuelan population, largely concentrated to the northern portion of the country, from the 36% of forest cover that was estimated to exist in this region by the mid twentieth century, some estimates place this number to as low as 10% in recent decades [24, 28], leaving the vast region of the Guiana Shield to the south of the Orinoco river as the main forested area in the country.

According to recent estimates from the Food and Agriculture Organization of the United Nations (FAO) [29] 287,500 ha, on average, were lost every year in Venezuela between 1990 and 2000 (−0.6% year^{-1}), with a decrease in forest cover during the 2000–2010 decade of about 164,600 ha per year (−0.3% year^{-1}). Updated statistics from public available data in Global Forest Watch (**Figure 2**) (www.globalforestwatch.org) that uses information from the study of Hansen et al. [6] indicates that from 2001 to 2018, Venezuela lost 1.95 million ha of forests (average of 108,333 ha per year), while only gained 191,000 ha of tree cover. In recent years, the spike in deforestation to the southern region in the Guiana Shield has been mostly driven by illegal gold mining [24, 28, 30]. Forests of the Western plains have been mostly cleared for agricultural purposes with current standing forests in this region being located mostly within protected areas and other inaccessible areas [24, 30], while agriculture also appeared to be a main driver of forest loss in the Andean region [31].

In Venezuela, the effects of deforestation and forest degradation in terms of carbon released have not been officially quantified. Lack of standardized methods for monitoring forest cover, the undermining of institutional capacities, and a

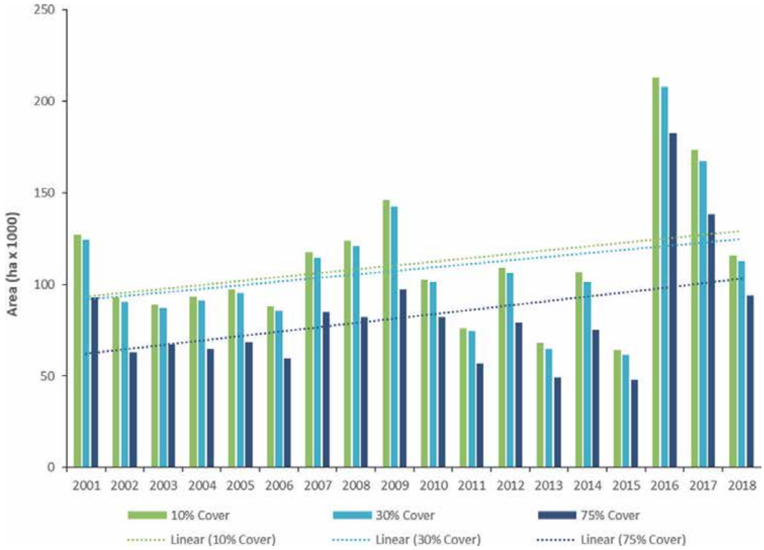

Figure 2.
Total annual forest cover loss (in thousands of hectares per year) in Venezuela for the 2001–2018 period using different proportions of forest canopy cover. Each category includes a linear trend. The figure was built using public data available from the global forest watch (www.globalforestwatch.org).

dramatic decline in professional training, among other factors, helps explaining this situation. However, reviewing the literature in this topic we find few studies that have shown that carbon emissions due to deforestation and forest degradation in Venezuela can be significant. Most of these studies have been conducted at global or pantropical scales and includes a reference for Venezuela. For example, Harris et al. [32] estimated that, between 2000 and 2005, about 9 Tg C year^{-1} (units are 10^{12} grams of carbon per year) were lost due to deforestation in Venezuela. This estimation might have represented between 9 and 28% of national emissions during the last decade [30]. Additionally, Pearson et al. [33] found for the 2005–2010 period that close to 10% of Venezuelan carbon emissions came from forest degradation, including selective logging, wood fuel harvest, fires and grazing as the main factors.

3. Environmental setting and forest-types in Venezuela

Located in the northern portion of South America, slightly above the equator, like much of the tropical region, Venezuela is largely subjected to the influence of the intertropical convergence zone (ITCZ), which affects rainfall patterns and results in the existence of wet and dry seasons in comparison to the cold and warm seasons of higher latitudes. ITCZ's position, structure, and migration influence the ocean-atmosphere and land-atmosphere interactions on a local scale, the circulation of the tropical oceans on a basin scale, and a number of features of the Earth's climate on a global scale [34]. Land form and relief, or the physiographical features in the land in Venezuela, largely expressed by the existence of three major mountain systems and different types of plains and savannas are a major driver of the seasonal and geographical patterns in rainfall in Venezuela at local and regional scales [35]. At least two major gradients in the distribution of precipitation in the country have been described: one from the Northeastern Atlantic to the Andes in the west, and a second one from the Caribbean Sea to the southern Amazonian flatlands. Annual precipitation in Venezuela ranges from less than 400 mm per year in some of the driest portions of the country up to about 4500 mm year^{-1}, which along with its seasonal distribution influence the type and characteristics of the vegetation [35].

From the standpoint of temperature, though much less variable than precipitation, there are important differences at regional scales, mostly as a response of elevation and latitude. With an elevation range from sea level up to close to 5000 m in the peaks of the Andean mountains, temperature varies accordingly [36]. Consequently, a highly diverse vegetation can be found across the country, where up to 18 different types have been identified with lowland evergreen forests largely dominating the nation's landscapes [35]. Other important formations include the cloud forests restricted to a rather narrow elevation gradient in different mountainous areas of the country, palm-dominated and swamp forests in the Orinoco Delta, mangroves, riparian and semi-deciduous forests across much of the savanna and the plains, and different expressions of shrub-like vegetation, grasslands and savannas. In fact, savannas might account for close to 25% of the country's land area with major continuous savannas located in the central Orinoco Plains (*Llanos*) limited by the Andean and Coastal Mountains to the west and north, respectively, and with a second large savanna in the Guiana Plateau (*Gran Sabana*) in the southeast of the country [37].

4. Management of natural forests in Venezuela

4.1 General overview

Venezuela is a tropical country having made one of the longest and continuous effort towards natural forest management (NFM), especially under long-term concession tracts in Tropical America. During the 1970s, the introduction of a forest concession system represented a significant advancement towards NFM at a regional level [24, 38, 39]. The first private concessions were allocated in 1970 and were probably the first known attempt to formally develop long-term management plans in the tropics, including silvicultural practices as a core component. By 1992, almost 3.2 million ha were allocated in more than 30 forest management units (FMUs) and had management plans approved by the national government [40]. In 1995, the national government planned a significant increase in the area under forestry concessions to 10 million ha over 5 years but the country's adoption of new policies and the rising criticism to forest management strategies prevented this from happening [24]. This process led to a significant decline in timber production coming from FMUs. For instance, in 1987 almost 40% of the national round wood production came from natural managed forests [40], while this proportion dropped to only 7% 20 years later [41], shifting the demand for timber products essentially to plantations. Albeit the lack of good quality indicators that has characterized the last decade in terms of forest statistics nationwide, the last available official data from 2018 indicates that only 2.5% of the wood legally consumed in the country came from FMUs in an estimated area of 246,313 ha of forests with formal management plans [42] (**Table 1**).

Several analyses of the forest management model applied in Venezuela have highlighted critical limitations in multiple aspects of the management process, including planning deficiencies, inadequate policies, and overall an insufficient application of sustainable management guidelines during forest operations. While some of the reasons behind this situation may fall outside the specific realms of forest management activities, there is consensus that forest management practices have not contributed to guarantee the long-term permanence of production forests nationwide [40, 43–45]. In a 2011 pantropical assessment led by the International Tropical Timber Organization (ITTO) [8], after almost four decades of NFM in

Description[a]	Total area (ha)	Relative proportion of the total area (%)
Areas under special administration (n = 382)[b]	67,883,078[c]	—
Natural forest production areas	11,183,202	16.5%
Forest reserves (n = 10)	6,742,047	9.9%
Forest lots (n = 4)	967,093	1.4%
Forest areas under protection (n = 43)	3,473,702	5.2%
Area with approved forest management plans	246,313	2.2%[d]
Area of forest plantations for wood production[e]	557,324	—
Wood production	Volume (m^3)	Relative proportion of total volume (%)
National roundwood production in 2017 (m^3)[e]	496,748	—
National roundwood production outside Forest Reserves in 2017 (m^3)	484,429	97.5%
National roundwood production inside Forest Reserves in 2017 (m^3)	12,319	2.5%

[a]*Modified from the last available official forest statistic report from 2018 [42]. Production forests are classified as category VI as per the International Union of Conservation of Nature (IUCN) guidelines to label protected areas with sustainable use of natural resources.*
[b]*Venezuela has a complex system of natural protected areas (NPAs). These are managed for specific purposes according to special laws and were designated as* "áreas de administración especial" *– ABRAE (Areas under Special Administration), which includes up to 25 different categories including National Parks, Natural monuments, wildlife refuge, among others. More info on Venezuela's protected areas can be found elsewhere (e.g. [24, 28]).*
[c]*Includes overlapping in some protected areas which implies the net area protected is lower.*
[d]*Relative proportion is given based on the total forest production area of 11,183,202 ha.*
[e]*Planted forests are not part of the ABRAE system. This area is based on 2014 data from the official government report [46] submitted to the 2015 Global Forest Resource Assessment program from FAO (http://www.fao.org/forest-resources-assessment/en/). Includes timber production from forest plantations mainly of exotics species such as Caribbean Pine (*Pinus caribaea*), Eucalyptus (*Eucalyptus sp.*) and Teak (*Tectona grandis*).*

Table 1.
General overview of the forest sector in Venezuela.

Venezuela, of the approximately 12 million ha of production forests, a very low proportion of close to 0.03% was considered as being sustainably managed. Moreover, Venezuela is one of the few tropical countries with no certified forest management operations in natural forests. In addition, although a modest progress has been made to establish a more inclusive approach to include local communities in the application and benefits of forest management, the few community-based cases that did exist have resulted in deforestation and degradation of forests [44, 47]. Also, there is no detailed information on the distribution of formal and informal employment in the Venezuelan forestry sector to quantify the social impact generated by this activity. Available data indicates that forestry's contribution to national gross domestic product (GDP) was between 0.5 and 1% in 2005 according to Carrero and Andrade [48], but historical economic trends suggest that this proportion is likely to have considerably reduced in recent years.

In recent years, with the enactment of the new *Forests and Forest Management Law* in 2008, later revised in 2013, a policy shift began with regards to how forest management should be planned and applied in Venezuela. Perhaps, the most novel aspect of this process was the creation in 2010 of a public government-based forest company (*Empresa Nacional Forestal* – National Forest Company). Broadly, the general objective is to promote the "*…sustainable production of forest goods and services, through the planning and management of the forest heritage (…) aimed at promoting the direct participation of local communities and other organizations…*" [49]. In practice, along with government agencies, this company currently oversees the guidelines for developing new forest management plans while slowly substituted the model of private concessions previously in place. At present, the company supervises the management for all production forests in the country and has an active management operation in the Imataca Forest Reserve in Eastern Venezuela. In addition, with the support of FAO, a 5-year project started in 2016 aiming at connecting multiple components (e.g. Reduced Impact Logging, ecological restoration, silviculture, research) to support the development of sustainable forest management guidelines at national scale.[1] The impacts of this initiative are yet to be assessed.

4.2 Historical perspective of silvicultural practices in Venezuela's production forests

With the beginning of major development projects in the second half of the twentieth century, it became clear that a significant area of forests in the country represented, on one hand, environments of high ecological value that had to be preserved and that is how the main foundations of the national system of protected areas (ABRAE) were laid. On the other, some of these areas also showed characteristics that made them important resources for the development of local and regional economies across the country and a significant component for a forest-based productive sector (**Figure 3**).[2]

As in many tropical countries, the beginnings of forestry in Venezuela were influenced by practices extrapolated mostly from experiences applied in European temperate forests [20, 50, 51], especially on topics related to methods to promote the regeneration of the harvested forest. In its early days, the forestry practice in Venezuela was grounded on the conceptual premise of the so-called "experimental management" through two variations upon different intensities in the prescription of silvicultural treatments [52]. This concept of experimental management was based on the need to manage production forests, even under conditions of lack of enough sound scientific information being available. Thus, conducting forest management as an "experiment" implied the existence of a set of guidelines in which there is a research program in place to test various silvicultural alternatives while management is simultaneously applied on a commercial scale [53].

In practice, this approach was later defined as a combination between active and passive management approaches [24, 54], and was influential during the early days of forest policy and management in the Forest Reserves of the Western Plains region (i.e. Ticoporo, Caparo, San Camilo – **Figure 3**). From the theoretical stand point, passive management meant that forests were managed via seemingly very low intensity treatments, on the assumption of natural and spontaneous production without silvicultural treatments, while timber harvest was properly regulated in intensity and environmental impacts [52, 54]. Generally speaking, this approach and its guidelines fit well with sustainable forest management practices that have been promoted based on reduced impact logging (RIL) in many tropical regions [45, 55–57]. However, as has been widely documented (e.g. [23, 24, 43, 44, 58]), its implementation was poorly executed often with negative environmental effects [45, 59, 60]. Secondly, the active management approach was characterized for concentrating intensive practices over relatively small areas. The main objective was the "improvement" of the forest composition mostly through directing the intervention towards species with high commercial value and with low natural abundance or that were completely absent in the forest stand. These practices ranged from relatively low-intensity practices (e.g. assisted natural

[1] http://www.fao.org/venezuela/programas-y-proyectos/lista-de-proyectos/es/

[2] http://sigot.geoportalsb.gob.ve/abrae_web/index.php

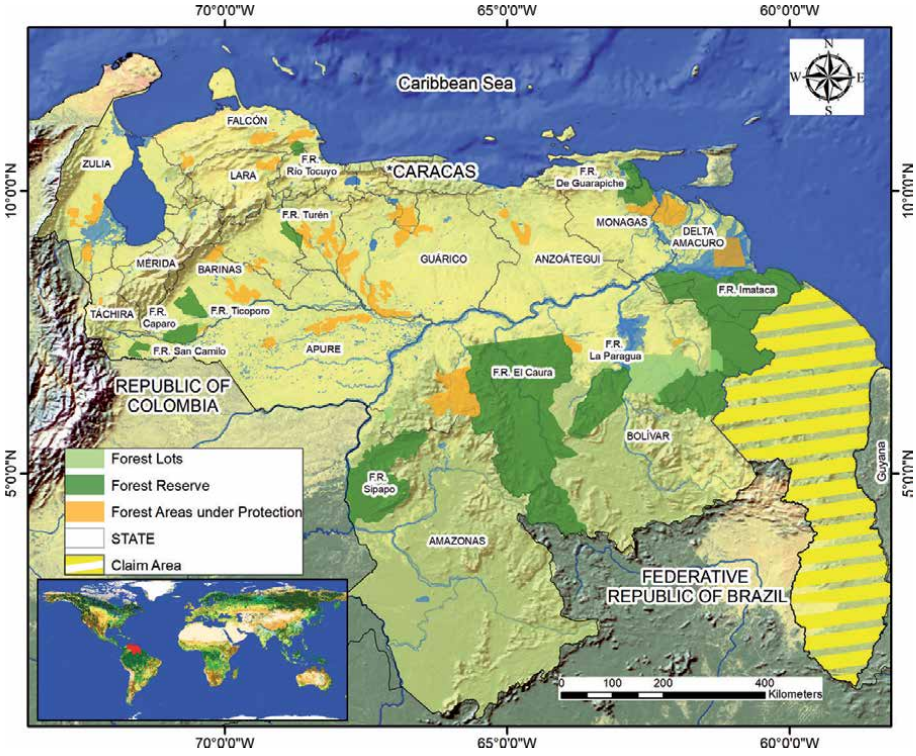

Figure 3.
General distribution of natural production forests in Venezuela. There are legal differences in terms of how forest reserves and lots are administered, but along with the group of 39 Forest areas under protection these are all public-managed areas. In 2018, El Caura *Forest Reserve in the Guiana Shield region, with more than 5 million ha officially became the 44th National Park. Therefore, the total area shown in* ***Table 2*** *excludes this reserve. Map elaborated by Carlos Pacheco based on publicly available data of Venezuela's protected areas.*

regeneration) to more intensive options including a total forest conversion of the selectively logged forest to fast-growing plantations mostly with exotic species [23, 52, 54].

Although no specific rules were set to determine how each approach should be spatially applied, Vincent [61] proposed that the selection and designation of the active management areas should be carried out in each annual compartment or, preferably, by sets of compartments only in the "best" sites (i.e. high productivity, mostly flat with relatively good drained soils). For instance, in the units of the Western Plains where this approach was implemented, active management was executed in close to 10% of the total managed area. The rest of the annual compartment, typically in areas on average of about 2000 ha in 25–30 years cutting cycles, was managed under the passive approach using selective logging upon a group of previously set minimum harvesting diameters (MHDs) for commercial timber species, and the marking and mapping of future commercial trees including those designated for seed production.

As has been documented in different analysis focusing on the tropics (e.g. [21, 22]), silvicultural practices applied in the management of Venezuela's production forests had the fundamental objective of solving the "problem" of a relatively low regeneration of many commercial species and, when possible, to secure sustained volumes of timber within cutting cycles typically of 25–30 years, although 40-year cycles were common in management plans for less productive forests of the Guiana Shield. Many of these practices have been strongly criticized [23, 24, 44, 62], because they frequently ignored the importance of pre-harvest planning operations and the complex ecology and dynamics of tropical forests. Thus, these practices generally neither increased the productivity of the system nor contributed to its sustainability [24].

Within the Forest Management Research Program that began in 1970 in the Caparo Forest Reserve in the Western Plains region, a great deal of effort was put into the understanding of the basic ecology of many forest species with timber value. Looking for effective and efficient systems, not only basic information on the ecology of the species was collected (i.e. phenology, reproduction, dispersal strategies, growth), but also a large number of these potential species was part of multiple experiments in which different planting and management conditions were tested, including open-field trials and enrichment planting in strips with different variations [63]. This is probably one of the most successful aspects of the Research Program, that is producing a baseline of applicable information on the ecology and management of several species in the Western Plains. The use of teak (*Tectona grandis*), an exotic species deserves a special mention as this species optimally acclimatized and resulted in highly productive stands specially in well-drained sites [24]. Teak plantations were often established in open field conditions in deforested areas, but several logging companies used teak after full conversion of logged forests as part of their "active" management plans [23].

Since 1970, and fundamentally since the creation of the Graduate Center for Forest and Environmental Studies (CEFAP) in 1968, a large cumulative experience in tropical silviculture exist, but with a limited application at the operational and commercial scales. Much of this experience is sustained on a research program that included multiple silvicultural trials and experiments that, for reasons analyzed later in this chapter, were not fully applied on a larger scale. As documented by Putz and Ruslandi [56] for the tropical region, between plantation conversion and single-tree selection using RIL, there is a wide variety of silvicultural interventions that tropical foresters can apply but these are rarely used outside of experimental plots. Silvicultural methods such as shelterwoods, group selection and others all have received considerable attention from research, but with a few exceptions, most have not been formally adopted at industrial or commercial scales. In the next sections, some of the most prominent silvicultural practices are described.

4.3 Common silvicultural systems and practices

An extensive literature review was conducted to compile and characterize the most common silvicultural systems used in Venezuela's production forests (**Table 2**). This process included peer-reviewed studies, but most of all was based on the analysis of numerous official reports from the national forest agencies, a review of different management plans from private companies and the results from surveys distributed widely among different stakeholders linked to forest management in Venezuela. The reader will find that the topic of plantation forestry has been purposely ignored (but see [24] for more information), while others such as the use of non-timber forest products (NTFP) and its management lacked of sufficient information to offer a comprehensive analysis. Nevertheless, the following section is by far the most updated review of the history and characteristics of silvicultural practices applied in natural production forests in Venezuela.

4.3.1 Minimum harvest diameters (MHDs)

Probably the oldest and most widespread management system applied in the tropics. After a pre-harvest inventory, a minimum harvest diameter is established to determine mature commercial trees and is the basis of the polycyclic management, a selection approach where, in theory, the objective is to control overexploitation of the forests by harvesting a relatively low number of commercial trees [64]. It is essentially a system based on natural forest production where the only direct intervention is selective harvesting repeated within moderately short cutting cycles [65]. It is a highly selective

Silvicultural system[a]	Reasons for its implementation	Region or area of the country where it is applied or may be applicable and main species used
Direct transformation (clearcutting): system that involves the complete replacement at once of the logged uneven-aged forest by another regular and homogeneous stand, established by even-aged plantation and usually with fast growing species.	• Forests poor in abundance of commercially valuable species (young or mature; logged or unlogged); • As a market need, to produce wood for homogeneous products; • Hardly applied today for environmental reasons.	• Used to be part of the "Active management" scheme applied in the Western Plains by using species such as *Tectona grandis, Gmelina arborea,* and other exotic species; • Applicable in the context of fast-growing plantations.
Enrichment in transversal strips: system of indirect transformation by introducing artificial regeneration *via* strip planting after selective logging mostly to increase commercial stocking of stands. Aims at maintaining the uneven-aged condition of the forest (**Figure 5**).	• Young or mature forests poor in abundance of commercial species; • Rich or relatively rich forests in which commercial species have limited natural regeneration; • Introduction of one or more species at special demand of an ecological, industrial or market nature;	• **Western Plains**: A local version of this system known as "Método Caimital" was developed with positive results. Species used: *Bombacopsis quinata, Cordia apurensis, Handroanthus rosea, Swietenia macrophylla, Cedrela odorata*; • **Guiana Shield**: Mureillo *Erisma uncinatum, Carapa guianensis, Tabebuia serratifolia*; • **Orinoco Delta:** *Euterpe oleracea*;
Modified selection thinning: system that seeks to transform a stand with an irregular structure and heterogeneous composition (e.g. 40–70 species/ha) to a more regular and less diverse stand (e.g. 20–30 species/ha).	• Suitable for the permanent treatment of mature natural forests where most of the species are shade tolerant with good natural regeneration; • System suitable for forests located on land with moderate to strong slopes prone to erosion that also meet the above;	• Applicable to irregular and heterogeneous forests. A large area of forests in the country meet these conditions, but no information is known about its practical application;
Strip clearcuttings: regeneration system applied to natural tropical forests to transform their heterogeneous structure to a more regular and less diverse structure.	• It has been suggested for primary or secondary forests (young or mature) where commercial species are predominately shade-intolerant species with abundant low-weight seeds; • It could be used for relatively regular and homogeneous forests where natural regeneration is high (e.g. Mangroves);	• In the early 1970s this system was applied in a management plan for flooded forests of the Orinoco Delta (Guarapiche Forest Reserve – **Figure 3**) dominated mainly by *Rhizophora mangle.*
Enhanced natural regeneration in strips: indirect transformation system to promote the establishment of the natural regeneration of valuable species (usually scarce) in previously open strips.	• Mature forests relatively rich in commercial species with limited natural regeneration to ensure long-term production;	• Applied at research scale in Caparo Forest Reserve ("Metodo Limba-Caparo) in the Western Plains with promising results for two important commercial species: *Cedrela odorata* and *Handroanthus rosea.*

Silvicultural system[a]	Reasons for its implementation	Region or area of the country where it is applied or may be applicable and main species used
Minimum Harvest Diameter (MHDs): the most widely applied system and is based only in natural forest production and is considered a "passive" approach (see Section 4.3.1 for further details).	• Its application is justified on the basis of the maintenance of enough crop trees to ensure a sustainable harvest under polycyclic schemes; • Typically accompanied by intermediate treatments to reduce competition on crop tress.	• Widely applied in the forest management of the Western Plains and today in the Guiana Shield. • If logging is the unique treatment, it is often not considered a formal silvicultural system.

Table adapted from [65] with inputs from [64, 66].
[a]This grouping of silvicultural systems is presented based on practical experiences, results of applied research or based on the analysis of the ecological conditions of forests that would make a particular system applicable. In all cases, wood production is the overall major objective.

Table 2.
Silvicultural systems applied or potentially applicable in Venezuela's production forests.

management system in terms of the spectrum of commercial species and the relatively low number of trees logged that is common in many tropical managed forests (1–20 trees per ha -[67]). Under these conditions, it was expected that with a proper planning with minimum standards for cutting and transportation activities, the impacts on the forest stand would be low and facilitate a consistent flow of timber in the next cutting cycles [20, 51, 64]. This system was the fundamental basis of the first forest harvesting permits granted in Venezuela more than 40 years ago [23], and remained relatively unaltered even at times when research evidences made clear that major modifications to this approach were urgent in support of long-term sustainable management [44, 68–70].

Most frequently, these MHDs are set by national authorities, and depending on the species groups and their commercial value, values are typically within the range of 30 up to 70 cm or more in diameter at breast height (DBH) [64]. However, these limits are mostly set to accommodate processing technologies and market demands, rather than the biology and persistence of the harvested species, limiting the possibility to provide ecologically sustainable forest management [71]. Remarkably high numbers of species that are common in many tropical differ in growth requirements, growth rates, marketability, ecological roles and other relevant traits. Thus, simple silvicultural guidelines such as fixed MHDs or cutting cycles are unlikely to be satisfactory [13]. In Venezuela's case, the lack of sound ecological information on growth patterns of commercial species, density and structure, along with limited long-term information has been highlighted as a major limitation [23, 24, 69, 72]. A relatively new official regulation related to MHDs enacted in 2009 [73] aimed at solving, at least partly, the lack of data on species growth by expanding the information on the number of species with MHDs. However, if no improvement is made to the overall process of forest planning, including the urgent implementation of reduced impact logging, and a rethinking of cutting cycles the negative perception towards timber harvest is likely to persist (**Figure 4**).

4.3.2 Post-logging silvicultural treatments

In many tropical managed forests, a group of standard practices are often applied after selective logging to reduce competition for future harvestable trees. These intermediate treatments might include *refining*, that is, the elimination of undesirable tree species or sick or damaged material, to the extent only that the

Figure 4.
General conditions of current logging practices in Venezuela's production forests in the Guiana shield. Unplanned road systems are major drivers of forest degradation. By using a RIL approach, the extension and size of logging roads is considerably lower compared to conventional harvesting which ultimately reduces the environmental impacts [55]. Photo: Emilio Vilanova.

(a) (b)

Figure 5.
Two examples of enrichment planting in strips in the Venezuelan Guiana shield. In less than 10% of the logged stands, strips of 5–6 m in width separated by 40–50 m are opened via clearcutting to establish artificial regeneration of commercial species. In these two examples, the main species planted is Carapa guianensis *(Meliaceae). Photos: Emilio Vilanova.*

structural stability of the stand is not weakened [64]. Also, *liberation* or the favoring of all valuable individuals (juveniles, candidates) through the elimination of competitors can be part of the silvicultural prescription. In practice, these activities can comprise cutting of vines and/or climbers – even before logging as a step for RIL – and the elimination of undesirable trees that might delay the establishment of regeneration of commercial species after harvest [66]. It is expected that with a periodical repetition and adequate monitoring, these practices would serve as complementary practices within the MHDs system. This approach has been known in Venezuela as *Management of the remnant stand* and was used mostly to improve growing conditions of the advanced regeneration of commercial species with diameters above 15 cm [74]. In some cases, to reduce multiple re-entries, girdling and occasionally poisoning of undesired trees with an arboricide were used [52]. However, although there are positive examples with regards to improving stand growth [e.g. 22], in the few cases where information was available from management plans in Venezuela, girdling often did not helped killing all selected trees and the use of chemicals in highly sensitive ecosystems was later discouraged and ultimately banned by government agencies overseeing these operations.

4.3.3 Natural and artificial regeneration enhancement systems

Results from early assessments that were carried out as part of the Forest Management Research Program of the Western Plains, and to some extent in the Guiana Shield region as well, led to the conclusion that in order to secure a sustained production of timber over multiple cutting cycles, natural regeneration of commercial species was limited [52]. It was suggested that this was a direct response of the predominately shade-intolerant condition of most commercial species, its reproductive biology and other limiting factors such as dispersion and germination in both undisturbed and logged stands, which drove the plan for practices aiming at the enhancement or improvement of regeneration [63]. Consequently, several alternatives were tested, from "simple" interventions such as the creation of small gaps or canopy openings to promote rapid colonization of natural regeneration of commercial species, to more intensive practices such as strip clearcutting, prescribed burning, and planting (see **Table 2**). For example, a modified version of the well-known uneven-aged system Shelterwood [56, 66] was applied in an experimental setting in the Western Plains and was described as a "promising" system [23, 52, 70] but the lack of an adequate financial analysis prevented this system for its potential application at the management scale [70].

Another variant of enrichment planting where the intensity of intervention was higher was the method known as "Limba-Caparo Method" [52, 63] and was considered as one of the few successful experiences for this type of system [70]. It is a type of plantation in strips where, once the commercial species are extracted via selective logging, often highly abundant palms, lianas and other minor competing vegetation with a diameter below 10 cm are removed to later facilitate the use of prescribed burning to facilitate establishment of natural regeneration. A synthesis of enrichment planting experiments indicates that this method was successful in promoting regeneration of an important local species (i.e. *Handroanthus rosea*) with rapid growth, high survival and adaptability to various environmental conditions, particularly in some flooded zones where growth of other valuable species is limited [63]. While this system was labeled as promising, the complexity behind the initial treatments, along with concerns for the use of fire in semi-dry forests and the potential impacts on biodiversity were major limitations for the application at larger scales.

Although several variants of this system were tested, it was the simpler versions of enrichment practices with artificial regeneration the ones that were predominately adopted at industrial scales. In fact, these ultimately became a requirement for the approval of many forest management plans in the country's production forests. It was considered a relatively simple approach yet with high environmental impacts [75] especially in highly sensitive ecosystems such as the Guiana Shield. Between 1987 and 2010 a total of 41,460 ha of enrichment strips were planted in Venezuela's production forests [76]. Furthermore, poor adaptability of many of the species used and lack of a solid monitoring plan led to very low growth rates and low survival [62] (**Table 3**). From the commercial standpoint, one of the few economic analyses conducted in Venezuela about the

Species	Survival (%)	Height growth (m/year)	Diameter growth (cm/year)
Jacaranda copaia	40.0	1.1	1.36
Parkia nitida	79.4	0.8	1.23
Loxopterygium sagotii	66.2	0.9	1.16
Ceiba pentandra	42.2	0.5	1.13
Simarouba amara	75.6	1.1	1.07
Spondias mombin	97.5	1.0	0.98
Cordia spp.	62.3	0.6	0.92
Terminalia amazonia	64.8	0.5	0.89
Swietenia macrophylla	54.7	0.5	0.84
Enterolobium cyclocarpum	26.0	0.4	0.82
Cedrela odorata	62.2	0.6	0.75
Handroanthus rosea	29.0	0.4	0.69
Gmelina arborea	100.0	1.0	0.68
Pera glabrata	42.6	0.8	0.65
Carapa guianensis	74.3	0.6	0.64
Caesalpinia coriaria	77.1	0.3	0.62
Handroanthus serratifolia	83.3	0.5	0.62
Anacardium giganteum	93.5	0.8	0.62
Hymenaea courbaril	84.6	0.6	0.58
Erisma uncinatum	47.2	0.5	0.54
Cattostema comune	43.0	0.5	0.48
Manilkara bidentata	69.4	0.4	0.48
Diplotropis purpurea	29.2	0.3	—
Mouriri huberi	12.5	0.1	—
Anacardium excelsum	33.3	0.2	—
Platymiscium pinnatum	—	0.8	—
Samanea saman	41.1	0.3	—
Tectona grandis	25.0	0.1	—
Peltogyne porphyrocardia	63.1	0.4	—
Mean	**66.7**	**0.60**	**0.80**

Table adapted from [13].

Table 3.
Results from multiple assessments of enrichment planting practices applied in managed forests of the Venezuelan Guiana shield region.

use of enrichment planting indicate that, even in scenarios of relatively high prices for timber, the net benefit–cost ratio (BCR) was very low, amidst the initial high costs of establishment and long cutting cycles needed to obtain reasonable wood volumes [74]. For many of these reasons, this practice has been officially abandoned, at least as a requirement for the approval of management plans since 2010. Nevertheless, as has been documented in other tropical regions, in recent years there has been a reawakening interest for the use of enrichment planting in a context of restoration of degraded and secondary forests [13, 63, 77] where this approach seems more feasible.

5. Overcoming the barriers and limitations of silviculture

In this chapter we have tried to synthetize some of the most important aspects of the silvicultural practices applied in Venezuela's natural production forests. Details on past and current practices and their impacts were offered with the idea of facilitating a much needed discussion about the compatibility of silviculture for enhanced timber production with the maintenance of other ecosystem services offered by tropical managed forests [18, 22]. In doing so, we made a thorough review of the available literature, most of which came in the form of gray literature produced by government agencies, academic institutions, forest companies and other sources. This last section aims at identifying relevant lessons learned over the course of 40+ years including a set of major recommendations to improve how silviculture could be applied in natural tropical forests in Venezuela.

5.1 The role of scientific research

The main product of more than four decades of silvicultural experience in Venezuela is perhaps the existence of an enormous amount of information available on the main silvicultural practices applied or with potential to be applied to natural forests in the country. Most, if not all, this information came from a pioneer effort starting during the early 1970s in support of the idea that natural tropical forests can be sustainably managed and reduce the risk of deforestation. Important elements on the basic ecology of commercial tree species and how these could be managed occupied a major proportion of this process. Yet, the capacity to fully influence forest management at industrial scales was limited.

Adequate communication has proven to be an urgent capacity that many scientists are acquiring as a matter to (successfully) transmit sound scientific information to the public and decision makers [78]. Despite this, a weak connection between science, policy and decision-making has been cited multiple times as a major limitation for sustainable management of tropical forests [13]. In terms of silviculture for instance, a limited adoption of some of the recommended systems at commercial scales can be attributed to some extent to failures of researchers to appropriately design their studies, or because some of these interventions are cost prohibitive and these implications are not properly considered during research, which ultimately reflects a failure to communicate their results effectively [13]. This requires an effort to invest resources in training and capacity development into novel approaches to further improve how scientists disseminate the results of research. This is particularly relevant in the context of the current complex political and economic crisis in Venezuela where research institutions are being disproportionally affected [79].

Another element linked to how silvicultural research is conducted has to do with the need to adapt to the recent shifts in the conditions and requirements for sustainable use of tropical forests. While timber production can - and should -continue being an objective in the management of production forests, the contribution of other elements such as non-timber forest products or ecosystems services including biodiversity conservation and mitigation of climate change must be part of any research agenda for a twenty-first century silviculture. Furthermore, research on topics beyond logging practices is needed to assure both sustainability and that these other values are not underestimated and unnecessarily compromised where timber production is the principal goal of management [13].

5.2 A need for better monitoring

Appropriate technical procedures for monitoring the process of forest management are critical to decision-making. While this goes beyond silvicultural practices, many of the treatments applied in Venezuela's production forests were insufficiently monitored or never monitored at all. The implications of this might be two-fold. On one hand, the absence of standard guidelines for monitoring could have severely limited the real potential for some of the most promising practices (e.g. "Caimital" enrichment system) or, on the other, could have helped in abandoning more quickly those that clearly showed negative results (e.g. forest conversion of logged forests). For a country with a silvicultural research program that started more than four decades ago, and with one of the oldest forestry schools in the tropics, it is remarkable that not a single formal process of forest monitoring has been part of the national forest policy. Examples such as the guidelines developed by the International Tropical Timber Organization (ITTO) [80] have been largely underestimated and can serve as a guide for the adoption of criteria and indicators for monitoring forest management including silvicultural practices.

5.3 Amplifying the objectives of silviculture

The need for a different perspective in silvicultural practices was previously addressed in item 5.1 when analyzing the role of scientific research. However, from the standpoint of policy and decision making, it is important to reinforce the idea that if we want to preserve the vast amount of production forests still available, a new vision of silviculture should be adopted. In many tropical regions, logged forests often retain substantial biodiversity, carbon and timber stocks [18]. Thus, increasing the overall value of production forests in the tropics compared to other more intensive land-uses often highly profitable and linked to deforestation [81] is not only a matter of applying reduced impact logging practices – although urgently needed in Venezuela [45]. It also requires major modifications to integrate the great diversity of products that can be obtained from these ecosystems. In this process, at least in the short and medium terms, updating forest education curriculum programs at different levels and directly connected to applied forestry practices can contribute to the formulation of new strategies for diversifying forest management [82]. Assessments of non-timber forest products to design silvicultural practices [83], use of silviculture for ecological restoration [64], or simply improving harvesting practices as a tool to mitigate climate change and conserve biological diversity [84] are all important steps towards a more inclusive practice of forest management.

6. Final remarks and conclusions

Venezuela has now more than 40 years of experience in the development of forestry practices for the country's forests, plantations and other forest lands, a long-term valued effort for the forestry sector in the tropical region and in Latin

America. The development of a conceptual and practical model for the management of the country's natural forests and the establishment of a significant area of forest plantations are all remarkable actions. However, available information reviewed here clearly indicates that the objectives originally set out primarily to increase forest productivity were not achieved. In addition, the negative environmental effects of timber harvest have been significant, and the overall role of management as has been applied up to now in the country's production forests should be questioned. While multiple drivers interacted, the total loss of most production forests in the Western Plains, the reduction in recent years of the area with formal management plans, and the limited participation of local communities in the practice and benefits of forest management, among others, indicate the urgent need to reformulate how Venezuelan forests have been managed.

While further analysis is required for additional technical details about the characteristics of the silvicultural practices applied in Venezuela, the main goal of this chapter was, first of all, to provide an historical perspective for one of the countries with the richest, yet largely unknown, silvicultural experience in the tropical region. Secondly, understanding the historical reasons that led to the design and implementation of the different forest management strategies in much of the country's forests, helps identifying the benefits, advantages and the limitations to improve the practice of silviculture in Venezuela's natural production forests. Despite the loss of some of the most productive ecosystems in the Western Plains, the existence of an important resource base for forest production, especially in the Guiana Shield region from which a large proportion of rural populations depends, represents a great opportunity for improving forest management based on principles of multiple use of forests.

Finally, the institutional and political changes that started in the early 2000s have undoubtedly impacted how forests resources have been managed in the last 20 years. From the concession model in the late 1990s, dominated by private companies often poorly managed with a very low degree of compliance to sustainable management guidelines, management of natural production forests slowly shifted towards a heavily government-dominated system. This transition included a general revision of how silviculture should be applied but the expected outcomes for a new and more sustainable model are far from being clear. Furthermore, the ongoing political and socioeconomic crisis in the country is putting at risk the long-term stability of many natural production forests. We firmly believe that these changes, if widely discussed and agreed upon by all actors involved in forest management, can facilitate the adoption of better practices and thus increase the strategic value of forests as tools for the sustainable development of the country.

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Conflict of interest

The author declares no conflict of interest.

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References

[1] FAO. The State of the World's Forests. Rome: FAO; 2018. p. 118

[2] Beer C, Reichstein M, Tomelleri E, Ciais P, Jung M, Carvalhais N, et al. Terrestrial gross carbon dioxide uptake: Global distribution and covariation with climate. Science. 2010;**329**(5993):834-838

[3] Mitchard ETA. The tropical forest carbon cycle and climate change. Nature. 2018;**559**(7715):527-534. DOI: 10.1038/s41586-018-0300-2

[4] Le QC, Andrew RM, Canadell JG, Sitch S, Korsbakken JI, Peters GP, et al. Global carbon budget. Earth System Science Data. 2016;**2016**:605-649

[5] FAO-UNEP. The State of the World's Forests 2020 [Internet]. Rome: Food and Agriculture Organization of the United Nations; 2020. p. 139. Available from: http://www.fao.org/documents/card/en/c/ca8642en

[6] Hansen MC, Potapov PV, Moore R, Hancher M, Turubanova SA, Tyukavina A, et al. High-resolution global maps of 21st-century forest cover change. Science. 2013;**342**(6160):850-853

[7] Baccini A, Walker W, Carvalho L, Farina M, Houghton RA. Tropical forests are a net carbon source based on aboveground measurements of gain and loss. Science. 2017;**358**(6360):230-234

[8] Blaser J, Sarre A, Poore D, Johnson S. Status of Tropical Forest Management 2011. Yokohama, Japan: ITTO Technical Series; 2011

[9] Petrokofsky G, Sist P, Blanc L, Doucet JL, Finegan B, Gourlet-Fleury S, et al. Comparative effectiveness of silvicultural interventions for increasing timber production and sustaining conservation values in natural tropical production forests. A systematic review protocol. Environmental Evidence. 2015;**4**(1):1-7

[10] Sist P, Rutishauser E, Peña-Claros M, Shenkin A, Hérault B, Blanc L, et al. The tropical managed forests observatory: A research network addressing the future of tropical logged forests. Applied Vegetation Science. 2015;**18**(1):171-174

[11] Piponiot C, Rutishauser E, Derroire G, Putz FE, Sist P, West TAP, et al. Optimal strategies for ecosystem services provision in Amazonian production forests. Environmental Research Letters. 2019;**14**(6):64014. DOI: 10.1088/1748-9326/ab195e

[12] Piponiot C, Rödig E, Putz FE, Rutishauser E, Sist P, Ascarrunz N, et al. Can timber provision from Amazonian natural forests be sustainable? Environmental Research Letters. 2019;**14**(12):124090. DOI: 10.1088/1748-9326/ab5eb1

[13] Putz FE, Romero C. Futures of tropical production forests. In: Futures of Tropical Production Forests. Occasional Paper 143. Bogor, Indonesia: Center for International Forestry Research (CIFOR); 2015. p. 51. DOI: 10.17528/cifor/005766

[14] Hérault B, Piponiot C. Key drivers of ecosystem recovery after disturbance in a neotropical forest: Long-term lessons from the Paracou experiment, French Guiana. Forest Ecosystems. 2018;**5**(1):1-15

[15] Rutishauser E, Hérault B, Baraloto C, Blanc L, Descroix L, Sotta ED, et al. Rapid tree carbon stock recovery in managed Amazonian forests. Current Biology. 2015;**25**(18): R787-R788. DOI: 10.1016/j.cub.2015.07.034

[16] Puettmann KJ, Coates KD, Messier C. A Critique of Silviculture: Managing for Complexity. Washington D.C: Island Press; 2009. p. 207

[17] Fahey RT, Alveshere BC, Burton JI, D'Amato AW, Dickinson YL, Keeton WS, et al. Shifting conceptions of complexity in forest management and silviculture. Forest Ecology and Management. 2018;**421**(January):59-71

[18] Putz FE, Zuidema PA, Synnott T, Peña-Claros M, Pinard MA, Sheil D, et al. Sustaining conservation values in selectively logged tropical forests: The attained and the attainable. Conservation Letters. 2012;**5**(4):296-303

[19] Yguel B, Piponiot C, Mirabel A, Dourdain A, Hérault B, Gourlet-Fleury S, et al. Beyond species richness and biomass: Impact of selective logging and silvicultural treatments on the functional composition of a neotropical forest. Forest Ecology and Management. 2019;**433**(November 2018):528-534. Available from: https://linkinghub.elsevier.com/retrieve/pii/S0378112718314944

[20] Dawkins H, Phillip M. Tropical Moist Forest Silviculture and Management: A History of Successes and Failures. Cambridge, UK: University Press; 1998. p. 351

[21] Weber M. Review: New aspects in tropical Silviculture. In: Gunter S, Weber M, Stimm B, Mosandl R, editors. Silviculture in the Tropics. Berlin Heidelberg: Springer-Verlag; 2011. pp. 63-89

[22] Finegan B. 21st century viewpoint on tropical Silviculture. In: Pancel L, Kohl M, editors. Tropical Forestry Handbook. 2nd ed. Berlin: Springer-Verlag Berlin Heidelberg; 2016. pp. 1605-1638

[23] Kammesheidt L, Lezama AT, Franco W, Plonczak M. History of logging and silvicultural treatments in the western Venezuelan plain forests and the prospect for sustainable forest management. Forest Ecology and Management. 2001;**148**(1-3):1-20

[24] Torres-lezama A, Ramírez-Angulo H, Vilanova E, Rodrigo B. Forest resources in Venezuela: Current status and prospects for sustainable management. Bois et Forets des Tropiques. 2008;**295**(295):21-34

[25] Pacheco C, Aguado I, Mollicone D. Las causas de la deforestacion en venezuela: un estudio restrospectivo. BioLlania. 2011;**5110**(10):281-292

[26] FAO. State of the World's Forests 2016. Forests and agriculture: land-use challenges and opportunities [Internet]. Vol. 45, State of the World'S Forests Forests and Agriculture: Land-Use Challenges and Opportunities. 2016. pp. 1-107. Available from: http://ccafs.cgiar.org/news/press-releases/agriculture-and-food-production-contribute-29-percent-global-greenhouse-gas

[27] Pacheco CE, Aguado MI, Mollicone D. Identification and characterization of deforestation hot spots in Venezuela using MODIS satellite images. Acta Amazonica. 2014;**44**(2):185-196. Available from: http://www.scielo.br/scielo.php?script=sci_arttext&pid=S0044-59672014000200004&lng=en&tlng=en

[28] Bevilacqua M, Cárdenas L, Flores AL, Hernández L, Lares E, Mansutti A, et al. The State of Venezuela's Forests. A Case Study of the Guayana Region. A world Resources Institute Report. Caracas, Venezuela: Fundación Polar; 2002. p. 132. Available from: http://pdf.wri.org/gfw_venezuela.pdf

[29] Food and Agriculture Organization of the United Nations, FAO. Global Forest resources assessment 2015.

Desk reference. Desk Reference. 2015;**2005**:244

[30] Pacheco-angulo C, Vilanova E, Aguado I, Monjardin S. Carbon emissions from deforestation and degradation in a forest reserve in Venezuela between 1990 and 2015. Forests. 2017;**8**:291. DOI: 10.3390/f8080291

[31] Aide TM, Grau HR, Graesser J, Andrade-Nuñez MJ, Aráoz E, Barros AP, et al. Woody vegetation dynamics in the tropical and subtropical Andes from 2001 to 2014: Satellite image interpretation and expert validation. Global Change Biology. 2019;**25**(6):2112-2126. DOI: 10.1111/gcb.14618

[32] Harris NL, Brown S, Hagen SC, Saatchi SS, Petrova S, Salas W, et al. Baseline map of carbon emissions from deforestation in tropical regions. Science. 2012;**336**(6088):1573-1576

[33] Pearson TRH, Brown S, Murray L, Sidman G. Greenhouse gas emissions from tropical forest degradation: An underestimated source. Carbon Balance and Management. 2017;**12**(1):3. DOI: 10.1186/s13021-017-0072-2

[34] Waliser DE, Jiang X. In: North GR, Pyle J, editors. Tropical Meteorology and Climate | Intertropical Convergence Zone. Oxford: Academic Press; 2015. pp. 121-131. Available from: http://www.sciencedirect.com/science/article/pii/B9780123822253004175

[35] Huber O, Oliveira-Miranda M. Ambientes terrestres de Venezuela. In: Rodríguez J, Rojas-Suárez F, Hernández D, editors. Libro Rojo de los Ecosistemas Terrestres de Venezuela. Caracas: PROVITA, Shell Venezuela, Lenovo Venezuela; 2010. pp. 29-89

[36] Arismendi J. Presentación geográfica de las formas de relieve. In: Fundación Empresas Polar. Caracas, Venezuela: GeoVenezuela, Tomo 2; 2007. pp. 128-183

[37] Baruch Z. Vegetation–environment relationships and classification of the seasonal savannas in Venezuela. Flora - Morphology Distribution Functional Ecology of Plants. 2005;**200**(1):49-64. Available from: http://www.sciencedirect.com/science/article/pii/S036725300500006X

[38] Dourojeanni M. El futuro de los bosques de América Latina. In: Keipi K, editor. Políticas forestales en América Latina. Washington D.C.: Banco Interamericano de Desarrollo (BID); 2000. pp. 89-104

[39] Kammesheidt L. Some autecological characteristics of early to late successional tree species in Venezuela. Acta Oecologica. 2000;**21**(1):37-48. Available from: https://www.sciencedirect.com/science/article/pii/S1146609X00001089

[40] Centeno J. Estrategia para el desarrollo forestal en Venezuela. Caracas, Venezuela: Fondo Nacional de Investigación Forestal; 1995. p. 83

[41] del Ambiente M. Anuario de Estadísticas Forestales 2007. Caracas, Venezuela: Ministerio del Ambiente-Venezuela; 2008

[42] MINEC. Anuario Estadísticas Forestales 2017. Caracas, Venezuela: Ministerio del Poder Popular para el Ambiente y el Ecosocialismo-Venezuela; 2018

[43] Aicher C. Los efectos del conocimiento forestal en la política forestal venezolana. In: SEFUT Working Paper No. 14. Freiburg, Germany: Albert-Ludwigs-Universität Freiburg; 2005. p. 39. Available from: https://freidok.uni-freiburg.de/fedora/objects/freidok:1747/datastreams/FILE1/content

[44] Lozada JR. Situación actual y perspectivas del manejo de recursos

forestales en Venezuela. Revista Forestal Venezolana. 2007;**51**(2):195-218

[45] Vilanova E, Ramírez-Angulo H, Ramírez G, Torres-Lezama A. Compliance with sustainable forest management guidelines in three timber concessions in the Venezuelan Guayana: Analysis and implications. Forest Policy and Economics. 2012;**17**:3-12

[46] Food and Agriculture Organization of the United Nations. Evaluación de los Recursos Forestales Mundiales 2015 - Informe Nacional Venezuela [Internet]. 2014. Available from: http://www.fao.org/3/a-az372s.pdf

[47] Rojas-López J. Regulación ambiental y colonización agraria en reservas de bosque. El drama de Ticoporo, estado Barinas-Venezuela. Revista Geográfica Venezolana. 2007;**48**(1):129-141

[48] Carrero GO, Andrade V. La contribución de las actividades del sector primario y secundario de la cadena forestal al PIB de Venezuela en los últimos 50 años y su relación con algunas variables macroeconómicas. Revista Forestal Venezolana. 2005;**49**(1):39-47

[49] de Venezuela RB. Ley de Bosques. Caracas, Venezuela: Asamblea Nacional; 2013. Gaceta Oficial No. 403.780

[50] Dawkins H. The Management of Natural Tropical High-Forest with Special Reference to Uganda. Oxford, UK: Imperial Forestry Institute, University of Oxford, Oxford, UK; 1958

[51] Lamprecht H. Silvicultura en los trópicos: Los ecosistemas forestales en los bosques tropicales y sus especies arbóreas -posibilidades y métodos para un aprovechamiento sostenido. Germany: Eschborn; 1990

[52] Vincent L, Rodriguez-Poveda L, Noguera O, Arends E, Lozada J. Evolución histórica, y desarrollos recientes de la silvicultura del Bosque Tropical Alto en América. In: Informe del Seminario-Taller "Experiencias prácticas y prioridades de Investigación en Silvicultura de Bosques Naturales en América Tropical." Pucallpa, Perú. Venezuela: Centro de Estudios Forestales y Ambientales de Postgrado; 1996

[53] Vincent L. A tropical forest minimum impact management system. Bois et Forets des Tropiques. 2004;**279**(1):88-90

[54] Plonczak M, Rodriguez-Poveda L. Conceptos, fundamentos y métodos del manejo forestal en Venezuela. Revista Forestal Venezolana. 2002;**46**(1):83-90

[55] Bicknell JE, Struebig MJ, Davies ZG. Reconciling timber extraction with biodiversity conservation in tropical forests using reduced-impact logging. Journal of Applied Ecology. 2015;**52**(2):379-388

[56] Putz FE. Ruslandi. Intensification of tropical silviculture. Journal of Tropical Forest Science. 2015;**27**(3):285-288

[57] Huth A, Kammesheidt L, Koehler P. Sustainable timber harvesting in Venezuela: A modelling approach. Journal of Applied Ecology. 2001;**38**(4):756-770

[58] Noguera LO, Pacheco AC, Plonczak Ratschiller M, Jeréz RM, Moret A, Quevedo RA, et al. Planificación de la explotación de impacto reducido como base para un manejo forestal sustentable en un sector de la Guayana venezolana. Rev For Venez. 2008;**51**(1):67-78

[59] Lozada JR, Arends E. Impactos ambientales del aprovechamiento forestal en Venezuela. Interciencia. 1998;**23**(2):74-83

[60] Vilanova-Torre E, Ramírez-Angulo H, Torres-Lezama A. Carbon storage in the aboveground biomass as indicator of logging impact in the Imataca Forest reserve, Venezuela. Interciencia. 2010;**35**(9):659-665

[61] Vincent L. Métodos cuantitativos de planificación silvicultural. Mérida, Venezuela: Centro de Estudios Forestales y Ambientales de Postgrado, Universidad de Los Andes; 1993. p. 237

[62] Lozada JR, Moreno J, Suescun R. Plantaciones de enriquecimiento. Experiencias en 4 unidades de manejo forestal de la Guayana venezolana. Interciencia. 2003;**28**(10):568-575

[63] Jerez-Rico M, Quevedo A, Moret AY, Plonczak M, Garay V, Silva JD, et al. Regeneración natural inducida y plantaciones forestales con especies nativas: potencial y limitaciones para la recuperación de bosques tropicales degradados en los llanos occidentales de Venezuela. In: Herrera F, Herrera I, editors. La Restauración Ecológica en Venezuela: Fundamentos y experiencias. Caracas, Venezuela: Instituto Venezolano de Investigaciones Científicas (IVIC); 2011. pp. 35-60

[64] Pancel L. Technical orientation of Silviculture in the tropics. In: Pancel L, Köhl M, editors. Tropical Forestry Handbook. 2nd ed. Berlin, Germany: Springer-Verlag Berlin Heidelberg; 2016. pp. 1639-1691

[65] Finol H. Sistemas silviculturales aplicados y aplicables al manejo de bosques tropicales en Venezuela. Venezuela: Mérida; 1983

[66] Ashton MS, Kelty M. The Practice of Silviculture: Applied Forest Ecology. 10th ed. Oxford, UK: John Wiley & Sons, Ltd; 2018. p. 776

[67] Burivalova Z, Şekercioğlu ÇH, Koh LP. Thresholds of logging intensity to maintain tropical forest biodiversity. Current Biology. 2014;**24**(16):1893-1898

[68] Ramirez-Angulo H, Ablan M, Torres-Lezama A, Acevedo MF. Simulación de la dinámica de un bosque tropical en los llanos occidentales de venezuela. Interciencia. 2006;**31**(2):101-109

[69] D'Jesus A, Torres Lezama A, Ramírez Angulo H. Consecuencias de la explotación maderera sobre el crecimiento y el rendimiento sostenible de un bosque húmedo deciduo en los llanos occidentales de Venezuela. Revista Forestal Venezolana. 2001;**45**:133-143

[70] Torres-Lezama A. La cuidada movilización de los recursos forestales. La industria forestal. In: Pantin A, Reyes A, Quintero A, Montero R, Cunill-Grau P, Márquez A, et al., editors. Geo Venezuela. Caracas, Venezuela: Fundación Empresas Polar; 2007. pp. 382-439

[71] Sist P, Fimbel R, Sheil D, Nasi R, Chevallier MH. Towards sustainable management of mixed dipterocarp forests of Southeast Asia: Moving beyond minimum diameter cutting limits. Environmental Conservation. 2003;**30**(4):364-374

[72] Kammesheidt L. Stand structure and spatial pattern of commercial species in logged and unlogged Venezuelan forest. Forest Ecology and Management. 1998;**109**(1-3):163-174

[73] de Venezuela RB. Norma técnica forestal sobre diámetros mínimos de cortabilidad. Caracas, Venezuela: Asamblea Nacional; 2009. Resolución No. 0000030

[74] Noguera LO, Carrero GO, Plonczak M, Jerez M, Kool G. Evaluación técnica y financiera de la silvicultura desarrollada en un bosque natural de la Guayana venezolana. Bois et Forets des Tropiques. 2007;**290**(4):81-91

[75] Ochoa GJ. Análisis preliminar de los efectos del aprovechamiento de maderas sobre la composición y estructura de bosques en la guayana venezolana. Interciencia. 1998;**23**(4):197-207

[76] Instituto Forestal Latinoamericano (IFLA). Evaluación del sistema silvicultural plantaciones en fajas de enriquecimiento (PFE) en las reservas forestales Imataca, San Pedro y Dorado Tumeremo, estado Bolívar. Mérida, Venezuela: Instituto Forestal Latinoamericano; 2010

[77] Hardt Ferreira dos Santos VA, Modolo GS, Ferreira MJ. How do silvicultural treatments alter the microclimate in a Central Amazon secondary forest? A focus on light changes. Journal of Environmental Management. 2020;**254**(October 2019):109816. Available from: https://www.sciencedirect.com/science/article/pii/S0301479719315348?dgcid=author

[78] Kappel K, Holmen SJ. Why science communication, and does it work? A taxonomy of science communication aims and a survey of the empirical evidence. Frontiers in Communication. 2019;**4**:55. Available from: https://www.frontiersin.org/article/10.3389/fcomm.2019.00055

[79] Paniz-Mondolfi AE, Rodríguez-Morales AJ. Venezuelan science in dire straits. Science. 2014;**346**(6209):559. Available from: http://science.sciencemag.org/content/346/6209/559.abstract

[80] International Tropical Timber Organization - ITTO. Criteria and indicators for the sustainable management of tropical forests. ITTO Policy Development Series No. 21. [Internet]. 2016. pp. 1-84. Available from: https://www.itto.int/direct/topics/topics_pdf_download/topics_id=4872&no=1&disp=inline

[81] Nepstad D, McGrath D, Stickler C, Alencar A, Azevedo A, Swette B, et al. Slowing Amazon deforestation through public policy and interventions in beef and soy supply chains. Science. 2014;**344**(6188):1118-1123

[82] Guariguata M, Evans K. Advancing tropical forestry curricula through non-timber Forest products. International Forestry Review. 2010;**12**(4):418-426. Available from: http://www.jstor.org/stable/24309824

[83] Sist P, Sablayrolles P, Barthelon S, Sousa-Ota L, Kibler JF, Ruschel A, et al. The contribution of multiple use forest management to small farmers' annual incomes in the eastern Amazon. Forests. 2014;**5**(7):1508-1531

[84] Griscom BW, Goodman RC, Burivalova Z, Putz FE. Carbon and biodiversity impacts of intensive versus extensive tropical forestry. Conservation Letters. 2018;**11**(1):1-16

Chapter 2

Mixed Forest Plantations with Native Species for Ecological Restoration in Cloud Forests of the Venezuelan Andes

Ana Quevedo-Rojas and Mauricio Jerez-Rico

Abstract

Tropical cloud forests play a fundamental role in the hydrological cycle of mountain watersheds having the largest biodiversity per unit area. In Venezuela, cloud forests are subject to intense deforestation and fragmentation by farming and cattle-ranching causing soil erosion, water cycle alteration, and biodiversity loss. Reforestation projects used exotic species as *Pines* and *Eucalyptus*, native species were rarely planted by lacking knowledge on species requirements and management. We report the performance of 25 native cloud forest species differing in shade-tolerance, planted in mixed assemblies on degraded areas. Tree survival and the individual tree variables: total height, root-collar diameter, tree-slenderness, and crown-ratio were evaluated at 1, 2, 4.5 and 7 years-old. Data was analyzed with a repeated measures analysis of variance mixed model considering species shade-tolerance, light intensity at planting and age as explanatory factors. Survival was over 80%. Shade-intolerant species displayed faster height and root-collar diameter growth. Shade-tolerant species had larger crown ratios due to persistence of lower branches; whereas, shade-intolerant showed signs of crown recession at age 7. Slenderness values from age 4.5 were indicative of good trees stability and health across treatments. The positive results have motivated landowners to establish native species plantations in critical areas with our support.

Keywords: active restoration, reforestation, species selection, successional status, shade tolerance, facilitation, survival, tropical forests, silviculture

1. Introduction

Reforestation is the action planting trees for recovering lands that were originally covered by forests that were cleared due to land change use (e.g. agriculture) or by natural phenomena. Worldwide, most planted forests are monocultures with native or exotic species, but mixed planted forests are increasingly established [1]. Reforestation together with afforestation is a way to reduce the pressure on natural forests and satisfy the demand of forest goods and services.

Ecological restoration, on the other hand, looks for overcoming the barriers for the recovery of degraded landscapes, creating forests that meet the ecosystem functionality and structure of the former primary forest [2, 3]. In the context of ecological restoration, reforestation fit within the category of "active restoration" in which, in addition to suppressing the causes of disturbances (passive restoration), strategies are implemented to accelerate recovery and, if possible influence the trajectory of the succession [4, 5]. Active restoration is important in places in which the natural regeneration is slow or hindered by biophysical factors [3].

Mixed plantations consist in establishing two or more species in a same stand. Species within the stand can be combined in many arrangements from alternate monospecific patches, to rows or intimate assemblages with various species. Species can differ in site requirements, successional stage (pioneer, late successional), architecture, growth habits, and uses. Also, trees within the mixture can be planted at different times to create uneven-aged stands. Mixed plantations are more accepted by communities, can have larger productivity and provide a wider array of goods and services [6]. From the ecological restoration point of view, mixed plantations with native species is a useful option for recovering the functionality and diversity of tropical forests. Petit and Montagnini [7] found that native species mixed plantations have favorable social and economic functions, because they provide a variety of timber and non-timber goods and services including soil recovery, carbon sequestration, and increase in biodiversity. Mixed plantations have been successful as a restoration option for creating a forest cover under certain conditions, where forests cannot regenerate by natural successional mechanisms as is the case in degraded pasturelands. However, the silviculture and management of mixed species with native tropical cloud forests species is complex, because lack of knowledge on growing these species, availability of plant, and difficult silvicultural management, particularly in intimate mixtures.

The Andean cloud forests are considered biodiversity "hotspots" and top the list of the most vulnerable ecosystems worldwide due to their small area and the high rates of deforestation by changes in land use from forest to agriculture and livestock that is accelerating the loss and degradation of these ecosystems. Cloud forests are characterized by a persistent cloudiness year around and occur in mountainous areas with abrupt topography. The complex combination of biotic and abiotic factors originate habitats with a high spatial and temporal heterogeneity leading to a high biodiversity [8].

Reforestation experiences in the cloud forest of Venezuela have been mostly carried out with exotic species (pine, cypresses, eucalyptus, ash, acacias) in monospecific cultures, and rarely in small tracts with several species. Native species plantation were included in past projects, but they were not successful due to failure to exclude main threats, lack of knowledge of site species requirements, improper planting methods, and absence of further care and monitoring in the initial stages. In general, seedlings of tree cloud forest species are characterized by slow growth and low tolerance to direct sunlight. In Venezuela, methods for production in nursery, plantation establishment, and management for these species are unknown. In 2007, our team began research projects aimed to study systematically cloud forest species requirements. Research included the distribution of seedlings of tree species in the forest understory along a light gradient [9, 10] and ecophysiological and morphological responses to contrasting light conditions [11, 12]. In 2012, we began a project aimed to develop and execute a plan for the landscape ecological restoration in the cloud forest of Paramo El Tambor with participation of stakeholders to restore degraded areas and help to protect the wildlife and their habitats. The plan includes several strategies of passive and active restoration. Among these, we initiated the establishment of small scale trials consisting of reforesting with native species in mixtures on deforested, degraded sites

The objective of this work was to report the performance of mixed species plantations, including more than 25 native species with different requirements of shade tolerance (shade-intolerant species; shade-tolerant species; partially shade species) established on degraded areas, originally covered by exotic pastures and some isolated trees. We describe the procedures for plant production, establishment, care and evaluated growth performance at ages 1, 2, 4.5, and 7 year-old for the different shade-tolerance groups growing in sites with different levels of shade.

2. Materials and methods

2.1 Study area

2.1.1 Physical environment

The study was carried out in Paramo El Tambor, an isolated mountain massif part of the Venezuelan Andes (8°37′00″N; 71°21′00″W) facing the Maracaibo Lake with unique environmental characteristics (**Figure 1**). Politically is located in the Mérida state. The area occupies around 150 km^2 covered by dense cloud forests, *paramo* (moorlands), and wetlands.

The tropical climate is influenced by altitude (2.000–3.000 m.a.s.l.) and topography. The trade winds coming from the NE penetrate through the piedmont area facing the lacustrine plain of Lake Maracaibo. These winds come loaded with water vapor forming a persistent cloudiness that when reaching its saturation point, triggers local precipitations in the slopes and valleys. The persistent thick cloudiness induces the existence of dense cloud forests from 2000 to 2500 m a.s.l. The annual average temperature is 14.9°C and average precipitation is 1400–1560 mm with a short dry season (December–February).

The landscape consists of rounded hills, with shallow to steep slopes. The soils are derived from the Colón Cretaceous formation characterized by stratified, massive, black, non-calcareous lutites with conchoidal fractures. Predominant soils are Ultisols and Inceptisols with and Udic regime of clay-silty, clay-silty-loam to clayey

Figure 1.
Working Area Location of Páramo El Tambor, Mérida, Venezuela (Images: Google Earth, 2020).

textures highly variable in depth, structural development, and drainage. Chemically, they are highly acid (pH 4–5) with a low base saturation, high exchangeable Al, low CEC and a high organic matter content (5.5 % C) in the upper horizon [13]. There are no large rivers coming from high moorlands, and horizontal precipitation captured by the forest plays and important role in the water balance [14].

2.1.2 Vegetation

The forests are rich in evergreen tree species densely covered by epiphytes, mosses, and lichens. Tree Diversity is high with 40-60 species ha^{-1} [15]. The main tree families are Lauraceae, Melastomataceae, Euphorbiaceae, Myrtaceae, and Podocarpaceae, constituting one of the few tropical forest in Venezuela where native conifers coexist with hardwoods. The forest comprises various plant communities ranging from dense high forest (DHF) with complex stratification and a canopy 25-30 m high to sparse low stature forests with canopies < 15 m tall. The DHF has three layers: the upper layer approximately 25-30 m in height with emergent trees up to 40 m tall (mainly *Retrophyllum rospigliosii*, a conifer), an intermediate layer 20-24 m in height, and a lower layer 10-19 m. The understory comprises tree seedlings and saplings, shrubs, vines, palms, and herbaceous plants. Tree ferns (*Cyathea spp*.) and bamboos (*Chusquea spp*.) are common, the latter forming dense scrubs [13].

2.1.3 Socio-economic aspects

The landscape is a mosaic in which traditional agricultural production methods and old agro-social structures coexist with intensive production systems, and legal figures of strict protection; whereas, land management with ecosystem approaches are less usual. The area have been subject to intense deforestation and forest fragmentation In recent times, local people are witnessing a rapid deterioration of their surrounding environment according to perception surveys carried as part of the community service project “Sensitization for the Conservation of the Andean Cloud Forest” (unpublished data). The areas in which deforestation was more intense are clearly suffering a reduction of water supply. In addition, they are more exposed to strong winds that damage the crops by mechanical and desiccation impacts. In recent times, the area is experiencing environmental changes such as longer dry seasons and extreme precipitation events. The later, together with exposed soils have caused landslides affecting roads and properties. Associated consequences of deforestation and degradation is the apparent extinction of some local wildlife species such as amphibians and monkeys.

2.2 Species selection and plant production

For selecting the species to produce in nursery, we took into account landowner’s preferences and the findings of previous studies that analyzed the spatial distribution of seedlings of cloud forest tree species along a light gradient [8] and experiments in which seedlings of various species were grown under controlled levels of light intensity, then subjected to sudden changes in irradiation to observe their photosynthetic acclimation capacity [12]. According to these studies, we categorized the species in functional groups based on shade tolerance. Species fitted into three categories: (a) shade intolerant (SI), (b) partially shade tolerant (PT), and (c) shade tolerant (ST) (**Table 1**).

To produce plants, we collected seedlings from the forest understory because collecting viable seeds was very difficult, as many species have a very irregular cycle of flowering and seed production [19]. Seedlings were transplanted in polyethylene

Scientific name	Family	Shade Tolerance[1]	Uses[2]
Tetrorchidium rubrivenium	Euphorbiaceae	SI	WDF, FD, FA
Alchornea grandiflora	Euphorbiaceae	SI	WDF, MTB, FA
Montanoa quadrangularis	Asteraceae	SI	WDF, FD; FA
Ruagea pubescens	Meliaceae	SI	MTB; OR, FA
Cedrela montana	Meliaceae	SI	MTB, OR, FA
Inga oerstediana	Leguminosae	SI	MTB; FA, NF
Ocotea macropoda	Laureaceae	SI	MTB; FA
Miconia meridensis	Melastomataceae	SI	MTB, FA
Cecropia telenitida	Urticaceae	SI	MTB; OR
Hieronyma moritziana	Euphorbiaceae	PT	MTB, FA
Billia columbiana	Hippocastanaceae	PT	MD, MTB, FA
Casearia tachirensis	Flacourtiaceae	PT	HWR, MTB, FA
Beilschmiedia sulcata	Laureaceae	PT	HWR, FA
Nectadra laurel	Laureaceae	PT	MTB, FA
Prunus moritziana	Rosaceae	PT	MTB, OR, FA
Retrophyllum rospigliosii	Podocarpaceae	PT	MTB, OR, FA
Myrcia acuminata	Myrtaceae	PT	TB, FW, FA
Vochysia meridensis	Vochysiaceae	PT	MTB, OR, FA
Eugenia tamaensis	Myrtaceae	ST	MWR; FA
Myrcianthes karsteniana	Myrtaceae	ST	HWR, OR, MD, FA
Miconia resimoides	Melastomataceae	ST	MTB; OR
Podocarpus oleifolius	Podocarpaceae	ST	MTB, OR
Myrcia fallax	Myrtaceae	ST	MTB; MD
Aegiphila terniflora	Verbenaceae	ST	MTB, HWR, FA
Eschweilera tenax	Lecytidaceae	ST	HWR, FA

[1]*Species shade tolerance group: shade intolerant (SI), partially tolerant (PT), shade tolerant (ST).*
[2]*Species uses: fodder (FD), fauna attraction (FA), medicinal (MD), firewood (FW), nitrogen fixing (NF), ornamental (OR), multipurpose timber (MTB), hardwood for building roofs (HWR), wooden fences (WDF).*

Table 1.
List of species used for restoration in Paramo El Tambor cloud forest according to their shade tolerance and potential uses.

bags (diameter = 17 cm, height = 25 cm), in a mix 1:1 forest soil and sand. Plantlets were grown for at least six months in a nursery covered by an 80 % shade mesh allowing the pass of 214–270 μmol m^2 s^{-1} of photosynthetic photon flux (PPF), approximately 20% PPF of daily full sunlight [12]. Plants were irrigated regularly to avoid water stress. Weeds were removed manually and pest control was needed against snails and slugs.

2.3 Site selection, plantation establishment, maintenance and monitoring

Sites for planting were chosen together with stakeholders on deforested, degraded areas limiting with springs and having strong limitations for farming activities. The sites were 1200 to 1500 m^2 each, at 2250 to 2300 m a.s.l. Areas were covered by grasses, usually kikuyu grass (*Pennisetum clandestinum*), shrubs, vines (e.g., *Rubus fruticosus*) and isolated trees. Sites varied in slope (30-80%) with soils poor in organic matter and nutrients. Most sites were exposed to direct sunlight; however, tree crowns close or within the planting areas projected shade that reduced the incident light at ground level. In the chosen sites, we delimited the area and made a topographic survey. Pastures and vines were cut with a grass-cutting machine. Trees and shrubs from natural regeneration were left. The areas were delimited with wired fences to exclude cattle. In addition, we estimated the light environment in the sites by using as a surrogate the percentage of canopy openness (%CO) [8]. This variable provides a good characterization of the potential penetration of solar radiation through canopies and its incidence on a given point [16]. For estimating the %CO, we took hemispherical photographs on the vertices of a superposed 5.0 m square-grid covering the planting areas (n= 140). We used a digital camera with a fisheye lens mounted on a tripod and leveled with the top of the lens standing 150 cm above the ground. The photos were processed with Gap Light Analyzer (GLA) v. 2.0 [17]. For detailed procedures see [9, 18].

For site preparation no tillage was done, so the soil was disturbed only when opening the planting holes. Around the holes, grasses and other weeds were removed from root. Planting was done during the rainy season.

In all areas, plants were established following a horizontal (slope corrected) triangular spacing of 1.5 m (5.128 trees ha^{-1}). A regular spacing was preferred over an irregular one for controlling variables of stand density and competition and easiness for monitoring. Also, landowners showed a marked preference for regular spacing. The selected planting density is very high for tropical plantations standards (600-2500 tree ha^{-1}), but we looked to ensure survival of sufficient trees to reach faster shading for controlling pastures, and to observe competition/facilitation interactions as soon as possible.

Planting areas were subdivided in plots (~200 m^2 each, ~100 trees per plot). Plots were assigned to levels of light intensity (LI) based on %CO at planting time. Three levels of LI were differentiated: (a) High Light (HL, above 50% CO); (b) Medium Light-(ML, 40–50 % CO); and, Low Light intensity (LL <40 % CO).

In plantation rows, within each plot, two trees of the same species were planted consecutively forming a "group". Groups were alternated randomly, but with the restriction that at least two groups from each shade tolerance category should be included within plots classified within a given light intensity level (**Figure 2**). Planting two trees looked to improve the probabilities of at least one tree surviving, so maintaining a regular distribution within the plantation. The less promising tree will be eliminated when thinning is needed. The approximate planting ratio of SI, PT, and ST was 5:3:2, as ST species were more difficult to grow in nursery until reaching the desired size (20–40 cm tall). Mechanical-manual control of weeds was needed at least every six months until age 2. Cleanings consisted of eliminating vines and weeds rooted around the tree stem in a circle of 50 cm radius. Weeding was done prior to each measurement time to facilitate access to the trees. No chemicals were used for controlling pests or plagues that periodically affected some species. Irrigation, was needed only during the first dry season after planting. A replanting was done six months after planting to replace dead plants.

2.4 Measurements and statistical design

We sampled 12 plots (~200 m^2 each, ~80–90 trees per plot) covering all combinations of light intensity (LI), species shade-tolerance (STOL), and ages. All plots included species of the three groups of shade tolerance; however, not all species were present in a given plot. The trials were re-measured at ages 1, 2, 4.5 and 7 years. Percent

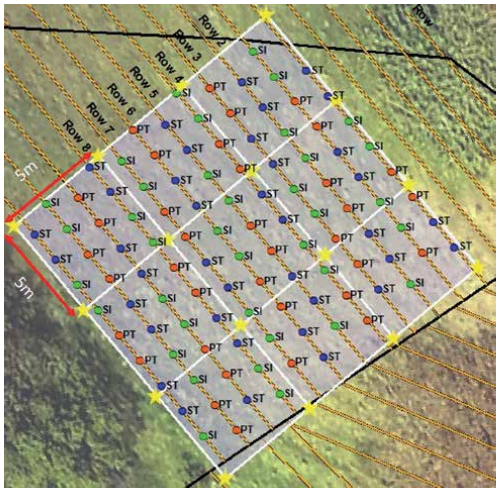

Figure 2.
Spatial arrangement of groups of plants within a plot. Spacing is triangular (1.5 m) and distance between rows is 1.3 m. Legend: hemispherical photographs (☆) taken on the vertices of the white grid, planted rows (▬), fences (▬), shade intolerant (●), partially tolerant (●), and shade tolerant plants (●).

tree survival was determined for each combination of light intensity and shade tolerance group as the percentage of surviving trees at a given age with respect to the initial number of planted trees. For each tree, we measured (a) total height in meters (HT), (b) root collar diameter in cm (RCD) and computed (a) tree slenderness coefficient (TSC) calculated as total height (m) divided by root collar diameter (m), and (b) crown ratio (CR) calculated as live crown length (m) divided by total height (m).

The TSC is an important indicator of tree stability. In general, a tree with a low TSC usually indicates lower center of gravity with a longer crown length, and a better developed root system increasing resistance to falling by strong winds and other factors [19–21]. The CR varies between 0 and 1, with a large live crown ratio indicating a healthy tree able to respond to favorable changes such as canopy openings [22].

To analyze the data, a factorial analysis of variance (ANOVA) mixed model was used in which STOL and LI factors were considered as fixed effects. Re-measurement age was considered a repeated effect, as the same plants and plots were re-measured at the various ages. Only the largest tree in each group of two of a same species was chosen for data analysis, as it was assumed that this expressed best the growth potential under the site conditions predominating in the plot. The second tree, if survived, usually grew less due to slow initial growth, or mechanical damage. Also, damaged trees were discarded from analysis (e.g., broken trees).

Data were stored in a spreadsheet and processed in SAS v. 9.1 [23]. For each variable, several models with varying structure of the variance-covariance matrix were tested. The best model was that with the lower value of the Akaike and Bayesian information criteria (AIC and BIC) [24]. For each variable, simple effects (shade tolerance, light intensity, and age) and resulting interactions within and across age were analyzed for shade tolerance and light intensity. The model takes into account the correlation between repeated measurements normality and variance heterogeneity along time [25]. As the design is unbalanced, a Tukey-Kramer test based on marginal means (least squares means) [26] allowed comparisons for differences among shade-tolerance groups and light intensity at similar re-measurement times. Probability values (p-values) indicated the statistical significance of effects and interactions. For simplicity in means comparisons, p-values ≥ 0.05 indicated no significant differences, p-value < 0.05, significant differences, and p-value < 0.01, highly significant differences.

3. Results and discussion

3.1 Survival

Survival was high for all categories of shade tolerance within different levels of light intensity averaging over 70% for all plots after year 4.5 (**Table 2**). Only one plot was discarded from analysis because flooding caused high mortality (>50%). In the remaining plots, the SI species had a survival above 85% after 4.5 years with a slightly better survival in LL. Also, PT species showed better survival in LL (above 90%), but only 73–77% in HL and ML. Shade Tolerant species presented the lower values of survival in HL; nonetheless, values remained above 70 % after year 4.5. In ML this group maintained over 80 % survival, and above 90% in LL. Only two species *M. resimoides* and *E. tenax* (n > 10 each) had 100% mortality before the second year. After age 2, most mortality was due to mechanical damage by the fall of large branches from the isolated trees and occasional herbivory and cramping by cattle that passed the fences. There are very few studies for mixed plantations in tropical cloud forests that report survival results of native species after four years-old.

In a Mexican cloud forest [27] evaluated mixed plantation trials with up to nine native species and found an average survival of 93% three years after planting. They concluded that if plants survive the establishment phase, mortality in further years is low. In the same study, the authors reported that in a trial with the native species *Alnus acuminata* and *Quercus xalapensis* growing on native and exotic abandoned grass, the combined survival was 92 and 48 % after 46 weeks since planting. Higher mortality on exotic grass varied between species due to competition and herbivory from rats. Also in a Mexican cloud forest [28] planted *Alnus acuminata* and *Trema micrantha* as facilitating species for the establishment of intermediate and late successional species (ILS). At age 2 (96 weeks), survival was 94 and 77% respectively for the facilitating species. On the other hand, the survival of the three ILS was significantly higher (>60%) under the shade of *A. acuminata*, and 40–50% under *T. micrantha*. When planted on open field survival was only 11 to 22% for the ILS. They attributed the low survival in the open field to water stress accompanied with herbivory. In the Atlantic semi-deciduous forest of Brazil, [29] compared the

Shade tolerance	Light intensity	Trees planted	Age (years)			
			1.0	2.0	4.5	7.0
IS	HL	121	92.1 ± 7	91.0 ± 9	88.4 ± 9	79.2 ± 10
	ML	223	95.9 ± 4	93.9 ± 6	89.4 ± 8	90.0 ± 5
	LL	132	100.0 ± 0	100.0 ± 0	97.8 ± 3	91.3 ± 4
PT	HL	82	84.3 ± 10	81.6 ± 6	73.6 ± 14	64.9 ± 18
	ML	120	91.2 ± 9	86.8 ± 12	76.9 ± 11	78.7 ± 4
	LL	55	98.6 ± 2	97.1 ± 4	94.6 ± 1	94.3 ± 2
ST	HL	51	76.0 ± 9	76.0 ± 9	73.5 ± 5	70.0 ± 5
	ML	101	98.5 ± 3	91.2 ± 8	82.9 ± 4	80.2 ± 8
	LL	55	97.5 ± 4	96.1 ± 2	94.6 ± 1	91.4 ± 1

[1]*Species shade tolerance group: shade intolerant (SI), partially tolerant (PT), shade tolerant (ST).*
[2]*Light intensity: high light (HL), medium light (ML), low light (LL).*

Table 2.
Percent survival ± standard deviation (n = 11 plots) for combinations of shade tolerance and light intensity at the ages of measurement.

survival of 36 native species from different successional stages established as mixed plantations on degraded sites. Before planting they plowed and fertilized the sites. Fifteen after planting survival was around 55% for pioneer and secondary species, and only 12% for the late successional ones. The high survival rates for most species in the present report was due to a relatively intense monitoring and maintenance within the first two years after planting. The main silvicultural treatments included avoiding water stress with irrigation in the first dry season after planting and controlling aggressive weeds and grass at least for two years. Keeping cattle exclusion was also critical. Although insects and pests attacked particular species at different times, they rarely caused mortality. Finally, in areas shaded by mature trees, the fall of large branches caused mechanical damage to some trees which eventually died.

3.2 Tree level variables

For all variables measured on individual trees, a mixed model with an autoregressive first-order residual variance-covariance matrix performed best than alternative structures (lowest AIC and BIC criteria). The type III test for fixed effects showed highly significant differences for simple effects, except for CR with $p > 0.05$ (**Table 3**). The interaction STOL × AGE was significant for all variables; whereas, LI × AGE was significant ($p < 0.05$) for TH, and highly significant for the rest of variables. The interaction STOL × LI was also significant for all variables except for TSC ($p > 0.05$). The interaction STOL×LI×AGE was not significant. The LI factor is based on values of %CO at the beginning of the plantations, and although changes in light incidence above the canopy (isolated trees) were minimum, the largest planted trees of SI species began to reduce the amount of light received by smaller trees included most of the ST. Mixtures of shade intolerant, early successional species and shade-tolerant, late-successional species, could facilitate the survival and growth of the latter [30].

These results indicate that trees from the three STOL groups underwent significant changes in the evaluated variables along age and across plots differing in initial LI levels, with the SI species showing faster growth at all levels of LI.

The most insightful results were for the STOL x AGE interactions where the averaged values of variables for each STOL group were compared within and along ages (**Figure 3a,b**). The STOL groups did not differ significantly in TH the first year after planting; but, after age two, the SI group presented significantly larger heights than the PT-ST groups (**Figure 3a**). Differences in TH among the three groups become larger at ages 4.5 and 7 years, with SI species growing faster than PT, and the ST species showing the lower height growth. Despite differences in total height the three groups appear to be growing at an increased rate with age. After 7 years

Variable[1]	STOL	LI	AGE	STOL x AGE	LI x AGE	STOL x LI
TH (m)	<.0001	0.0001	<.0001	<.0001	0.0152	0.0005
RCD (cm)	<.0001	<.0006	<.0001	<.0001	<.0001	0.0200
TSC	0.0015	<.0001	<.0001	<.0001	<.0001	*0.0917*
CR	<.0001	*0.4635*	<.0001	<.0004	0.0083	0.0168

[1]*Variables: total height (TH), root collar diameter (RCD), tree slenderness coefficient (TSC), crown ratio (CR).*
[2]*Factors and interactions: shade tolerance (STOL), light intensity (LI).*

Table 3.
Simple effects and interactions for the analyzed variables. Triple interaction not included. Means comparisons p-values ≥ 0.05 indicated no significant differences, p-value < 0.05, significant differences, and p-value < 0.01, highly significant differences.

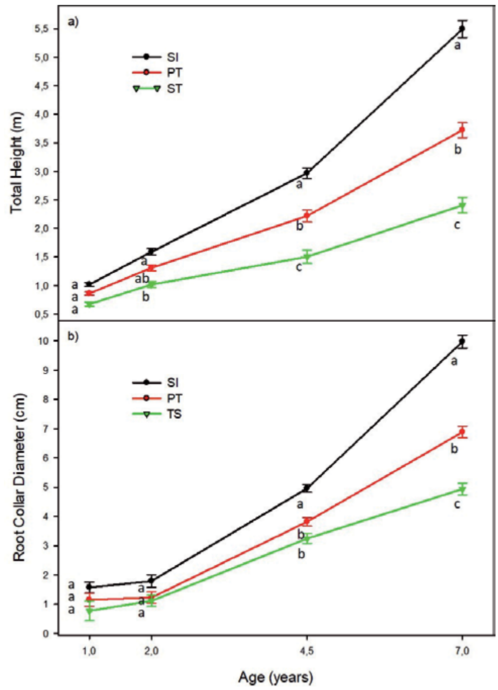

Figure 3.
Changes in total height and root collar diameter among shade tolerance species groups with age. (a) Total height, (b) root collar diameter. [1]Species shade tolerance group: shade intolerant (SI), partially tolerant (PT), shade tolerant (ST). [2]Similar letters below the lines indicate statistically not significant differences (p>0.05) among shade tolerance groups; whereas, different letters indicate significant differences (p < 0.05) according to the Tukey-Kramer test.

average height increments can be considered low (< 1 m yr^{-1}) when compared with the growth increments for lowland tropical species (2–4 m yr^{-1}) and the growth of exotic species such as cypresses and pines. However, SI species had many trees close to 10 m tall at age 7. These trees also developed large, wide crowns (e.g., *M. quadrangularis, T. rubrivenium, M. meridensis*). Likewise, for RCD there were no differences among STOL groups for ages 1–2; whereas at ages 4.5 and 7, in which the three groups, with SI having a larger RCD than the other groups (**Figure 3b**).

When considering the STOL × LI interaction effect on the performance of TH and RCD, the Tukey test indicated no statistically significant differences ($p > 0.05$) at ages 1 and 2 years among the STOL groups, independently of LI for any of these variables. However, at ages 4.5 and 7, SI species showed significantly better performance in both variables than PT-ST species. Likewise, no significant differences were found for HT and RCD between PT and ST species across LI levels.

The TSC had not significant differences among groups at age 1 (**Figure 4a**). At age 2, all species increased their TSC (above 1.0), with a significantly higher increment for the ST-PT species over the SI species. Possibly, TSC at age 1 reflected the values that trees had in the nursery. Larger TSC for the second year could be due to a faster growth in height relative to RCD growth, indicating a faster stem elongation and formation of leaves at the top of the trees. By ages 4.5 and 7, TSC decreased again and stabilized for all groups, with no significant differences among them. The lower TSC indicates that the trees have more stability and are showing an adequate growth pattern [20].

Finally, the live crown ratio (**Figure 4b**) was significantly higher (>0.6) for the ST group than for PT-SI. By age 4.5 however, there was a considerable increase in CR for the PT and SI groups with no significant differences among STOL groups. Nonetheless, SI species had the lowest CR. At age 7, CR was similar (above 0.7) for

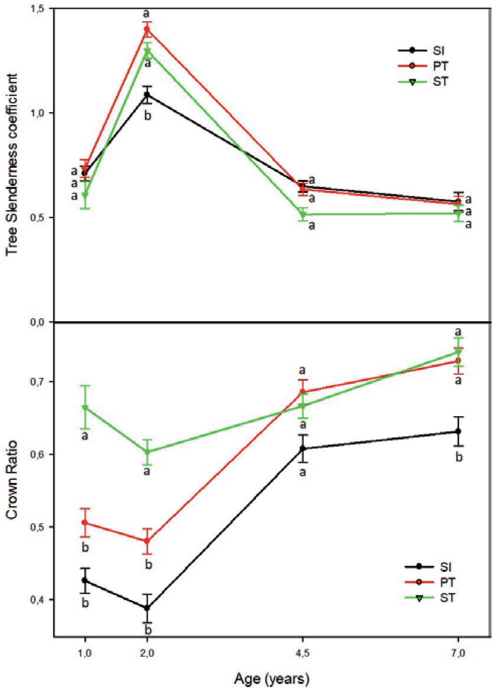

Figure 4.
Changes in tree slenderness coefficient and crown ratio among shade tolerance species groups with age. (a) Tree slenderness coefficient, (b) crown ratio. [1]Species shade tolerance group: Shade Intolerant (SI), Partially Tolerant (PT), Shade Tolerant (ST). [2]Similar letters below the lines indicate statistically not significant differences (p>0.05) among shade tolerance groups; whereas, different letters indicate significant differences (p < 0.05) according to the Tukey-Kramer test.

ST and PT and clearly higher than for SI species (0.6). The large CR for ST species is explained by the trend to have persistent leaves and branches along most of the stem along the years. This finding is supported by [31] who working with the plasticity traits of saplings from the tropic humid forest in the French Guiana determined that crown depth in shade tolerant species was a trait that depended on the leaf lifespan rather than crown elongation. In PT, and specially SI, the rapid height growth was accompanied by a large development in crown length; however by age 7, SI trees displayed signs of crown recession, as shaded leaves on the lower branches were dying quickly. Conversely to shade tolerant trees, [31] shorter crowns in SI species was attributed to a shorter lifespan of leaves. Usually, trees showing CR above 0.5 can be considered healthy; whereas, values below 0.3 indicate trees that are under strong competition. In addition, a characteristic shade intolerant species is the trend to reduce faster their CR under competition, because lower limbs tend to die due to insufficient light for having a net positive C assimilation.

At age 1, the TSC coefficient was significantly larger at ML for the three STOL groups than at HL or LL; differences were significantly higher slenderness in ML than in HL or LL; whereas, PT and ST species. At age 2, TSC remained stable for all STOL groups in the ML level, but the values increased significantly for all groups in HL and LL. At ages 4.5 and 7 no differences in TSC were found for any of the STOL groups in any LI level. Finally, at age 1 and 2, CR had significantly larger values in ML than in HL or LL. For ages, 4.5 and 7, there were no significant differences within light levels for any of the STOL groups.

The relatively low effect of LI levels on the performance of SI species support findings by [32] who suggest these species had an inherent fast growth rate determined mainly by morphological traits, but these growth rates are maintained at the expense of defense and storage allocation. As we could observe, SI species suffered from selective herbivory by cattle (i.e., leaves of trees from SI species were eaten; whereas those of PT-ST were not). On the other hand, as mentioned by the same author, survival of SI species cannot be attributed to a high net C balance

By age 7, the crown of SI species were in contact and forming an upper layer of dominant trees; whereas, PT and ST species conformed an intermediate layer. Although competition for light was not evident from the observed crown ratio values; the SI trees showed a large loss of leaves on their lower branches; whereas these persisted in trees of PT and ST, keeping large crown ratios despite being shaded. Only, between ages 5 and 7, the shade created by the new plantation has eradicated pastures, except in sites canopy holes created by tree mortality or in the borders of the planted sites. Natural regeneration of some tree and shrub species began after year two, and when taken into account increase stand density above 100%. In highly shaded areas, the presence of herbaceous plants is rather scarce and a fine layer of litter is forming.

4. Conclusions

Our research with tree native species and establishment of demonstrative trials in cloud forests show promising that reforesting with native species is a viable alternative for restoring degraded areas and the recovery of tree biodiversity given adequate planning, management and monitoring (**Figure 5**).

The most critical aspects for the initial success of these plantations consists on maintaining cattle exclusion for at least five years after establishment, irrigating during the first dry season, and keeping control of grasses and vines at least for two years.

Different mixed shade tolerance groups of tree species can stand conditions of sites dominated by pastures under high level of sunlight exposure; however, SI species appear to have faster growth rates than PT or ST species independently of shade conditions.

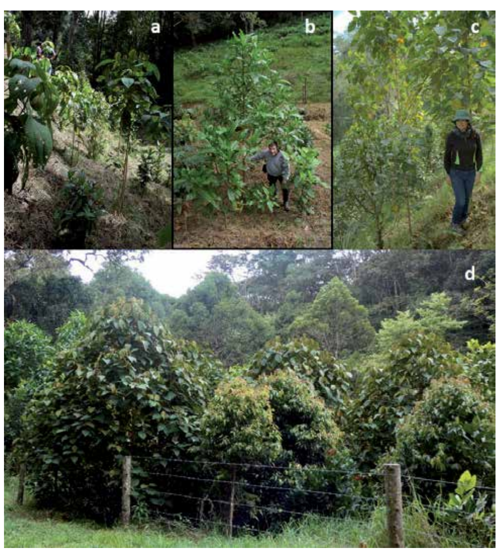

Figure 5.
State of development of one of the mixed plantations: (a) one year old, (b) 2 years old, (c) 4.5 years old, (d) 7 years old.

Planting mixed forests is a good option for recovering degraded sites in which the forest has disappeared and conditions for unassisted tree regeneration is not possible. More than scaling up to cover larger extensions, many landowners can establish small plantations in critical areas. Many of these plantations can act as small nucleus for maintaining and dispersing rare, shade-tolerant, late successional species that usually are difficult to regenerate without specific silvicultural treatments.

Mixed planted forests have a very positive response from local people who view these plantations as a very satisfying way of recover the forests as pointed by [33] in a similar work in the Andean cloud forest of Ecuador.

Among other benefits mixed planted forests with native species, provide food and shelter for wildlife, have better social acceptance because they are part of the natural landscape, and provide a variety of goods and services (e.g., shade, soil protection, medicinal properties).

Mixed plantations are complementary with other methods such as those of assisted regeneration in degraded forests or passive restoration used to recover large areas.

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Conflict of interest

The authors declare no conflict of interest.

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References

[1] Paula RR, de Oliveira IR, Gonçalves JL de M, de Vicente Ferraz A. Why Mixed Forest Plantation? In: Bran Nogueira Cardoso EJ, Gonçalves JL de M, Balieiro F de C, Franco AA, editors. Mixed Plantations of Eucalyptus and Leguminous Trees [Internet]. Cham: Springer International Publishing; 2020 [cited 2020 Oct 12]. p. 1-13. Available from: http://link.springer.com/10.1007/978-3-030-32365-3_1

[2] Lamb D, Erskine PD, Parrotta JA. Restoration of degraded tropical forest landscapes. Science. 2005;310(5754):1628-1632.

[3] Chazdon RL. Beyond Deforestation: Restoring Forests and Ecosystem Services on Degraded Lands Science. 2008;320:1458-1460.

[4] Trujillo-Miranda AL, Toledo-Aceves T, López-Barrera F, Gerez-Fernández P. Active versus passive restoration: Recovery of cloud forest structure, diversity and soil condition in abandoned pastures. Ecol Eng. 2018 Jul;117:50-61.

[5] Holl KD, Aide TM. When and where to actively restore ecosystems? For Ecol Manag. 2011 May;261(10):1558-63.

[6] Liu CLC, Kuchma O, Krutovsky KV. Mixed-species versus monocultures in plantation forestry: Development, benefits, ecosystem services and perspectives for the future. Glob Ecol Conserv. 2018 Jul;15:e00419.

[7] Petit B, Montagnini F. Growth in pure and mixed plantations of tree species used in reforesting rural areas of the humid region of Costa Rica, Central America. For Ecol Manag. 2006;233(2-3):338-343.

[8] Schwarzkopf T, Riha SJ, Fahey TJ, Degloria S. Are cloud forest tree structure and environment related in the Venezuelan Andes?: CLOUD FOREST STRUCTURE IN THE ANDES. Austral Ecol. 2011 May;36(3):280-9.

[9] Quevedo-Rojas A, Jerez-Rico M, Schwarzkopf Kratzer T, García-Núñez C. Distribution of juveniles of tree species along a canopy closure gradient in a tropical cloud forest of the Venezuelan Andes. IForest - Biogeosciences For. 2016 Jun 1;9(3):363-9.

[10] Quevedo-Rojas AM, Schwarzkopf T, García C, Jerez-Rico M. Ambiente de luz del sotobosque de una selva nublada andina: efectos de la estructura del dosel y la estacionalidad climática. Rev Biol Trop [Internet]. 2016 Jul 20 [cited 2020 Oct 12];64(4). Available from: http://revistas.ucr.ac.cr/index.php/rbt/article/view/21861

[11] García-Núñez C, Azócar A, Rada F. Photosynthetic acclimation to light in juveniles of two cloud forest tree species. Trees. 1995 Dec 1;10(2):114-24.

[12] Quevedo-Rojas A, García-Núñez C, Jerez-Rico M, Jaimez R, Schwarzkopf T. Leaf acclimation strategies to contrasting light conditions in saplings of different shade tolerance in a tropical cloud forest. Funct Plant Biol. 2018;45(9):968.

[13] Ramos MC, Plonczak M. Dinámica sucesional del componente arbóreo, luego de un estudio destructivo de biomasa, en el bosque universitario San Eusebio, Mérida-Venezuela. Rev For Venez. 2007;51(1):35-46.

[14] Ataroff M, Rada F. Deforestation Impact on Water Dynamics in a Venezuelan Andean Cloud Forest. AMBIO J Hum Environ. 2000 Nov;29(7):440-4.

[15] Ataroff M. VENEZUELA. In: BOSQUES NUBLADOS DEL NEOTROPICO. INBIO; 2001. p. 397-442.

[16] Gonsamo A, D'odorico P, Pellikka P. Measuring fractional forest canopy

element cover and openness - definitions and methodologies revisited. Oikos. 2013 Sep;122(9):1283-91.

[17] Frazer GW, Canham CD, Lertzman KP. Gap Light Analyzer (GLA), Version 2.0: Imaging software to extract canopy structure and gap light transmission indices from true-colour fisheye photographs, users manual and program documentation. Simon Fraser Univ Burnaby Br Columbia Inst Ecosyst Stud Millbrook N Y. 1999;36. Available from: http://rem.sfu.ca/forestry/downloads/Files/GLAV2UsersManual.pdf

[18] Zhang Y, Chen JM, Miller JR. Determining digital hemispherical photograph exposure for leaf area index estimation. Agric For Meteorol. 2005;133(1):166-181.

[19] Ezenwenyi JU, Chukwu O. Effects Of Slenderness Coefficient In Crown Area Prediction For Tectona Grandis Linn. F. In Omo Forest Reserve, Nigeria. Current Life Sciences. 2017 Sep 25;3(4):65-71

[20] Zhang X, Wang H, Chhin S, Zhang J. Effects of competition, age and climate on tree slenderness of Chinese fir plantations in southern China. For Ecol Manag. 2020 Feb;458:117815.

[21] Soto DP, Donoso PJ, Vásquez-Grandón A, González-Chang M, Salas-Eljatib C. Differential Early Performance of Two Underplanted Hardwood Tree Species Following Restoration Treatments in High-Graded Temperate Rainforests. Forests. 2020 Apr 3;11(4):401.

[22] Zhao D, Kane M, Borders BE. Crown Ratio and Relative Spacing Relationships for Loblolly Pine Plantations. Open J For. 2012;02(03):101-15.

[23] SAS Institute, SAS/STAT user's guide, version 9.1. Cary, NC: SAS Institute; 2004.

[24] Di Rienzo JA, Macchiavelli R, Casanoves F. Modelos Mixtos en InfoStat. Córdoba, Argentina: Grupo INFOSTAT; 2012.

[25] Stroup WW. Rethinking the Analysis of Non-Normal Data in Plant and Soil Science. Agron J. 2015 Mar;107(2):811-27.

[26] Littell RC, Milliken GA, Stroup WW, Wolfinger RD, Schabenberger O. SAS for Mixed Models, Second Ed. SAS Institute, Cary, NC. 2006.

[27] López-Barrera F, Rojas-Soto O, Montes-Hernández B, Aguirre A, Aguilar-Dorantes K, Landgrave R, et al. Ecología de la restauración del bosque nublado en el centro de Veracruz. In: Experiencias mexicanas en la restauración de los ecosistemas (Mexican experiences in ecosystem restoration). 1st ed. UNAM-CRIM, UAEM, CONABIOEditors: Eliane Ceccon, Cristina Martínez-Garza; 2016. p. 30.

[28] Avendaño-Yáñez M de la L, Sánchez-Velásquez LR, Meave JA, Pineda-López M del R. Is facilitation a promising strategy for cloud forest restoration? For Ecol Manag. 2014 Oct;329:328-33.

[29] Coelho GC, Benvenutti-Ferreira, G, Schirmer, J, Luchesse OA. Survival, growth and seed mass in a mixed tree species planting for Atlantic Forest restoration. AIMS Environ Sci. 2016;3(3):382-94.

[30] Martínez-Camilo R, González-Espinosa M, Ramírez-Marcial N, Cayuela L, Pérez-Farrera MÁ. Tropical tree species diversity in a mountain system in southern Mexico: local and regional patterns and determinant factors. Biotropica. 2018 May;50(3):499-509.

[31] Laurans M, Vincent G. Are inter- and intraspecific variations of sapling crown traits consistent with

a strategy promoting light capture in tropical moist forest? Ann Bot. 2016 Oct;118(5):983-96.

[32] Kitajima K. Relative importance of photosynthetic traits and allocation patterns as correlates of seedling shade tolerance of 13 tropical trees. Oecologia. 1994 Aug;98(3-4):419-28.

[33] Wilson SJ, Rhemtulla JM. Acceleration and novelty: community restoration speeds recovery and transforms species composition in Andean cloud forest. Ecol Appl. 2016 Jan;26(1):203-18.

Chapter 3

Thinning: An Overview

Ana Cristina Gonçalves

Abstract

Thinning is one of the primordial silvicultural practices. It has been analysed by its methods and intensities, associated to the tree selection criteria. Yet, while some methods are of generalised use, others were developed for specific purposes. The goal of this review is to compile the existing information regarding tree selection, thinning methods and intensity as well as their effects on trees and stands. The effects of thinning indicate a reduction of density and a trend towards an increase of growth rates at tree level for a short time after thinning. Biomass and volume show similar or smaller values when compared to unthinned stands. Mortality and growth stagnation, especially in stands with low stability or vigour, can also occur. The modifications in stand structure can enhance its role as an adaptive measure.

Keywords: method, intensity, stand structure, growth, adaptive measure

1. Introduction

Stand and forest management encompasses a set of silvicultural practices which are designed according to its goals. Among these, thinning is of primordial importance as it influences stand structure, tree and stand growth, products, yields and diversity.

In time, the trees of a stand occupy gradually the available growing space, developing simultaneously facilitation and competitive interactions [1]. The balance between these two interactions is dynamic, but competition increases with the decrease of the growing space. The result is that individuals with competitive advantages reallocate the growing space formerly occupied by other individuals with less competitive advantages and suppress them. This originates from the development of a social structure which, when growing space is fully occupied, derives in the death of the suppressed individuals, that is, self-thinning [2].

Thinning implies always the removal of trees with the main goal of allocating the growing space to those better suited to the desired productions and yields [2–6]. The removal of trees can have both positive and negative effects. The positive, are related to the reduction of competition, anticipation of volume losses due to self-thinning, increase of diameter growth rate, increase of timber value and revenue and reduction of the damages due to the abiotic and biotic disturbances. The negative, are associated with the reduction of total volume, risk of mortality or growth stagnation, cost of the operation, damages in the remaining trees and risk of damages by abiotic and biotic agents [2, 6].

In literature, thinning has been analysed according to its method and intensity as well as with the tree selection criteria. Yet, while some methods are of generalised use, others were developed for specific purposes. The main goal of this review is to compile the existing information regarding tree selection, thinning methods and

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intensity as well as their effects on trees and stands. The chapter is organised in four sections. Section 2 describes and characterises tree selection criteria. Section 3 analyses the thinning methods and intensity. Section 4 analyses the effects of thinning on stand structure, growth, products and as an adaptive measure.

2. Tree selection

Tree selection plays one of the key roles in thinning, as one of its main objectives is the reallocation of growing space to a set of trees in the stand. Care has to be taken so that the trees maintained in the stand are able to use the growing space made available [2, 6, 7]. Thus, it has to be thought at two complementary levels: (i) at tree level, reallocating the growing space to the trees kept so that they reach the desired growth rates, yields and product quality and (ii) at stand level, optimising yield for the desired production cycle, which is also related to density, spatial arrangement and site quality. These levels derive from two tree development traits, namely, the intrinsic and the external. The former is mainly driven by genetics, preponderant when trees grow isolated. In the latter, the growing space availability determines trees development [1, 2, 8]. These complementary objectives enable balancing interactions to achieve growing space use optimization and improve the overall stand quality and yield.

The need of selecting trees enhanced the development of tools to evaluate growing stock, stability, potential photosynthetic ability and growth rate [1–3, 6, 9]. One of the mostly used is the tree classification system. Their main advantages are that tree and stand description, evaluation and monitoring (both spatial and temporal) can be carried out with a set of qualitative criteria, needless of forest inventories. The stand evaluation is quick with low costs and helps to implement silvicultural practices. The disadvantages are related to their development or adaptation to stand structure and management goals, to not enabling a quantitative evaluation and to the need of skilled practitioners [2, 6–8, 10].

Due to the variety of stand structures and management goals, many tree classification systems were developed. They evolved in time, increasing in complexity, as more criteria were included to increase accuracy and precision [7, 8, 11]. Typically, tree classification systems are grouped in two broad classes according to the stand structure and production goals. One is directed towards pure even-aged stands with one main production (timber), for example, of kraft [10], of 1902 [10], English [12], of Assmann [2], Belgian [3], of Meadows and Skojac [11]. The other is directed to pure or mixed uneven-aged systems with one main production or several ones, for example, of Assmann [2], of Florence [8], of IUFRO [7], of Meadows [13] and of Perkey classification [14].

The concept of future trees is related to tree characteristics, moment of selection, number of trees per unit area and their spatial arrangements.

The criteria associated with the future tree characteristics referred in independent studies (e.g. [3, 6, 7, 15–19]) are similar, and eight criteria can be pointed out: (i) vigour and good sanitary conditions, (ii) social position (dominant or codominant), (iii) suitability of the species mixture, (iv) vertical straight stem, (v) without stem deformation (forks) up to 6–8 m for timber and 2–4 m for bark and fruit production, (vi) wood without serious defects, (vii) final ramification, few small branches in the case of timber production and to be promoted to increase production in the case of bark and fruit and (viii) balanced crown. These criteria can be totally or partially used depending on the management and production goals.

The moment to designate the future trees is not consensual, however, some guidelines have been reported [3, 7, 17, 19]: (i) social position maintenance – young trees have higher probability of social regression than adult ones; (ii) species tolerance to shade – shade-tolerant trees are able to live suppressed and, after release, are able to ascend to dominant positions whether intolerant are not; (iii) low risk of sudden death or break and free of wounds – trees should be vigorous, stable, in good sanitary conditions and without injuries; (iv) stability – trees should be stable enough so that after release they are able to develop with low probability of falling down and (v) longevity – it should be ensured that they are able to reach the end of the production cycle. The selection should then be made as earlier as possible, as soon as the probability of changing social status is low. It can be done when trees reach 10–25 cm of diameter at breast height [3, 17], or 20 m of dominant height [17] or 10–14 m of stem height [17]. When it is convenient to designate future trees very early in time, a preselection of the future trees is recommended, followed by their selection later, for example, after 30–40 years [3, 4, 7].

The number of trees per unit area is determined by the release from the competition of the future trees during the entire production cycle, to optimise their development. A density between 80 and 250 trees ha^{-1} is suggested [4, 7]. The better the site quality and the shorter the production cycle, the higher their number. The larger the crowns and the lower the shade tolerance, the lower their number [7].

The spatial arrangement of the future trees should be uniform to enable a more efficient and complete use of the growing space while maintaining the growth rate at highest desired levels, which corresponds to a mean spacing of ≈7–12 m, depending on the species ecological and cultural characteristics [4, 7].

3. Thinning method and intensity

Thinning method or type can be defined by the classes and social position of the removed trees, although other parameters such as stem and crown characteristics are also important [2–6]. Nine methods have been identified, namely from below, from above, selective or Schädelin, of dominants, mechanical, free, compensation, crown release and variable density thinning.

Thinning from the below main goal is to favour the best trees of the upper layer, of better dimensions and crowns. The removal of the individuals starts with the dead, dying and dominated, and only if necessary the codominant and dominant individuals (mainly, individuals of bad characteristics) are removed. It has low effect in the subsequent growth of the remaining stand. Thus, it only anticipates the normal pattern of tree senescence and dead in an unthinned stand. It is suited for sites where water is a limiting factor [2–6]. The best results are attained with intolerant species, where the stems of the inferior layers do not have or only have a limited reaction to release [6].

Thinning from the above main goal is to favour the best trees of the upper layer until the end of rotation. The trees to be removed are predominantly in the upper layer and in direct competition with the best trees. The inferior layers are maintained with the objectives of enhancing natural pruning, soil protection, reducing spontaneous vegetation development, increasing resistance to wind and maintaining or enhancing wildlife habitat. However, the tree removal in the inferior layers can be considered for aesthetical reasons or to reduce the risk of fire, creating vertical discontinuity [2–6]. It is better suited for shade or semi-shade-tolerant species, in pure and mixed stands, especially when quality trees are found in an adequate number in the superior layers. It is not suited for shade-intolerant species, especially in the later stages of development [6].

Selective or *Schädelin thinning's* main assumption is the selection of the future trees. They can be selected in all social classes, according to a set of criteria (cf. Section 2). The thinning is focused on the release of the future trees, with the removal of all competitors and the maintenance of trees that can be useful or do not interfere with them. Also, the future trees should have, as much as possible, a uniform spacing. Its selection is not static in time, especially in young development stages. Thus, before each thinning they have to be checked and, if necessary, reselected [7, 20, 21]. The main goal of this thinning is the optimization of the production in value rather than in volume, favouring at the same time the mechanical and ecological stability [7, 20].

Thinning of dominants is focused on the upper layers. The dominant and the codominant trees are removed, including the more promising and those of the intermediate and inferior layers are favoured. It is suited for a reduced set of objectives, and care should be taken so that it does not derive in the harvest of the best trees. Three approaches can be considered, as a function of the objectives and number interventions [6]:

i. Thinning of dominants *with temporary character*: The goal is to improve the overall stand, with the promotion of lower layers that have individuals with good characteristics, both in growth and quality. It is suited for stands where the irregular or low density has originated dominants of bad quality; for shade-tolerant species, as the stems in the lower layers maintain their vigour and ability to react to release and it is less suited for intolerant species, yet it can be used in young stands where trees have not lost their vigour. It should be done as earlier as possible and should be replaced by the thinning from below as soon as the trees reach the superior layer.

ii. Thinning of dominants *with permanent character*: The goal is the production of small- and medium-dimension timber. The objective is to promote canopy gaps that enhance regeneration with the largest possible number of individuals with the removal of dominant trees. When the stand is dense and uniform, this method is replaced by the thinning from below. The rotation length is considerably shortened.

iii. Thinning of dominants *combined with thinning from below*: The goal is forming a superior layer with codominant individuals. It minimises the negative effects of the thinning of dominants, especially in very dense unthinned stands. Its main disadvantage is the tendency to increase the losses due to biotic and abiotic disturbances.

Mechanical or *geometric thinning* is associated to large spacing silviculture with selected material, in which the removal of a tree is more related with its location than with its position, because the goal is to maintain a regular cover. It is advantageous in young, very dense unthinned stands. Two subtypes can be identified: *spacing*, where the trees at a certain distance of the selected tree are removed; and *row* or *strip*, where the individuals of one or several lines are removed [6].

Free thinning goal is the selection of a set of trees which are maintained in free growth until the end of the rotation in order to produce high quantity of timber with high quality [22]. It is directed to oak species and rotations of less than 100 years. It begins with the selection of the future trees with a density of 60–80 trees ha^{-1}, uniformly spaced. It is followed by the removal of all the trees whose crown distance from a future tree is less than 25% of the mean crown width (assumption to be maintained throughout rotation, to keep future trees in free growth). As a secondary silvicultural practice, pruning is recommended up to a height of 6 m as well as removing the epicormic branches [22, 23]. It has also been used in mixtures of conifers and oaks, and oaks in pure or mixed stands with *Fraxinus excelsior*, *Acer pseudoplatanus* and *Prunus avium* [24].

Compensation thinning's (*éclairci de ratrapage*) aim is favouring the future trees that have sufficient stability and well-balanced crowns, being less important in their spatial distribution and their optimal distance. It is frequently linked with the goal of keeping stand stability and should have light intensity. It is preferred in stands without or with thinnings of low intensity in the past. In these cases, only the dominant stems are able to react to release, and thus codominant individuals should be preferably removed. It is especially suited for stands in steep slope areas where it is easy to overestimate distances due to crown overlapping and asymmetry [7].

Crown releasing thinning's (*éclairci misse en lumiére*) main goal is regulating future trees in old growth development stage. The trees' metabolism and growth, capacity of reaction to release and ability to redo their crowns are lower than in the mature ones. The social positions are nearly definite, and the probability of individuals of the inferior social classes to ascend to upper classes is low. Future trees are released from competition from those of the intermediate layers as the dominant competitors were removed in the former thinnings. The objective is to maintain or increase diameter growth, to allow the largest possible increase of productivity in value. The intensity should be light and periodicity should be long, according to the trees' growth rates [7].

Variable density thinning's goal is to promote variability and heterogeneity, both spatial (horizontal and vertical) and structural [25–28], as well as stimulate late-successional forest structures, reduce stand density, alter species composition [25, 26, 29] and be a restauration tool [27, 30, 31]. It assumes the unevenly removal of trees, creating gradients of density in the stand. This is implemented with the creation of patches with variable spatial distribution, where canopy gaps and patches of different densities coexist [26, 27, 32]. The proportion of each patch type is also variable according to the intended complexity of stand structure, the existing stand structure, the species composition and spatial arrangement [31, 32]. Six protocols are referred, which due to their similarities were grouped in four types [32]:

i. *Randomised grid*: Stand area is divided in a grid with cells of equal area and the number of individuals to be maintained is randomly sorted. Two target densities can be chosen, low (≈185 trees ha^{-1}) and moderate (≈370 trees ha^{-1}).

ii. *Dx rule*: Selected trees define density depending on site variability and tree dimensions. The area of influence of each selected tree is defined by a circle proportional to the diameter at breast height by k-fold (e.g. k = 2). In this area, trees between a diameter range are removed, while those outside it are kept. The upper and lower thresholds can be defined per stand or per species. The areas outside the circles are not thinned.

iii. *Spacing thinning*: Stand area is divided in a square point grid (e.g. ≈5 × 5 or 6 × 6 m) as well as a buffer for each point (e.g. ≈1.2 m). For each buffer area, the best tree larger than a diameter threshold is selected and released from the competition.

iv. *Localised release*: Stand area is divided in a point square grid associated to a buffer (e.g. 7.6 m of radius). In each circular area, three trees are selected irrespective of their spatial arrangement with the same rules as the former type, while the areas outside the buffers are either thinned with about 3.6 m spacing in the row space outside the buffers or remain unthinned (e.g. third or fourth row).

Thinning *intensity* or *degree* is more frequently evaluated by the number of trees or basal area, as function of the amount of removed in relation of the total number of stems ($RN = \frac{Nrem}{Nt}$, where Nrem is the number of removed trees, and Nt is the total number of trees) or basal area ($RG = \frac{Grem}{Gt}$, where Grem is the basal area removed, and Gt is the total basal area). It is frequently grouped in three classes: light, moderate and heavy. The most consensual ranges for light intensity are ≤25% of the number of trees and <20% of the basal area, for moderate 50% of the number of trees and 20–35% of the basal area and for heavy >50% of the number of trees and >35% of the basal area [33, 34].

4. Thinning effects

The main goals of thinning are to improve residual trees' efficiency, encompassing to concentrate growth on a selected subset of trees, thus controlling density and reallocating the growing space and reducing competition among trees while promoting their growth [35–39]. It is also considered as a mean of capturing tree mortality, providing early financial return, increasing future merchantable volume and financial value of timber [39–41]. Moreover, density threshold (or thinning intensity) enables to keep volume growth [42], which depends too on the species, stand development stage and site [41, 43, 44]. The thinning effects will be discussed for density, stand structure, stability, mortality and growth stagnation, growth, wood, biomass and carbon stocks, soil and understorey and as an adaptive measure.

Regardless of the method and intensity of thinning, *density* decreases always. All thinning methods increase individual tree growth due to the increase in growing space and reduction of competition [45–48], especially in the long term [47]. The method affects differently tree interactions post thinning. In the thinning from below, the interactions in the upper layer are maintained [7, 49], while in the thinning from above, compensation and crown releasing thinning are reduced [7, 34, 50]. In the thinning of dominants, there is a change in the interactions in the intermediate layer, which is a consequence of the removal of the upper layer [6, 51, 52]. In the free and mechanical thinnings, the spatial pattern of interactions is kept [6, 22, 23]. In the Schädelin and variable density thinnings, future trees are released from competition, resulting in the alteration of the spatial patterns of interaction, both horizontal and vertical [7, 32].

In general, the heavier the thinning intensity, the higher the decrease of density and the lower the competition. This derives in different growth reactions post thinning, which is also related to stand development stage and site quality, with growth rates increasing with thinning intensity and site quality and decreasing from initial to old growth development stages [41, 43, 50, 53].

Stand structure is affected by thinning, producing more or less accentuated changes depending on its method and intensity. This changes with the increase of light, water and nutrients levels (e.g. [54, 55]). These alterations can be observed in the diameter and height distributions, canopy stratification and spatial arrangements of trees. Thinning from below and of dominants narrow the range of diameter and height distributions and decrease canopy stratification due to the preferential removal of trees with smaller and larger dimensions, respectively [52]. Thinning from above and of Schädelin keep the range of diameter and height distributions and maintain or increase canopy stratification [20, 39, 52, 56, 57]. Compensation and crown releasing, free and mechanical thinnings tend to keep diameter and height distribution ranges and canopy stratification [6, 7, 23]. Variable density thinning increases diameter and height distribution ranges and maintains or increases canopy stratification [29, 58–60]. The tree spatial arrangements after thinning depend on those prior to thinning. A regular spacing was reported for thinnings from below, while from above and of dominants have been referred as a trend towards cluster one at low and intermediate distances [52]. Schädelin thinning tends to derive in a uniform or cluster distribution when future trees are at uniform distances or in clusters [20].

Stand stability depends on individual tree morphology and their spatial arrangement. It is frequently evaluated by diameter at breast height, total height, *hd* ratio (defined as quotient between total height and diameter at breast height, with both variables in the same units), stem taper, crown dimensions, crown eccentricity and crown inclination as well as root architecture [61–63]. For the same diameter at breast height, the taller the total height, the higher the *hd* ratio and the lower the stability. The increase of stability can be achieved with thinning, as it promotes diameter growth, the *hd* ratio reduction and stem taper increase [21, 61, 63]. The crown dimensions (width and length), eccentricity and inclination depend on stand structure and species traits, which are determined primordially by the amount of light. The higher the light level and the wider the spacing, the higher crown volume, and the higher the shade tolerance, the higher the crown dimensions. The constellation of neighbours reducing or promoting irregular available aerial space can promote the development of eccentric crowns and stem inclination and thus reducing stability [64, 65]. Stability is attained more efficiently with thinnings at younger ages as it enables a more favourable above- and below-ground morphology [21, 66]. Also, heavy thinnings promote tree morphologies that are more stable than moderate or light ones [21, 41, 66]. Thinnings from below increase stability by the removal of trees with the less suited morphologies (e.g. higher *hd* ratio), and increase with the increase of intensity due to the reduction of *hd* ratio and increase of stem taper [37, 67, 68] and crown length and crown ratio (the coefficient between crown length and total height) [69, 70]. Thinnings from above and of dominants removing trees from the upper layer may decrease stability, especially when associated to trees with high *hd* ratio and due to unbalance of aerial/root systems, eccentric crowns and the swaying of trees [35, 52, 71]. Schädelin, variable density and free thinning maintain or improve stability of thinning as the trees more stable are selected [20, 22, 32, 72]. Compensation and crown release thinning maintain stability [7].

Mortality after thinning can be caused by increased tree swaying due to wind or snow [71, 73] or is associated to shallow root systems' water stress [35, 36]. In general, it is higher in the thinning from above and of dominants than from below [52] or Schädelin [72]. *Growth stagnation* [74] is linked to the reduction of growth increment [75, 76], sometimes associated to drought events [77–79].

The thinning effects on *growth* are related to a suite of factors such as method and intensity, age, species traits, density (cf. density section), stand structure (cf. stand structure section) and time after thinning.

In general, thinning increases growing space and favours certain classes of trees. This results in asymmetric competition [80, 81], that is, the share of resources used by larger trees is disproportionally larger than those used by smaller trees, resulting in the growth suppression of the latter [35, 80]. Thus the increase in growth at tree level is maintained or enhanced by the methods where release occurs in the upper layers and/or favours future trees (from above, Schädelin, free, compensation, crown release or variable density thinnings) [7, 20, 23, 30, 31, 39, 59, 60, 82, 83]. This is especially true if the thinning is carried out before canopy closure and crown recession [47]. It can be explained by two factors: the available growing space and the individual tree growth strategies. In closed canopy stands, the upper layers absorb most radiation, the taller trees cast shade on their smaller neighbours and tree swaying may derive branch abrasion [1]. These three factors promote height growth and constrain crown lateral development [84]. In fact, it has been reported that the dominant height increases with density for broadleaved species [41, 85], due its effects on epinastic control and specificities of stand development at early ages [41]. The inverse has been reported for the conifers [86, 87].

Thinning intensity influences directly density and thus the availability of growing space for individual stems, which in turn affects height, stem and crown growth [69, 88, 89]. Growth, at tree-level basis, increases with thinning intensity [37, 50, 71, 89, 90]. In general, smaller/medium and/or younger trees react faster and with higher growth rates than larger ones [50, 75, 91]. After moderate and heavy thinnings, especially those carried out early, dominant and codominant trees have higher radial growth than supressed ones [37, 38, 67, 71, 75]. Stand age and site also curtails the response to thinning, the younger the stand and the better site quality, the larger the diameter increments [37]. Moreover, dominant trees have higher diameter growth [37] and need less time to react to release [91]. This is especially true with mature trees that have reached their maximum growth potential [35, 71]. However, a positive trend in above-ground biomass has been reported for large trees [92–94]. This trend seems to be linked with shade tolerance. Shade-intolerant species increase growth with light increase, though reaching maximum annual growth at younger ages conversely to shade-tolerant species [35, 77]. Also, according to Bose et al. [35], thinning intensity plays a less important role in growth increase after thinning in very shade-tolerant species than in shade-intolerant ones as the former are not able to use the increased growing space after thinning (especially light) efficiently.

Differences in tree reaction to thinning with age also depend on the stand history. While in unthinned stands, recovery decreased with stand age, it did not decrease in thinned stands [95], at least partially explained, by the larger crowns in thinned stands that enable a faster recovery of growth [95, 96].

After thinning, the trees increase their growth, frequently 1–3 years after thinning [50, 82] due to the availability of growing space. The growth increase derives in the increase of crown volume (width and length), crown cover [90, 97] and foliar mass enhancing photosynthetic capacity, as the lower parts of the crowns receive more light than unthinned stands [98]. As trees occupy gradually the available growing space, the growth rate decreases [35, 36, 99] after reaching the maximum (about 3 years after thinning), attaining 7–8 years after thinning, growth levels similar to the unthinned stands [50]. Primicia et al. [99] reported that magnitude and duration of thinning effects on growth (stem and crown diameter) as well as on mortality seem to be more related to thinning intensity than with thinning method.

Wood quantity and *quality* can be improved by thinning. In general, quantity per tree increases with thinning intensity [100], though it depends also on the species and their ecological and cultural traits. Light thinnings favour more regular growth rings, especially interesting for timber, though with overall smaller diameter growth, while heavy thinnings promote larger annual growth rings [77, 101]. The counterpart of thinning is the development of large branches that reduce the wood quality. Consequently, a compromise has to be equated between low and high densities as function of the species and its traits (e.g. epinastic control and natural pruning ability), associated frequently to future tree selection, pruning and early silvicultural operations [21, 37, 38, 83], especially in what regards wolf trees, which should be removed as early as possible [102].

In general, thinning reduces *biomass* and *carbon storage* when compared with unthinned stands, the decrease is higher in thinning from above and of dominants [39, 103] than from below [104, 105]. Yet, the effects of thinning are dependent also on the individual tree development stage, their growth rates and density after thinning [106, 107]. In the short term, thinning of young trees results in a reduction of above-ground carbon even if there is an increase of the individual tree's growth rate, because they are not able to use all the growing space available, that is, they do not fully occupy the site [104, 108]. In general, increasing thinning intensities result in decreasing standing and deadwood biomass and literfall [109, 110]. At stand level, it seems that biomass and carbon storage is the result of the interaction between the density and size of the overstorey trees. Though with an inverse relationship, they balance each other, resulting in a rather constant above-ground carbon stock, regardless of whether stands are thinned or not [39, 41], especially with early thinning from below [41].

Soil and *understory* vegetations are affected by thinning. If biomass residues are kept in the stand, their decomposition incorporates carbon in the soil. Thinning increasing decomposition rates decreases the soil carbon stocks. In the 0–10 cm of the soil layer, carbon stock is higher than in the 10–20 cm layer [39, 40], due to higher decomposition rates [111]. Yet, with time and tree growth, soil carbon stocks tend to be similar in thinned and unthinned stands [112]. Zhang et al. [39] reported that soil carbon stocks prior and 5 years after thinning were similar. Thinnings originate higher light levels in the lower storeys, which can result in higher transpiration and water loss by evaporation (e.g. [54, 55]) and increase of the understorey vegetation, the higher the thinning intensity [39]. This is particularly negative if it is composed mainly by shrub vegetation [39]. Yet, heavy thinnings favour pasture production, a suitable option for agroforestry systems [90, 113].

Thinnings are considered primordial adaptive measures as they can reduce vulnerability to climate change, fires, droughts and increase diversity.

Thinning can reduce vulnerability to climate change [95] as it controls stand density [41]. It can improve tree and stand growth by releasing growing space (e.g. [41, 77, 114]), including increasing water availability and their use efficiency [77, 101], thus mitigating the effects of the droughts (i.e. water deficits) [77, 95, 114, 115].

Fire prevention is enhanced by thinnings (including also pruning) as they reduce the quantity and horizontal and vertical continuity of fuel, [116] enabling stands to withstand surface fires [117] and increase the canopy seed bank storage [118].

Thinning intensity has a primordial effect on the magnitude and duration of the drought effects on trees and stands. The higher the proportion of crown cover removed, the longer the effects of thinning, that is, more water reaches the soil, enabling drought effects' mitigation [95, 101, 119]. Less-intensive thinnings reach pre-thinning transpiration levels in a few years, [55, 114] while in heavy ones they last longer [119], occasioning tree growth rates' increase [95, 101, 120]. Broadleaved species seem to have developed resistance mechanisms, mitigating diameter growth reduction [121]. This can be due to the deeper root systems [122] and spring radial growth (especially in the ring-porous species) is much larger than in the autumn one [123]. Conifers seem to have improved recovery and resilience mechanisms, probably due to more precipitation reaching the soil and transpiration reduction [95, 121]. Regardless of tree species, the stronger the drought severity, the longer the recovery period in diameter growth [95], which is probably related to the longer period needed to restore the soil water and to rebuild fine root system [124]. Thus, species with higher expansion rates, increase of leaf area (assimilation ability) and fine roots (water absorption ability), are expected to have more benefits from thinning [115, 125].

An increased diversity in stand structure, especially in pure even-aged stands, particularly in plantations, can be derived from thinning in tree' dimensions [126] and/or their variability [20, 39, 52, 57] as well as in species and their proportions [127] and produces greater trade-offs with other ecosystem services [128, 129]. This is especially valid with methods that promote variability, such as Schädelin or variable density thinnings.

5. Conclusion

Stand structure and production goals influence the thinning method and its intensity, which in turn affects stand structure and the quantity and quality of the products. Thus, it is of primordial importance the selection of the most suitable methods (that can be more than one during the production cycle) as well as their intensities (which can vary too along the production cycle) that have also to be suited to the products and services desired to the forest stand. Thinning is frequently linked to tree selection. Tree classification systems are quick, low-cost tools that enable thinning implementation and are also a monitoring tool that enables the evaluation of the dynamics of the forest stands.

All thinnings reduce density, however, their effects on density, stand structure, growth, soil, understorey vegetation and diversity depend on the method and intensity of thinning, stand development stage and site quality. The positive effects of thinning are the increase in growth and production, especially in value, and the reduction of the vulnerability of the forest systems to climate change, droughts and fire. The negative effects are related to the reaction of the trees to release, which can cause mortality, growth stagnation or no increase of the growth rates.

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References

[1] Oliver CD, Larson BC. Forest Stand Dynamics. Update ed. New York: John Wiley & Sons, Inc; 1996. pp. 544

[2] Assmann E. The Principles of Forest Yield Study. Pergamon Press; 1970. pp. 506

[3] Boudru M. Forêt et Sylviculture. Le Traitement des Forêts. Vol. Tome 2. Gembloux: Presses Agronomiques de Gembloux; 1989. pp. 344

[4] Lanier L, Badré M, Delabraze P, Dubourdieu J, Flammarion JP. Précis de Sylviculture. Engref: Nancy; 1986. pp. 468

[5] Matthews JD. Silvicultural Systems. Oxford: Claredon Press; 1989. pp. 284

[6] Smith DM, Larson BC, Kelty MJ, Ashton PMS. The Practice of Silviculture. Applied Forest Ecology. 9th ed. New York: John Wiley & Sons, Inc.; 1997. pp. 560

[7] Schütz JP. Sylviculture 1. Principes d'Éducation des Forêts. Lausanne: Presses Polytechniques et Universitaires Romandes; 1990. pp. 245

[8] Florence RG. Ecology and Silviculture of Eucalyptus Forests. Csiro Publishing; 1996. pp. 413

[9] O'Hara KL. Multiaged Silviculture Managing for Complex Forest Stand Structures. Oxford: Oxford University Press; 2014. pp. 213

[10] Pretzsch H. Forest dynamics, growth, and yield. Berlin: Springer; 2009. pp. 664

[11] Meadows JS, Skojac DAA. New tree classification system for Southern Hardwoods. Southern Journal of Applied Forestry. 2008;**32**(2):69-79

[12] Kerr G, Haufe J. Thinning Practice—A Silvicultural Guide. 2011. p. 54

[13] Meadows JS, Burkhardt EC, Johnson RL, Hodges JD. A numerical rating system for crown classes of southern hardwoods. Southern Journal of Applied Forestry. 2001;**25**:154-158

[14] Perkey AW, Wilkins BL, Smith HC. Crop Tree Management in Eastern Hardwoods. USDA, Forest Service; 1993. pp. 58 (NA-TP-19-93)

[15] Gonçalves AC. Modelação de povoamentos adultos de pinheiro bravo com regeneração de folhosas na Serra da Lousã. Lisboa: Instituto Superior de Agronomia, Universidade Técnica de Lisboa; 2003

[16] Gonçalves AC. Estabelecimento de ensaios para avaliação do efeito do primeiro desbaste no desenvolvimento de montados de sobro. 1996

[17] Oswald H. Résultats principaux des places déxpérience de chêne du centre national de recherches forestières. Revue Forestière Française. 1981;**XXXIII** (no sp):65-85

[18] Polge H. Prodution de chênes de qualité. Revue Forestiére Francaise. 1984;(no sp):34-48

[19] Roy FX. La désignation des arbres de place dans les futaies de chêne destinées à fournir du bois de tranchage. Revue Forestiére Francaise. 1975;**XXVII**(1):50-60

[20] Boncina A. Comparison of structure and biodiversity in the Rajhenav virgin forest remnant and managed forest in the Dinaric region of Slovenia. Global Ecology and Biogeography. 2000;**9**(3):201-211

[21] Cameron AD. Importance of early selective thinning in the development of

long-term stand stability and improved log quality: A review. Forestry. 2002;**75**(1):25-35

[22] Jobling J, Pearce ML. Free-growth of oak. Forest Record. 1977;**113**:3-17

[23] Kerr G. The effect of heavy or "free growth" thinning on oak (*Quercus petraea* and *Q. robur*). Forestry. 1996;**69**(4):303-317

[24] Kerr G, Evans J. Growing broadleaves for timber. In: Forestry Commission Handbook. London: H.M.S.O; 1993. pp. 95

[25] Carey AB. Active and passive forest management for multiple values. Northwestern Naturalist. 2006;**87**(1):18

[26] Carey AB. Biocomplexity and restoration of biodiversity in temperate coniferous forest: Inducing spatial heterogeneity with variable-density thinning. Forestry. 2003;**76**(2):127-136

[27] Comfort EJ, Roberts SD, Harrington CA. Midcanopy growth following thinning in young-growth conifer forests on the Olympic Peninsula western Washington. Forest Ecology and Management. 2010;**259**(8):1606-1614

[28] Sullivan TP, Sullivan DS, Lindgren PMF, Ransome DB. Stand structure and small mammals in intensively managed forests: Scale, time, and testing extremes. Forest Ecology and Management. 2013;**310**:1071-1087

[29] Puettmann K, Ares A, Burton J, Dodson E. Forest restoration using variable density thinning: Lessons from Douglas-Fir stands in Western Oregon. Forests. 2016;**7**(12):310

[30] O'Hara KL, Nesmith JCB, Leonard L, Porter DJ. Restoration of old forest features in Coast Redwood Forests using early-stage variable-density thinning. Restororation Ecology. 2010;**18**:125-135

[31] Roberts SD, Harrington CA. Individual tree growth response to variable-density thinning in coastal Pacific Northwest forests. Forest Ecology and Management. 2008;**255**(7):2771-2781

[32] O'Hara KL, Leonard LP, Keyes CR. Variable-density thinning and a marking paradox: Comparing prescription protocols to attain stand variability in Coast Redwood. Western Journal of Applied Forestry. 2012;**27**(3):143-149

[33] del Río M, Bravo-Oviedo A, Pretzsch H, Löf M, Ruiz-Peinado R. A review of thinning effects on Scots pine stands: From growth and yield to new challenges under global change. Forest Systems. 2017;**26**(2):eR03S

[34] Gradel A, Ammer C, Ganbaatar B, Nadaldorj O, Dovdondemberel B, Wagner S. On the effect of thinning on tree growth and stand structure of White Birch (*Betula platyphylla* Sukaczev) and Siberian Larch (*Larix sibirica* Ledeb.) in Mongolia. Forests. 2017;**8**(4):105

[35] Bose AK, Weiskittel A, Kuehne C, Wagner RG, Turnblom E, Burkhart HE. Does commercial thinning improve stand-level growth of the three most commercially important softwood forest types in North America? Forest Ecology and Management. 2018;**409**:683-693

[36] Kuehne C, Weiskittel AR, Wagner RG, Roth BE. Development and evaluation of individual tree- and stand-level approaches for predicting spruce-fir response to commercial thinning in Maine, USA. Forest Ecology and Management. 2016;**376**:84-95

[37] Mäkinen H, Isomäki A. Thinning intensity and long-term changes in increment and stem form of Scots pine trees. Forest Ecology and Management. 2004;**203**(1-3):21-34

[38] Peltola H, Kilpeläinen A, Sauvala K, Räisänen T, Ikonen V-P. Effects of early thinning regime and tree status on the radial growth and wood density of Scots pine. Silva Fennica. 2007;**41**(3):489-505

[39] Zhang H, Zhou G, Wang Y, Bai S, Sun Z, Berninger F, et al. Thinning and species mixing in Chinese fir monocultures improve carbon sequestration in subtropical China. European Journal of Forest Research. 2019;**138**(3):433-443

[40] Strukelj M, Brais S, Paré D. Nine-year changes in carbon dynamics following different intensities of harvesting in boreal aspen stands. European Journal of Forest Research. 2015;**134**(5):737-754

[41] Trouvé R, Bontemps J-D, Collet C, Seynave I, Lebourgeois F. When do dendrometric rules fail? Insights from 20 years of experimental thinnings on sessile oak in the GIS Coop network. Forest Ecology and Management. 2019;**433**:276-286

[42] Zeide B. Thinning and growth. Journal of Forestry. 2001:20-25

[43] Giuggiola A, Ogée J, Rigling A, Gessler A, Bugmann H, Treydte K. Improvement of water and light availability after thinning at a xeric site: Which matters more? A dual isotope approach. New Phytologist. 2016;**210**(1):108-121

[44] Taeger S, Zang C, Liesebach M, Schneck V, Menzel A. Impact of climate and drought events on the growth of Scots pine (*Pinus sylvestris* L.) provenances. Forest Ecology and Management. 2013;**307**:30-42

[45] Dwyer JM, Fensham R, Buckley YM. Restoration thinning accelerates structural development and carbon sequestration in an endangered Australian ecosystem: Restoration thinning in natural regrowth. Journal of Applied Ecology. 2010;**47**(3):681-691

[46] Hoover C, Stout S. The carbon consequences of thinning techniques: Stand structure makes a difference. Journal of Forestry. 2007;**105**(5):266-270

[47] Horner GJ, Baker PJ, Nally RM, Cunningham SC, Thomson JR, Hamilton F. Forest structure, habitat and carbon benefits from thinning floodplain forests: Managing early stand density makes a difference. Forest Ecology and Management. 2010;**259**(3):286-293

[48] Schaedel MS, Larson AJ, Affleck DLR, Belote RT, Goodburn JM, Page-Dumroese DS. Early forest thinning changes aboveground carbon distribution among pools, but not total amount. Forest Ecology and Management. 2017;**389**:187-198

[49] Lei X, Lu Y, Peng C, Zhang X, Chang J, Hong L. Growth and structure development of semi-natural larch-spruce-fir (*Larix olgensis–Picea jezoensis–Abies nephrolepis*) forests in northeast China: 12-year results after thinning. Forest Ecology and Management. 2007;**240**(1-3):165-177

[50] Juodvalkis A, Kairiukstis L, Vasiliauskas R. Effects of thinning on growth of six tree species in north-temperate forests of Lithuania. European Journal of Forest Research. 2005;**124**(3):187-192

[51] Bradford JB, Palik BJ. A comparison of thinning methods in red pine: Consequences for stand-level growth and tree diameter. Canadian Journal of Forest Research. 2009;**39**(3):489-496

[52] Kuehne C, Weiskittel A, Pommerening A, Wagner RG. Evaluation of 10-year temporal and spatial variability in structure and growth across contrasting commercial thinning treatments in spruce-fir forests of northern Maine, USA. Annals of Forest Science. 2018;**75**(1):20

[53] Pretzsch H. Stand density and growth of Norway spruce (*Picea abies* (L.) Karst.) and European beech (*Fagus sylvatica* L.): Evidence from long-term experimental plots. European Journal of Forest Research. 2005;**124**(3):193-205

[54] Brooks JR, Mitchell AK. Interpreting tree responses to thinning and fertilization using tree-ring stable isotopes. New Phytologist. 2011;**190**(3):770-782

[55] Lagergren F, Lankreijer H, Kučera J, Cienciala E, Mölder M, Lindroth A. Thinning effects on pine-spruce forest transpiration in central Sweden. Forest Ecology and Management. 2008;**255**(7):2312-2323

[56] Gagné L, Sirois L, Lavoie L. Comparaison du volume et de la valeur des bois résineux issus d'éclaircies par le bas et par dégagement d'arbres-élites dans l'Est du Canada. Canadian Journal of Forest Research. 2016;**46**(11):1320-1329

[57] Schütz J-P, Ammann PL, Zingg A. Optimising the yield of Douglas-fir with an appropriate thinning regime. European Journal of Forest Research. 2015;**134**(3):469-480

[58] Curtis RO, Harrington CA, Brodie LC. Stand development 18 years after gap creation in a uniform Douglas-Fir plantation. In: Report No.: PNW-RP-610. USDA Forest Service; 2017. p. 38

[59] Davis LR, Puettmann KJ, Tucker GF. Overstory response to alternative thinning treatments in young Douglas-fir forests of Western Oregon. Northwest Science. 2007;**81**(1):1-14

[60] Willis JL, Roberts SD, Harrington CA. Variable density thinning promotes variable structural responses 14 years after treatment in the Pacific Northwest. Forest Ecology and Management. 2018;**410**:114-125

[61] Gardiner B, Blennow K, Carnus J-M, Fleischer P, Ingemarson F, Landmann G, et al. Destructive Storms in European Forests. Joensuu: European Forest Institute; 2010. p. 138

[62] Lindström A, Rune G. Root deformation in plantations of container-grown Scots pine trees: Effects on root growth, tree stability and stem straightness. In: Stokes A, editor. The Supporting Roots of Trees and Woody Plants: Form, Function and Physiology. Dordrecht: Springer Netherlands; 2000. pp. 31-39

[63] Schmidt M, Hanewinkel M, Kändler G, Kublin E, Kohnle U. An inventory-based approach for modeling single-tree storm damage—Experiences with the winter storm of 1999 in southwestern Germany. Canadian Journal of Forest Research. 2010;**40**(8):1636-1652

[64] Binkley D, Campoe OC, Gspaltl M, Forrester DI. Light absorption and use efficiency in forests: Why patterns differ for trees and stands. Forest Ecology and Management. 2013;**288**:5-13

[65] Lee MJ, García O. Plasticity and extrapolation in modeling mixed-species stands. Forest Science. 2016;**62**(1):1-8

[66] Keskitalo E, Bergh J, Felton A, Björkman C, Berlin M, Axelsson P, et al. Adaptation to climate change in Swedish Forestry. Forests. 2016;**7**(2):28

[67] Mäkinen H, Isomäki A. Thinning intensity and long-term changes in increment and stem form of Norway spruce trees. Forest Ecology and Management. 2004;**201**(2-3):295-309

[68] Nilsson U, Agestam E, Ekö P-M, Elfving B, Fahlvik N, Johansson U, et al. Thinning of Scots pine and Norway spruce monocultures in Sweden effects of different thinning programmes on standlevel gross- and net stem volume

production. Studia Forestalia Suecica. 2010;**219**:1-46

[69] Sullivan TP, Sullivan DS. Acceleration of old-growth structural attributes in lodgepole pine forest: Tree growth and stand structure 25 years after thinning. Forest Ecology and Management. 2016;**365**:96-106

[70] Varmola M, Salminen H, Timonen M. Thinning response and growth trends of seeded Scots pine stands at the arctic timberline. Silva Fennica. 2004;**38**(1):71-83

[71] Peltola H, Miina J, Rouvinen I, Kellomäki S. Effect of early thinning on the diameter growth distribution along the stem of Scots pine. Silva Fennica. 2002;**36**(4):813-825

[72] Bachofen H, Zingg A. Effectiveness of structure improvement thinning on stand structure in subalpine Norway spruce (*Picea abies* (L.) Karst.) stands. Forest Ecology and Management. 2001;**145**(1-2):137-149

[73] Peltola H. Swaying of trees in response to wind and thinning in a stand of Scots pine. Boundary Layer Meteorology. 1996;**77**(3-4):285-304

[74] Sharma M, Smith M, Burkhart HE, Amateis RL. Modeling the impact of thinning on height development of dominant and codominant loblolly pine trees. Annals of Forest Science. 2006;**63**(4):349-354

[75] Mäkinen H, Isomäki A. Thinning intensity and growth of Scots pine stands in Finland. Forest Ecology and Management. 2004;**201**(2-3):311-325

[76] Moulinier J, Brais S, Harvey B, Koubaa A. Response of Boreal Jack Pine (*Pinus banksiana* Lamb.) stands to a gradient of commercial thinning intensities, with and without N fertilization. Forests. 2015;**6**(12):2678-2702

[77] Aldea J, Bravo F, Bravo-Oviedo A, Ruiz-Peinado R, Rodríguez F, del Río M. Thinning enhances the species-specific radial increment response to drought in Mediterranean pine-oak stands. Agricultural and Forest Meteorology. 2017;**237-238**:371-383

[78] Fernández-de-Uña L, Cañellas I, Gea-Izquierdo G. Stand competition determines how different tree species will cope with a warming climate. Liang E, editor. PLoS One. 2015;10(3):e0122255.

[79] Martín-Benito D, Del Río M, Heinrich I, Helle G, Cañellas I. Response of climate-growth relationships and water use efficiency to thinning in a *Pinus nigra* afforestation. Forest Ecology and Management. 2010;**259**(5):967-975

[80] Schwinning S, Weiner J. Mechanisms determining the degree of size asymmetry in competition among plants. Oecologia. 1998;**113**(4):447-455

[81] Thomas AD, Walsh RPD, Shakesby RA. Nutrient losses in eroded sediment after fire in eucalyptus and pine forests in the wet Mediterranean environment of northern Portugal. CATENA. 1999;**36**(4):283-302

[82] Dobner M, Nicoletti MF, Arce JE. Influence of crown thinning on radial growth pattern of *Pinus taeda* in southern Brazil. New Forests. 2019;**50**(3):437-454

[83] Liziniewicz M, Ekö PM, Klang F. Effects of five tree-selection strategies when thinning spruce (*Picea abies*) stands: A case study in a field trail in southern Sweden. Scandinavian Journal of Forest Research. 2016;**31**(5):495-506

[84] Trouvé R, Bontemps J-D, Seynave I, Collet C, Lebourgeois F. Stand density, tree social status and water stress influence allocation in height and diameter growth of *Quercus petraea* (Liebl.), Mäkelä A, editor. Tree Physiology. 2015;35(10):1035-1046.

[85] Weiskittel A, Kenefic L, Seymour R, Phillips L. Long-term effects of precommercial thinning on the stem dimensions, form and branch characteristics of red spruce and balsam fir crop trees in Maine, USA. Silva Fennica. 2009;**43**(3):397-409

[86] Anton-Fernandez C, Burkhart HE, Strub M, Amateis RL. Effects of initial spacing on height development of Loblolly Pine. Forest Science. 2011;**57**(3):201-211

[87] Zhao D, Kane M, Borders BE. Growth responses to planting density and management intensity in loblolly pine plantations in the southeastern USA Lower Coastal Plain. Annals of Forest Science. 2011;**68**(3):625-635

[88] Schaedel MS, Larson AJ, Affleck DLR, Belote RT, Goodburn JM, Wright DK, et al. Long-term precommercial thinning effects on *Larix occidentalis* (western larch) tree and stand characteristics. Canadian Journal of Forest Research. 2017;**47**(7):861-874

[89] Valinger E, Sjögren H, Nord G, Cedergren J. Effects on stem growth of Scots pine 33 years after thinning and/or fertilization in northern Sweden. Scandinavian Journal of Forest Research. 2019;**34**(1):33-38

[90] Martínez Pastur G, Soler R, Lencinas MV, Cellini JM, Peri PL. Long-term monitoring of thinning for silvopastoral purposes in *Nothofagus antarctica* forests of Tierra del Fuego, Argentina. Forest Systems. 2018;**27**(1):e01S

[91] Mehtätalo L, Peltola H, Kilpeläinen A, Ikonen V-P. The response of basal area growth of Scots pine to thinning: A longitudinal analysis of tree-specific series using a nonlinear mixed-effects model. Forest Science. 2014;**60**(4):636-644

[92] Sillett SC, Van Pelt R, Kramer RD, Carroll AL, Koch GW. Biomass and growth potential of *Eucalyptus regnans* up to 100 m tall. Forest Ecology and Management. 2015;**348**:78-91

[93] Sillett SC, Van Pelt R, Koch GW, Ambrose AR, Carroll AL, Antoine ME, et al. Increasing wood production through old age in tall trees. Forest Ecology and Management. 2010;**259**(5):976-994

[94] Stephenson NL, Das AJ, Condit R, Russo SE, Baker PJ, Beckman NG, et al. Rate of tree carbon accumulation increases continuously with tree size. Nature. 2014;**507**(7490):90-93

[95] Sohn JA, Saha S, Bauhus J. Potential of forest thinning to mitigate drought stress: A meta-analysis. Forest Ecology and Management. 2016;**380**:261-273

[96] Guiterman CH, Seymour RS, Weiskittel AR. Long-term thinning effects on the leaf area of *Pinus strobus* L. as estimated from Litterfall and individual-tree allometric models. Forest Science. 2012;**58**(1):85-93

[97] Peri PL, Martínez Pastur G, Monelos L. Natural dynamics and thinning response of young lenga (*Nothofagus pumilio*) trees in secondary forests of Southern Patagonia. Bosque Valdivia. 2013;**34**(3):5-6

[98] Aussenac G. Interactions between forest stands and microclimate: Ecophysiological aspects and consequences for silviculture. Annals of Forest Science. 2000;**57**(3):287-301

[99] Primicia I, Artázcoz R, Imbert J-B, Puertas F, Traver M-C, Castillo F-J. Influence of thinning intensity and canopy type on Scots pine stand and growth dynamics in a mixed managed forest. Forest Systems. 2016;**25**(2):e057

[100] Pretzsch H, Rais A. Wood quality in complex forests versus even-aged monocultures: Review and perspectives.

Wood Science and Technology. 2016;**50**(4):845-880

[101] D'Amato AW, Bradford JB, Fraver S, Palik BJ. Effects of thinning on drought vulnerability and climate response in north temperate forest ecosystems. Ecological Applications. 2013;**23**(8):1735-1742

[102] Fahlvik N, Ekö P-M, Pettersson N. Influence of precommercial thinning grade on branch diameter and crown ratio in *Pinus sylvestris* in southern Sweden. Scandinavian Journal of Forest Research. 2005;**20**(3):243-251

[103] D'Amato AW, Bradford JB, Fraver S, Palik BJ. Forest management for mitigation and adaptation to climate change: Insights from long-term silviculture experiments. Forest Ecology and Management. 2011;**262**(5):803-816

[104] Jimenez E, Vega JA, Fernandez C, Fonturbel T. Is pre-commercial thinning compatible with carbon sequestration? A case study in a maritime pine stand in northwestern Spain. Forestry. 2011;**84**(2):149-157

[105] Skovsgaard JP, Stupak I, Vesterdal L. Distribution of biomass and carbon in even-aged stands of Norway spruce (*Picea abies* (L.) Karst.): A case study on spacing and thinning effects in northern Denmark. Scandinavian Journal of Forest Research. 2006;**21**(6):470-488

[106] Keyser TL. Thinning and site quality influence aboveground tree carbon stocks in yellow-poplar forests of the southern Appalachians. Canadian Journal of Forest Research. 2010;**40**(4):659-667

[107] Skovsgaard JP. Analysing effects of thinning on stand volume growth in relation to site conditions: A case study for even-aged Sitka spruce (*Picea sitchensis* (Bong.) Carr.). Forestry. 2009;**82**(1):87-104

[108] De las Heras J, Moya D, López-Serrano FR, Rubio E. Carbon sequestration of naturally regenerated Aleppo pine stands in response to early thinning. New Forests. 2013;**44**(3):457-470

[109] Blanco JA, Imbert JB, Castillo FJ. Influence of site characteristics and thinning intensity on litterfall production in two *Pinus sylvestris* L. forests in the western Pyrenees. Forest Ecology and Management. 2006;**237**(1-3):342-352

[110] Finkral AJ, Evans AM. The effects of a thinning treatment on carbon stocks in a northern *Arizona ponderosa* pine forest. Forest Ecology and Management. 2008;**255**(7):2743-2750

[111] Jobbágy EG, Jackson RB. The distribution of soil nutrients with depth: Global patterns and the imprint of plants. Biogeochemistry. 2001;**53**:51-77

[112] Hoover CM, Heath LS. A commentary on 'Mineral soil carbon fluxes in forests and implications for carbon balance assessments': A deeper look at the data. GCB Bioenergy. 2015;**7**(2):362-365

[113] Peri PL, Bahamonde HA, Lencinas MV, Gargaglione V, Soler R, Ormaechea S, et al. A review of silvopastoral systems in native forests of *Nothofagus antarctica* in southern Patagonia, Argentina. Agroforestry Systems. 2016;**90**(6):933-960

[114] Bréda N, Granier A, Aussenac G. Effects of thinning on soil and tree water relations, transpiration and growth in an oak forest (*Quercus petraea* (Matt.) Liebl.). Tree Physiology. 1995;**15**(5):295-306

[115] Sohn JA, Hartig F, Kohler M, Huss J, Bauhus J. Heavy and frequent thinning promotes drought adaptation in *Pinus sylvestris* forests. Ecological Applications. 2016;**26**(7):2190-2205

[116] Fulé PZ, Waltz AEM, Covington WW, Heinlein TA. Measuring forest restoration effectiveness in reducing hazardous fuels. Journal of Forestry-Washington. 2001;**99**(11):24-29

[117] Fernandes PM, Vega JA, Jiménez E, Rigolot E. Fire resistance of European pines. Forest Ecology and Management. 2008;**256**(3):246-255

[118] Verkaik I, Espelta JM. Post-fire regeneration thinning, cone production, serotiny and regeneration age in *Pinus halepensis*. Forest Ecology and Management. 2006;**231**(1-3):155-163

[119] Bren L, Lane P, Hepworth G. Longer-term water use of native eucalyptus forest after logging and regeneration: The Coranderrk experiment. Journal of Hydrology. 2010;**384**(1-2):52-64

[120] Bréda N, Huc R, Granier A, Dreyer E. Temperate forest trees and stands under severe drought: A review of ecophysiological responses, adaptation processes and long-term consequences. Annals of Forest Science. 2006;**63**(6):625-644

[121] Sohn JA, Gebhardt T, Ammer C, Bauhus J, Häberle K-H, Matyssek R, et al. Mitigation of drought by thinning: Short-term and long-term effects on growth and physiological performance of Norway spruce (*Picea abies*). Forest Ecology and Management. 2013;**308**:188-197

[122] Christina M, Laclau J-P, Gonçalves JLM, Jourdan C, Nouvellon Y, Bouillet J-P. Almost symmetrical vertical growth rates above and below ground in one of the world's most productive forests. Ecosphere. 2011;**2**(3):art27

[123] Foster TE, Schmalzer PA, Fox GA. Seasonal climate and its differential impact on growth of co-occurring species. European Journal of Forest Research. 2015;**134**(3):497-510

[124] Mainiero R, Kazda M. Depth-related fine root dynamics of *Fagus sylvatica* during exceptional drought. Forest Ecology and Management. 2006;**237**(1-3):135-142

[125] Dieler J, Pretzsch H. Morphological plasticity of European beech (*Fagus sylvatica* L.) in pure and mixed-species stands. Forest Ecology and Management. 2013;**295**:97-108

[126] Crecente-Campo F, Pommerening A, Rodríguez-Soalleiro R. Impacts of thinning on structure, growth and risk of crown fire in a *Pinus sylvestris* L. plantation in northern Spain. Forest Ecology and Management. 2009;**257**(9):1945-1954

[127] Peri PL, Dube F, Varella A. Silvopastoral Systems in Southern South America. Heidelberg, New York: Springer; 2016. pp. 270

[128] Martínez Pastur G, Peri PL, Huertas Herrera A, Schindler S, Díaz-Delgado R, Lencinas MV, et al. Linking potential biodiversity and three ecosystem services in silvopastoral managed forest landscapes of Tierra del Fuego, Argentina. International Journal of Biodiversity Science, Ecosystem Services & Management. 2017;**13**(2):1-11

[129] Peri PL, López DR, Rusch V, Rusch G, Rosas YM, Martínez Pastur G. State and transition model approach in native forests of Southern Patagonia (Argentina): Linking ecosystem services, thresholds and resilience. International Journal of Biodiversity Science, Ecosystem Services & Management. 2017;**13**(2):105-118

Chapter 4

Differentiation of the Forest Structure as the Mitigation Action of Adverse Effects of Climate Change

Janusz Szmyt

Abstract

For several decades, the attention of societies has been focused on potential environmental changes due to climate change. Although climate change is not a new phenomenon, in the recent two decades, there has been a growing interest of scientists trying to determine scenarios of trends and their potential impact on forest ecosystems and forestry. Despite the uncertainties of climate change and the response of forest ecosystem to change, the forest management must deal with these uncertainties. There is no single prescription on how to manage forest resources under climate change in order to fulfill all demands from society. Various strategies in forest management are developed to counteract the adverse effects of climate change on forests and forestry. The future forest management should implement the following three main strategies: create forests which are resistant to change, promote their greater resilience to change, and enable forests to respond to change. It is expected that the more the structured forest, the higher the adaptive capacity is expected. Experiment focused on the influence of different silvicultural procedures on the structure of Scots pine in Poland is presented. Achieved results indicated that the process of stand structure conversion is a long-term process and different structural elements can be modified to different extents.

Keywords: stand structure, adaptive management, stand diversity, adaptive silviculture, *Pinus sylvestris*

1. Forests and forestry under climate change

For several decades, the attention of societies has been focused on the information about potential changes in our environment due to the changing climate system. Although the climate change is not a new phenomenon, in the recent two decades, there has been a growing interest of scientists trying to determine trends in climate change and their potential impact on a number of areas of human life. The impact of these changes is also studied in the context of forest ecosystems and forestry [1–3].

As the Intergovernmental Panel on Climate Change (IPCC) reports indicate, one of the significant reasons for the observed climate change is the increasing content of greenhouse gases in the atmosphere and the human activity attributed to them. Apart from determining the causes of the increasing content of these gases and their origin, the increase in average air temperature, changes in precipitation regimes, and changes in the natural disturbance regimes observed in recent years raise concern among scientists dealing with forest ecosystems as well as foresters and forest owners [4].

In addition to the uncertainty of the scale and rate of climate change and the different nature of these impacts on forest ecosystems, the response of forest ecosystems to these changes is subject to high uncertainty too [1, 5]. This problem is not easy to solve because it must be underlined that the projections derived from global circulation (climatic) models and ecological models are not predictions of future climate conditions, but they are rather description of possible conditions resulting from certain scenarios [4, 5]. In other words, climate models represent the range of probable features of the future environmental conditions, and here we are dealing with uncertainties. Therefore, the forest management under climate change must deal with uncertainties. Up to now, there is no single prescription on how to manage forest resources under climate change in order to fulfill all the demands from society.

Due to the growing concern about the future of the forests around the world, various strategies in forest management are developed to counteract the adverse effects of climate change on forests and forestry [4, 6]. Novel environmental conditions resulting from climate change might result in changes of forest tree species distribution (change in natural ranges) through changes in forest productivity and the economic value of managed forests [3, 7].

Up to now, different paradigms of forest management are suggested as the potential solution, that is, close-to-nature forestry, adaptive forestry, systemic forestry [8–11]. It is expected that the future forest management should implement the following three main strategies [6, 12]:

- create forests which are resistant to change,
- promote their greater resilience to change, and
- enable forests to respond to change.

Adaptive management can be defined as a systematic and iterative approach for improving forest resource management by learning from management outcomes. It can be done by exploring alternative ways to meet the management objectives [13].

Modern forest management, taking into account the multifunctionality of forests and uncertainty of future climate conditions, will then require the introduction of innovative ways of management to ensure all services provided by forest ecosystems under the future unpredictable environmental conditions. Different approaches of short-term and long-term strategies are assumed to be required [13].

Three ways of adaptation strategies concerning forests are indicated [14, 15]. The so-called *business as usual* (no intervention) relies on today's practices and management targets. It is based on the assumption that forests themselves can adapt to changing environmental conditions as they did in the past. The second strategy called *reactive adaptation* takes place in the moment just after the fact. This strategy takes in account salvage cutting, updated harvest scheduling, recalculating allowable cuttings, etc. The third strategy is called *planned adaptation*, and it involves redefining goals and practices in advance taking into consideration climate change risk and uncertainties. This strategy will require new thinking of foresters taking into account the considerations of the global implications of local operations. Of course, the planned adaptation for climate change involves greater uncertainty and novel risk. Among the activities related to planned adaptation strategy, one can consider the planting of new provenances or species capable of growing in environment under projected climatic conditions, reaping the benefits of new products from forests (i.e., carbon sequestration). It is expected that the adaptation strategy will increase the resilience of the forests while simultaneously decreasing their vulnerability. Other operations within planned adaptation include the silviculture of mixed stands, use of clones better suited for novel conditions, modification of thinning regimes, etc.

Bolte et al. [12] indicated three other strategies in adaptation of forest ecosystems to change to meet the management goals: *conservation of forests structure*, *active adaptation*, and *passive adaptation*. The first can be treated as business as usual and it relies on the maintenance of the structural consistency independently on the pressure due to environmental change. Active adaptation means the use of silvicultural methods to change the structure of the stand in a way that the resulting forest is better adapted to a new climatic condition than it would happen by natural succession. Passive adaptation uses spontaneous adaptation processes in terms of natural succession and natural species migration.

2. Different silvicultural tools for increasing adaptive capacity of forests to change

To understand the importance of silviculture, it is worth to recall its goals. They are defined as related to creating and maintaining the forest that will best fulfill all objectives of both owner and society. As they stated, the wood production is neither the only nor necessarily the dominant goal. At present the benefits of the forest are manifold, and all of them, for example, recreation, esthetics, or habitat protection, must be taken into account in modern forestry. The biggest problem, however, in modern silviculture, is getting the owners and society to define the management objectives which should aim to ensure all services and functions provided by the forest for a long time despite the impact of the potential climate change.

While the priority of timber production was clearly seen in the past, one can observe that the forest management focused mainly on the providing economic benefits is no longer possible. Ecological and cultural services seem to be more and more desirable by society even when their provision is mostly possible due to the timber harvesting. Therefore, it is obvious that protection and production functions of the forest are both important to society and the conflict between these two functions must be avoided or, at least, mitigated [16, 17].

The changing needs of society also require a change in forest management which must provide more services than wood production. In Europe, such management, called *continuous cover forestry* (CCF) is only one option for that, and it is now successfully implemented in practice in many countries [18–20]. The concept of CCF mostly relies on close-to-nature silviculture (CTNS) or natural silviculture [21, 22]. Different aspects of the implementation of CTNS to increase the stability of forest ecosystems can be recommended: avoidance or limitation of clear-fellings, promotion of highly structured forests, and promotion of native tree species and selective individual tree silviculture are among the most important. Two basic principles of CTNS should be implemented: (1) reducing silvicultural risk and (2) reducing its spreading. Both are extremely important to mitigate the potential adverse impact of climate change on forests and forestry as well. Under the first principle, the following activities should be promoted [23, 24]:

- full use of genetic diversity of forest tree species (natural regeneration promoted),
- species composition adapted to the local site conditions,
- tending treatments aiming at the increase of tree vitality and ensuring better use of growing space, and
- forest site cultivation.

Among activities within the second principle, the most important is associated with the promotion (creation) of complex forest structure in terms of their species composition (mixed stands), vertical profiles (multilayered and multicohort stands), and horizontal patterns (patchy stands) [25, 26].

3. Which adaptation strategy is better? A case study from Scots pine (*Pinus sylvestris* L.) stands in Poland

Why forest structure matters? Shortly—it is a key to the forest ecosystem, its function, and diversity [27, 28]. Understanding the forest structure dynamics allows us to better understand the history, functions, and future of the forest ecosystem. The stand structure of the stand can be described by lots of elements, for example, species composition, tree age, tree size, and dead wood amount. If we manage the forest structure, we will affect the forest functions. Potential benefits and limitations of different silvicultural regimes on the structuring forest stand are presented here on the base of the experiment in Scots pine (*Pinus sylvestris* L.) forests in Poland.

Forests cover in Poland ca. 9,200,000 hectares (29.6% of area) and *P. sylvestris* is the most economically important tree species in Poland. In Poland, this tree species has optimal climatic and site conditions within its Euro-Asiatic natural range. While conifers dominate the species structure of Polish forests, pine accounts for 58% of the area of forests. It also accounts for 56.5% in the volume structure of timber resources [29]. Most Scots pine forests in Poland are managed according to even-aged silviculture, and thus they represent rather structurally homogenous stands in terms of species composition, vertical and horizontal structures. The Department of Silviculture, Faculty of Forestry of the Poznan University of Life Sciences, has been involved for decades in research projects aiming at finding opportunities to change the even-aged silviculture of pine forest into more complex management, for example, shelterwood cuttings or selection cuttings [26]. One example of such studies is presented below.

3.1 Methodological considerations of the experiment

Experiment has been established in three stands where *P. sylvestris* shares 90% or more in abundance. Admixture tree species is silver birch (*Betula pendula* Roth.). Till the initialization of experiment in the 1980s of the last century, each stand (experimental object) has been managed according to even-aged silviculture and they could be characterized as monocultures, even-aged and single-layer stands. Three experimental objects, reflecting different status of silvicultural treatments, has been applied: control (C), experimental (Ex), and economic (E) of sizes 35.78, 37.88, and 41.01 ha, respectively. In the control object, no logging operation has been allowed and it represents *passive* adaptation strategy. In case of the experimental object, only selective thinning has been allowed and it can be treated as the *active* strategy of adaptation. In the economic object, a *business as usual* strategy has been planned and conducted according to the low thinning rules indicated in the management plan elaborated for this forestry district.

In 1988, the net of permanent circular measurement plots of size 0.05 ha each was laid out in the nodes of the rectangular grid of size 100 × 50 m. On each plot, the stem diameter at 1.3 m (dbh, cm), total tree height (H, m) of 2–3 trees, and polar coordinates (x, y calculated from the azimuth and distance to each tree from the plot center) were measured. Also, tree species and tree status (dead, live) were recorded. The first survey was done in 1988 and the second after 15 years, in 2003. The mean stand parameters (tree density, tree diameter, and basal area) coupled with spatially explicit structural indices (**Table 1**) describing different aspects of the stand structure were calculated. Tree diameter and basal area distributions were checked for their normality using Shapiro-Wilk test of normality. If the distributions were not significantly different from the normal distribution ($\alpha = 0.05$), the analysis of variance (ANOVA) was applied to analyze the differences among treatment means in terms of both characteristics. If these differences are significant, the *post hoc* Tuckey's range test is applied to find out which treatments differed significantly from each other.

Spatial pattern of tree distribution was evaluated on the basis of the Clark-Evans (CE) index. For random distribution of trees, the index gets the value of 1.00. If CE < 1.00, trees are distributed in smaller or larger clumps while if CE > 1.00, they are more or less regularly dispersed in the stand. The significance of the departures from unity is estimated using the standard z-test value [30].

Size differentiation indices TD (for tree diameter) and TH (for tree height) are calculated for each tree in the plot in relation to three neighbors. The higher the value of the index, the more is the diversity in terms of tree size observed. Apart from the mean value of these indices, it is possible to analyze the distribution of them in five differentiation classes [30]: very low (<0.20), low (0.20–0.40), moderate (0.40–0.60), large (0.60–0.80), and very large (>0.80) differentiation among the nearest neighbors. Small value of size differentiation index means homogenous in size group of trees, and large value indicates heterogeneous groups of trees.

Index	Formulation	Explanations
Aggregation index, CE	$CE = \frac{r_A}{r_E} = \frac{\frac{1}{N}\sum_{i=1}^{N} r_i}{0.5 \cdot \left(\frac{A}{N}\right)^{1/2} + 0.0514 \cdot \frac{P}{N} + 0.041 \cdot \frac{P}{N^{3/2}}}$	r_A—Observed mean distances between trees A—Area (m^2) N—Total number of trees P—Circumference of the plot
Mingling index, SM	$SM = \frac{1}{k}\sum_{i=1}^{k} v_{ij}$	k—Numbers of nearest neighbors $v_{ij} = 1$, if reference tree and neighbor are different species, otherwise $v_{ij} = 0$ n—Number of neighbors ($n = 3$)
Size differentiation index, T	$T = \frac{1}{n}\sum_{i=1}^{n}\sum_{j=1}^{3}\left[1 - \frac{\min(\text{size}_i, \text{size}_j)}{\max(\text{size}_i, \text{size}_j)}\right]$	size_i—Diameter or height of ith tree size_j—Diameter or height of jth tree n—Number of neighbors ($n = 3$)

Table 1.
Structural indices calculated in each of the objects analyzed.

Species mingling (SM) is calculated for each tree in the plot and their three nearest neighbors. The lower the value of the index, the more is the homogeneous group of trees in terms of their species. In the case of three neighbors, the index can take four values: 0.0, 0.3, 0.7, and 1.0, indicating no mingling, low mingling, large mingling, and full mingling, respectively [30].

All structural indices were calculated for plots with the number of trees ≥10.

Afterward, the change in stand structure over the 15-year period of the stand development was evaluated. To find out the differences in structural diversity between objects (C, Ex, and E), the Kruskal-Wallis nonparametric test followed by the Dunn's multi-comparison test were applied to test significant differences ($\alpha = 0.05$).

Statistical calculations were done in R environment [31] and Siafor 1.0 software [32].

3.2 Results

3.2.1 Stand parameters

Scots pine was the dominant tree species in the stand independently on the experimental object. In 1988, the average number of trees per plot was for the control object 44.5 (SD = ±8.4), experimental—35.3 (SD = ±7.3), and economic—34.3 (SD = ±8.0). After 15 years, the density decreased in each object and, in 2015, the average number of trees per plot was reached in the control object 38.3 (SD = ±7.5), in experimental—23.8 (SD = ±4.7), and in the economic—21.0 (SD = ±6.1).

The highest number (on average) of individuals (per 1 ha) of this tree species was observed in the control object, where no logging was conducted. Experimental and economic objects showed similar number of trees of Scots pine. In case of birch, the highest number was observed in the economic object and the lowest in the control one (**Table 2**). Similar trend was observed in terms of basal area—Scots pine was the dominant tree species reaching the share by more than 90% in each of the object analyzed. In absolute numbers, however, the highest basal area was observed in 1988 in the economic object and the lowest in the control one (**Table 2**). In 2003, the highest basal area was obtained in the control object and the lowest in the economic one. The share of birch at the beginning of the experiment was the highest in the

Year	Pinus sylvestris		Betula pendula		Picea abies		Robinia pseudoacacia	
	N (%)	BA (%)	N (%)	BA (%)	N (%)	BA (%)	N (%)	BA (%)
Control object								
1988	817 (92)	25.4 (91)	50.4 (6)	1.8 (6)	6.3 (1)	0.3 (1)	12.4 (1)	0.5 (2)
2003	709 (93)	35.3 (92)	41 (5)	2.1 (5)	6 (1)	0.4 (1)	5 (1)	0.4 (1)
Experimental object								
1988	638 (90)	26.8 (91)	68 (10)	2.6 (9)	—			
2003	432 (92)	29.9 (92)	37 (8)	2.4 (8)				
Economic object								
1988	610 (89)	27.7 (92)	76 (11)	2.4 (8)	—			
2003	383 (92)	28.8 (94)	33 (8)	2 (6)				

Table 2.
Average tree number (N ha^{-1}) and basal area (BA ha^{-1}) and the corresponding percentage (in brackets) of tree species present in the objects in 1988 and 2003.

economic object and the lowest in the control one. After 15 years, this trend was still observed (**Table 2**). Other tree species, that is, Norway spruce (*Picea abies* (L.) Karst.) and black locust (*Robinia pseudoacacia* L.) were present only in the control object and both are excluded from the further analysis.

Figure 1 shows that the mean diameter of living trees was the highest in the economic object, followed by the experimental one. The lowest dbh was observed in the case of the control object. Similar trend can be observed for the basal area.

Coefficient of variation calculated for dbh was similar in all objects, however, it was slightly higher in case of the economic object (25.7%) than in the others.

The diameter distribution of living trees and their basal area were not different significantly from the normal distribution (data not shown). Analysis of variance, followed by Tukey's *post hoc* test, revealed that the mean tree diameter and basal area were significantly different in the analyzed objects in 1988 and 2003 (α = 0.05, **Figure 2** for tree diameter).

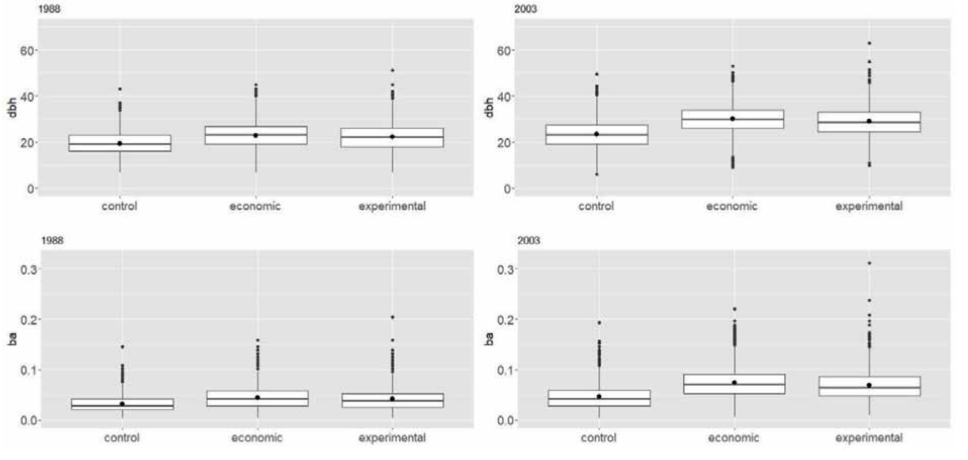

Figure 1.
Boxplots (mean, median, min, max, outliers, and first and third quartiles) for tree diameter (dbh) and basal area (ba) in the objects in two inventories.

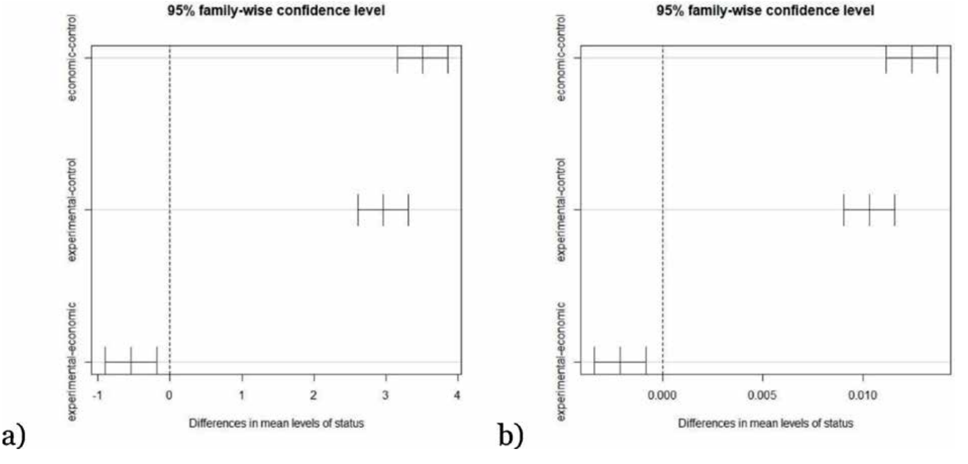

Figure 2.
Differences in the mean of dbh between objects in 1988 (a) and 2003 (b). If the confidence level does not include 0 value, then two means are significantly different at α = 0.05.

3.2.2 Structural parameters

3.2.2.1 Control object

3.2.2.1.1 Spatial distribution

The average value of the CE index for the object at the beginning reached the value of 1.14 and was significantly different from the random expectation. The index ranged from 0.81 to 1.35 with its variation among plots at the level of 8% (**Figure 3**). There were 32 plots (51% of all plots) in the control object on which trees showed regular pattern of their distribution (CE > 1.0) and only on one plot in this index was significantly lower than CE < 1.0, indicating clumped distribution of trees. On the rest of the plots (48%), the deviations from the random expectation were not statistically proved and trees were randomly distributed. After 15 years—in 2003—the mean value of CE index did not change (CE = 1.14). The value of this index varied among plots between 0.83 and 1.30. The number of plots on which the index was significantly higher than 1.0 indicating regular pattern decreased to 23 (36%). No plot indicating clumped distribution of trees was observed. Therefore, the random pattern was still dominant in 2003 and this type of tree distribution was observed on 40 plots (63%).

3.2.2.1.2 Tree size diversity

Just after the initialization of the experiment, in 1988, the mean value of diameter differentiation index, TD, in the case of the control object reached TD = 0.19. This index ranged from 0.00 to 0.54 depending on the plot (**Figure 3**). Coefficient of variation for TD index between plots was large (cv = 50%). The mean index showed that, in general, the variation in dbh among neighboring trees was low. This was confirmed by the distribution of the index in differentiation classes (**Table 3**). The dominant classes were these of very low and low diameter differentiation, which indicates that the diversity in diameter between the nearest neighbors was lower than 40%. After 15 years, the situation did not change much. The average TD index took the value of TD = 0.20 with much smaller range: 0.17–0.31 than in 1988. The variation of TD among plots clearly decreased to 20%. Again, the distribution of TD index in diameter differentiation classes confirmed that trees were mostly similar in their diameter at the small spatial scale (**Table 3**).

In the case of tree height differentiation, the mean value of the TH index was much smaller than for tree diameter and it reached TH = 0.10. The index ranged from 0.06–0.16 (**Figure 3**) depending on plot, and the coefficient of variation between plots was 28%. This indicates that neighboring trees were very similar in their height (**Table 3**). Homogenous groups of trees are indicated also by the share of trees belonging to the lowest differentiation class (92.9% of trees). After 15 years, the mean value of the TH index did not change (TH = 0.10), with the range varying between 0.06 and 0.15. The share of trees in the lowest differentiation class increased to more than 95% (**Table 3**).

3.2.2.1.3 Species mingling

In 1988, the species mingling index, SM, reached the mean value of SM = 0.08, indicating very homogenous conditions, on average, in terms of species diversity at the small spatial scale. The index ranged from 0.00 to 0.54 and its variation between plots was at very high level, cv = 103% (**Figure 3**). There were 17 plots (27%) in the control object on which the index was equal to 0.00, indicating the

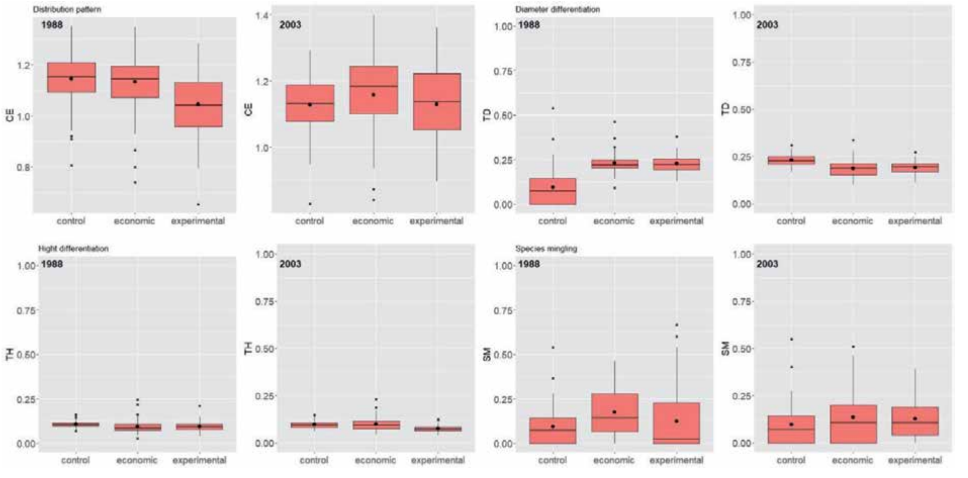

Figure 3.
Statistical characteristic (mean, median, max, min, outliers, and first and third quartiles) of structural indices describing spatial pattern (CE), tree size differentiation (TD), and species mingling (SM) in the experimental objects in 1988 and 2003.

Index	Year	Differentiation classes				
		Very low	Low	Moderate	Large	Very large
TD	1988	45.5	48.5	5.8	0.18	0
	2003	44.8	48.0	6.6	0.46	0.12
TH	1988	92.9	6.98	0.04	0	0
	2003	95.5	4.14	0.33	0	0

Table 3.
The share (%) of diameter (TD) and height (TH) differentiation classes in the control object.

lack of species mingling at all. On other plots, the mingling index varied from low to moderate. Analysis of the index distribution in mingling classes pointed out that the neighborhood of most trees in the control object was homogeneous (83% of trees). Only in the case of 9% of trees, their neighborhood was more heterogeneous in terms of species, meaning that 1–2 neighbors were different in species. In 2003, the mean SM index increased to SM = 0.12 and ranged from 0.00 to 0.55 depending on the plot. The coefficient of variation between plots in the control object decreased slightly after 15 years and got the value of 97%. On 19 plots (30%), the index value was equal to 0.00, indicating the lack of species mixture. Again, the dominance of very low mingling class can be observed on most plots in the object (84% trees). Both tree species showed the opposite behavior (**Figure 4**). Scots pine formed large homogenous groups of trees, while silver birch was present in the stand mostly as a single mixture.

3.2.2.2 Experimental object

3.2.2.2.1 Spatial distribution

In 1988, the mean CE index for the experimental object took the value of CE = 1.18, indicating regular pattern in tree distribution. The value of the index ranged from 0.65 to 1.28 and the coefficient of variation among plots reached the

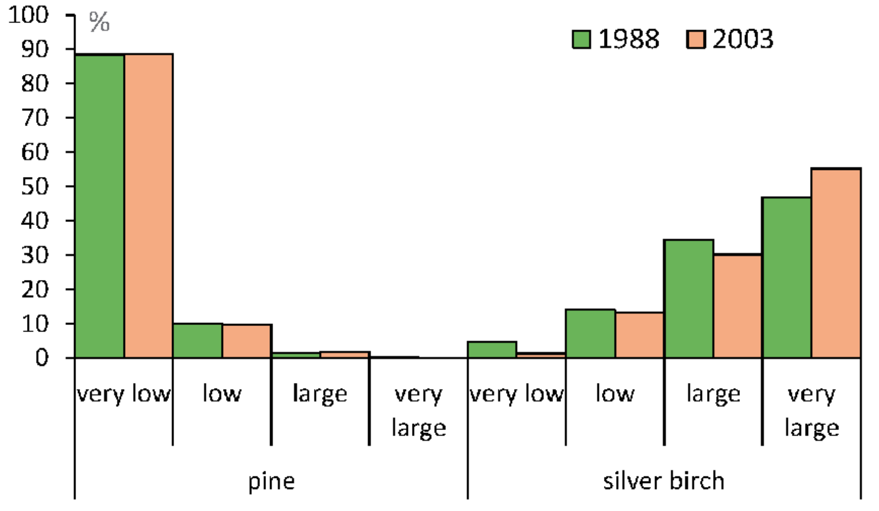

Figure 4.
Spatial mingling of Scots pine and silver birch in the control object in two inventories.

level of 12% (**Figure 3**). In the case of 19 plots (25% of all plots), the value of the index was significantly larger than that for the random expectation and on the others the distribution pattern was random. After 15 years, in 2003, the mean value of the CE index increased to CE = 1.22 and it varied from 0.89 to 1.29 depending on the plot. The coefficient of variation among plots was at the level of 8%. In the case of 26 plots (35%), the value of this index was significantly different from the randomness, indicating clear regularity in the spatial distribution of trees.

3.2.2.2.2 Tree size diversity

In 1988, the mean value of the diameter differentiation index, TD, was TD = 0.23 which pointed to the low diversity in diameter of trees at small spatial scale. This index ranged in this object from 0.12 to 0.38, and the variation on it among all plots was at the level of 20% (**Figure 3**). Most trees in the experimental object could be characterized by very low and low differentiation (95% of all trees), which confirmed that trees were similar in their diameter at the nearest-neighbor spatial scale (**Table 4**). In 2003, the average value of the TD index decreased to TD = 0.19. The lowest value of the index was 0.14 and the largest was 0.31. Coefficient of variation of the index among plots decreased to the level of 15%. Up to 97% of trees were characterized by very low and low differentiation in diameter at small spatial scale (**Table 4**). At the beginning of the experiment, the mean index describing the differentiation of tree in terms of their height took the value of TH = 0.10. It pointed to a large similarity of trees in tree height. The index ranged from 0.04 to 0.21 with the coefficient of variation among plots at the level of 28%. Up to 96% of trees showed similarity in height with their nearest neighbors (**Table 4**). In 2003, the mean value of the TH index decreased to TH = 0.07, with the minimum value of 0.05 and maximum one of 0.15. Variation in the TH index among plots was at the level of 21%. The share of trees which showed large similarity in their total height with the nearest neighbors increased to 98% (**Table 4**).

3.2.2.2.3 Species mingling

In 1988, the mean value of the species mingling index got SM = 0.13, indicating rather low species mixture in the experimental object. The value of this index varied

Index	Year	Differentiation classes				
		Very low	Low	Moderate	Large	Very large
TD	1988	43	52	5	0.3	0
	2003	60	37	2	0.4	0
TH	1988	96	3.5	0.3	0	0
	2003	98	1.5	0.5	0	0

Table 4.
The share (%) of diameter (TD) and height (TH) differentiation classes in the experimental object.

among plots from 0.00 to 0.60 (**Figure 3**) and the coefficient of variation was very high, cv = 78%. Species homogenous neighborhood, expressed by the index SM = 0.00, was observed in the case of 28 plots (39%), and the others showed higher mingling level. After 15 years, the mean value of the index was almost the same like in 1988—SM = 0.12. The minimum value of SM was 0.00 and the highest one was 0.55, with variation among plots reaching the level of 89%. The number of plots with the index SM = 0.00 decreased in 2003–2018 (25%), that is a 34% decrease. Similar to the control object, Scots pine and silver birch showed opposite behavior (**Figure 5**). Homogenous neighborhood was observed in the case of Scots pine, while birch was present most often as a single mixture.

3.2.2.3 Economic object

3.2.2.3.1 Spatial distribution

At the beginning of the experiment, the mean value of CE describing the spatial pattern of living trees in the economic object took the value CE = 1.21, pointing to a clear regular pattern. The lowest value of the index was CE = 0.74 and the highest was CE = 1.35 (**Figure 3**). The coefficient of variation between plots for this index was low—10%. The index differed significantly from randomness (CE = 1.00) in the case of 33 plots (43%). The dominant spatial pattern of living trees was therefore a random pattern. No clumping was observed on any plot. After 15 years, in 2003, the mean value of the index increased to CE = 1.27. In the object, the index ranged from 0.84 to 1.40 depending on the plot, with the coefficient of variation at the level of 11%. The number of plots with the CE index significantly larger than 1.0 was 34 (48%). The dominance of the random pattern was still observed in this object.

3.2.2.3.2 Tree size diversity

In 1988, the mean value of diameter differentiation index reached TD = 0.23, indicating rather low diversity in tree diameter among the nearest neighbors. The index ranged from 0.09 to 0.47 (**Figure 3**), with the coefficient of variation among plots at the level of 22%. Most trees showed very low or low diameter differentiation at the small spatial scale (92% of trees) and only few (7.8%) showed larger variation in diameter (**Table 5**). In 2003, the mean value of the index decreased to TD = 0.19, varying between 0.10 and 0.34 depending on the plot. The variation of the index among plot was at the level of 23%. After 15 years of stand development, the number of trees in the lowest two classes of diameter differentiation clearly increased (**Table 5**). As much as 97% of trees belonged to both these classes.

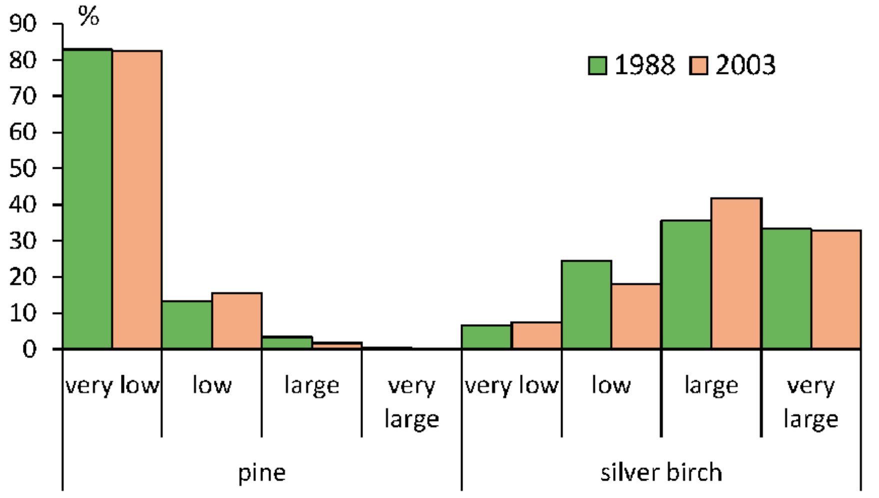

Figure 5.
Spatial mingling of Scots pine and silver birch in the experimental object in two inventories.

Index	Year	Differentiation classes				
		Very low	Low	Moderate	Large	Very large
TD	1988	45.6	46.4	7.0	0.8	0
	2003	63.9	32.9	2.6	0.6	0
TH	1988	93.2	6.2	0.7	0	0
	2003	93.9	4.9	1.1	0.1	0

Table 5.
The share (%) of diameter (TD) and height (TH) differentiation classes in the economic object.

The height differentiation of trees in the economic object was clearly lower than the diameter. The mean value of the index, TH, was 0.10 and it ranged from 0.03 to 0.25, with cv = 35%. Up 93% of trees showed very low differentiation in height among their neighbors (**Table 5**). In 2003, the TH index reached the same mean value as in 1988 (TH = 0.10). The index varied from 0.05 to 0.23 depending on plot, and the coefficient of variation of TH index among plots was at the level of 32%. Again, the most abundance class was the one indicating very low height differentiation (**Table 5**).

3.2.2.3.3 Species mingling

In 1988, the spatial mingling index, SM, reached the mean value of 0.17, with its range from 0.00 to 0.46 (**Figure 3**). The coefficient of variation for the index was at the level of 74%. The relative low mean value of the index pointed to rather homogenous neighborhoods in terms of tree species. In the case of eight plots (10%), the index showed no mingling and in the case of the others, the species diversity was slightly higher. The low species mingling in the economic object was also confirmed by the distribution of this index in the mingling classes. Trees belonging to the lowest mingling class accounted for 68.8%, but 6.4% belonged to the mingling class showing very large mingling. In 2003, the mean value of SM index dropped to 0.13, varying between 0.00 and 0.51 among plots (cv = 97%). The abundance of very low mingling class increased to 75.8% at the expense of the classes of higher species mingling. Both tree species showed more complex situation in terms of spatial mingling comparing to the other objects (**Figure 6**). However, Scots pine formed large homogenous groups of trees, while silver birch was mixed in the form of groups or a single mixture.

3.2.3 Difference in structural diversity of the stand between objects

3.2.3.1 Spatial distribution

At the beginning of the experiment, the analyzed stands in the control, experimental, and economic objects showed significant differences in terms of spatial distribution of trees (α = 0.05). It was indicated by the Kruskal-Wallis test (KS) (χ^2 = 27.6787, P = 0.00). The Dunn's test, applied to find out which objects differed, showed that such significant differences were observed between the control and experimental objects (P = 0.00) as well as between the experimental and economic ones (P = 0.00). No significant difference in terms of spatial pattern was observed between the control and economic objects (P = 0.57). In 2003, the KS test confirmed the significant differences between the objects (χ^2 = 5.8092, P = 0.05). However, the Dunn's test indicated the only significant differences between the control object and economic one (P = 0.02). No differences have been observed between other pairs of the objects.

3.2.3.2 Tree size diversity

In 1988, the differences in the diameter differentiation index between the objects have been statistically proven by the KS test (χ^2 = 87.6834, P = 0.00). They have been observed in the case of economic and control objects (P = 0.00) as well as the experimental and the control ones (P = 0.00). The experimental and economic objects were not different in terms of diameter differentiation of trees at the neighborhood spatial scale (P = 0.96). After 15 years, these differences were still significant (χ^2 = 52.4553, P = 0.00) and they were observed in the case of the same pairs of

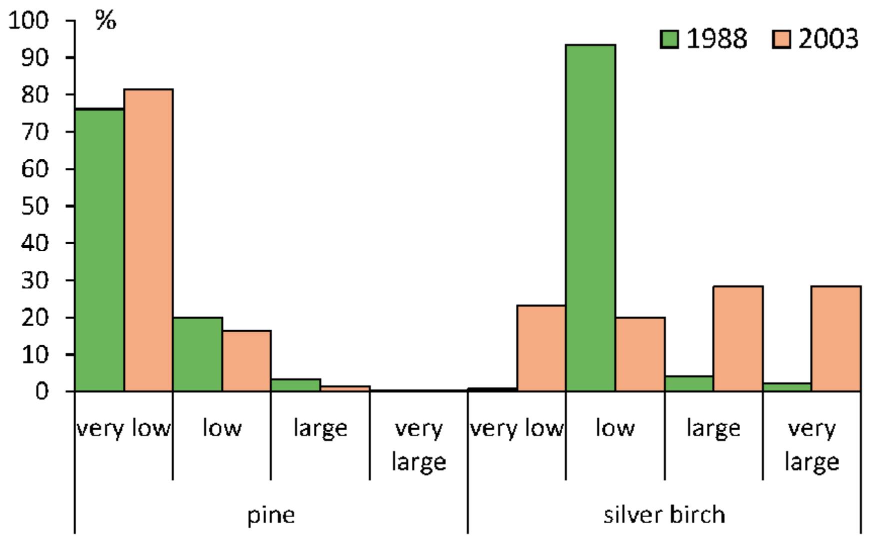

Figure 6.
Spatial mingling of Scots pine and silver birch in the economic object in two inventories.

objects. In 1988, the differences in tree height differentiation between objects were statistically significant (χ^2 = 20.5312, P = 0.00). Dunn's test proved the significance of the differences between the control and economic objects (P = 0.00) as well as for control and experimental objects (P = 0.001). Trees in the economic object and experimental one were not significantly different in terms of total tree height (P = 0.81) at the beginning of the experiment. While the KS test showed significant differences between the objects after 15 years, the pairs of them for which Dunn's test pointed out the significant differences were different. No significant differences in tree height at the small spatial scale were confirmed for the economic and control objects (P = 0.66), but in the case of the other pairs of objects they were significant (P = 0.00).

3.2.3.3 Species mingling

Species diversity expressed in the form of species mingling index showed that the objects differed significantly (χ^2 = 28.6449, P = 0.00) but only at the beginning of the experiment (in 1988). The Dunn's test showed that such differences could be observed between the control and economic objects (P = 0.004) and between the economic and experimental ones (P = 0.01).

4. Conclusion

The structure of Scots pine stands has been shaped by the historical management system, that is, even-aged silviculture. This system results in the homogenous stand structure what is confirmed by the analysis of the stand structure based on different structural metrics. Just after the initialization of the experiment with different silvicultural strategies and their impact on the stand structure, the common stand parameters (dbh, basal area) were quite similar in each of the objects being analyzed. Fifteen years after, these parameters changed clearly, and the objects differed significantly. The highest mean tree diameter was reached in the economic object followed by the experimental one. The lowest was in the case of the control object. The total stand basal area was the highest in the control object.

While the spatial pattern of tree distribution was regular, on average, the silvicultural strategies influenced clearly in the number of plots for which the regularity was statistically proved. Active strategy led to the increase of regularity and passive strategy favored the random pattern occurrence in the stand.

The previous even-aged silviculture favored low diameter differentiation of trees in each of the object. Fifteen years of the experiment, passive and active silvicultural strategies resulted in more differentiation between objects. Each of the strategies led to a lower tree diameter diversity, but business as usual strategy favored diameter homogeneity to much more extent than other strategies. Passive strategy supported higher diversity of tree diameter. In the case of tree height diversity, all strategies considered here were associated with decreasing of tree height diversity. There was no clear impact of any strategy on creating tree height diversity in Scots pine stands.

The dominance of Scots pine in the stands was confirmed by the structural metrics in each of the object. Species homogenous plots were favored by two strategies: passive and business as usual. The share of homogeneous plots decreased after 15 years of experiment only in case of the experimental object.

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References

[1] Lindner M, Fitzgerald JB, Zimmermann NE, Reyer C, Delzon S, van der Maaten E, et al. Climate change and European forests: What do we know, what are the uncertainties, and what are the implications for forest management? Journal of Environmental Management. 2014;**146**:69-83

[2] Bonan GB. Forests and climate change: Forcings, feedbacks, and the climate benefits of forests. Science. 2008;**320**:1444-1449

[3] Hanewinkel M, Cullmann DA, Schelhaas MJ, Nabuurs GJ, Zimmermann NE. Climate change may cause severe loss in the economic value of European forest land. Nature Climate Change. 2013;**3**:203-207

[4] Lindner M, Maroschek M, Netherer S, Kremer A, Barbati A, Garcia-Gonzalo J, et al. Climate change impacts, adaptive capacity, and vulnerability of European forest ecosystems. Forest Ecology and Management. 2010;**259**:698-709

[5] Harris RMB, Grose MR, Lee G, Bindoff NL, Porfirio LL, Fox-Hughes P. Climate projections for ecologists. Wiley Interdisciplinary Reviews: Climate Change. 2014;**5**:621-637

[6] Millar CI, Stephenson NL, Stephens SL. Climate change and forests of the future: Managing in the face of uncertainty. Ecological Applications. 2007;**17**:2145-2151

[7] Dyderski MK, Paź S, Frelich LE, Jagodziński AM. How much does climate change threaten European forest tree species distributions? Global Change Biology. 2018;**24**:1150-1163

[8] Corona P, Scotti R. Systemic silviculture, adaptive management and forest monitoring perspectives. L'Italia For. e Mont. 2011:219-224

[9] Messier C, Puettmann K, Chazdon R, Andersson KP, Angers VA, Brotons L, et al. From management to stewardship: Viewing forests as complex adaptive systems in an Uncertain World. Conservation Letters. 2015;**8**:368-377

[10] Puettmann KJ. Silvicultural challenges and options in the context of global change: "Simple" fixes and opportunities for new management approaches. Journal of Forestry. 2011;**31**:855-856 discussion 856

[11] Puettmann KJ, Wilson SMG, Baker SC, Donoso PJ, Drössler L, Amente G, et al. Silvicultural alternatives to conventional even-aged forest management—what limits global adoption? Forest Ecosystem. 2015;**2**:8. DOI: 10.1186/s40663-015-0031-x

[12] Bolte A, Ammer C, Löf M, Madsen P, Nabuurs GJ, Schall P, et al. Adaptive forest management in Central Europe: Climate change impacts, strategies and integrative concept. Scandinavian Journal of Forest Research. 2009;**24**:473-482

[13] Nocentini S, Buttoud G, Ciancio O, Corona P. Managing forests in a changing world: The need for a systemic approach. A review. Forest Systems. 2017;**26**:1-15

[14] Sousa-Silva R, Ponette Q, Verheyen K, Van Herzele A, Muys B. Adaptation of forest management to climate change as perceived by forest owners and managers in Belgium. Forest Ecosystem. 2016;**3**:22. DOI: 10.1186/s40663-016-0082-7

[15] Lexer MJ, Jandl R, Nabernegg S, Bednar-Friedl B. Forestry IPCC Assess Rep. 2015. pp. 147-167

[16] Temperli C, Bugmann H, Elkin C. Adaptive management for competing forest goods and services under climate

change. Ecological Applications. 2012;**22**:2065-2077

[17] D'Amato AW, Bradford JB, Fraver S, Palik BJ. Forest management for mitigation and adaptation to climate change: Insights from long-term silviculture experiments. Forest Ecology and Management. 2011;**262**:803-816

[18] Pommerening A, Murphy ST. A review of the history, definitions and methods of continuous cover forestry with special attention to afforestation and restocking. Forestry. 2004;**77**:27-44

[19] Puettmann K, Coates KD, Messier C. A Critique of Silviculture. Managing for Complexity. Washington-Covelo-London: Island Press; 2009. ISBN: 978-1-59726-145-6

[20] Pukkala T, Lähde E, Laiho O, Salo K, Hotanen JP. A multifunctional comparison of even-aged and uneven-aged forest management in a boreal region. Canadian Journal of Forest Research. 2011;**41**:851-862

[21] Brang P, Spathelf P, Larsen JB, Bauhus J, Bončína A, Chauvin C, et al. Suitability of close-to-nature silviculture for adapting temperate European forests to climate change. Forestry. 2014;**87**:492-503

[22] O'Hara KL. What is close-to-nature silviculture in a changing world? Forestry. 2016;**89**:1-6

[23] Bernadzki E. Gospodarka leśna w obliczu zmian klimatu. Sylwan. 1995;**1**:19-32

[24] Bernadzki E. Cele hodowli lasu wczoraj i dziś. Sylwan. 1997;**4**:23-31

[25] Schütz J. Close-to-nature silviculture: Is this concept compatible with species diversity? Forestry. 1999;**72**:359-366

[26] Brzeziecki B. Podejście ekosystemowe i półnaturalna hodowla lasu (w kotekście zasady wielofunkcyjnej lasu). Stud. i Mater. Cent. Edukac. Przyr. 2008;**19**:41-54

[27] Franklin JF, Spies TA, Van Pelt R, Carey AB, Thornburgh DA, Berg DR, et al. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. Forest Ecology and Management. 2002;**155**:399-423

[28] Szmyt J. Ocena Dynamiki Struktury Drzewostanów Sosnowych (*Pinus sylvestris* L.) Objętych Różnym Postępowaniem Hodowlanym. Wydawnictwo Uniwersytetu Przyrodniczego w Poznaniu; 2019. ISBN: 9788371609312

[29] Rozkrut D. Statistical Yearbook of Forestry. Zakład Wydawnictw Statystycznych Warszawa; 2018. ISSN 2657-3199

[30] Kint V, Meirvenne MV, Nachtergale L, Geudens G, Lust N. Spatial methods for quantifying forest stand structure development: A comparison between nearest-neighbor indices and Variogram analysis. Forest Science. 2003;**49**:36-49

[31] R Core Team. R: A Language and Environment for Statistical Computing. Vienna, Austria: R Foundation for Statistical Computing; 2019

[32] Kint V. SIAFOR 1.0—User Guide. Laboratory of Forestry, Ghent University; 2004

Chapter 5

Basic Theory and Methods of Afforestation

Jie Duan and Dilnur Abduwali

Abstract

Afforestation is an important practice in silviculture. This chapter outlines the forest site, site preparation, selection of afforestation materials in the process of afforestation. The life cycle of forests is very long, and it is difficult to change them once afforested. Therefore, the forest site must be analyzed in depth before afforestation to maintain the success of afforestation and the healthy growth of forests later. Forest sites are mainly affected by environmental and human activities. To facilitate afforestation, it is necessary to evaluate and classify the forest site factors and achieve a suitable species planted on the right site. Site preparation is also based on site classification. It is usually carried out after determining the type of afforestation land, divided into mechanical land preparation and chemical methods. An essential task of site preparation is to maintain soil moisture and promote seedlings' survival and growth. Afforestation materials are mainly divided into three categories: seed, seedling, and cutting. The choice of these three types of afforestation materials and methods is related to site conditions, tree species, and age.

Keywords: afforestation, forest site, site preparation, afforestation material

1. Introduction

One of the most important afforestation principles is to adapt the trees to the site [1, 2]. In a narrow sense, a forest site refers to afforestation land. In a broad sense, it refers to all factors that affect forest growth, including natural factors such as climate, soil, vegetation, and human activities. These factors constitute the forest site factor. From an ecological point of view, these factors interact with the forest and will change over time. From this perspective, forest factors affect the survival rate of afforestation and affect the forest's entire life cycle. The systematic study of forest sites has a history of over 200 years and is still continuing. Most forest site research objects are mountain forests. With the continuous development of urban forestry, urban forest site research also appears [3]. The forest site conditions of mountain forests (**Figure 1**) are entirely different from urban forests (**Figure 2**).

Forest site factors can have many combinations, each of which determines the corresponding suitable tree species and its afforestation methods, and even subsequent management methods. Therefore, the site factors of the forest should be scrutinized and analyzed before afforestation to avoid afforestation failure. After the type of afforestation land is devised, afforestation tree species and afforestation methods suitable for the type are selected according to the cultivation objectives.

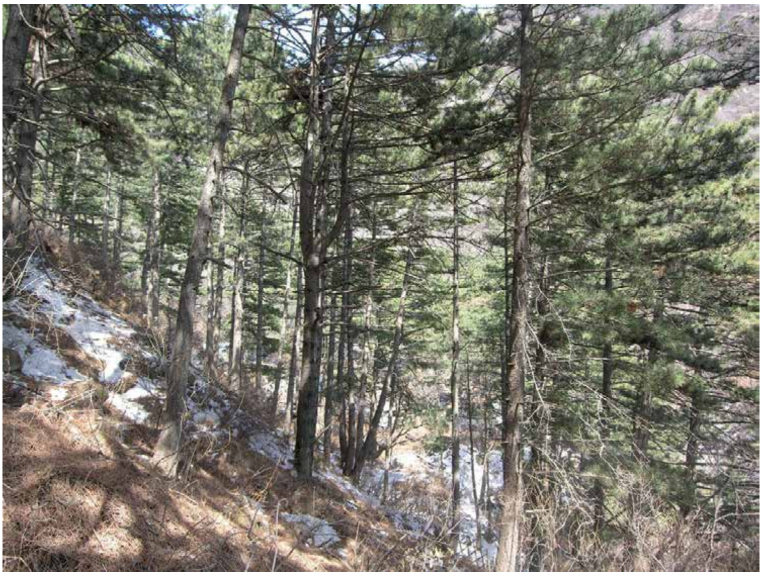

Figure 1.
Pinus tabulaeformis *forest in Song Mountain, Beijing.*

Figure 2.
Pinus tabulaeformis *forest in city plain area, Beijing.*

Another critical factor affecting the success or failure of afforestation is the healthy growth of the root system. Whether it is a seed or a seedling, only rooting or rooting after transplanting can form a forest [4]. Site preparation promotes and ensures that the root system can be closely integrated with the soil through different methods. Furthermore, promote the root system to absorb enough water.

Common afforestation materials include seeds, seedlings, and cuttings. Each material has its advantages and disadvantages. The selection of suitable afforestation materials should fully consider the characteristics of the tree species. Many studies have shown that the age of planting materials, planting season and time, and methods all affect the survival rate [5–7].

In summary, this chapter mainly introduces the concept of forest site, analyzes different site factors, and summarizes forest site evaluation and classification.

Next, the types of afforestation land and standard land preparation methods are introduced. Finally, we outline the three different afforestation materials and their planting characteristics.

2. Forest site

2.1 Concept of forest site

Forest is an important part of the ecosystem, which means silviculture is ecosystem management. Afforestation must be carried out from an ecosystem perspective, specifically light, water, carbon dioxide, and various nutrients. In the traditional sense, forest site refers to the overall environment of an area [8]. Generally speaking, the forest site has two meanings. First, it refers to geographic location; second, it refers to integrating environmental conditions (biology, soil, and climate) in a particular location [1]. The forest site remains unchanged for a certain period, especially climate conditions, and irrelevant with the tree species growing on it. Meantime, some experts refer to the forest site potential and productivity are not constant but change over time [9, 10]. Forest site and its quality should be considered first in afforestation activities.

Forest Site research in various countries around the world mainly focuses on on-site classification and site productivity evaluation. In the late 18th century, European silviculturists tried to classify forestry's productivity by compiling stand yield tables [11, 12]. In 1946, multifactor forest site classification developed into a comprehensive multifactor classification based on climate, geography, soil, and vegetation, namely the Baden-Württemberg forest site classification [13]. Since the 1950s, multifactor site classification methods have been widely used in Canada and the United States [14, 15]. Skovsgaadr and Vanclay's [16] review paper mentioned that there are two methods to assess forest site productivity include geocentric (earth-based) or phytocentric (plant-based) methods.

With the intervention of mathematical methods such as remote sensing, geographic information system, computer technology, and multivariate statistical analysis, forest site classification has gradually moved from qualitative to quantitative or a combination, from the single-factor to the ecological multifactor classification for multi-purpose forest resource management [17].

2.2 Forest site factors

2.2.1 Environmental factor

Environmental factors include climate, topography, soil, and hydrology factors. Climate factors determine the water and heat conditions that plants depend on, thus forming vegetation types. Meteorologists divided climate into macroclimate and microclimate base on the ecological scale. The macroclimate has often been referred to as that climate resulting from air masses' passage [18]. The microclimate is the suite of climatic conditions measured in the localized areas near the earth's surface [19]. Macroclimate mainly affects tree distribution [20]. For afforestation, the foresters pay more attention to microclimate factors. Light, temperature, rainfall, solar radiation, wind speed, and other factors affect tree growth and forest productivity [21, 22].

The topography factors include elevation, aspect, slope, position, slope type, etc. Elevation and aspect appeared to be fundamental variables in the assessment of forest site quality [20]. In mountainous areas, tree height and forest productivity decreased as elevation increased [23, 24]. The increase in elevation within a specific area can reduce temperature, decrease evaporation, shorten the frost-free period, increase precipitation and atmospheric and soil moisture, increase soil fertility, dense vegetation, or change vegetation types [23–25]. The light of different aspect is usually different, which can indirectly influence the soil moisture content; the south has more soil moisture than the north [14]. Furthermore, sites with lower slopes have better soil quality and higher nutrients and soil moisture than sites with steep slopes [14].

Soil is the substrate for tree growth and the forest site's essential factor, and it can influence root distribution and the ability to take up water and nutrients exchange [26]. Before afforestation, it is essential to assess the soil factors, mainly soil type, soil layer depth, soil texture, soil structure, soil nutrients, pH, and soil erosion [27]. South African forest site classification system has six soil variables (parent material, soil classification, effective soil depth, depth limiting material, topsoil organic matter, and topsoil texture), and these variables are dynamically changing [28]. It is necessary to collect many soil samples to represent the actual site situation despite this is very expensive.

Hydrology factors include groundwater depth and seasonal changes, groundwater salinity and salt composition, the presence or absence of seasonal stagnant water, and its duration. For some forests in plain areas, hydrology plays a significant role. Such as Ningxia, China, has a high groundwater level and heavy soil salinization [29]. Controlling the rise of the groundwater level is the key to forest site improvement in the irrigation area. When afforestation in mountainous areas, generally does not consider the groundwater level because it is difficult for the tree roots to reach the groundwater layer, and more consideration is streamflow. Moisture tends to increase with elevation and gets wetter on the northern aspects [30].

At present, there are many monitoring instruments that can monitor forest meteorological factors, including radiation, temperature, humidity, wind speed, wind direction, etc. (**Figure 3**). It provides an important data source for afforestation and future forest management.

Figure 3.
A weather station that can monitor a variety of climate factors.

2.2.2 Vegetation factor

Forest vegetation types and the distribution comprehensively reflect different site conditions, especially for soil conditions. Many studies indicated that vegetation factors could refect soil fertility, soil moisture, soil nutrient, and indirectly site quality [31, 32]. In the cold-temperate forests of Russia, Northern Europe, Canada, plant species or plant communities are widely used to evaluate sites [33–35]. Zhang Wanru [36] researched the use of vegetation types as the basis for forest site classification systems in China. However, in China, many plantations are damaged, and it is not easy to use indicator plants to evaluate these forest sites.

2.2.3 Human activity factor

Human activity can affect the forest; some are negative, such as removing litter from forest land and mining groundwater seriously, which will deteriorate the site, cause soil erosion, and lower groundwater levels. Some human activities can severely impact forests, such as the destruction of forests caused by the slashing and burning of agricultural activities in Europe [37]. From an ecological perspective, forest management is also a disturbance to forest growth, but most of them are positive effects, such as afforestation and reforestation [38] (**Figure 4**). Human activity factors are generally analyzed in forest site assessment as one of the driving forces for forming or changing other site factors, not as a constituent factor of site condition types.

2.2.4 Forest site dominant factor

Many factors affect the forest site and tree growth; see 2.2.1, 2.2.2, and 2.2.3. However, some factors have little effect on the growth and development of trees, and some factors play a decisive role. These decisive factors are called dominant factors. Generally, the climate is the dominant factor at the regional scale, and landform and soil are the dominant factors at the management unit scale [39]. There are two methods to determine the dominant factors. One is to analyze the relationship between each environmental factor and the essential living factors (light, heat, air,

Figure 4.
Reforestation site after clearcutting.

water, and nutrition) of trees to determine the most significant impact on living factors. On the other hand, it is to find out those environmental factors in extreme conditions and restrict plant growth [1]. Generally, the most restrictive factors play a leading role, such as drought, severe cold, strong wind, and extreme weather. For example, in Saihanba Forest farm, some afforestation area has very thin soil; the dominant factor is water (**Figure 5**).

2.3 Forest site classification

The forest site classification refers to a traditionally used method to determine the suited tree species in the right site and perform macro-classification and micro-classification [1, 40]. The system generally consists of multiple (level) taxa depend on the scale. The climate is the primary effect factor at the landscape and regional scale, whereas topography and soil at the local scale [27]. Usually, afforestation always tends to pay more attention to micro-forest sites because it directly relates them to the tree survival rate and have similar management properties. As climate change is concerned, afforestation is increasingly considering the macro-regional scales and the forest life cycle.

Figure 5.
Young Pinus sylvestris *forest of Saihanba Forest farm is planted on a thin site.*

Site classification methods are divided into two groups, single factor and multifactor methods [41]. Single factors classification systems depend on one factor to express a forest site, such as soil, indicator plant, or climate. The soil classification system was wildly used in many countries, like the United States and Ireland, to quantify site quality and determine its suitability for afforestation or timber yields [42–44]. Indicator plants or plant communities could also indicate a forest site's fertility and moisture status, especially in some humid climate regions or coastal countries, such as Scotland, Britain, Ireland, Finland, and British Columbia in Canada [31, 33, 34, 45].

Recently, with the rise of sustainable forest management, multifactor forest site classifications system has developed rapidly. The biogeoclimatic ecosystem classification in British Columbia has combined the climate, vegetation, and soil factors to assess the site productivity and guide the afforestation and forest management [46, 47]. The Finland upland forest site classification system consists of six clusters depend on the vegetation types and the site water conditions [34]. In Germany, the Baden-Wurttemberg silviculture forest site classification has three levels, landscape level, regional level, and local level. The landscape level contains subunit called growth districts, divided into smaller areas at the regional level depending on the climate and topographic; the basic ecological units are called site units according to soil and vegetation [48].

In China, the research group of "China Forest Site Classification," headed by Zhan Zhaoning, proposed a site classification system in 1989 [49]. Zhang Wanru formally established a site classification system based on timber forests [36]. The Chinese Forest Site Classification and Chinese Forest Site are national classification systems. Forest Site Classification system in China can be divided into six levels [49]: site area, site region, site sub-region, site type district, group of site type, and site type. According to this classification system, China divides forest sites into 8 site regions, 50 site areas, 166 site sub-areas, 494 site type communities, 1716 site type groups, and 4463 site types.

2.4 Forest site quality and assessment

Site quality refers to a given forest's production potential on a forest site or forest land's ability to grow trees [1, 42]. Site quality impact factors include climate factors, soil factors, and biological factors, determining forest growth quality and quantity. Generally, forest sites' potential productivity should be predicted and evaluated before afforestation, and the same or similar forest sites should be classified.

Forest site quality evaluation methods can be simplified into direct and indirect methods. The direct evaluation method refers to using the forest's harvest and growth data to evaluate site quality, such as volume, tree height, site index. In 1881, the German forest scientist Von Baur used the stand average tree height to indicate site class; Assmann recommended using top tree height instead later in 1961 [50, 51]. In the United States, from about 1910 to 1925, there were three different site evaluation methods: some people strongly agreed to express by volume; another group of people favored using the "forest site type system", which is based on the plant to indicate site types; the third part support the use of site index [42]. The indirect evaluation method refers to assess site quality with the characteristics of physiographic, climate, edaphic variables, and understory [52].

Site index (SI) is the most commonly used, relatively density-independent quantitative indicator of site productivity [53]. It was defined as the top height of the trees at a specified (index) age [54]. Many countries used site index to evaluate the site quality among different species, like *Picea abies* in Germany, *Quercus suber* in Portugal, *Eucalyptus Grandis* in South Africa, *Pinus tabuliformis* in China [55–58]. Site index established by the multiple regression analysis methods indicates the relationship between the average height of the dominant tree or tallest trees (also called the upper canopy height). We can clearly see the highest height from the site index table that Chinese pine can grow on different sites and at different ages [57] (**Table 1**).

Some scientists used edaphic or physiographic variables in site quality models [59, 60]. In contrast, some scientists have combined the site index with climate data to establish a stable site index that evaluates site quality under climate change [61] (**Table 2**). Using a site index to test the site quality of uneven-aged-mixed forest stands has low accuracy. McNab et al. [62] used the indicator species method combined with the site index to evaluate the hardwood stands' site productivity in Western North Carolina and pointed out that the good quality site's predicting accuracy is higher (85 percent accuracy) than the poorer site (60 percent accuracy).

Tree age/a	**Site index**						
	4	**5**	**6**	**7**	**8**	**9**	**10**
15	2.2 ~ 2.9	3.0 ~ 3.6	3.7 ~ 4.3	4.4 ~ 5.0	5.1 ~ 5.7	5.8 ~ 6.5	6.6 ~ 7.2
20	3.0 ~ 4.0	4.1 ~ 5.0	5.1 ~ 6.0	6.1 ~ 7.0	7.1 ~ 8.0	8.1 ~ 9.0	9.1 ~ 10.0
25	3.5 ~ 4.7	4.8 ~ 5.9	6.0 ~ 7.0	7.1 ~ 8.2	8.3 ~ 9.4	9.5 ~ 10.5	10.6 ~ 11.7
30	3.9 ~ 5.1	5.2 ~ 6.4	6.5 ~ 7.7	7.8 ~ 9.0	9.1 ~ 10.3	10.4 ~ 11.5	11.6 ~ 12.9
35	4.1 ~ 5.5	5.6 ~ 6.8	6.9 ~ 8.2	8.3 ~ 9.6	9.7 ~ 10.9	11.0 ~ 12.3	12.4 ~ 13.7
40	4.3 ~ 5.7	5.8 ~ 7.1	7.2 ~ 8.6	8.7 ~ 10.0	10.1 ~ 11.4	11.5 ~ 12.8	12.9 ~ 14.3
45	4.4 ~ 5.9	6.0 ~ 7.4	7.5 ~ 8.8	8.9 ~ 10.3	10.4 ~ 11.8	11.9 ~ 13.3	13.4 ~ 14.7
50	4.5 ~ 6.0	6.1 ~ 7.6	7.7 ~ 9.1	9.2 ~ 10.6	10.7 ~ 12.1	12.2 ~ 13.6	13.7 ~ 15.1
55	4.6 ~ 6.2	6.3 ~ 7.7	7.8 ~ 9.3	9.4 ~ 10.8	10.9 ~ 12.3	12.4 ~ 13.9	14.0 ~ 15.4

Table 1.
Site index table of Pinus tabulaeformis *Carriese plantation.*

Variables	**Subvariables**
Summer/annual temperature	Mean annual temperature (°C)
	Mean temperature warmest quarter (°C)
	Mean temperature May to Sept. (°C)
	Max. temperature warmest month (°C)
	Mean July temperature (°C)
Winter temperature	Mean January temperature (°C)
	Min. temperature coldest month (°C)
Precipitation	Annual precipitation sum (mm)
	Precipitation sum warmest quarter (mm)
	Precipitation sum May to Sept. (mm)
Continentality	Continentality index
	Tmax_wm-Tmin_cm (°C)
	T_wq-T1 (°C)
Elevation	Elevation (m)

Table 2.
Characterization of the environmental variables for spruce and beech plots used for site index model fitting.

3. Site preparation

Site preparation is a crucial activity affecting the survival rate of afforestation. It is the step after forest classification and before planting. Site preparation is various among afforestation land types. The treatment applied to slash, ground story vegetation, forest floor, and soil to exclude or reduce competing vegetation, pests, fire, and make the site suitable for afforestation or natural regeneration [2]. Site preparation usually includes mechanical and chemical methods, include mounding, scalping, trenching, bedding, chopping, herbicide, prescribed burning, et al.

3.1 Types of afforestation lands

The types of afforestation land are different in each country. Some countries have diverse terrains, such as China and the United States, and some countries have a few terrain types, such as some European countries [1]. This is the primary reason that affects the type of afforestation land. There are five types of afforestation land in china, namely barren mountains and wasteland, farmland, logging, and burning land, and secondary forest land [1]. The site quality of farmland is high, and the site quality of other afforestation sites is poor. In Ireland, the country's afforestable land was divided into four types based on biophysical factors, biological factors, national and EU designations and policies, and potential afforestation, respectively [63]. Kadam et al. [17] used the Land Suitability Analysis (LSA) method to divide the afforestation land types of Western Ghat in India into four classes (highly, moderately, marginally, and not suitable); the main dividing factors are topographical factors, soil factors, and meteorological factors.

3.2 Mechanical methods

The primary purpose of mechanical methods is to remove undesirable plants, reduce their growth, protect the surface soils, and improve site quality [2]. Mechanical methods can redistribute the dead vegetation, like slashing or chopping; they also can reshape the soil surface, like bedding, plowing, and mounding.

Mechanical site preparation can influence the species diversity, quantity, composition of underground vegetation. Sebesta et al's research showed that mechanical site preparation decreased the species richness of the understorey and increased the number of non-native species coursed by soil disturbance [64]. Newmaster et al. revealed no differences in the frequency of native species and composition in mechanical site preparation [65].

Proper site preparation methods, either mechanical or chemical methods, can improve both conifers and hardwoods' survival rate and growth [66–68]. However, in coniferous and broad-leaved mixed forests, different site preparation methods lead to different effects. Cain et al's research showed that mechanical or chemical site preparation methods reduced the density and stocking of Oak in a pine-hardwood mixed forest [69]. Mohler et al's study also mentioned that red Oak trees had benefited most from the larger gaps without site preparation [70]. Therefore, mechanical methods should be applied carefully and adapted to site conditions.

The cost of mechanical site preparation should be a consideration. Such as slash can be expensive or course diseases, but it can reduce forest fires' risk, protect the seedlings, and provide organic and inorganic nutrients [71–73]. The equipment and the labor cost is expensive for afforestation (**Figure 6**).

Figure 6.
Use a tractor for site preparation.

3.3 Chemical methods

Chemical methods usually refer to herbicides, pesticides, and fertilization. Chemical site preparation, like herbicides or pesticides, are both harmful and beneficial to site quality. The herbicide can promote trees' early survival rate and have a long-term effect on maintaining forest growth. However, people are also concerned about the environmental effect and cost [74]. It should be noted that herbicide use is related to the length of time of different research. Much long-term research of longleaf pine showed that herbicide applied for site preparation increased seedling growth and had a lasting improvement effect. However, some short time studies reported that the longleaf pine seedling survival was unaffected or reduced by herbicide [68, 75, 76]. Compared with the mechanical methods, herbicides' cost looks more efficient; it was wildly used in South American, especially in pine plantations [66]. Callaghan et al's study showed that herbicide could reduce the hardwood competition and improve the *Pinus taeda* growth, as it costs less [77].

Fertilization can supplement the nutrient loss of the soil caused by logging and increase the seedlings' survival rate. Nitrogen, phosphorus, and potassium (NPK) are the main nutrient elements in fertilizers [78]. Although the fertilization costs a lot, it can improve either hardwood or conifer seedling growth [79, 80]. Fertilization is also used for larger trees. A fertilization study for conifers in Finland showed that after ten years of fertilization if harvested as sawlogs or pulpwood, the additional volume increment was 25% and 75%, respectively, higher than non-fertilized forests [81].

3.4 Site water management

Site water management also plays an essential role in site preparation in certain areas. If drainage is not considered in some low-lying afforestation lands, the afforestation will fail (**Figure 7**). Flooding is a treatment that use channel or dikes to guide the water from afforestation site with high moisture, like coastal and riparian lands, shrimp ponds, swamps. Flooding or irrigation also can reduce the salt and alkali content of saline soil [82]. Irrigation is an essential way in improving site water conditions in water-deficient areas, such as the Middle East, South Africa, China, India, especially for the cultivation of timber forests [83–85].

Figure 7.
After a rain, the accumulation of water in afforestation land caused the death of seedlings.

4. Afforestation materials

4.1 Afforestation with seeds

Direct seeding is a widely used afforestation method. Compared with the seedling method, it has the merits of simple operation, high efficiency, low cost, and can be used over hard to reach areas. Direct seeding was considered the 'best practice' for producing seedlings, regeneration, and afforestation [8]. Sowing seeds directly to forest land without lifting seedlings, packaging, transportation, and planting, the root system of seedlings will not be damaged. Therefore, direct seeding is more "close-to-nature." It can keep intact natural distribution and expansion of the root system, especially of the pivot root the tree species. The seeds that germinate and grow on the forested land are better adapted to the climate and soil conditions [86]. However, it has strict requirements on the water, heat, and vegetation conditions. Compared with seedling afforestation, seedlings formed by direct seeding grow slowly at the initial stage, so it takes a longer time to reach crown closure [86]. Sometimes, the seeds after sowing are easily damaged by birds and animals, trampled by livestock, and human destruction, so it is necessary to strengthen the management and protection.

Generally, before sowing, the seeds should be disinfected, soaked, sprouted, dressed, coating, and gluing [1]. The purpose of pre-sowing treatment is to shorten the time of seeds in the soil before germination, ensure the emergence of seedlings orderly, and prevent the harm of birds, mammals, and diseases. The germination rate after sowing is related to seed size and weight. Moreover, the establishment rate is accord to the timing of seeding, planting practices, microsite environment, competitive vegetation, and seed predation [87–90].

Seed afforestation methods include seed burial, spot, and broadcast [91–93]. Seed burial refers to put the seeds under the soil to store water, preserve moisture, create conditions for germination, and protect seeds. Some seeding experiments conducted in the nursery show that the suitable spot seeding depth is between one and two times the seed width [94, 95]. Broadcast seeding has the advantages of small workload, simple construction, great flexibility in site selection, and is widely used for barren mountains and wasteland (including desert) and cutting and burning slash site [1]. No matter which seeding method, it is required that the covering soil thickness is appropriate (except for broadcast sowing) [1].

Different sowing season can affect the seed germination rate, which should be determined according to the tree species' characteristics and environmental conditions [96]. Many studies were conducted about the suitable sowing season, such as some pine species in southern US and Finnland (spring seeding), *Pinus palustris* in the US (fall seeding), *Fraxinus excelsior,* and *Acer pseudoplatanus* in the UK (winter seeding) [5, 96, 97]. Some species can be sowed in multiple seasons, like temperate hardwoods in the US [98].

4.2 Afforestation with seedlings

There are two types of seedlings, bare root and containerized. Compared with the direct seeding, seedlings have a complete or partial root system. It can be planted in almost all suitable sites, and site conditions requirements are not high. In general, container seedlings are used for afforestation under difficult site conditions. Furthermore, seedlings are usually grown in nurseries or a controlled greenhouse environment, transplantation more or less damaged the root system, and both bare root seedlings and container seedlings can be produced all year round [6, 99].

Bareroot seedlings have always been promoted for reforestation projects because it can be easily hand-carried by forester and less expensive than containerized seedlings [100]. The survival ratio of bare root seedlings is affected by the seedling vitality, planting time, or season, especially the soil moisture and temperature [101]. The water content in the seedling is the most critical factor affecting the seedling vitality. To maintain the seedlings' water balance, appropriate treatment measures should be taken before planting, such as pruning and partial-root cutting, which can remove most of the seedling leaves, branches, trunk, and roots, reducing water evaporation [102, 103].

Compared with seeding afforestation, container seedlings show better environmental adaptability and stress resistance because of their protected root systems (**Figure 8**). Container seedlings have increased survival rates of or more than other transplant types and show improved growth on adverse sites, though they cost more than bare-root seedlings [104]. Under droughty conditions, container seedlings survived and grew better than bare root seedlings [105]. Furthermore, some researchers mentioned no difference between bare root seedlings and container seedlings when soil moisture was adequate at the planting time [106].

Figure 8.
Container seedlings of Larix principis-*rupprechtii in a greenhouse.*

Afforestation with seedlings requires a series of practices include lifting, storage, transport to the site, and planting. All of these operations can affect seedling performance [107]. The protection of seedling roots during those operations is critical to maintaining the water content and seedlings' vitality. To this end, it is advantageous to shorten the operation time of each process; grading and packaging should be carried out in a shady, wet, and cold environment. Some studies have shown that exposing the seedlings' roots to air can limit their growth [108, 109]. The seedling's roots should be closely contacted with the surrounding soil during planting. The planting depth of each seedling should be the same in the nursery; sometimes, a little deeper is more favorable. After planting, the bare root seedlings generally have a process of root restoration and adaptation.

Hole planting is a common method that is suitable for all kinds of bare root seedlings. Digging tools can be large machinery, shovel, mattock, spade (**Figure 9**). The depth and width of the hole are determined according to the seedling root's length and width [110]. Generally, the planting depth should be about 3 cm above the original soil seal at the seedling's ground path [1]. The planting method can also be divided into a single plant and cluster plant according to one or more plants per hole. Recently, seedlings with root-ball were widely used in afforestation, especially in urban afforestation; it can maintain a relatively complete root system, and the planting survival rate is high, but the weight is massive, so the afforestation cost is relatively high.

To ensure planting seedlings, it is necessary to select the appropriate season and time according to the climate and soil conditions. Bareroot seedlings and container seedlings are produced in one to four growing seasons or one to two years [6, 111]. Theoretically, the appropriate planting time should be when the physiological activity of the aboveground part of the seedling is weak (deciduous broadleaf tree species are in the deciduous stage), and the physiological activity of the root is vital, so the root healing ability is strong [112]. Generally, hardwood seedlings must be planted in late winter or early spring, when the seedlings are dormant and the ground has thawed [113].

4.3 Afforestation with cutting

Seeds and seedlings are sexual afforestation method, and many trees also have asexual reproduction ability. Cutting is a piece of a plant that can be used in

Figure 9.
Use excavators to dig a hole for the afforestation.

Figure 10.
Root sprouting of black locust.

afforestation. It can be taken from stems, branches, leaves, roots and directly plant on forest land. Cuttings can maintain the parent tree's target characteristics, such as high yield, fast growth, and stress resistance ability [114, 115]. Some research showed that cuttings' behavior varies with age, genotypes, the parent plant's physiological status, cutting position, and temperature [116, 117]. Besides, sprouting is another afforestation method that can produce a new forest. Sprouts are more resistant to disturbance than seed-origin seedlings and grow fast [7, 118]. Compared with the plant with seed and seedling, cutting afforestation is labor-saving, time-saving, and low-cost.

Stump or root adventitious sprouts are commonly used sprouting materials, and it can rapidly produce many adventitious roots with strong water-absorbing ability, such species include *Populus*, *Robinia pseudoacacia*, *Salix*, *Cunninghamia lanceolata*, *Ziziphus jujuba*, *Paulownia tomentosa*, and *Toxicodendron vernicifluum* [119–121] (**Figure 10**). It is believed that the sprouting ability is related to species, stem size, age, management intensity [7, 122–125]. It was reported that different damage treatments on beech roots could cause different sprouting results [126]. Some sproutings come from the stump after cutting, like *Cercidiphyllum japonicum* and some tropical species [127, 128]. Recently, sprouting has been widely used in coppice forest cultivation. In China, Europe, and the Americas, this method is used to develop short rotation energy forests to get biomass raw materials [129, 130].

The cutting plant or sprouting harvest season varies with the tree species and region. Generally, the most suitable time is the same as planting seedlings, like fall, winter, and early spring [127, 131]. However, some studies showed that harvest season had no effect on sprout number but can affect the dominant sprout height in the first year [132, 133].

5. Conclusion

Afforestation and reforestation activities must be considered systematically and integrally. More and more studies have shown that making afforestation plans from the perspective of forest ecosystems is the future trend. Using multi-factor methods to analyze forest site characteristics will become the primary site evaluation and classification method. Although the cost is high, with the continuous advancement of technology, afforestation machinery will be popularized in the afforestation process, such as land preparation and planting. At the same time, many future afforestation activities will fully consider climate change dynamics on the forest site and the forest itself and determine afforestation tree species, materials and methods.

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References

[1] Sheng G. Silviculture. Beijing: China Forestry Press; 2001.

[2] Bassman JH, Smith DM. The Practice of Silviculture. 10th ed. Vol. 40, Journal of Range Management. Oxford: Wiley; 1987. 289 p.

[3] Yao N, Konijnendijk van den Bosch CC, Yang J, Devisscher T, Wirtz Z, Jia L, et al. Beijing's 50 million new urban trees: Strategic governance for large-scale urban afforestation. Urban For Urban Green [Internet]. 2019 Aug 1 [cited 2019 Aug 6];44:126392. Available from: https://www.sciencedirect.com/science/article/pii/S1618866719300378

[4] Nyland RD. Silviculture: concepts and applications. Plant Biosystems. 2011.

[5] Barnett JP. Direct Seeding Southern Pines : Development of Techniques , Use , and Current Status. Tree Plant Notes. 2014;57(1):35-48.

[6] Daniels TD, Simpson DG. Seedling production and processing: bareroot. Regen Br Columbia's For Univ Br Columbia Press Vancouver. 1990;206-226.

[7] Bond WJ, Midgley JJ. Ecology of sprouting in woody plants: The persistence niche. Vol. 16, Trends in Ecology and Evolution. 2001. p. 45-51.

[8] Bassman JH, Smith DM. The Practice of Silviculture. 9th ed. Vol. 40, Journal of Range Management. New York: Wiley; 1987. 289 p.

[9] Valentine HT. Height growth, site index, and carbon metabolism. Silva Fenn. 1997;31(3):251-263.

[10] Spiecker H, Mielikäinen K, Köhl M, Skovsgaard JP. Growth Trends in European Forests. Growth Trends in European Forests. 1996.

[11] Vannière B. Tables de production pour les forêts françaises. Nancy: Engref; 1984. 159 p.

[12] Schober R. Ertragstafenln wichtiger baumarten bei verschiedener durchforstung. 4th ed. Frankfurt: Sauerlander; 1987. 116 p.

[13] Sebald O. Ökologische Artengruppen für den Wuchbezirk "Orberer Neckar." 1964.

[14] Barnes B V., Pregitzer KS, Spies TA, Spooner VH. Ecological forest site classification. J For. 1982;80(8):493-498.

[15] Hills GA, Pierpoint G. Forest site evaluation in Ontario. Research Report. 1960.

[16] Skovsgaard JP, Vanclay JK. Forest site productivity: A review of spatial and temporal variability in natural site conditions. Forestry. 2013;86(3):305-315.

[17] Kadam A, M R, Umrikar B, Bhagat V, Wagh V, Sankua RN. Land Suitability Analysis for Afforestation in Semi-arid Watershed of Western Ghat, India: A Groundwater Recharge Perspective. Geol Ecol Landscapes. 2020;1-13.

[18] Byers HR, Landsberg HE, Wexler H, Haurwitz B, Spilhaus AF, Willett HC, et al. Climate—The Synthesis of Weather. In: Compendium of Meteorology. 1951. p. 967-975.

[19] Oke TR. Boundary Layer Climates. Boundary Layer Climates. Routledge; 2002.

[20] Moreno-Fernández D, Álvarez-González JG, Rodríguez-Soalleiro R, Pasalodos-Tato M, Cañellas I, Montes F, et al. National-scale assessment of forest site productivity in Spain. For Ecol Manage. 2018;417:197-207.

[21] Babst F, Poulter B, Trouet V, Tan K, Neuwirth B, Wilson R, et al. Site- and species-specific responses of forest growth to climate across the European continent. Glob Ecol Biogeogr. 2013;22(6):706-717.

[22] Pretzsch H, Biber P, Schütze G, Uhl E, Rötzer T. Forest stand growth dynamics in Central Europe have accelerated since 1870. Nat Commun. 2014;5(1):1-10.

[23] Worrell R, Malcolm DC. Productivity of sitka spruce in northern britain 1. The effects of elevation and climate. Forestry. 1990;63(2):105-118.

[24] Moser G, Leuschner C, Hertel D, Graefe S, Soethe N, Iost S. Elevation effects on the carbon budget of tropical mountain forests (S Ecuador): The role of the belowground compartment. Glob Chang Biol. 2011;17(6):2211-2226.

[25] Looby CI, Maltz MR, Treseder KK. Belowground responses to elevation in a changing cloud forest. Ecol Evol. 2016;6(7):1996-2009.

[26] Bronick CJ, Lal R. Soil structure and management: A review. Geoderma. 2005;

[27] Quichimbo P, Jiménez L, Veintimilla D, Tischer A, Günter S, Mosandl R, et al. Forest site classification in the Southern Andean region of ecuador: A case study of pine plantations to collect a base of soil attributes. Forests. 2017;8(12):473.

[28] Louw JH. The value of six key soil variables for incorporation into a South African forest site classification system. Vol. 78, Southern Forests. 2016. p. 81-95.

[29] Du LI, Bao Ziyun HW. Experimental study on groundwater regulation in Yinbei Irrigation District. Northwest water Resour water Eng. 1995;6(3):77-83.

[30] Schnepf C. Forest Habitat Types: A Plant-Based Clue to Better Forestry [Internet]. Woodland NOTES. 2001 [cited 2020 Dec 11]. p. 1-3. Available from: https://www.uidaho.edu/-/media/UIdaho-Responsive/Files/Extension/topic/forestry/FM34-Forest-Habitat-Types-A-Plant-Based-Clue-to-Better-Forestry.pdf?la=en&hash=5798A8D3BA4632045F5C305285F783B4FC4EBE81

[31] Wilson SMG, Pyatt DG, Ray D, Malcolm DC, Connolly T. Indices of soil nitrogen availability for an ecological site classification of British forests. For Ecol Manage. 2005;220(1-3):51-65.

[32] Greller AM, Klinka K, Krajina VJ, Ceska A, Scagel AM. Indicator Plants of Coastal British Columbia. Bull Torrey Bot Club. 1990;117(2):185.

[33] DeLong C. A Field Guide to Site Identification and Interpretation for the Southeast Portion of the Prince George Forest Region. B.C. Min. Forests. Land Management Handbook No. 28. 2003. 28-111 p.

[34] Kuusipalo J. An ecological study of upland forest site classification in southern Finland. Acta For Fenn. 1985;192.

[35] Krestov P V. Forest Vegetation of Easternmost Russia (Russian Far East). In: Forest Vegetation of Northeast Asia. 2003. p. 93-180.

[36] Zhang W. Forest sites in China. Beijing: Science Press; 1997.

[37] Taavitsainen JP, Simola H, Grönlund E. Cultivation history beyond the periphery: early agriculture in the North European boreal forest. J World Prehistory. 1998;12(2):199-253.

[38] Osman KT. Forest soils: Properties and management. Vol. 9783319025, Forest Soils: Properties and Management. 2013. 1-217 p.

[39] Bailey RG. Ecosystem geography. Ecosystem geography. 1996.

[40] Farrelly N, Fealy RM, Radford T. The use of site factors and site classification methods for the assessment of site quality and forest productivity in Ireland. Irish For [Internet]. 2009;64(1&2):21-38. Available from: http://www.teagasc.ie/forestry/docs/research/2009 Irish Forestry 21-38 A1.pdf

[41] Savill PS. Land classification for plantation forestry. Irish For. 1983; 40(2):78-91.

[42] Carmean WH. Forest site quality evaluation in The United States. Adv Agron. 1975;27(C):209-269.

[43] Bulfin M. Availability of land for forestry in Ireland and its suitability for Sitka spruce (Picea sitchensis (Bong.) Carr). Irish For. 1987;44(1):18-31.

[44] O'Carroll C, Farrell EP. Relationships between soil characteristics and yield for Norway spruce on fen peat in central Ireland. For Ecol Manage. 1993;57(1-4):61-70.

[45] Dickson DA, Savill PS. Early growth of Picea sitchensis (Bong.) Carr. On deep oligotrophic peat in Northern Ireland. Forestry. 1974;47(1):57-88.

[46] Kayahara GJ, Klinka K, Schroff AC. The relationship of site index to synoptic estimates of soil moisture and nutrients for western redcedar (Thuja plicata) in southern coastal British Columbia. Northwest Sci. 1997;71(3):167-173.

[47] Pyatt G, Suarez J. An ecological site classification for forestry in Great Britain with special reference to Grampian, Scotland. In: Forestry Commission Technical Paper. 1997. p. 1-100.

[48] Spathelf P. Seminatural silviculture in Southwest Germany. For Chron. 1997;73(6):715-722.

[49] Zhan Z, Wang G. Forest site classification in China. Beijing: China Forestry Press; 1989.

[50] F von B. Das Forstl. In: Versuch-swesen. Augsburg; 1881. p. 359.

[51] Assmann E. Waldertragskunde. Organische Produktion, Struktur, Zuwachs und Ertrag von Waldbeständen. München: BLV Verlagsgesellschaft; 1961. 490 p.

[52] Marques CP. Evaluating site quality of even-aged maritime pine stands in northern Portugal using direct and indirect methods. For Ecol Manage. 1991;41(3-4):193-204.

[53] Weng Y. Forest Measurements. 5th ed. Vol. 117, Journal of Forestry. Long Grove: Waveland Press; 2019. 528-529 p.

[54] Sharma M, Amateis RL, Burkhart HE. Top height definition and its effect on site index determination in thinned and unthinned loblolly pine plantations. For Ecol Manage. 2002;168(1-3):163-75.

[55] Brandl S, Falk W, Klemmt HJ, Stricker G, Bender A, Rötzer T, et al. Possibilities and limitations of spatially explicit site index modelling for spruce based on national forest inventory data and digital maps of soil and climate in Bavaria (SE Germany). Forests. 2014;5(11):2626-2646.

[56] Paulo JA, Palma JHN, Gomes AA, Faias SP, Tomé J, Tomé M. Predicting site index from climate and soil variables for cork oak (Quercus suber L.) stands in Portugal. New For. 2015;46(2):293-307.

[57] Jie D, Luyi M, LiMing J, YaQin H, NingNing G. Establishment and application of site index table for Pinus tabulaeformis plantation in the low elevation area of Beijing. [Internet]. Vol. 45, Scientia Silvae Sinicae. 2009. p. 7-12. Available from: http://lyke.chinajournal.net.cn

[58] Louw JH. A site-growth study of eucalyptus grandis in the mpumalanga escarpment area. South African For J. 1997;180(1):1-14.

[59] Bravo F, Lucà M, Mercurio R, Sidari M, Muscolo A. Soil and forest productivity: A case study from Stone pine (Pinus pinea L.) stands in Calabria (southern Italy). IForest. 2011;4(JANUARY).

[60] Bueis T, Bravo F, Pando V, Turrión MB. Relationship between environmental parameters and Pinus sylvestris L. site index in forest plantations in northern Spain acidic plateau. IForest. 2016;9(JUNE2016):394-401.

[61] Brandl S, Mette T, Falk W, Vallet P, Rötzer T, Pretzsch H. Static site indices from different national forest inventories: harmonization and prediction from site conditions. Ann For Sci. 2018;75(2):56.

[62] Mcnab WH, Lloyd FT, Loftis DL. Preliminary Evaluation of Methods for Classifying Forest Site Productivity Based on Species Composition in Western North Carolina by W. Henry McNab', F. Thomas Lloyd', and David L. Loftis'. In: Doruska PF, Bragg DC, editors. Methods. Monticello, AR: Arkansas Forest Resources Center; 2002. p. 10-18.

[63] Group CLAW. Land Availability for Afforestation: Exploring opportunities for expanding Ireland's forest resource. COFORD; 2016. 49 p.

[64] Šebesta J, Rogers PC, Maděra P, Koutecký T, Dufour S, Řepka R. Long-term effects of mechanical site preparation on understorey plant communities in lowland floodplain forests. For Ecol Manage. 2021;480.

[65] Newmaster SG, Parker WC, Bell FW, Paterson JM. Effects of forest floor disturbances by mechanical site preparation on floristic diversity in a central Ontario clearcut. For Ecol Manage. 2007;246(2-3):196-207.

[66] Fox TR, Jokela EJ, Allen HL. The development of pine plantation silviculture in the Southern United States. Journal of Forestry. 2007.

[67] Knapp BO, Wang GG, Walker JL, Cohen S. Effects of site preparation treatments on early growth and survival of planted longleaf pine (Pinus palustris Mill.) seedlings in North Carolina. For Ecol Manage. 2006;226(1-3):122-128.

[68] Crouch CD, Knapp BO, Cohen SA, Stambaugh MC, Walker JL, Wang GG. Site preparation for longleaf pine restoration on hydric sites: Stand development through 15 years after planting. For Ecol Manage. 2020;461.

[69] Cain MD, Shelton MG. Effects of opening size and site preparation method on vegetation development after implementing group selection in a pine-hardwood stand. In: Proceedings, Southern Weed Science Society: New Century: New Opportunities, 54th Annual Meeting. 2001. p. 191-196.

[70] Mohler C, Bataineh M, Bragg DC, Ficklin R, Pelkki M, Olson M. Long-term effects of group opening size and site preparation method on gap-cohort development in a temperate mixedwood forest. For Ecol Manage. 2021;480.

[71] Pollet J, Omi PN. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forests. Int J Wildl Fire. 2002;11(1):1-10.

[72] Devine WD, Harrington CA. Influence of harvest residues and vegetation on microsite soil and air temperatures in a young conifer plantation. Agric For Meteorol. 2007;145(1-2):125-138.

[73] Eisenbies MH, Vance ED, Aust WM, Seiler JR. Intensive utilization of harvest

Residues in southern pine plantations: Quantities available and implications for nutrient budgets and ustainable Site Productivity. Bioenergy Res. 2009;2(3):90-98.

[74] Thompson ID, Baker JA, Ter-Mikaelian M. A review of the long-term effects of post-harvest silviculture on vertebrate wildlife, and predictive models, with an emphasis on boreal forests in Ontario, Canada. For Ecol Manage. 2003;177(1-3):441-469.

[75] Freeman JE, Jose S. The role of herbicide in savanna restoration: Effects of shrub reduction treatments on the understory and overstory of a longleaf pine flatwoods. For Ecol Manage. 2009;257(3):978-986.

[76] Addington RN, Greene TA, Elmore ML, Prior CE, Harrison WC. Influence of herbicide site preparation on longleaf pine ecosystem development and fire management. South J Appl For. 2012;36(4):173-180.

[77] Callaghan DW, Khanal PN, Straka TJ, Hagan DL. Influence of forestry practices cost on financial performance of forestry investments. Resources. 2019;8(1):28.

[78] Binkley D, Fisher RF. Ecology and Management of Forest Soils: Fourth Edition. Ecology and Management of Forest Soils: Fourth Edition. 2013. 1-347 p.

[79] Lesko J, Jacobs DF. Conversion of conifer plantations to native hardwoods: influences of overstory and fertilization on artificial regeneration. New For. 2018;49(6):829-849.

[80] Duan J, Xu C, Jacobs DF, Ma L, Wei H, Jiang L, et al. Exponential nutrient loading shortens the cultural period of Larix olgensis seedlings. Scand J For Res. 2013;28(5).

[81] Pukkala T. Optimal nitrogen fertilization of boreal conifer forest. For Ecosyst. 2017;4(1).

[82] Zhang J, Liu M, Qian H, Zhang Q, Wang Z. Spatial-temporal variation characteristics of water-salt movement in coastal saline soil improved by flooding and subsurface drainage. Nongye Gongcheng Xuebao/Transactions Chinese Soc Agric Eng. 2018;34(6):98-103.

[83] Mahmoud SH, Gan TY. Irrigation water management in arid regions of Middle East: Assessing spatio-temporal variation of actual evapotranspiration through remote sensing techniques and meteorological data. Agric Water Manag. 2019;212:35-47.

[84] Yan XL, Dai TF, Jia LM. Evaluation of the cumulative effect of drip irrigation and fertigation on productivity in a poplar plantation. Ann For Sci. 2018;75(1):5.

[85] Balasubramanian A, Prasath C, Radhakrishnan S, Manivasakan S. Comparision of Growth Performance in Timber Tree Species Cultivated under Drip Irrigation on Farm Lands. Int J Plant Soil Sci. 2017;16(4):1-4.

[86] Grossnickle S, Ivetić V. Direct Seeding in Reforestation – A Field Performance Review. Reforesta. 2017;(4):94-142.

[87] González-Rodríguez V, Navarro-Cerrillo RM, Villar R. Artificial regeneration with Quercus ilex L. and Quercus suber L. by direct seeding and planting in southern Spain. Ann For Sci. 2011;68(3):637-646.

[88] Hossain F, Elliott S, Chairuangsri S. Effectiveness of Direct Seeding for Forest Restoration on Severely Degraded Land in Lampang Province, Thailand. Open J For. 2014;04(05):512-519.

[89] Wendel GW. Converting hardwoods on poor sites to white pine by planning

and direct seeding. Vol. Res. Pap. Northeastern Forest Experiment Station, Forest Service, US Department of Agriculture; 1971. 19 p.

[90] Fraser JW. Experimental seed-spotting trials with black spruce on upland cutovers. 1981;23.

[91] Doust SJ, Erskine PD, Lamb D. Restoring rainforest species by direct seeding: Tree seedling establishment and growth performance on degraded land in the wet tropics of Australia. For Ecol Manage. 2008;256(5):1178-1188.

[92] Leverkus AB, Castro J, Puerta-Piñero C, Rey Benayas JM. Suitability of the management of habitat complexity, acorn burial depth, and a chemical repellent for post-fire reforestation of oaks. Ecol Eng. 2013;53:15-22.

[93] Haywood JD, Barnett JP. Comparing methods of artificially regenerating loblolly and slash pines: container planting, bareroot planting, and spot seeding. Tree Plant notes - US Dep Agric For Serv [Internet]. 1994;45(2):63—67. Available from: http://europepmc.org/abstract/AGR/IND20462211

[94] Pigott JP, Brown PH, Williams MR. Direct seeding trees on farmland in the Western Australian wheatbelt. CALMScience. 1994;1(2):131-158.

[95] Johnson RL, Krinard RM. Regeneration of Oaks By Direct Seeding. In: In: Third symposium of southeastern hardwoods 1985 April 16-17; Dothan, AL 56-65 [Internet]. 1985. p. 56-65. Available from: https://www.srs.fs.usda.gov/pubs/ja/1985/ja_1985_johnson_004.pdf

[96] Jinks RL, Willoughby I, Baker C. Direct seeding of ash (Fraxinus excelsior L.) and sycamore (Acer pseudoplatanus L.): The effects of sowing date, pre-emergent herbicides, cultivation, and protection on seedling emergence and survival. For Ecol Manage. 2006;237(1-3):373-386.

[97] De Chantal M, Leinonen K, Ilvesniemi H, Westman CJ. Effects of site preparation on soil properties and on morphology of Pinus sylvestris and Picea abies seedlings sown at different dates. New For. 2004;27(2):159-173.

[98] Stanturf JA, Schweitzer CJ, Gardiner ES. Afforestation of marginal agricultural land in the lower Mississippi River Alluvial Valley, U.S.A. Silva Fenn. 1998;32(3):281-297.

[99] Dey DC, Jacobs D, McNabb K, Miller G, Baldwin V, Foster G. Artificial regeneration of major oak (Quercus) species in the eastern United States - A review of the literature. Vol. 54, Forest Science. 2008. p. 77-106.

[100] Anella L, Hennessey TC, Lorenzi EM. Growth of balled-and-burlapped versus bare-root trees in Oklahoma, U.S. Arboric Urban For. 2008;

[101] Kļaviņa D, Gaitnieks T, Menkis A. Survival, growth and ectomycorrhizal community development of container- and bare-root grown Pinus sylvestris and Picea abies seedlings outplanted on a forest clear-cut. Balt For. 2013;19(1):39-49.

[102] South DB. Effects of Top-Pruning on Survival of Southern Pines and Hardwoods. Proc Ninth Bienn South Silvic Res Conf. 1998;623.

[103] Sayer MAS, Sung SJS, Haywood JD. Longleaf pine root system development and seedling quality in response to copper root pruning and cavity size. South J Appl For. 2011;35(1):5-11.

[104] Stein WI, Edwards JL, Tinus RW. Outlook for Container-Grown Seedling Use in Reforestation. J For. 1975;73(6):337-341.

[105] Grossnickle SC, El-Kassaby YA. Bareroot versus container stocktypes: a performance comparison. Vol. 47, New Forests. 2016. p. 1-51.

[106] South DB, Barnett JP. Herbicides and planting date affect early performance of container-grown and bare-root loblolly pine seedlings in Alabama. New For. 1986;1(1):17-27.

[107] Parker B. Ecophysiology of Northern Spruce Species: The Performance of Planted Seedlings. Tree Physiol. 2001;21(7):415-416.

[108] Deans JD, Lundberg C, Tabbush PM, Cannell MGR, Sheppard LJ, Murray MB. The influence of desiccation, rough handling and cold storage on the quality and establishment of sitka spruce planting stock. Forestry. 1990;63(2):129-141.

[109] McKay HM. Frost hardiness and cold-storage tolerance of the root system of picea sitchensis, pseudotsuga menziesii, larix kaempferi and pinus sylvestris bare-root seedlings. Scand J For Res. 1994;9(1-4):203-213.

[110] Region USFSS. A Guide to the Care and Planting of Southern Pine Seedlings [Internet]. USDA Forest Service, Southern Region; 1996. (Management bulletin R8). Available from: https://books.google.com.sg/books?id=pgpFKH-zN9AC

[111] Landis TD, Tinus RW, McDonald SE, Barnett JP. Containers and Growing Media, Vol. 2. Contain Tree Nurs Manual Agric Handb 674. 1990;88.

[112] Margolis HA, Brand DG. An ecophysiological basis for understanding plantation establishment. Can J For Res. 1990;20(4):375-390.

[113] Teskey RO, Grier CC, Hinckley TM. Change in photosynthesis and water relations with age and season in Abies amabilis. Can J For Res. 1984;14(1):77-84.

[114] Cox DA. Hartmann and Kester's Plant Propagation Principles and Practices, 9th Edition. HortScience. 2018;53(5):741.

[115] Kauffman JB. Survival by Sprouting Following Fire in Tropical Forests of the Eastern Amazon. Biotropica. 1991;23(3):219.

[116] Henning RK. The Jatropha Booklet A Guide to the Jatropha System and its Dissemination in Africa. Bagani Gbr. 2003;37.

[117] Hansen J. Influence of cutting position and temperature during rooting on adventitious root formation and axillary bud break of Stephanotis floribunda. Sci Hortic (Amsterdam). 1989;40(4):345-354.

[118] Bellingham PJ, Sparrow AD. Resprouting as a life history strategy in woody plant communities. Vol. 89, Oikos. 2000. p. 409-416.

[119] Isikawa H. Generation of Adventitious Plant Organs by Tissue Culture Methods in Forest Trees. Bull For For Prod Res Inst. 1987;(343):119-154.

[120] Morgenson G. Vegetative Propagation of Poplar and Willow 1. Growth (Lakeland). 1992;2-4.

[121] Wan Gaiping (Nanjing Forestry Univ. J (China)). Nitrogen nutrition in sprouting buds from chinese fir stamps. Vol. 1995, Journal of Zhejiang Forestry College. 1995. p. 133-138.

[122] Lust N, Mohammady M. Regeneration of coppice. Silva Gandav. 1973;39.

[123] Zimmerman JK, III EME, Waide RB, Lodge DJ, Taylor CM, Brokaw NVL. Responses of Tree Species to Hurricane Winds in Subtropical Wet Forest in Puerto Rico: Implications for Tropical Tree Life Histories. J Ecol. 1994;82(4):911.

[124] Ito S, Gyokusen K. Analysis of the multi-stem clump structure of Litsea japonica Juss. growing in a coastal dwarf forest. Ecol Res. 1996;11(1):17-22.

[125] Keeley JE. Recruitment of seedlings and vegetative sprouts in unburned chaparral. Ecology. 1992;73(4):1194-1208.

[126] Jones RH, Raynal DJ. Root sprouting in American beech (Fagus grandifolia): Effects of root injury, root exposure, and season. For Ecol Manage. 1988;25(2):79-90.

[127] Kubo M, Sakio H, Shimano K, Ohno K. Age structure and dynamics of Cercidiphyllum japonicum sprouts based on growth ring analysis. For Ecol Manage. 2005;213(1-3):253-260.

[128] Negreros-Castillo P, B. Hall R. Sprouting capability of 17 tropical tree species after overstory removal in Quintana Roo, Mexico. For Ecol Manage. 2000;126(3):399-403.

[129] Dickmann DI. Silviculture and biology of short-rotation woody crops in temperate regions: Then and now. Biomass and Bioenergy. 2006;30(8-9):696-705.

[130] Meifang Y, Lu W, Honghui R, Xinshi Z. Biomass production and carbon sequestration of a short-rotation forest with different poplar clones in northwest China. Sci Total Environ [Internet]. 2017;586:1135-1140. Available from: http://dx.doi.org/10.1016/j.scitotenv.2017.02.103

[131] Pyttel PL, Fischer UF, Suchomel C, Gärtner SM, Bauhus J. The effect of harvesting on stump mortality and re-sprouting in aged oak coppice forests. For Ecol Manage. 2013;289:18-27.

[132] Johansson T. Sprouting of 10- to 50-year-old Betula pubescens in relation to felling time. For Ecol Manage. 1992;53(1-4):283-296.

[133] Xue Y, Zhang W, Zhou J, Ma C, Ma L. Effects of stump diameter, stump height, and cutting season on Quercus variabilis stump sprouting. Scand J For Res. 2013;28(3):223-231.

Chapter 6

Afforestation in Karst Area

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Abstract

In order to study the afforestation technology in rocky desertification area and provide guidance for the cultivation and management of artificial forest in the later stage, an experimental study was carried out on the artificial forest in National long term scientific research base for comprehensive control of rocky desertification in Wuling Mountain, Western Hunan Province. The experiences of afforestation, land preparation and forest management in this area were summarized. The result show that: 1. Through appropriate afforestation land preparation and forest management measures, the forest in rocky desertification area can be successfully restored. 2. Vegetation restoration in rocky desertification area has formed relatively healthy and stable multi tree species and multi-level forest communities. 3. The biological yield of each afforestation tree species was significantly different with different tree species. 4. The diversity index and evenness index of undergrowth plants in different stands were significantly different. 5. Young trees of dominant species dominated the undergrowth vegetation of different stands, and the natural regeneration of each stand has been stabilized. 6. There are some differences in soil chemical properties under different stands. There were significant differences in SOM, TN, NO3-N, NH4-N and AP contents in the soil of the eight stands.

Keywords: karst area, afforestation, site preparation, growth pattern, biodiversity

1. Introduction

Rocky desertification land is one of the difficult forestation areas faced by human beings. 12% of the world's land is facing the problem of rocky desertification. The area of rocky desertification in China is 50 million ha. From Sinian to Triassic, the underlying strata deposited thick carbonate rocks, which laid the material foundation for the formation of rocky desertification in this area. Early studies have shown that the species diversity of vegetation communities will gradually increase with the improvement of environmental conditions and the development of succession stages and the community structure will become better and better (see [1]). The karst area has strong spatial heterogeneity, poor anti-interference ability, low ecosystem function, and very fragile environment. In addition, it is affected by backward productivity and unreasonable human activities. Vegetation is gradually degraded, vegetation coverage is reduced, and the ability of soil to retain water and soil is reduced. It restricts the growth of plants, makes soil erosion present a vicious circle, and slows down the process of ecological civilization construction in karst areas (see [2, 3]).

Xiangxi Autonomous Prefecture is located in the hinterland of Wuling Mountain, with a forest area of 633,200 hectares and a forest coverage rate of 61%. The territory is rich in biological species resources, with many rare species, which can be called a natural treasure house of wild animal and plant resources and a gene bank of biological species. 19 species of world-famous relict plants such as *Cyclops*, *Metasequoia*, *Davidia involucrata*, *Ginkgo biloba*, *Gastrodia*, *camphor*, and *turmeric* are preserved; more than 230 species of oily plants with seed oil content greater than 10%; 216 ornamental plants in 91 families 383 species; There are more than 60 kinds of vitamin plants; 12 kinds of pigment plants. It is the main producing area of *Tung Oil, Camellia oleifera, lacquer* and Chinese medicinal materials, especially in the prefecture, there is the most complete and largest low-altitude evergreen broad-leaved primary secondary forest in the subtropical zone. In the past thousand years, the forests in this area were cut down and the hillsides were used for farming. As a result, the soil erosion in this area was accelerated and the rocky desertification was intensified. Vegetation restoration is the key to ecological reconstruction, and the restoration of plant diversity is an important part of vegetation restoration. The zonality and succession of vegetation should be followed by the selection of suitable economic tree species, and the optimal allocation of forest should have configured shrub and grass. Therefore, it is of great significance to explore the technology of forest vegetation restoration in rocky desertification area, and Vigorously promote the use of rocky desertification management models based on locally suitable native tree species, continue to increase comprehensive conservation efforts, and ultimately create a near-natural growth community environment for vegetation growth, so as to achieve the expected results of rocky desertification vegetation restoration.

The purpose of this study is to restore the near natural forest ecosystem with multi tree species and multi canopy in the rocky desertification area with serious vegetation degradation through silviculture. This experimental study preliminarily achieved the goal, improved the soil production capacity, reduced soil erosion, improved the microclimate of afforestation in rocky desertification area, produced a certain amount of wood, and it has improved the living environment, also increases the income of the people in the area. This effort caused the social production activities into a sustainable virtuous circle.

2. Materials and methods

2.1 Overview of the study area

Under the influence of subtropical monsoon and mountain control, the national long-term scientific research base of Wuling Mountain has obvious Subtropical monsoon climate characteristics. The four seasons are distinct, the precipitation is abundant. The annual average sunshine hours are 1240-1440 h, the annual average temperature is 15.8–16.9°C, the annual active accumulated temperature is 4835–5200°C, the frost free period is 269–292 days, and the annual average rainfall is 1300–1500 mm.

Since 1964, the local forestry department has carried out the artificial afforestation movement on the mountain with serious rocky desertification. After 55 years of hard work, 126 native tree species of 39 families and more than 10 exotic tree species have been successfully used to carry out forest vegetation restoration test on 386.7$hm^2$of serious rocky desertification mountain. Here, from the past chaotic rock slope with overgrown weeds, it has become today's lush and green mountains Linhai has formed a modern forestry construction demonstration base integrating forest management and forestry scientific research in rocky desertification areas.

The national long-term scientific research base for comprehensive management of rocky desertification in Wuling Mountain is selected as the research object. The research base is located in Qingping Town, Yongshun County, Xiangxi Autonomous Prefecture, Hunan Province, 110° 13′40.296 "E, 29 ° 3'21.59"N, belonging to the central area of Wuling Mountain Area. The highest altitude is 820 meters and the lowest altitude is 320 meters (**Figures 1** and **2**).

The parent rock is limestone, which belongs to severe rocky desertification area. There are three main methods of land preparation in this area:

a. The artificial trench is suitable for sites with more than 90% rock exposure.

b. The artificial bund is suitable for slope land with less than 90% rock exposure.

c. Cave shaped site preparation is suitable for the site of stone bud pile.

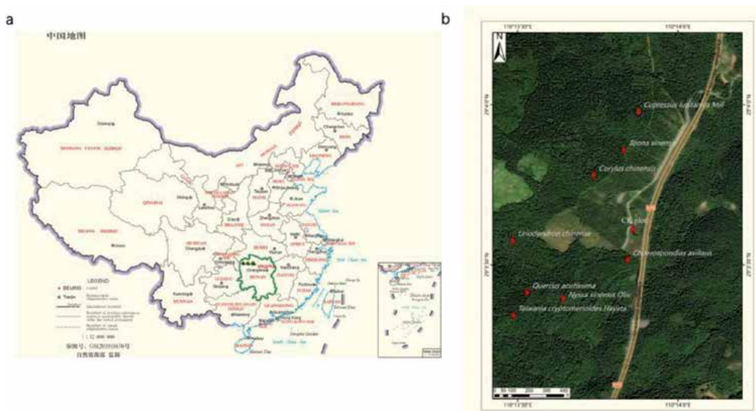

Figure 1.
Research location; a. Research location in China b. The plot distribution map.

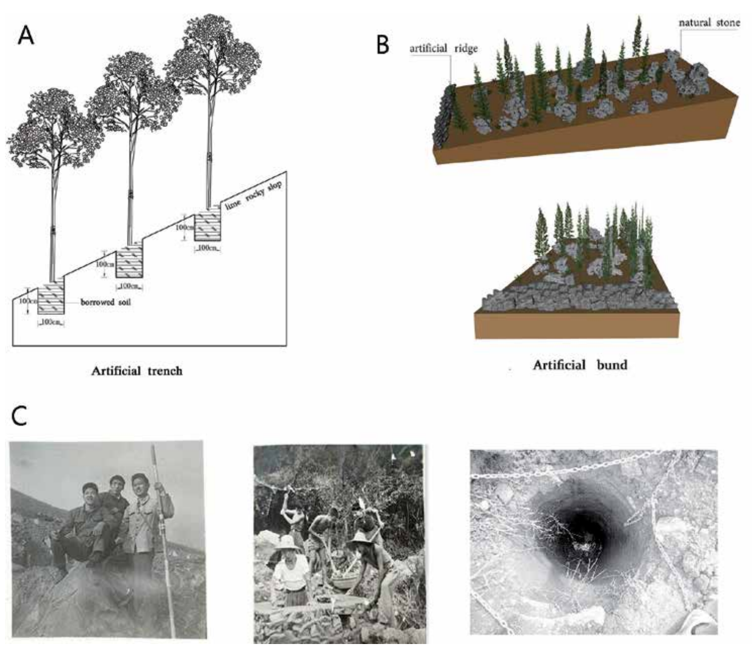

Figure 2.
Land preparation; A. The level artificial trench B. The level artificial bund C. The local forestry workers carried out land cave-shaped soil preparation in 1973.

Plot name	Afforestation patterns	Plot area/m^2	Canopy closure	Stand age/a	Percentage of total forest area/%
Cupressus lusitanica (Mill.)	Pure forest	20 m*30 m	0.68	25	3%
Taiwania cryptomerioides (Hayata.)	Pure forest	20 m*30 m	0.7	40	5%
Quercus acutissima (Carruth.)	Pure forest	20 m*30 m	0.69	20	4%
Corylus chinensis (Franch.)	Pure forest	20 m*30 m	0.8	42	2%
Toona sinensis (Juss.)	Pure forest	20 m*30 m	0.82	40	5%
Nyssa sinensis (Oliver.)	Pure forest	20 m*30 m	0.65	40	3%
Liriodendron chinensis (Sarg.)	Pure forest	20 m*30 m	0.67	37	8%
Choerospondias axillaris (Roxb.)	Pure forest	20 m*30 m	0.77	35	5%

Table 1.
The basic situation of monitoring sample plots.

In terms of tree species selection, afforestation mode and stand tending management, the selection principle of tree species follows the principle of local tree species and suitable tree species, and In order to increase local species resources and land biodiversity, a small number of exotic species are introduced. For example, *Liriodendron chinense* (Hemsl.) Sarg. All the afforestation methods are seedling planting. Young forest tending combined with crop interplanting was used to loosen soil and weed. In the early stage of afforestation, Corn was the main crop in early interplanting, After the stand was closed, crop interplanting was stopped, and the stand density was adjusted by artificial pruning and thinning.

2.2 Experimental design

The fixed standard plot survey method was adopted in January 2019. In the study area, eight representative native precious tree species were selected: *Toona sinensis* (Juss.), *Choerospondias axillaris* (Roxb.), *Corylus chinensis* (Franch.), *Taiwania cryptomerioides* (Hayata), *Cupressus lusitanica* (Mill.), *Nyssa sinensis* (Oliver.) and *Liriodendron chinensis* (Sarg.). One is unplanted shrub and grassland as a control plot, Three20 m * 30 m sample plots were set up for each stand. The DBH, tree height, height of the beginning of the crown, crown diameter and stem straightness were recorded, and the average tree height and DBH were calculated. Calculation of average DBH and average tree height of sample plot: the average DBH of sample plot is the DBH corresponding to the average cross-sectional area, so the cross-sectional area of each tree should be calculated, and then the average cross-sectional area should be calculated to calculate the average DBH. The average tree height is to find out the corresponding tree height with the average DBH on the basis of the DBH tree height curve. The basic conditions of the monitored plots are shown in **Table 1**.

3. Research contents

Eight artificial forests were selected for the study. The main research contents are as follows:

3.1 Growth patterns of plantation

Each tree was investigated in the sample plot, and one standard tree was selected for stem analysis in each standard plot. Through the measurement of DBH, tree height and volume growth process, the measured data were obtained, and the total growth, annual growth and average growth curve of each tree species were drawn to analyze their growth pattern.

3.2 Biodiversity of plantation

Three 2 m * 2 m shrub plots were set up in 8 fixed sample plots of *Toona sinensis* (Juss.), *Choerospondias axillaris* (Roxb.), *Corylus chinensis* (Franch.), *Taiwania cryptomerioides* (Hayata), *Cupressus lusitanica* (Mill.), *Nyssa sinensis* (Oliver.) and *Liriodendron chinensis* (Sarg.). Three 1 m × 1 m small plots were set up to investigate the shrub and grass diversity under the forest.

3.3 Biomass survey of tree layer in plantation

The biomass of standard wood was measured by stratified harvest method. 500 g samples were taken from the upper, middle and lower layers of branches and stem. The underground part was excavated in three layers of 0–20 cm, 20–40 cm and 40–60 cm within the radius range of 1 m of sample tree, and were divided into coarse roots (d > 5) three levels of roots (5 cm > d > 1 cm), medium root (5 cm > d > 1 cm) and fine root (d < 1 cm) were placed by classification, and 500 g samples of their fresh weights were weighed. The fresh weights of leaves, stem, bark and roots were measured, and then dried in 85°C oven to constant weight. The water content of each part and the biomass of standard tree were calculated, and the biomass of the whole tree layer was calculated. Calculate the dry mass of each component, calculate the dry mass of the sample wood, and then convert the dry mass per unit area and stand biomass.

3.4 Regeneration patterns of plantation

The ground diameter, DBH, tree height, crown diameter and stem straightness of all young trees in the plot were recorded, and the average tree height and DBH were calculated.

3.5 Soil sampling and analysis

The soil physical properties were mainly measured for *Taiwania cryptomerioides* (Hayata.), *Liriodendron chinensis* (Sarg.), and *Taiwania cryptomerioides-Liriodendron chinensis* mixed forest, and soil nutrients were measured for eight forests. Three 20 m* 20 m sample plots were selected as the sample plots in the fixed sample plots, and the soil samples were randomly selected from three points in each sample plot. The visible animal and plant residues and small stones were carefully removed, and then were mixed evenly through a 2 mm sieve. The samples were taken back to the laboratory for analysis. The rocky desertification unforested shrub grassland was taken as the research sample plot, and each sample was determined three times.

Soil samples were dried by natural air to remove impurities. 5-10 g samples were screened by 2 mm soil sieve to determine the contents of C, N and P in soil. Soil C was determined by potassium dichromate external heating sulfuric acid oxidation method (LY / T 1237–1999), while soil N and P were determined by semi micro Kjeldahl method (LY / T 1228–1999) and molybdenum antimony resistance Colorimetry (LY/ T 1232–1999) (see [4]).

3.6 Statistical analyses

3.6.1 Species diversity calculation method

1. The calculation formula of species importance value is as follows:

Important value = (relative density + relative dominance + relative frequency)/3 × 100% (see [5]).

2. Species diversity calculation method

$$\text{Berger} - \text{Parker index}: \text{d} = \text{Nmax} / \text{N}$$

$$\text{Simpson index}: \text{D} = 1 - \Sigma\left\{ni\left(ni - 1\right) / \left[\text{N}\left(\text{N} - 1\right)\right]\right\}$$

$$\text{Shannon} - \text{wiener index}: \text{H} = -\sum_{i=1}^{s} pi \ln pi$$

$$\text{Pielou index}: \text{P} = \frac{H}{\ln S}$$

Note: In the formula, N_{max} is the number of individuals of the most dominant species; N is the total number of individuals; ni is the number of individuals of the i-th species. Pi is the ratio of the number of individuals of the i-th species to the number of individuals of all species in the community; S is the total number of species in the community (see [6]).

3.6.2 Calculation method of stand average DBH

1. Quadratic mean diameter at breast height

The quadratic mean diameter at breast height of the stand is calculated based on the section area of the stand height at breast height, as follows: $D_g = \sqrt{\frac{1}{n}\sum_{i=1}^{n} d_i^2}$

Dg——Stand quadratic mean diameter at breast height
di——Diameter at breast height of the i-th tree
n——Total number of trees in the plot

2. Average stand height

The average height of forest stands adopts the weighted average height of section area, and the calculation formula is: $\bar{H} = \frac{\sum_{i=1}^{k} \bar{h}_i G_i}{\sum_{i=1}^{k} G_i}$

$\bar{H}$ —— Average stand height
$\bar{h}_i$ —— The arithmetic average height of the i-th diameter tree in the forest stand
$\overline{G_i}$ —— The cross-sectional area of the breast height of the i-th diameter forest tree in the stand
k——Number of stand diameter steps

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3. Volume per plant

The volume per plant is calculated using the average stem profile. The specific calculation formula is as follows: $V_{average} = g_{1.3}(h+3)\times f_c \times 667$

$V_{average}$——Average forest accumulation per m^2
fc——Average form factor
$g_{1.3}$——Average wood breast height section area
h——Average tree height

3.6.3 Data analysis

Use Excel to calculate the standard tree's height (H(t)), diameter at breast height(D(t)), and volume per plant(V(t)), volume average growth (V_θ(t)), volume annual growth (V_Z(t)), etc. Statistical analysis was performed with SPSS25.0 (see [7]), single-factor analysis of variance was used to test the significant differences in soil physical and chemical properties of different forest stands, and Pearson correlation was used to study the correlation between plant community diversity and soil nutrients; origin8.0 was used for mapping.

4. Results and analysis

4.1 Growth patterns of different plantations

4.1.1 The growth pattern of plantation tree height

According to the survey data of fixed sample plots, the age variation curve of tree height was drawn. The height and growth of each tree species increase with age (**Figure 3**), but the rapid growth period of each tree species is different. The specific performance is as follows: *Taiwania cryptomerioides* 1 ~ 15 years is the fast-growth period, and the growth rate gradually slows down after 15 years, and the tree height growth reaches 19.7 m at 40 years; *Quercus acutissima* 1 ~ 2 years is the fast-growth period, and the growth rate gradually slows down after 2 years. The height growth

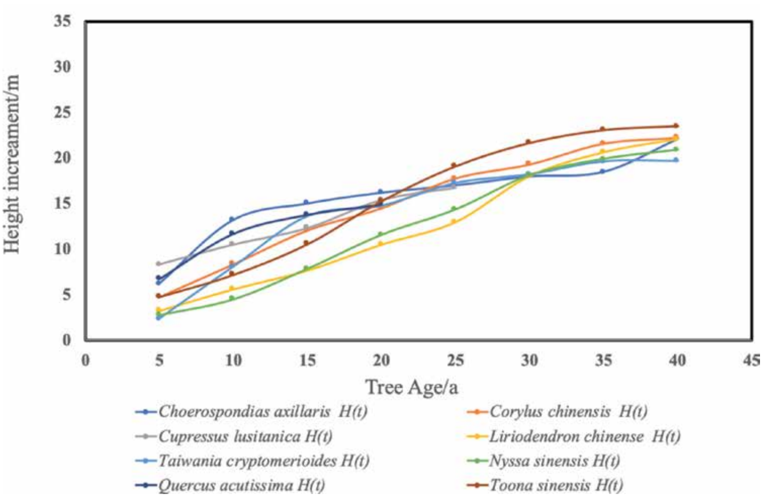

Figure 3.
Height growth curve.

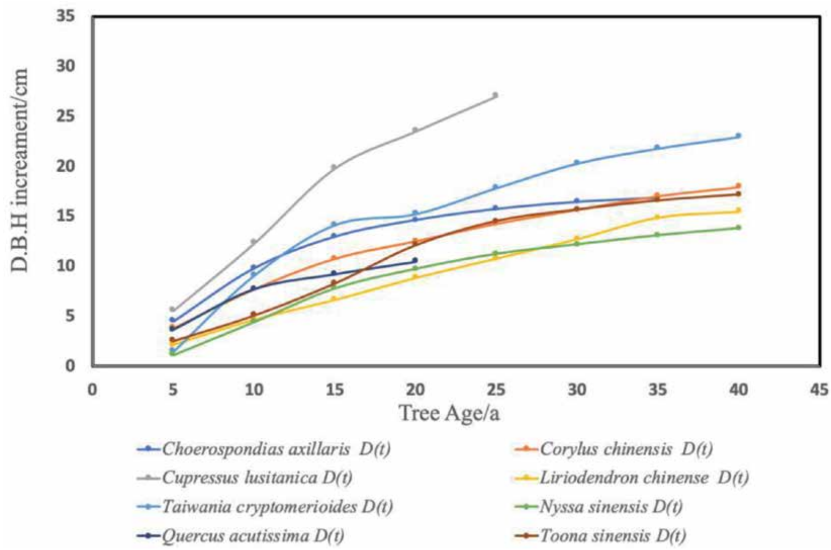

Figure 4.
DBH growth curve.

reaches 14.9 m; *Cupressus lusitanica* 1 ~ 4a is the fast growth period, the growth rate gradually slows down after 15 years, and the tree height growth reaches 16.8 m at 24 years (**Figure 4**). *Corylus chinensis* 1 ~ 27 years is the fast-growing period. After 27 years, the growth rate gradually slows down. After 37 years, the height of the tree grows extremely slowly, and the height of the tree reaches 22.3 m at 42 years. For *Choerospondias axillaris*,1-10a is the fast-growing period, and the growth rate gradually slows down after 10a, and the tree height grows to 18.5 m at 35 years. *Toona sinensis* 1 ~ 35 years is the fast-growing period. After 35 years, the growth rate gradually slows down, and the tree height grows to 23.5 m at 40 years. *Nyssa sinensis* 1–30 years is the fast-growing period, the growth rate gradually slows down after 30a, and the tree height growth reaches 20.9 m at 40 years. *Liriodendron chinensis* 1 to 27 years is a fast-growing period, after 27 years, the growth rate gradually slows down, and the tree height grows to 22.1 m at 37 years.

4.1.2 Growth pattern of diameter at breast height of plantation

The growth of diameter at breast height of each tree species increases with age. Specifically, it shows that: the first 1–5 years after planting of *Taiwania cryptomerioides* grows slowly, the growth enters the fast growth period after 5 years, the growth of diameter at breast height begins to slow down after 15 years, and the growth of diameter at breast height reaches 23 cm at the 40th year; the rapid growth period after the plantation of *Cupressus lusitanica*, The growth slowed down after 14th year, and the breast diameter growth reached 27 cm at 24th year. After afforestation, *Corylus chinensis* entered the fast-growing period, and the growth slowed down after 17 years, and the diameter at breast height reached 17.9 cm at 42 years; *Choerospondias axillaris* quickly entered the fast-growing period after afforestation, the growth slowed down after 15 years, and the diameter at breast height reached 16.8 cm at 35 years. *Toona sinensis* grows slowly in the first 5 years after afforestation. After a slow growth period, it enters the fast-growth period after 5 years and slows down after 25 years. The diameter at breast height reaches 17.2 cm at 40th year. *Nyssa sinensis* grow slowly in the first 5 years after afforestation. After a slow growth period, they enter the fast-growth period after 5 years, and the growth slows down after 25 years. The diameter at breast height reaches 13.8 cm at 40th year. After afforestation, the first 2 years of *Liriodendron chinensis* grows slowly. After the slow growth stage, it enters the fast growth stage after 2 years. The growth slows down after 32 years. The diameter at breast height reaches 15.5 cm at 37 years.

4.1.3 Volume growth pattern of plantation forest

It can be seen from **Figure 5** that the volume growth of each tree species increases with age. Specifically, it shows that *Nyssa sinensis* go through the first 10 years of slow growth period after afforestation, and then enter the fast growth period after 10 years, and the volume growth reaches 0.1476m^3 at 40th year. Its volume annual growth $V_Z(t)$ reached its maximum value of 0.0037m^3 at $V_Q(t)$ at 40th year. In the first 2 years of *Liriodendron chinensis*, the forest was in a slow growth period, and after 2 years, it entered the rapid growth period. At 37th year, the volume growth reached 0.2004 m^3. The continuous annual growth of the volume $V_Z(t)$ reached the maximum value of 0.01344 m^3 at 32th year, and the average growth volume reached the maximum value of 0.00542 m^3 at $V_Q(t)$ at 37th year. *Quercus acutissima* was in a slow growth period 5 years ago, and entered a fast growth period 5 years later, and the volume growth reached 0.0814m^3 at 17th year. The volume of continuous annual growth $V_Z(t)$ reached the maximum value of 0.0083 m^3 at 17th year, and the average growth volume reached the maximum value of 0.0048 m^3 at $V_Q(t)$ at 17th year. *Cupressus lusitanica* was in the slow growth period 4 years ago, and entered the fast growth period after 4 years. The volume growth reached 0.3749 m^3 at 24th year. The continuous annual growth volume $V_Z(t)$ of *Cupressus lusitanica* reached the maximum value of 0.0264 m^3 at 24th year, and the average growth volume reached the maximum value of 0.0156 m^3 at $V_Q(t)$ at 24th year; none of the four tree species has reached quantitative maturity and theoretically the optimal cutting age.

After afforestation, the growth of *Toona sinensis* growth slowly in the first 10 years, then growth accelerated to 40 years of volume growth reached 0.2540 m^3. The annual volume growth of *Toona sinensis* reached the maximum value of 0.0120 m^3 at 25th year, and the average growth $V_Q(t)$ reached the maximum value of 0.0061m^3 at 40th year. V_Z (t) and $V_Q(t)$ intersected at the 39th year, when *Toona sinensis* reached the best cutting age. After afforestation, the growth rate of *Choerospondias axillaris* was slow in the first five years, and entered the fast-growing stage after five years. The volume growth reached 0.1868 m^3 in 35 years. The annual volume growth of *Choerospondias axillaris* reached the maximum

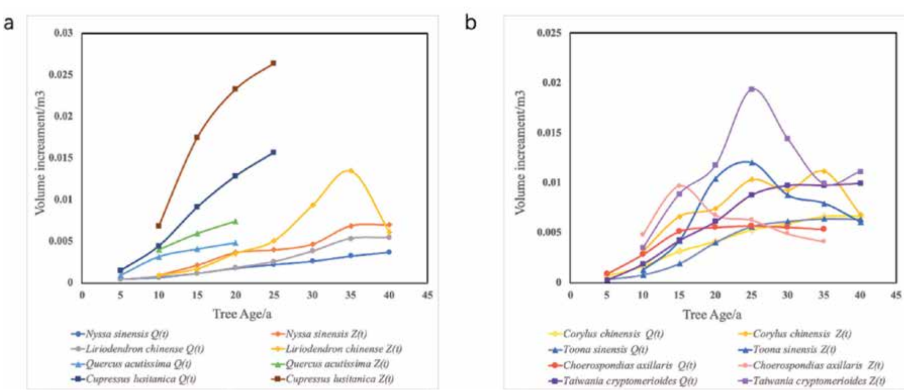

Figure 5.
Volume growth curve; a. The continuous annual growth volume $V_Z(t)$ of plantation forest; b. The average growth volume $V_Q(t)$ of plantation forest.

value of 0.0097 m^3 at 20 years, and the average growth reached the maximum value of 0.0057 m^3 when $V_Q(t)$ was 25 years. $V_Z(t)$ and $V_Q(t)$ intersect at 27th year, which is the best cutting age of *Choerospondias axillaris*. After afforestation, the growth of *Corylus chinensis* was slow in the first seven years, and entered the fast-growing stage after seven years. The volume growth reached 0.2783m^3 at 42th year. The annual volume growth of *Corylus chinensis* reached the maximum value of 0.0112 m^3 at 37 years, and the average growth reached the maximum value of 0.0068 m^3 when $V_Q(t)$ was 42 years. $V_Z(t)$ and $V_Q(t)$ intersect, and the best cutting age at 42th year. After afforestation, *Taiwania cryptomenoides* experienced slow growth period in the first 10 years, and entered the fast-growing stage after 10 years, and the volume growth reached 0.3959 m^3 at 40 years. V_Z (t) reaches the maximum value of 0.0194 m^3 at 25th year and 0.0099m^3 at $V_Q(t)$ 40th year. $V_Z(t)$ and $V_Q(t)$ do not intersect before 40th year. Therefore, the best cutting age of *Taiwania cryptomenoides* is at least 40th year.

4.2 Biomass per tree and its distribution

The biomass of individual tree was significantly different with different tree species (**Table 2**). The order of biomass per plant of eight tree species was as follows: *Cupressus lusitanica* (382.483 kg/plant) > *Taiwania cryptomenoides* (239.907 kg/plant) > *Corylus chinensis* (205.245 kg /plant) > *Toona sinensis* (167.054 kg/ plant) > *Quercus acutissima* (149.734 kg/ plant) > *Choerospondias axillaris* (126.345 kg/ plant) > *Nyssa sinensis* (124.824 kg/ plant) > *Liriodendron chinensis* (117.456 kg/ plant). The results showed that the biomass of each component of tree species was as follows: stem > branches> roots of *Cupressus lusitanica* and *Taiwania cryptomenoides*; the biomass of *Corylus chinensis*, *Toona sinensis*, *Quercus acutissima*, *Choerospondias axillaris*, *Nyssa sinensis* and *Liriodendron chinensis* shows:Stem > roots> branches.

Tree species	Biomass per plant /(kg/plant)			
	Stem	Branches	Tree root	Total
Cupressus lusitanica	247.09	76.34	59.06	382.48
%	64.60	20.00	15.40	100.00
Taiwania cryptomenoides	154.68	49.47	35.75	239.91
%	64.50	20.60	14.90	100.00
Corylus chinensis	166.16	4.72	34.37	205.25
%	81.00	2.30	16.70	100.00
Toona sinensis	140.95	5.73	20.38	167.05
%	84.40	3.40	12.20	100.00
Quercus acutissima	111.9	17.15	20.69	149.73
%	74.70	11.50	13.80	100.00
Choerospondias axillaris	97.88	6.74	21.72	126.35
%	77.50	5.30	17.20	100.00
Nyssa sinensis	97.72	9.35	17.76	124.82
%	78.30	7.50	14.20	100.00
Liriodendron chinensis	93.02	4.38	20.06	117.46
%	79.20	3.70	17.10	100.00

Table 2.
Biomass comparison of different tree species.

4.3 Stand biomass and its distribution pattern

The biomass of individual tree is converted into stand biomass as shown in **Table 3**. The biomass of each stand is as follows: *Cupressus lusitanica* (319.171 t·ha^{-1}) > *Quercus acutissima* (281.197 t·ha^{-1}) > *Corylus chinensis* (210.264 t·ha^{-1}) > *Taiwania cryptomenoides* (186.601 t·ha^{-1}) > *Toona sinensis* (18 5.386 t·ha^{-1}) > *Choerospondias axillaris* (181.875 t·ha^{-1}) > *Liriodendron chinensis* (161.548 t·ha^{-1}) > Nyssa sinensis (158.32 t·ha^{-1}).The biomass of tree layer, understory vegetation layer and litter layer of different stands were compared and analyzed under the condition of similar forest age.

1. Tree layer: *Cupressus lusitanica* > *Quercus acutissima* > *Corylus chinensis* > *Toona sinensis* > *Taiwania cryptomenoides* > *Nyssa sinensis* > *Liriodendron chinensis* > *Choerospondias axillaris.*

2. Undergrowth vegetation layer: *Taiwania cryptomenoides* > Choerospondias axillaris > *Cupressus lusitanica* > *Corylus chinensis* > *Liriodendron chinensis* > *Toona sinensis* > *Quercus acutissima* > *Nyssa sinensis.*

3. Litter layer: *Corylus chinensis* > Nyssa sinensis > *Cupressus lusitanica* > *Taiwania cryptomenoides* > *Liriodendron chinensis* > *Toona sinensis* > *Quercus acutissima* > *Choerospondias axillaris*.

4. Total biomass:*Cupressus lusitanica* > *Quercus acutissima* > *Coryl*us chinensis > *Taiwania cryptomenoides* > *Toona sinensis* > *Nyssa sinensis* > *Liriodendron chinensis* > *Choerospondias axillaris.*

5. Tree layer > litter layer > understory vegetation layer

The results showed that: the total biomass of *Cupressus lusitanica* forest was the largest, the biomass of understory vegetation layer and litter layer was also higher than that of other forests, and the growth trend was better than that of other tree species. It can be seen that there are some problems in the regeneration of evergreen

Tree species	Age of forest /a	Stand biomass /t·ha^{-1}					
		Stem	Branch	Tree root	Undergrowth vegetation	Litter	Total
Cupressus lusitanica	25	201.79	62.34	48.23	2.65	4.16	319.17
Taiwania cryptomenoides	40	116.01	37.10	26.82	2.98	3.69	186.60
Quercus acutissima	20	207.02	31.72	38.27	1.71	2.48	281.20
Corylus chinensis	42	162.01	4.60	33.51	2.06	8.09	210.26
Toona sinensis	40	152.70	6.20	22.08	1.81	2.60	185.39
Nyssa sinensis	40	138.43	13.24	25.16	0.76	4.28	181.88
Liriodendron chinensis	37	124.02	5.84	26.75	1.94	3.00	161.55
Choerospondias axillaris	35	119.09	8.20	26.43	2.79	1.81	158.32

Table 3.
Stand biomass of different tree species.

broad-leaved trees, which need to be paid attention to. It is not our ultimate goal to build artificial pure forest. We need to use artificial afforestation technology to restore its ecological function, carry out natural regeneration, and finally form a complex and stable ecological community structure.

4.4 Differences of undergrowth plant diversity in different stands

Habitat heterogeneity and plant biological characteristics are the main factors affecting the diversity of understory plants (see [8]). The diversity index and evenness index of different plantations were significantly different (**Figure 6**), indicating that there were differences in the diversity level of understory plants in different plantations. The Berger Parker index of shrub layer in different stands was the largest in *Liriodendron chinensis*. The Simpson index of is the largest of *Corylus chinensis* forest, and that of *Cupressus lusitanica* is the smallest among 8 stands. For Shannon Wiener index, *Cupressus lusitanica* is the largest, *Liriodendron chinensis* is the smallest. The Berger Parker index of herbaceous layer in different stands was the largest in *Toona sinensis* forest and the smallest in *Taiwania cryptomenoides* forest; Simpson index of eight stands was the largest in *Taiwania cryptomenoides* forest and the smallest in *Quercus acutissima* forest. For Shannon Wiener index*, Nyssa sinensis* forest is the largest, *Taiwania cryptomenoides* forest is the smallest. The analysis of variance of undergrowth shrub and herb diversity in each stand shows that the diversity of undergrowth plants is significant, but there is no significant difference in Pielou index among the eight stands, indicating that the evenness of plants in the eight stands is basically similar. Secondly, the Shannon Wiener index of undergrowth shrub in *Cupressus lusitanica* was higher than that in other forest stands, indicating that the plant diversity under *Cupressus lusitanica* was richer than that in other stands, but the diversity index of shrub grassland without afforestation was higher than that of other stands ($P < 0.05$).

4.5 The regeneration difference of young trees in different stands

The horizontal spatial distribution of seedlings and young trees is often reflected by the spatial distribution pattern, which will change with the biological characteristics of plants and the comprehensive influence of environmental conditions (see [9]). There are many factors that affect the spatial distribution of seedlings and saplings, and the main factors are seed dispersal and different habitats (see [10]). It can be seen from **Table 4** that saplings of dominant species dominate the undergrowth vegetation of different stands. The regeneration of saplings under *Taiwania cryptomenoides* and *Toona sinensis* stands has gradually appeared other tree

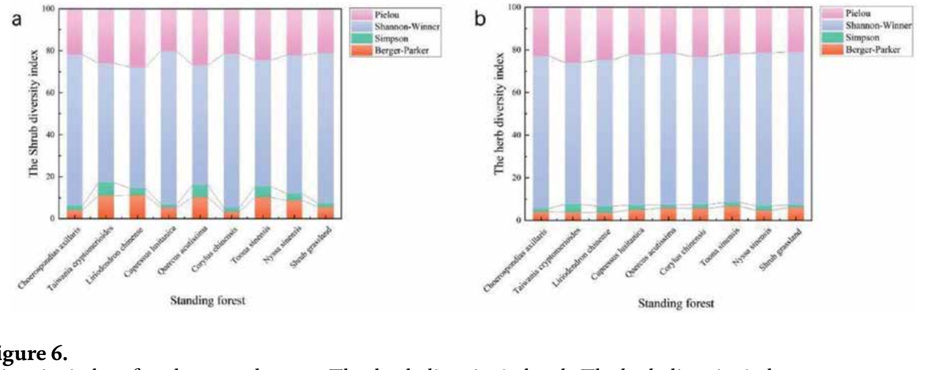

Figure 6.
Diversity index of understory plants; a. The shrub diversity index; b. The herb diversity index.

Sample plot	Seedling species	Tree height /m	DBH/cm	Average crown diameter /m
Choerospondias axillaris	*Choerospondias axillaris*	3.47 ± 1.21	3.15 ± 0.88	3.47 ± 2.81
	Cinnamomum camphora	4.32 ± 1.17	3.32 ± 0.75	4.35 ± 2.02
	Lindera communis	2.87 ± 0.06	1.93 ± 0.55	2.74 ± 0.61
Corylus chinensis	*Corylus chinensis*	1.73 ± 0.11	2.90 ± 0.28	36.5 ± 2.12
	Cinnamomum camphora	1.40 ± 0.14	2.11 ± 0.43	28.6 ± 20.67
Cupressus lusitanica	*Zanthoxylum bungeanum*	1.17 ± 0.32	33.2 ± 55.95	0.14 ± 0.19
	Camellia japonica	1.10 ± 0.83	18.22 ± 52.38	0.41 ± 0.65
	Cupressus lusitanica	2.40 ± 0.28	3.65 ± 0.21	0.09 ± 0.11
	Eriobotrya japonica	1.25 ± 0.35	0.65 ± 0.21	0.04 ± 0.00
	Rhus chinensis	1.55 ± 0.66	2.18 ± 2.07	0.09 ± 0.08
	Cinnamomum camphora	1.02 ± 0.6	1.43 ± 0.15	0.11 ± 0.08
	Liriodendron chinensis	1.27 ± 0.43	3.16 ± 1.81	0.03 ± 0.01
	Vernicia fordii	3.25 ± 0.21	9.95 ± 7.14	0.16 ± 0.00
Liriodendron chinensis	*Liriodendron chinensis*	6.21 ± 2.82	3.22 ± 1.10	0.97 ± 0.36
	Cinnamomum camphora	5.22 ± 2.36	3.02 ± 0.90	0.92 ± 0.32
Taiwania cryptomenoides	*Acer davidii*	6.68 ± 1.72	3.05 ± 0.56	2.77 ± 2.67
	Taiwania cryptomenoides	3.77 ± 1.10	3.19 ± 0.75	2.51 ± 0.89
	Vernicia fordii	4.81 ± 0.57	3.15 ± 0.49	3.24 ± 1.19
Nyssa sinensis	*Nyssa sinensis*	1.95 ± 0.82	2.07 ± 1.11	0.46 ± 0.28
	Phoebe bournei	1.48 ± 0.82	1.61 ± 0.82	0.37 ± 0.27
	Camellia japonica	1.15 ± 0.48	1.71 ± 0.66	0.56 ± 0.25
Quercus acutissima	*Quercus acutissima*	7.37 ± 3.32	3.63 ± 1.12	3.41 ± 1.84
	Taiwania cryptomenoides	2.51 ± 1.13	2.41 ± 1.43	0.75 ± 0.74
Toona sinensis	*Acer davidii*	4.25 ± 1.07	2.82 ± 0.52	2.88 ± 1.02
	Cinnamomum camphora	4.33 ± 1.73	3.35 ± 1.17	3.65 ± 3.61

Note: Mean ± standard error.

Table 4.
Relationship between tree growth and natural regeneration of young forest under the forest.

species, and the natural regeneration of each stand has become stable. The growth of undergrowth plants is closely related to the growth of trees. The composition and structural characteristics of understory plants are closely related to the internal environmental conditions of plantations. On the one hand, because the growth and management of plantation affect the soil, water, light intensity, temperature and humidity and other micro environmental conditions, the growth and development of understory plants is limited, which directly affects the species, coverage, biomass and diversity of understory plants. On the other hand, after nearly 40 years of development, the plantations in the study area have formed a relatively stable understory environment. The pattern of understory plants is mainly formed by natural competition, which fully reflects the advantages and disadvantages of internal environmental conditions of different artificial forests and the intermediate relationship of understory niche. Therefore, most of the plantations are shade tolerant plants with strong adaptability.

4.6 The difference of soil physical and chemical properties among different stands

Soil density and total porosity are not only the basic physical characteristics of forest soil, but also important indicators of soil and water conservation, which affect the growth and development of understory plants (see [11]). The soil physical properties of typical *Taiwania cryptomenoides* (coniferous forest), *Liriodendron chinensis* (broad-leaved forest) and *Taiwania cryptomenoides-Liriodendron chinensis* mixed forest (coniferous and broad-leaved forest) were determined (**Table 5**). The average soil density in 0 ~ 30 cm depth soil layer was as follows: *Taiwania cryptomenoides* > *Liriodendron chinensis* > mixed forest > shrub grassland (CK)Plot.

In the depth of 0 ~ 30 cm, the average soil porosity was mixed forest (*Taiwania cryptomenoides- Liriodendron chinensis*) > shrub grassland > *Taiwania cryptomenoides* pure forest >*Liriodendron chinensis* pure forest. The soil density in 0–15 cm and 15–30 cm soil layers of the shrub grassland was significantly lower than that of the pure *Taiwania cryptomenoides* forest ($P \leq 0.05$), but there was no significant difference in soil density between the mixed forest of *Taiwania cryptomenoides* and pure forest. The soil density of each stand decreased with the increase of soil depth. The soil total porosity of different stands increased with the increase of soil depth, and with the increase of soil layer, the soil porosity of different stands showed significant difference. On the whole, the soil water holding capacity of the mixed forest of *Taiwania cryptomenoides* and *Liriodendron chinensis* was higher than that of pure *Taiwania cryptomenoides* and pure *Liriodendron chinensis*. Compared with pure forest, the maximum water holding capacity of *Taiwania cryptomenoides- Liriodendron chinensis* mixed forest was significantly increased, and the field water holding capacity of 0 ~ 15 cm soil layer in different stands did not reach significant difference. Except for *Taiwania cryptomenoides*, the maximum water holding capacity and field water holding capacity of other stands increased with the increase of soil depth. Among them, the maximum water holding capacity and field water holding capacity of 15 ~ 30 cm soil layer of *Taiwania cryptomenoides -Liriodendron chinensis* mixed forest and *Liriodendron chinensis* mixed forest were significantly higher than those of *Taiwania cryptomenoides* pure forest and Liriodendron chinensis pure forest ($P \leq 0.05$), but there was no significant difference between 0 ~ 15 cm soil layer.

Soil is the matrix of plant growth, and its physical and chemical characteristics determine the distribution of plant community types. At the same time, the plant community reacts on the soil to improve its habitat conditions and make the community develop. Through the analysis of soil chemical properties under different stands, the results show that there are some differences in soil properties under different stands (**Table 6**). Among the eight stands, the contents of TP, SOM and TN in the soil of *Choerospondias axillaris* forest were the highest, the contents of NH_4-N and NO_3-N in *Quercus acutissima* forest were higher than those in other stands, the AP content of *Taiwania cryptomenoides* was the highest, and the SOM content of shrub grassland was significantly lower than that of plantation. There were significant differences in SOM, TN, NO_3-N, NH_4-N and AP contents in the soil of the eight stands.

Thickness of soil layer (cm)	Stand type	Moisture content (100%)	Bulk density (g/cm^3)	Maximum water holding capacity (100%)	Minimum water holding capacity (100%)	Total porosity (100%)
0-15 cm	*Liriodendron chinensis*	0.15 ± 0.05b	1.39 ± 0.02ab	0.24 ± 0.02a	0.22 ± 0.03a	0.33 ± 0.03b
	Taiwania cryptomenoides- Liriodendron chinensis	0.32 ± 0.03a	1.29 ± 0.03ab	0.36 ± 0.02a	0.33 ± 0.03a	0.46 ± 0.02a
	Taiwania cryptomenoides	0.18 ± 0.01b	1.47 ± 0.01a	0.24 ± 0.01a	0.22 ± 0.01a	0.36 ± 0.01b
	shrub grassland	0.23 ± 0.03ab	1.21 ± 0.13b	0.35 ± 0.07a	0.28 ± 0.04a	0.41 ± 0.04a
15-30 cm	*Liriodendron chinensis*	0.16 ± 0.01b	1.38 ± 0.04a	0.28 ± 0.01b	0.23 ± 0.01b	0.34 ± 0.01d
	Taiwania cryptomenoides- Liriodendron chinensis	0.33 ± 0.01a	1.22 ± 0.02ab	0.40 ± 0.02a	0.34 ± 0.02a	0.49 ± 0.01a
	Taiwania cryptomenoides	0.17 ± 0.01b	1.39 ± 0.06a	0.24 ± 0.02b	0.20 ± 0.01b	0.38 ± 0.01c
	shrub grassland	0.29 ± 0.05a	1.13 ± 0.11b	0.40 ± 0.06a	0.33 ± 0.05a	0.43 ± 0.02b

Note: Mean ± standard error; the same letter means no significant difference; no same letter means significant difference.

Table 5.
Soil physical properties of different stands.

Tree species	TP g/kg	SOM g/kg	TN g/kg	NH4-N Mg/kg	NO3-N Mg/kg	AP Mg/kg
Choerospondias axillaris	0.40 ± 0.01a	74.13 ± 0.46a	3.31 ± 0.08a	29.87 ± 0.37c	15.46 ± 0.33 g	1.79 ± 0.05d
Corylus chinensis	0.32 ± 0.01b	45.16 ± 0.20e	2.29 ± 0.03d	26.83 ± 0.03e	23.66 ± 0.06d	1.31 ± 0.02f
Cupressus lusitanica	0.35 ± 0.01b	41.58 ± 0.01f	2.09 ± 0.01e	20.87 ± 0.04 h	19.73 ± 0.12e	0.50 ± 0.01 h
Liriodendron chinensis	0.35 ± 0.01b	57.96 ± 0.08c	2.83 ± 0.03c	26.00 ± 0.11f	17.76 ± 0.06f	1.90 ± 0.03c
Taiwania cryptomenoides	0.34 ± 0.00b	65.83 ± 0.19b	3.09 ± 0.01b	20.85 ± 0.01 h	12.44 ± 0.06 h	2.26 ± 0.02a
Nyssa sinensis	0.27 ± 0.02c	48.56 ± 0.04d	2.28 ± 0.01d	31.00 ± 0.06b	25.27 ± 0.07c	2.08 ± 0.03b
Quercus acutissima	0.35 ± 0.02b	41.65 ± 0.20f	2.08 ± 0.02e	35.83 ± 0.14a	33.85 ± 0.03a	0.67 ± 0.04 g
Toona sinensis	0.27 ± 0.02c	33.70 ± 0.10 g	1.90 ± 0.03f	27.69 ± 0.06d	32.41 ± 0.27b	1.49 ± 0.01e
shrub grassland	0.40 ± 0.01a	25.92 ± 0.03 h	1.61 ± 0.04 g	21.68 ± 0.33 g	32.86 ± 0.19b	0.43 ± 0.02 h

Note: Mean ± standard error; the same letter means no significant difference; no same letter means significant difference; TP: soil total phosphorus; SOM: soil organic matter; TN: Soil total nitrogen; NH_4-N: Soil ammonium nitrogen; NO_3-N: Soil nitrate nitrogen; AP: Soil available phosphorus.

Table 6.
Soil nutrient difference analysis of different afforestation tree species.

5. Discussion

5.1 Growth patterns of different plantations

According to the analysis of the growth patterns of the eight tree species in the Xiangxi Rocky Desertification Area from three aspects, (1) the total growth of DBH of 8 tree species increased with age. In contrast, the growth of DBH of 8 tree species in this area is slightly less than that in other areas, which may be due to the single community structure, barren soil, uneven thickness of soil layer, and lack of nitrogen, phosphorus, potassium and other elements to promote plant growth, root growth is hindered, resulting in a smaller DBH growth and lower productivity. (2) With the growth and development of trees, the canopy density gradually increased, the competition among individuals was obvious, the growth space was insufficient, and the growth rate of successive years was significantly slowed down, which led to the differences in the growth of various tree species. (3) The total volume growth of 8 tree species increased with the growth of age, but the time when each stand reached the main cutting age was different. Therefore, it can be seen that in the rapid growth period of 8 kinds of stands, water and fertilizer management and appropriate thinning should be strengthened to control the stand density (see [12]). The rapid growth period should be fully utilized to effectively promote the rapid growth of tree height and DBH, so as to improve the productivity.

5.2 Stand biomass and its distribution pattern

According to the stand productivity of the eight tree species, it can be seen that broad-leaved branches and leaves are more developed than coniferous trees (see [13]). For example, due to its own biological characteristics, flexible material and low shrinkage rate, the stand productivity of *Choerospondias axillaris* is larger than that of *Taiwania cryptomerioides*, although its age is smaller than that of *Taiwania cryptomerioides*. Therefore, in the process of vegetation restoration in karst areas, priority should be given to broadleaved trees such as *Choerospondias axillaris* and *Toona sinensis* or mixed afforestation with coniferous and broad-leaved trees. Studies have shown that the average individual biomass of broad-leaved tree species decreases with increasing altitude, while the biomass of coniferous tree species gradually increases (see [14]). According to the niche in the area, the trees, shrubs and grasses should be arranged reasonably to make full use of the favorable conditions of the microclimate environment in the forest land to promote the biomass accumulation (see [15]). At the same time, proper assessment and management of these forest stands is essential to ensure the health of the forest ecosystem (see [16]).

5.3 Differences of undergrowth plant diversity in different stands

Species richness can be used to measure the quantitative characteristics of species in the community, and the overall diversity index of plant species under different tree species is not high. The Shannon Wiener index of unforested shrub grassland is higher than that of woodland, which is due to the fact that most vegetation biodiversity is caused by herbaceous plants. There is no tall tree layer in the shrub grassland, and its light environment conditions are better than those under the forested forest, which is conducive to the growth and development of shrubs and herbs; the dominant species of shrub layer in the shrub grassland are *Rhus chinensis*, *Rubus tephrodes* and other light loving plants, while the herb layer is mainly perennial herbs such as *Erigeron acris*, etc., which are light loving, semi shade, wet and drought tolerant; plant diversity index and organic matter with the physiological characteristics of bacteria at the soil level of the soil layer (o layer) are related (see [17]). The adaptability of each species to different environments is different, and the differences of light environment on the landing surface of different groups lead to significant differences in dominant species of shrubs and herbs (see [18]). This shows that the environment is heterogeneous and complex, Highly diverse plant combinations can better stimulate plant–soil feedback, and increasing plant diversity is an important strategy to improve the stability of fragile ecosystems. At the same time, related research shows that certain plants (such as *Miscanthus sinensis* Anderssons and *Leguminosae sp.*) should be used first to establish a diversified plant community for rapid vegetation restoration (see [19]).

5.4 The regeneration difference of young trees in different stands

Natural regeneration of multiple tree species occurred under all native tree species, *Taiwania cryptomerioides* and *Toona sinensis* is more prominent, and the natural regeneration of each stand has become stable. There are few species of undergrowth plants in a few rare tree species stands, but they grow well, which is closely related to the growth of trees. Before afforestation, the area was generally wasteland, and there were almost no shrub and herb species. After 40 years of afforestation, the species and quantity of shrubs and herbs increased significantly, and the natural succession of forest ecosystem was an endothermic and spontaneous reaction. Finding key species is essential for understanding the role of species diversity in ecosystem functions (see [20]), and the balance between intraspecific and interspecific competition plays a major role in the functional relationship of biodiversity ecosystems. Although light is a key environmental factor that affects understory communities and diversity, other environmental variables (such as soil nutrients and soil moisture) are also important (see [21, 22]).

5.5 The difference of soil physical and chemical properties among different stands

Soil organic matter, nitrogen and phosphorus are the main nutrient indicators of soil, and organic matter is also an important factor in the formation of soil structure (see [23]). In this study, the SOM, TN, NH4-N and NO_3-N of *Liriodendron chinense* forest were higher than those of *Taiwania cryptomenoides* forest, It is consistent with the conclusion of Geng, that is: the soil organic carbon content of broad-leaved forest was significantly higher than that of coniferous forest (see [24]). Compared with coniferous forest, the concentration of organic carbon and total nitrogen in the aggregates of broad-leaved forest increased, and the stability of aggregates increased, which was beneficial to soil total organic carbon and soil activity Organic carbon accumulation. Studies have shown that the planting density of mixed *larch* plantations significantly affects soil bulk density, soil porosity, total nitrogen, total phosphorus, available nitrogen and available phosphorus (see [25]), and the establishment of artificial *Haloxylon ammodendron* forest can prevent sand damage, It can also change the physical and chemical properties of soil and improve soil fertility (see [26]), which further shows that soil organic matter plays a very important role in improving soil physical and chemical properties and promoting nutrient cycling (see [27, 28]).

6. Conclusions

This study proved that silviculture can quickly realize forest restoration in rocky desertification area. Afforestation technology should focus on afforestation land preparation, tree species selection and forest protection. In order to realize the sustainable forest with multi tree species and multi canopy, the rational application

Figure 7.
Aerial view of afforested land.

of mixed forest in the process of forest management should be paid more attention. This study is only the first step of forest vegetation restoration in rocky desertification area, and the future work will focus on how to cultivate the next generation of sustainable near natural forest (**Figure 7**).

Due to the poor site conditions and poor water distribution in rocky desertification areas, many areas have failed in the process of planting pure forest or mixed forest. For example, the survival rate of young forest is very low because the ecological and physiological relationship between species is not satisfied. The main reason is that the ecological and physiological relationship between species is properly handled. At the same time, the cost is saved and the probability of improper tending is reduced. Not only the pioneer tree species are successful, but also the saplings of multi tree species begin natural succession, which finally forms multi tree species and multi canopy in rocky desertification area, The experimental site provides a good reference template for vegetation restoration in rocky desertification areas.

It is suggested that trees form the families such as *Fagaceae*, *Lauraceae*, *Magnoliaceae*, *Camelliaceae* and *Cerambycidae* should be selected as drought resistant, (see [29]). barren trees with strong regeneration ability, high economic and ecological value and high water and light energy utilization rate in the rocky desertification area, and multi tree species, multi-layer and stable mixture of different ages should be built by mixing forest.

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References

[1] Yu LF, Zhu Shou Q, Zhu XK, et al. Restoration evaluation and restoration technology of degraded karst forest[J]. Guizhou Science, 2002,20(1):7~13. DOI:10.3321/j.issn:1001-7488.2000.06.002

[2] Yao XH, Ren HD, Li S, *et al*. Research on the restoration of rocky desertification vegetation[M]. Beijing: Science Press, 2013.

[3] Xiong KN, Li P, Zhou ZF, *et a*l. Remote sensing-GIS typical research on karst rocky desertification: Taking Guizhou Province as an example[M]. Beijing: Geological Publishing House, 2002.

[4] Bao SD. soil agrochemical analysis [M]. China Agricultural Press, Beijing,2000.

[5] Li Bo. Ecology. Beijing: Higher Education Press. 2000.

[6] Li YP, Xu H, Li YD, *et al*. Scale effect on the spatial distribution pattern of species diversity in tropical montane rain forests in Jian feng ling, Hainan[J]. Chinese Journal of Plant Ecology, 2016, 40(09):861-870. DOI:10.17521/cjpe.2015.0400

[7] Zhang YQ, Li ZC, Hou LY, *et al.* Effects of Stand Density on Understory Species Diversity and Soil Nutrients in Chinese Fir Plantation[J]. Acta Pedologica Sinica, 2020, 57(1): 239-250. DOI:10.11766/trxb201904080633

[8] Huang RX, Xu MF, Liu T, *et al*. Understory plant species diversity and environmental interpretation of five subtropical forest types [J]. Journal of Southwest Forestry University: *Natural Science*, 2020 (2): 53-62. DOI: 10.11929/j.swfu.201901016

[9] Kutter M, Gratzer G. Current methods for estimation of seed dispersal of trees - An example of seed dispersal of *Picea abies*, *Abies alba* and Fagus sylvatica[J]. Austrian Journal of Forest Science, 2006, 123(3): 103-120. DOI:10.1016/j.biocon.2011.09.018

[10] Zhang YQ, Li ZC, Hou LY, *et al*. Effects of stand density on species diversity and soil nutrients under Chinese fir plantations [J]. Acta Sinica Sinica, 2020, 57 (1): 239-250 DOI:10.11766/trxb201904080633

[11] Jing YR, Deng XW, Deng DH, *et al.* 2016. Soil physical and chemical properties and correlation analysis of different rocky desertification degrees in Southwest Hunan [J]. Journal of soil and water conservation, 30 (01): 189-195. DOI: 10.13870/j.cnki.stbcxb.2016.01.035

[12] Lin LB, Li TH, Wen SZ, *et al.* Study on growth regularity and biomass distribution feature of *Phoebe bournei-Schima superba* young mixed forest [J]. Journal of central south university of forestry and technology, 2019,39 (4) :79-84,98. DOI: 10.14067/j.cnki.1673-923x.2019.04.014

[13] Wei ZL, Le K, Huai C, *et al.* Regional scale variation in forest structure and arborous layer aboveground biomass along altitudinal gradient in the Qinling Mountains[J]. African Journal of Agricultural Research, 2012, 7(30): 4305-4316. DOI.org/10.5897/AJAR12.521

[14] Zakaria Rozainah Mohamad, Sofawi Ahmad Bakrin, Joharee Nur Adibah, *et al.* Stand structure and biomass estimation in the Klang Islands Mangrove Forest, Peninsular Malaysia[J]. Environmental Earth Sciences, 2018, 77(13): 486. DOI.org/10.1007/s12665-018-7636-7

[15] Tran Van Do, Phung Dinh Trung, Yamamoto Mamoru, *et al.* Aboveground biomass increment and stand dynamics in tropical evergreen broadleaved forest[J]. Journal of Sustainable Forestry, 2018, 37(1): 1-14. DOI.org/10.1080/10549811.2017.1375959

[16] Augusto L, Ranger J, Ponette Q, *et al.* Relationships between forest tree species, stand production and stand nutrient amount[J]. Annals of Forest Science, 2000, 57(4): 313-324. DOI.org/10.1051/forest:2000122

[17] ZHANG LH, QI JQ, LI TT *et al.* Effects of stand density on understory plant diversity and biomass in a *Pinus massoniana* plantation in Wenfeng Mountain Xinjin County[J]. ACTA ECOLOGICA SINICA, 2019,39(15): 5709-5717.DOI: 10.5846 /stxb201808151747

[18] Klimek Beata, Chodak Marcin, Jazwa Malgorzata, *et al.* The relationship between soil bacteria substrate utilisation patterns and the vegetation structure in temperate forests[J]. European Journal of Forest Research, 2016,135(1), 179-189. DOI.org/10.1007/s10342-015-0929-4

[19] Jia P, Liang JL, Yang SX, *et al.* Plant diversity enhances the reclamation of degraded lands by stimulating plant-soil feedbacks[J]. Journal of Applied Ecology, 2020, 57(7): 1258-1270. DOI.org/10.1111/1365-2664.13625

[20] Mahaut Lucie, Fort Florian, Violle Cyrille, *et al.* Multiple facets of diversity effects on plant productivity: Species richness, functional diversity, species identity and intraspecific competition[J]. Functional Ecology, 2020, 34(1): 287-298. DOI.org/10.1111/1365-2435.13473

[21] de Carvalho DC, Pereira MG, Toledo LO, *et al.* Nutrient cycling in the eucalyptus plantation with a regeneration of native species in understory[J]. Floresta, 2017, 47(1): 17-27. DOI.org/10.5380/rf.v47i1.43652

[22] Duan WJ, Ren H, Fu SL, *et al.* Community Comparison and Determinant Analysis of Understory Vegetation in Six Plantations in South China[J]. Restoration Ecology, 2010, 18(2): 206-214. DOI.org/10.1111/j.1526-100X.2008.00444.x

[23] Diaz S, Cabido M. Vive la difference: plant functional diversity matters to ecosystem processes[J]. Trends in Ecology & Evolution, 2001, 16(11), 646-655. DOI.org/10.1016/S0169-5347(01)02283-2

[24] GENG YQ, YU XX, YUE YJ *et al.* Soil active organic carbon pool of coniferous and broadleaved forests in the mountainous area of Beijing[J]. Journal of Beijing Forestry University 2009,31(5):19-24. DOI :10.13332/j.1000-1522.2009.05.015

[25] Wang SL, Ma C, Yuan WB. Soil physical and chemical properties under four densities of hybrid Larch Plantations[J]. Advanced Materials Research, 2012, 524-527: 2139-2142. DOI.org/10.4028/www.scientific.net/AMR.524-527.2139

[26] Chen F, Zhou DQ, Bai XY, *et al.* Responses of soil physical and chemical properties to karst rocky desertification evolution in typical karst valley area[J]. 2017 3RD INTERNATIONAL CONFERENCE ON ENVIRONMENTAL SCIENCE AND MATERIAL APPLICATION (ESMA2017), VOLS 1-4, 2018, 108. DOI.org/10.1088/1755-1315/108/3/032012

[27] Liu NJ. Effect of Artificial Haloxylon Ammodendron Forest on the Physical and Chemical Properties of Sand Soil[J]. Soil Bulletin, 2008, 39(6): 1480-1482. DOI.org/10.3321/j.issn:0564-3945.2008.06.052

[28] Qin J, Tang XH, Yang XM. Effects of soil physical and chemical properties on different forest types of *Pinus massoniana*[J]. Journal of Eco-Environment[J]. 2013, 22(4): 598-604. DOI.org/10.3969/j.issn.1674-5906.2013.04.009

[29] Zhu NH, Li JX, Zhang B. The collection of precious and special plants in the Wuling mountainous area [M].2015.

Chapter 7

Legal and Administrative Aspects of Forest Pest and Disease Control in Japan

Koji Matsushita

Abstract

Approximately 40% of Japanese forests are softwood plantations consisting of trees such as Japanese cedar (*Cryptomeria japonica*), Japanese cypress (*Chamaecyparis obtusa*), and several varieties of pine (*Pinus* spp.). Policies and programs related to forest pests and diseases are important for growing forest plantations. Damage caused by the pine bark beetle (*Monochamus alternatus*) has been a long-standing problem in Japan. Forest damage caused by the pine bark beetle was first found in Nagasaki Prefecture in 1905. Since then, the area of damage has expanded gradually to all prefectures. Damage caused by pine bark beetles became serious during and just after the end of the Second World War. In 1950, the Natural Resource Section of the General Headquarters of the Allied Forces, Supreme Commander for the Allied Powers (GHQ/SCAP) made recommendations for how to control forest pests and diseases. The first act was enacted in 1950, although the control of forest pests was initially addressed as part of the first Forest Act of 1897. Several important reasons for why the Japanese government has failed to stop the expansion of the damaged area can be found in GHQ recommendations: the lack of coordinated programs, underutilization of damaged trees, and shortcomings of forest-management plans.

Keywords: cutting damaged trees, Forest Act, forest management plan, Forest Pest Control Act, GHQ, pine bark beetle, usage of damaged trees

1. Introduction

Measures to respond to natural disasters and various kinds of pests and diseases are necessary for sustainable forest management. There have been numerous natural disasters in Japan because of its steep terrain, abundant rainfall, and the susceptibility of its islands to typhoons. The area of damaged forests changes annually, with respective values, for example, of 7,023 ha, 4,831 ha, 5,686 ha, 14,575 ha, and 3,766 ha over the 5-year period from 2013 to 2017, respectively [1]. This constitutes a total area of 35,881 ha, equal to 26.7% of the total area of new forest planted during the same period (134,531 ha).

The area of forests damaged by pests and disease is significant, although it is difficult to measure the damaged area precisely due to the constant spread of damage over a long period. On the other hand, climate-related damage occurs over short periods. As a result, statistics on damaged forests sometimes use the volume of damaged trees instead of the area of damaged forests.

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The most serious and long-term damage in Japan has been caused by the pine bark beetle [2]. The first pine bark beetle damage[1] was recorded around 1905 in Nagasaki Prefecture[2] on Kyushu Island, which is the western-most of Japan's four main islands [2]. Since then, the damaged region has gradually extended to the east. According to the most recent statistics, the volume of damage was 399,000 cubic meters in fiscal year 2017 [1], equivalent to 62.2% of the volume of pine production in the same year (641,000 cubic meters).

Sustainable forest management, especially in the case of coniferous plantation forests, necessitates protection against various kinds of damage, such as that from climate change, pests, disease, and forest fires.[3] Countermeasures to protect against damage include both technical and social actions, for example, forest-management systems, forest-product consumption, and legal systems of forest management. This chapter focuses on the legal and administrative aspects of social countermeasures.

The contents of this chapter are as follows. Section 2 discusses the method and statistical overview of damage caused by the pine bark beetle. Section 3 covers the development of the legal system and related topics. Section 4 explores the legal and administrative reasons why damage from the pine bark beetle has continued unabated over the last 100 years, in the context of the present-day GHQ recommendations. Concluding remarks are made in the final section.

2. Methods and overview of damage

2.1 Methods

Primary sources of information were literature surveys and statistical data from the Forestry Agency of the Ministry of Agriculture, Forestry, and Fisheries (MAFF). Information from the literature surveys was divided into three categories.

The first category is the Forest Act and related enforcement orders. The first Forest Act was enacted in 1897, with amendments in 1907, 1939, and 1951. In 1950, articles on pest and disease control were transferred from the Forest Act to a new act dealing only with pest and disease control.

The second category is the legal system. There have been several acts relating to the pine bark beetle.

The third category includes literature and documents from the Natural Resource Section (NRS) of the General Headquarters of the Allied Forces, Supreme Commander for the Allied Powers (GHQ/SCAP). In 1950 and 1951, GHQ made two recommendations on pine bark beetle management.

Most of the literature related to the first and second categories is owned by the Library of Forest Resources and Society, Graduate School of Agriculture, Kyoto University. Some was retrieved from the website of the Forestry Agency[4]. Documents by the NRS of the GHQ/SCAP were obtained from the digital collections of the National Diet Library and microfiche materials collected by the Modern Japanese Political History Materials Room of the National Diet Library. Most of the

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1 Pine bark beetle damage in the area currently known as Chiba Prefecture was recorded in 1804 (Edo Period) [3].

2 According to the Forestry Agency [2], the first record was reported by Yano [4].

3 The number of forest fires and the combined area of burned forest in fiscal year 2017 were 1,284 fires and 938 ha, respectively [1]. Generally, the damage caused by forest fires is not very large in Japan.

4 https://www.rinya.maff.go.jp/j/kouhou/toukei/index.html, 2020/11/05

statistical data were obtained from the Annual Statistics of Forestry, edited by the Forestry Agency. Statistical data from and immediately after the Second World War were based on the former administrative organization of the Forestry Agency[5] and documents compiled by NRS of the GHQ/SCAP.

The essential Japanese literature defining pest control policies in Japan includes [5–8]. The relevant English literature, published during occupation, is limited to [9, 10]. These publications were used as sources for all explanations of policy and legislation covered in section 3.

English translation of the current Forest Pest Control Act (Act No. 53 of 1950), Order for Enforcement of the Forest Pest Control Act (Cabinet Order No.87 of 1997), Forest and Forestry Basic Act (Act No. 161 of 1964), was based on the Japanese Law Translation webpage[6] prepared by the Ministry of Justice, Government of Japan.

2.2 Statistics on pine plantation forest damage by the pine bark beetle

The volume of pine trees damaged by the pine bark beetle since 1932 is shown in **Figure 1**. The volume of national forest damage in 1956, 1957, and 1958 is unclear. The volume of forest damage was 670,000, 635,000, and 802,000 cubic meters in 1956, 1957, and 1958, respectively [3]. However, these amounts are larger than the total volume of damaged national and non-national forest shown in [7]; therefore, we have omitted national forest damage volume data for these three years. For 1948 and 1949, GHQ reports greater volumes of damaged forest than reported by the ministry [15]: 225,000 cubic meters for national forest and 1,267,000 cubic meters for non-national forest in 1948, and 192,000 cubic meters for national forest and 1,487,000 cubic meters for non-national forest in 1949.

There are two peaks in the volume of damage; the first occurred in 1948 and 1949, when the volume of damage was 1.3 million cubic meters. Subsequently, the volume tended to decrease. The annual volume of damage was almost 0.4 million cubic meters to 0.5 million cubic meters by 1971, after which time it increased again, reaching a second peak of 2.4 million cubic meters in 1979. The volume of damage was almost double that in the first peak year. Since the second peak year,

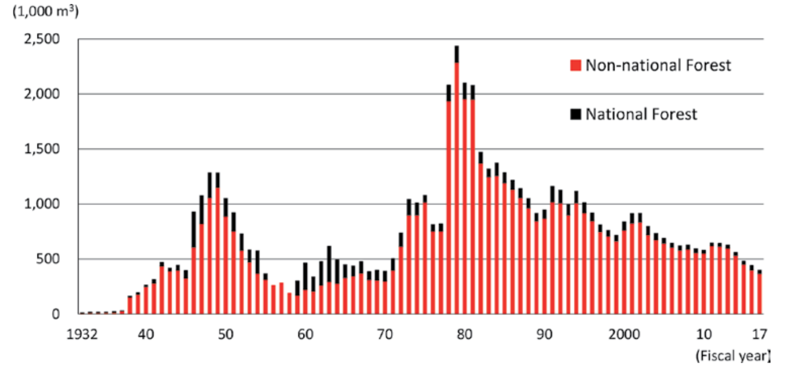

Figure 1.
Volume of forests damaged by pine bark beetles. Source: [3, 11–14]. Note: The Japanese forest classification system defines non-national forests to include private forests and public forests owned by prefectural governments, municipalities, etc.

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[5] Before 1947, the national forest was managed by three ministries. One of the bureaus was the Bureau of Forest, Ministry of Agriculture and Forestry.

[6] http://www.japaneselawtranslation.go.jp, 2020/11/05

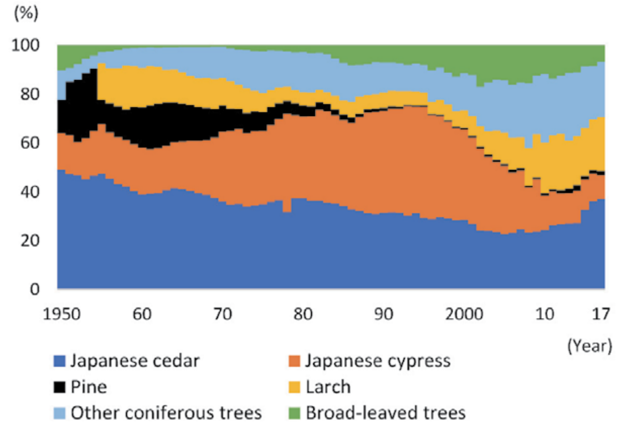

Figure 2.
Percentage of planted area classified by species. Source: [12–14, 16]. Note: Before 1954, pine include larch. After 1989, data include the area of underplanting.

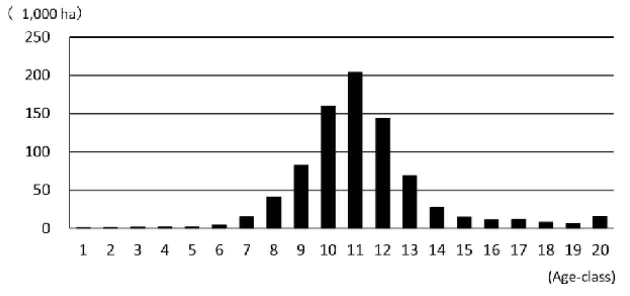

Figure 3.
Age–class distribution of pine plantation forests. Source: [17]. Note: In Japanese forest resource tables, one age-class equals 5 years. These data are calculated for forests under the forest planning system. The data in age-class 20 include age-class 21 and over.

the volume of damage has gradually decreased. After 2015, it was less than 0.5 million cubic meters, which was almost the same as in the 1960s.

Due to such long-term damage by the pine bark beetle, the annual area of pine trees planted has decreased. **Figure 2** shows the percentages of planted trees, classified by species. In 1955, the percentage of pine was 10.1%. The highest percentage of pine was recorded in 1962 at 18.4%. Since 1962, the percentage has decreased. It was less than 5% in 1980 and has been less than 2% since 1987. Pine has almost completely lost its role as the major plantation species as a result of the pine bark beetle. As the annual area of pine planted has decreased, the age distribution of pine plantation forests from age-class 1 to age-class 6 is exceedingly small (see **Figure 3**).

3. The legal system related to pine bark beetle management

3.1 The Forest Act prior to the Second World War

The first Forest Act (Act No. 46 of 1897) was enacted in 1897. The Forest Act was first proposed by a bill in 1872, but it, along with several succeeding bills, was killed [18]. The most important chapters of the first Forest Act were Chapter 2, on the regulation of forest practices, and Chapter 3, on forest protection.

Only one article on pests and diseases was included in the first Act. Article 36 required that people who discovered fire, pests, or disease in or near a forest, or who knew of someone that had committed a crime or attempted to commit a crime against a forest, should immediately report the details to foresters, police, or local government staff. In this version, forest pests and diseases were treated in the same way as fires and crimes.

The first Forest Act was amended in 1906 (Act No. 43 of 1906). It included the introduction of the Forest Owners' Association (FOA). Because it was difficult to prevent damage by pests and diseases using the same reporting systems as for fires and crimes [5], the Forest Act of 1906 contained three additional articles (Articles 80, 81, and 82) that directly addressed forest pests and diseases.

Article 80 stipulated that forest owners had a legal obligation to prevent and exterminate forest pests and diseases. Forest owners could even enter another owner's land to exterminate or prevent pests or diseases with the permission of foresters or police authorities. Article 81 stated that local governors could order interested forest owners to enact necessary extermination or prevention measures under sufficient threat. The extermination and prevention costs were incurred by the forest owners, depending on the forest area or value. Administrative subrogation was available, and the expense was compulsorily collected. The Act on the Extermination and Prevention of Pest Insects[7] (Act No. 17 of 1896), which defined the extermination of pest insects in the context of agricultural products, imposed a duty on farmers to attend to pest and disease management, whereas no such duty existed in the case of forest pests and diseases [19]. Article 82 defined that the right to demand compensation was denied for extermination and prevention of forest pests.

Under the Forest Act of 1907, the extermination and prevention of forest pests and diseases was the obligation of forest owners, who incurred related expenses[8]. This was an important feature of the act. When it was difficult for forest owners to control the spread of pests and diseases, local governors could order or carry out extermination and prevention measures.

The Order on Enforcement Procedures of the Forest Act (Ministry of Agriculture and Commerce, Order No. 30 of 1907) laid out the procedure for responding to the spread (or threat thereof) of forest pests and diseases into neighboring prefectures. Article 24 defined immediate reporting of the spread, and Article 25 defined the cooperative extermination and prevention measures [20].

Under the Forest Act of 1906, four types of FOAs were introduced. These included FOAs for planting, forest practices, forest roads, and forest protection. Members of each FOA acted cooperatively to plant trees, create forest management plans (FMPs), construct and maintain forest road systems, and prevent and manage forest fires, illegal cutting, and pests. While the establishment of FOAs was voluntary, once one was established all forest owners living within the jurisdiction of a FOA were required to join [21].

According to an example of the articles of incorporation of a FOA for protection [22], the contents of the business field defined in Article 2 were defense against forest fire, defense against theft, extermination and prevention of forest pests, and other detriments [20]. The expense distribution was to depend on the forest area holding (Article 9). There were numerous FOAs dedicated to forest practices, but

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7 This act excluded pest insects in forests.

8 In the prewar legal system, pest control was the responsibility of forest owners; less than half of the total cost was subsidized by the prefectural government [6].

not the other three types. Many FOAs did not work substantially after their establishment [23]. Thus, it was not clear how many protection FOAs worked diligently towards the extermination of forest pests.

In comparison to the 1897 Forest Act, the 1907 Forest Act developed a clear legal system to deal with forest pests and diseases. However, there were only three articles, and, as the Forestry Agency pointed out, they were problematic [5]. The articles were too simplistic and were lacking in detail, there were many ambiguities as far as procedures, and the rule on forest owners' expenses was problematic from the standpoint of a workable system. Although the major objective of the 1897 Forest Act was forest resource management based on the regulation of forest practices and protection, a chapter on FOAs was added in the 1907 amendment that added three articles relating to pest and disease control. However, the Forestry Agency pointed out that the articles in the 1907 Forest Act had very few controls and that there had been only one prosecution for violation of the order, based on Article 81, in 1931 [5].

The 1907 Forest Act was partially amended in 1939 [24]. The most important part of the amendment was that FMPs had to be developed not only for public forests and forests owned by shrines and temples but also for all non-national forests. The government supervision system for non-national FMPs was contained in the amendment. Forest owners who held 50 ha or more were required to make individual FMPs, whereas for forest owners holding less than 50 ha the plans were the responsibility of the FOAs. The four types of FOAs were removed and new FOAs were established in all municipalities. No changes were made to the three articles relating to forest pests and disease control.

3.2 Pre-war acts other than the Forest Act

In the early Meiji period, before the establishment of FOAs in the 1907 Forest Act, local cooperative organizations were established based on common forests that originated in the mid-Edo period. Articles on forest protection were sometimes included in the rules and regulations of the cooperative organization. Local governments started to establish rules for non-national forests. For example, Mie Prefecture (1884), Fukushima Prefecture (1885), and Shiga Prefecture (1886) instituted a new regulatory rule for non-national forests [23]. In the case of Article 5 of Fukushima Prefecture's rule, protections against forest fires, illegal cutting, and pest insects were included [23]. Rules on forest protection, including pest control, existed in common forests and prefectural regulations, but it is unclear whether such rules were part of a workable system.

The utilization of trees damaged by forest pests in national forests was considered during wartime. Before the Second World War, basic management rules were determined by FMPs based on the ordinances of the national FMP (Ministry of Agriculture and Commerce, Order No. 42 of 1899; Ministry of Agriculture and Commerce, Order No. 9 of 1914). The central policy was sustained yield management, but it was difficult to maintain the sustained yield policy in national forests after 1940 because of a timber supply shortage resulting from a decrease in log imports and a shortage of forestry workers [25]. In 1940, the chief of the Bureau of Forest, Ministry of Agriculture and Forestry, sent a circular notice to the chief of the Regional Office of the Bureau of Forest regarding a temporary increase in timber production in national forests. As a result of the notice, 11 types of cutting activities were authorized that were not previously authorized in the national FMP. One was cutting trees when there was no possibility of recovery from the damage inflicted by pests and diseases. Effective control of pests and diseases required cutting and immediate removal of damaged trees, which could then be used.

3.3 Before the Forest Pest Control Act

The first pine bark beetle damage in Japan was found in Nagasaki Prefecture on Kyushu Island. The damaged area began to spread east (**Figure 4**). In the first year that statistical records of pine bark beetle damage began, only four prefectures were recorded as having damage: Nagasaki, Miyazaki, Hyogo, and Shiga. In that year, 93.8% of the total damaged volume in Japan was located in Hyogo Prefecture. In 1946, the final year in **Figure 4**, the number of prefectures containing damage increased to 17. At that time, damage spread to numerous western prefectures. By 1947, the number of prefectures containing damage had doubled to 34. By 1949, damage had spread to all of Japan except Hokkaido, and by 1950, all of Japan was affected.

Although the reliability of wartime record keeping is questionable, it is certain that damaged areas and the volume of damage began to increase during and after the Second World War. Several causes were identified. Shinrin Byogaichutou Boujo Kenkyukai pointed out the increase in cut-over pine forests at the time, which resulted from cutting for military uses and increased wood demand for post-war reconstruction [7]. Cut-over pine forests became the source of the pine bark beetle. Furthermore, forest products made of pine, which included mine timber and timber for ships and vehicles during the war as well as fuelwood, mine timber, and construction wood after the war, moved widely. This transfer contributed to the spread of the area subject to damage.

Before the enforcement of the new act on the pine bark beetle in 1950, Articles 80–82 of the Forest Act of 1907 were effective. The legal system had not worked well to prevent and deal with damage since around 1946 and that there were almost no measures to deal with the damage except in a few prefectures [27].

In 1948, a circular notice prohibiting transfer of pine logs with bark was sent from the Director-General of the Bureau of Forestry to the prefectural governors. Transfer of pine logs with bark from areas where pine bark beetle damage occurred to outside the prefecture was prohibited, and complete barking was required. In 1949, the Director-General sent another circular notice to the prefectural governors prohibiting fuel production from untreated damaged pine trees.

3.4 Enforcement of the Forest Pest Control Act

After the spread of the pine bark beetle, new legislation related to forest pests and diseases was enacted in 1950, called the Act on the Extermination and

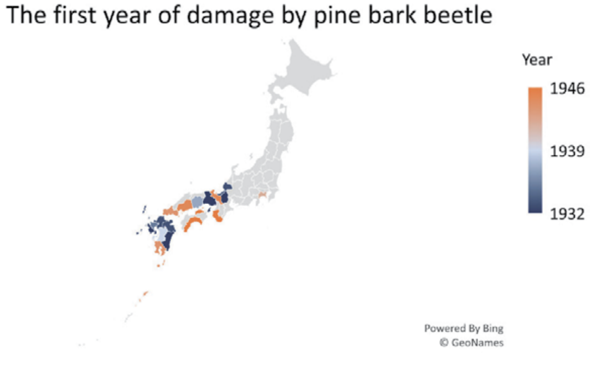

Figure 4.
First year of pine bark beetle damage (1932–1946). Source: [11, 26]. Note: Year means the first year when damaged volume was recorded. Record keeping of damaged areas first began in 1932.

Prevention of Pine Bark Beetle and Other Similar Borers and Other Destructive Forest Pests and Diseases[9] (Act No. 53 of 1950). Articles 81 and 82 of the Forest Act of 1907 were removed, and Article 80 was amended to say that forest owners could enter other owners' forests for the purpose of extermination and prevention. In the current Forest Act, the contents of this article are included in Article 49, which defines entry and surveying. The original purpose of Article 49 was to establish that, when necessary, forest owners could enter other owners' forest for measurement and onsite surveys related to forest practices with the permission of the municipal mayor. Article 49 would be applied mutatis mutandis when pests, animals, fungi, and viruses threatened to cause serious damage to a forest.

The new act of 1950 contained 16 articles and differed in several ways from the Forest Act of 1907. Prior to the enforcement of the act, the first GHQ recommendation[10] was made by Furniss [5].

As can be seen from the name of the act, the main objective was to establish countermeasures against the pine bark beetle. The Minister of Agriculture and Forestry or prefectural governors could order exterminations (Articles 3 and 5). For example, they could prohibit transfers of cut trees, which became a problem just after the war. A new finance system was introduced to provide compensation for prevention costs when ordered by the government (Article 8). A national subsidy was also determined in Article 9. Concerned officials or control members could conduct spot inspections (Article 6) and had the right to direct forest owners to take measures (Article 7).

The second GHQ recommendation [28] was issued in 1951, and in 1952, the act of 1950 was amended and renamed the Forest Pest Control Act (Act No. 26 of 1952). The species of pests and diseases were determined by related cabinet order, including seeds and seedlings for forestry. Six species were first identified; one was removed in 1958, and then three were added between 1952 and 1962, bringing the current total to nine. The original six species are borers that attach to and impede the growth of trees: pine caterpillars, pine needle gall midges, gypsy moths, *Diprion nipponica*, and field mice. *Diprion nipponica* was removed in 1958. Chestnut gall wasps, cryptomeria needle gall midges, cedar spider mites, and guignardia laricina were added in 1952, 1955, 1956, and 1962, respectively.

The Forest Pest Control Act was amended several times, and a significant amendment was made in 1967 that added control through pesticide application to the extermination order. Article 6 laid out the extermination order by the Minister of Agriculture and Forestry. There were six types of orders listed in the article, and (4) is as follows: to order a person who owns or takes care of trees or designated seeds and seedlings to which damage is being caused or is likely to be caused by forest pests to carry out control through pesticide application.

3.5 Enactment of The Act on Special Measures Concerning the Control of Pine Bark Beetle

As shown in **Figure 1**, the damaged volume began to increase again, reaching a second peak of 2.4 million cubic meters in 1979. The Forestry Agency identified three causes for the increase during this period.

[9] The name of the act was taken from the name of the bill, including the following GHQ/SCAP documents: GHQ/SCAP Records (RG 331, National Archives and Records Service), Box No. 3007, Folder title: Pine Bark Beetle Program.

[10] A summary of the first recommendation is included in [28].

The first cause was a decrease in the demand for fuelwood spurred by a fuel revolution [7]. The percentage of the total cutting volume of wood used for fuel decreased linearly (see **Figure 5**). In 1972, it decreased to 3.0%. As a result, demand for damaged pine trees decreased significantly, and dead trees and branches left in forests became sources of new damage. The second cause was a decrease in the chip price of pine, which resulted in worsening profits from the production of pine and pine forest management [7]. Some owners abandoned pine forest management, along with pine bark beetle control. The third cause was typhoon damage in 1971 and extraordinary climatic conditions in 1973 [7]. Except for climatic causes, economic factors such as a decrease in pine tree demand and decreases in the price of pine chips contributed to the expansion of the volume damaged in the 1970s. Under these conditions, the Act on Special Measures Concerning the Control of Pine Bark Beetle (Act No. 18 of 1977) was enacted as 5-year temporary legislation.

The objective of this act was to protect important forest resources from the increasing damage caused by the pine bark beetle. During the 5 years from 1977 to 1981, the government planned to conduct intensive extermination to halt pine bark beetle damage. As seen in **Figure 1**, the annual volume of damage from 1978 to 1981 was over 2 million cubic meters. The effective period of the Act on Special Measures was almost equal to the period when the volume of damage was at a record high. When the temporary act expired on 31 March 1982, the effective period was extended for an additional 5 years under the new name Act on Special Measures Concerning the Damage of Pine Bark Beetle (Act No. 21 of 1982).

Under the extended act, five integrated methods were identified to exterminate and prevent pine bark beetle damage. The methods were pesticide application from aircraft and above ground, applying pesticides to felled trees, special felling (after which crushing, incineration, or charring could be necessary), and tree species replacement. Tree species replacement was a new policy. Paragraph 4 of Article 3 of the act outlined replacement. To protect pine forests and maintain their various functions, the policy called for replacement with species other than pine that were unlikely to be affected by the pine bark beetle.

The area in which new species[11] were planted is shown in **Figure 6**. A maximum of about 9,000 ha was planted in 1982 (the first year in which the method was implemented), decreasing gradually to about 4,000 ha in 1987. In 1987, the area of

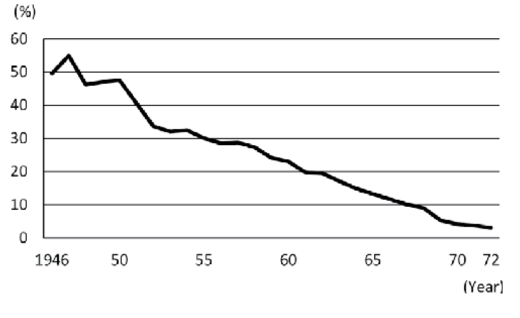

Figure 5.
Fuelwood as a percentage of total cutting volume. Source: [12]. Note: The statistical survey on cutting volume for fuelwood ended in 1972.

[11] Most of them are broad-leaved trees. In the case of plantation forest, the recommended species by the Forestry Agency is Japanese cypress, *Quercus acutissima*, etc. (Forestry Agency, https://www.maff.go.jp/j/kokuji_tuti/tuti/pdf/t0000211_2.pdf, 2020/11/05).

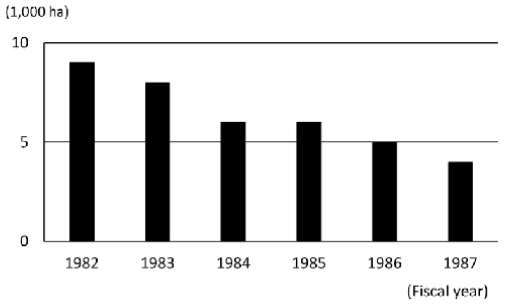

Figure 6.
Area of plantation forests replaced with other species. Source: [7].

damaged pine forest was about 610,000 ha[12] [7], and the area of plantation forests planted with new species under the replacement policy was equal to 0.7% of the total damaged area. This area was limited because of the lack of motivation to cut pine trees due to the economic situation of the forest products market (for example, decreased timber prices) [7]. It was difficult to replace tree species without clear-cutting, and even after a clear cut, the trees had to be removed. Without hauling, it is difficult to plant new species.

For example, in Iwate Prefecture, tree species replacement was conducted on 93 ha in 2016 [29]. Iwate Prefecture is located in the northern part of Honshu Island and pine bark beetle damage is now spreading. It was pointed out that there is a cooperative policy regarding the use of damaged trees by woody biomass powerplants, as well as encouragement of forest owners by municipal offices and local FOAs to cut damaged trees. It is difficult to conduct tree species replacement without a system for utilizing the damaged trees.

As shown in **Figure 2**, newly planted pine forests decreased substantially after the 1980s. The area of pine forests decreased as a result of the damaged forests and the replacement policy.

Expiration of the Act on Special Measures of 1982 was postponed in 1987 and 1992, finally lapsing on 31 March 1997. At that time, the Forest Pest Control Act (Act No. 26 of 1952) was amended to follow the control methods[13]. In the official document,[14] the amendment stated that the damaged volume had decreased by almost 60%, from 2.43 million cubic meters in fiscal year 1979 to 1.01 million cubic meters in fiscal year 1995, and that the spread of damaged areas had almost been stopped. The volume of damage was still high, and the document highlighted the possibility of damage again increasing due to climatic conditions. It also highlighted the need to strengthen forest pest control systems for early detection and quick extermination. Levels of forest management were declining in general due to changes in the business environment of the forestry industry. A special felling method included in the expired act was carried over into the amended Forest Pest Control Act.

[12] 590,000 ha of the total damaged forest area (610,000 ha) was non-national forests. As the percentage of non-national forest is almost 70% of the total area, the likelihood of pine bark beetle damage in non-national forests was extremely high.

[13] When the Act on Special Measures lapsed in 1997, treatment of trees in which the degree of damage was not serious could become part of forest improvement projects. This practice was identified in the Forest Improvement Conservation Project Plan based on Article 4 of the Forest Act of 1951.

[14] On the enforcement of the amendment of the Forest Pest Control Act (Document No. 9-Rinya-Zou-100, April 1 of 1997) https://www.maff.go.jp/j/kokuji_tuti/tuti/pdf/t0000205.pdf, 2020/08/23

4. Discussion

4.1 GHQ recommendations and their present-day significance

The first peak of damage caused by the pine bark beetle occurred just after the end of the Second World War when GHQ occupied Japan (from 1945 to 1952). GHQ/SCAP pointed out challenges with prevention during wartime [15]. The damaged volume began to increase during the war, and it continued to increase substantially afterwards, especially in non-national forests (see **Figure 1**). Even now, 75 years after the end of the war, the damage has not been contained. Analyses of policies and related reports, especially those written by GHQ, are therefore of great importance.

During the occupation of Japan, GHQ made various recommendations in the field of forestry. The Natural Resource Section (NRS) published many reports and made many recommendations, including two by Furniss regarding forest pest control. This section discusses the present-day importance of these recommendations. While many of the recommendations were technical in nature[15], the focus of this section is primarily on four aspects of their legal and administrative contents (including the first recommendation by Furniss in 1950, hereafter referred to as "the recommendation").

The first point concerns issues with the pre-war legal system, especially Article 80 of the Forest Act of 1907. This article set out the obligation of forest owners to exterminate and prevent pine bark beetles. The recommendation also required the government to make a unified extermination plan. The basic policy framework differed between the recommendation and the Forest Act of 1907. When pest control is the responsibility of the national government, differences in damage to national and non-national forests will not occur. According to **Figure 1**, the problem occurred mainly in non-national forests. When pest control is the responsibility of private forest owners, extermination and prevention measures depend on the will of the forest owners and their management situation. After the end of the war, the three organizations formerly responsible for national forest management were unified into one organization, and new national forests were to be managed by an independent accounting system subject to the National Forest Management Special Account Act[16] (Act No. 38 of 1947). Under the special accounting system, regional and local national forest offices conducted pest control only in national forests. If adjacent non-national forest was damaged by the pine bark beetle, it was difficult for the national forest to conduct certain extermination measures.

In the 2012 amendment of the Forest Act, a new system of cooperation between national and non-national forests concerning the public interest was created (Article 10.15–10.19). In the case of thinning, a national forest could sign an agreement with a private forest owner, whereby the national forest could outsource the forest management of both areas to a forestry company with a copayment by the private forest owner. Another case of cooperation was in the extermination of non-native species at World Natural Heritage sites and candidate sites. To date, only these two types of cooperation exist, and this system has not been applied to pine bark beetle extermination. It remains difficult to conduct forest practices, including joint pest

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[15] For example, Nakano [30] of the Government Forestry Experiment Station commented that some of the technical contents were different from his experience. The different opinion by the Chief of the Department of Forestry in Yamanashi Prefecture was also referred to in [30].

[16] This special account act was abolished by the enactment of the Act on Special Accounts (Act No. 23 of 2007).

control, even now. In Japan, the basic policy framework clearly delegates management of national forests to the Forestry Agency and management of non-national forests to forest owners, local governments, and FOAs. Generally, in the case of national forests, a public bidding system determines which companies will perform work, while in non-national forests, negotiated contracts are used. With regard to pest control, this difference seems to be a considerable barrier.

A similar problem was pointed out in GHQ/SCAP [10]: "Further, the outstanding characteristics of the present bark beetle control activities in Japan have been lack of a coordinated program and lack of an adequate organization to carry on control." While the situation has changed greatly over the past 70 years, generally speaking, the vertically segmented administrative system[17] still exists in Japan. Individual countermeasures against damage in national and non-national forests is a typical example[18].

Second, it is worth noting that the recommendation referred to non-economic forest and trees. In the Forest Act (Act No. 249 of 1951), three articles relate to the definition of a forest. Article 2 prepares the definition of a forest, excluding trees on land that was mainly utilized as farmland, residential land, etc. Trees on residential land, including housing shelter trees, windbreak trees on farmland, and trees in Japanese gardens, are excluded. The basic Japanese forest resource program is a forest planning system. The forests to which the forest planning system applies is defined by Article 5, which excludes isolated small forests (less than 0.3 ha) in city areas in the Regional Forest Plan prepared by prefectural governments. Article 10 (4) excludes from the Regional Forest Plan experimental forest sites in university forests and forest on the grounds of a temple or shrine. Forests excluded in Articles 2, 5, and 10(4) are generally small and are not primarily managed for timber production.

As for pine forests, both timber production and non-production forests are important for measures against pine bark beetle damage. The definition of forests to which the Forest Pest Control Act applies is unclear from its articles, meaning that all pine forests may be included. The total forest management plan, which includes all forest practices, including extermination and prevention of pine bark beetle damage, has some importance for controlling damage.

Third, the recommendation pointed out the necessity of amending the Forest Act and prefectural forestry regulations. The Forestry Agency stated that the recommendation created momentum for amending the prewar legal system concerning damage by forest pests and diseases [5]. An amendment of the legal system could be realized by using the outside political pressure that the GHQ recommendation represented. The increase around the second peak in 1978 followed the enforcement of the Act on Special Measures Concerning the Control of Pine Bark Beetle in 1977. As shown in **Figure 1**, the volume of damage has decreased since the second peak, although it remains high and almost equal to that in the 1960s. Although there is no outside pressure, the necessity of improving the legal

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[17] GHQ/SCAP [10] pointed out the situation of forest research, including forest insect research. In the paragraph on this topic, the following sentence was included; "entomologists who were unable to translate their detailed biological findings into terms of practical control." This means that there was a communication problem between the research and administrative sectors.

[18] Furniss [28] noted the following passage as a part of "progress and problems in 1950": "The responsibility of the central government for control of forest insects, both on national forest and private forest lands, should be placed in one organization in the Forestry Agency." This recommendation has still not been followed.

measures related to pine bark beetle damage exists, at least regionally, especially in the northern Honshu Island prefectures where the volume of damage is increasing.

Finally, the necessity of an investigative organization and correct data collection[19] should be noted. This recommendation applies not only to pine bark beetle damage but also forest resource statistics. For example, GHQ/SCAP [10] pointed out the lack of dependable statistics and a comprehensive forest survey for forest resource management. GHQ/SCAP [10] also pointed out the lack of dependable statistics on tourism and recreational activities in national parks. After the end of the Second World War, the forestry sector statistical system developed, but development in the field of statistics regarding damage to forests remains poor. It is difficult to assess the damaged area, volume and value correctly, particularly in the case of forest pests and diseases. The level of concern on the part of plantation forest owners has significantly decreased for various reasons, such as low profitability, aging, and an increase in absentee forest owners. As a result, developing statistics on damaged trees is difficult. Since it is difficult to make an adequate control plan without reliable statistics, the recommendations made by GHQ regarding the need for investigative organizations and data-collection mechanisms still apply.

4.2 Utilization of damaged pine trees

Effective utilization of damaged trees is another important issue related to extermination and prevention. Furniss' first recommendation in 1950 was to use the damaged trees as much as possible[20]. As previously mentioned, at around the first peak in 1948, one of the causes of the increase in the volume of damage was abandoned damaged trees or logs. At the time, there was demand for fuelwood and various types of industrial round wood, but moving the damaged logs became a problem. By the time of the second peak in 1978, the decreased demand for pine logs due to the decreased demand for fuelwood and the price of chips was the precursor to increases in damage. Finding uses for damaged trees motivates removal, which helps to prevent the further spread of the pine bark beetle.

Although the data are old, statistics on the utilization of damaged pine logs from the mid-1980s are available. The 1980s was a period of decreasing damage just after the second peak of 1978, and the Act on Special Measures Concerning the Control of Pine Bark Beetle (Act No. 18 of 1977) was postponed and renamed in 1982. Figures on the total damaged and utilized volumes are shown in **Figure 7**. Both damaged and utilized volumes decreased in the mid-1980s. The percentage of utilized volume decreased from 13.1% in 1983 to 10.5% in 1987. Unutilized damaged volume should be treated in some way or left in the forest, and it is possible that damaged trees and logs left in the forest preceded subsequent damage.

As shown in **Figure 1**, even though the damaged volume has decreased, it is still large at 0.4–0.5 million cubic meters. Although the statistical details are unclear, it seems that utilization of damaged trees and logs is insufficient. Although the

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[19] GHQ emphasized data collection. For example, a summary of results from the Japan Pine Bark Beetle Conference held in several cities in August 1950 showed key points addressed by Mr. D. J. Haibach of NRS, GHQ/SCAP. The first point was accurate surveying and reporting of infestation, which was the foundation of building an effective pine bark beetle control program (GHQ/SCAP Records (RG 331, National Archives and Records Service), Box No. 3105, Folder title: Pine Bark Beetle Control: by Kessler, Jeneye).

[20] At that time, labor-intensive countermeasures against pine bark beetle damage established immediately after the Second World War have had an economic impact, as a component of unemployment relief projects [31].

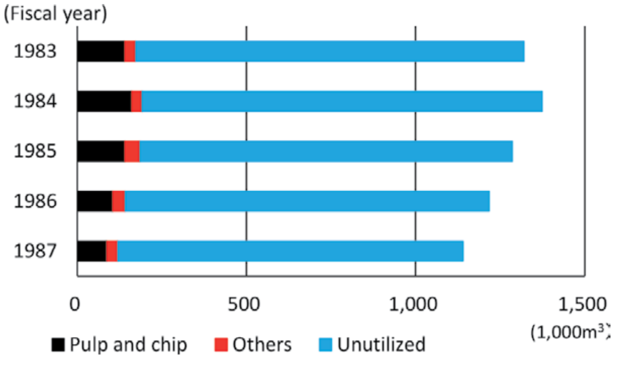

Figure 7.
Utilization of damaged pine trees. Source: [7] for utilized volume; [13] for damaged volume. Note: "Others" is the total logs for fuelwood, lumber, etc. "Unutilized" volume was calculated by subtracting the total volume used for "pulp and chips" and "others" was calculated from the total damaged volume.

utilized volume is not specified, the Forestry Agency [2] stated that examples of utilized damaged trees include fuelwood and chips supplied to paper mill companies.

The current significant change in domestic timber demand comes from increased demand from woody biomass powerplants (see **Figure 8**). It is worth noting that the self-sufficiency rate increased in 2017 and decreased in 2018. The increase in the volume of chips supplied to woody biomass powerplants from 2017 to 2018 was 0.23 million cubic meters from domestic sources, while 1.05 million cubic meters were imported. Thus, the demand for domestic fuel decreased while that for imported fuel accelerated.

While poor quality limits their utilization in construction, damaged trees can be used in woody biomass plants. For example, in the mid-1980s, pulp and chips represented between 73.3% and 83.9% of the total utilized volume. The current increase in the number of woody biomass powerplants might be expected to follow in lock step with the increase in utilization of trees damaged by the pine bark beetle.

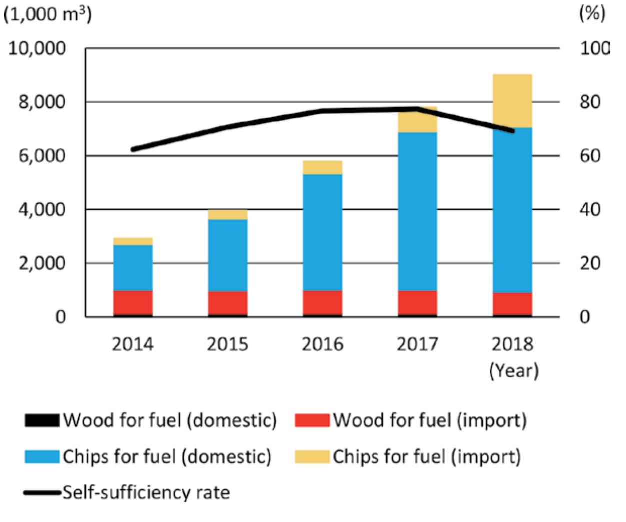

Figure 8.
Consumption and self-sufficiency rate of fuelwood. Source: [2]. Note: The vertical axis on the left side is fuelwood consumption and the vertical axis on the right side is the self-sufficiency rate.

However, there are three reasons why the increase in demand for fuelwood may not result in an increase in the use of damaged trees.

The first is the price of fuelwood used by woody biomass powerplants. The standard price is set by the feed-in tariff (FIT) system for renewable energy. To obtain the highest fuelwood price, logs should be supplied from forests with FMPs developed by forest owners in non-national forests. This plan was introduced with the amendment of the Forest Act in 2011, and the planning period is 5 years. A notable characteristic of the plan is that timber production should be planned in coordination with subsidized forest road or spur road construction. Although it is possible to design an FMP for forests damaged by the pine bark beetle, in practice, the plan is geared towards Japanese cedar and Japanese cypress plantations. For most FMPs, the objective is thinning associated with subsidized road construction, and forests where thinning has occurred will be expected to conduct final cutting thereafter. Namely, it is assumed that the forest with normal growth; very few FMPs for pine forest include preparation by forest owners for the possibility of future damage by the pine bark beetle. Moreover, many forest owners cease pine forest management in the face of continuing long-term pine bark beetle damage. At the end of fiscal year 2014 and 2018, 28% and 29%, respectively, of all non-national forests had FMPs [2, 32].[21] The area covered by FMPs has not increased recently.

The original policy motivation for promoting woody biomass powerplants with the FIT system was to make use of unutilized domestic wood resources. There are many types of unutilized wood resources,[22] such as thinned trees that are left in the forest, construction waste materials from the demolition of old wooden structures, residue from sawmills, remnants left behind after removing logs from forests, and damaged trees.

In the case of trees damaged by pests and diseases, the price of logs for fuelwood should be given special treatment. A new financial distribution system from the central government to municipal governments started in April 2019, and the use of the funds as regards forests, forestry, and wood should be determined. For example, municipal governments may add grants to the price determined by the FIT system. This type of pricing policy should be considered to increase the use of damaged trees.

The second cause is a problem common in the Japanese forestry sector, which is the shortage and aging of forestry laborers. Given the limited labor availability, forestry companies and FOAs prioritize forest practices linked to the production of timber or that are more profitable. Otsuka [33] analyzed the problems surrounding the use of trees damaged by pine bark beetle at the biomass powerplant in Iwate Prefecture. One problem is that tree felling is a countermeasure to the damage and that the practice is not based on the premise that the felled trees should be taken out. Another problem is that supply of felled trees due to damage is unstable because the felling period is limited. Hayashi et al. [34] analyzed the use of damaged trees in a paper and wood-pellet company in Yamagata Prefecture and pointed out that there is a benefit for wood companies to get damaged materials at a cheaper price compared to healthy trees[23]. At the same time, the instability of the volume of damaged trees is a problem.

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21 There are no statistics on the relationship between FMPs and plantation tree species.

22 In 2018, utilized residues from sawmills and construction waste materials were extremely high at 98% and 96%, respectively, while forest residue utilization was only 24% in 2017. The low utilization rate is a problem in all forests, including those damaged by pine bark beetle [2].

23 Hayashi *et al.* [34] pointed out that the increased utilization of pine bark-damaged trees by woody biomass powerplants would make business sense since they are cheaper. Although there is a possibility to

In these two recent analyses, supply stability was commonly pointed out as being a problem. As shown in **Figure 1**, annual changes in the volume damaged are not large at the national level, however, they tend to be large at the prefectural level because the damaged area shifts from year to year. The damaged volume should decrease in the future due to various countermeasures. Because moving damaged trees long distances is undesirable, it is difficult to collect trees from a wide area, for example, across several prefectures. This must be considered in analyzing the effective utilization of damaged trees.

From the discussion above in Section 4.2, the mismatch between the large volume of damage and the potential volume for fuelwood and chips should be pointed out. Utilizing damaged logs, including those damaged by the pine bark beetle, should be taken into consideration in policy development.

In the current forest planning system designated in the Forest Act, the utilization of damaged trees is not clearly specified. This needs to be done in the national government's National Forest Plan and the prefectural Regional Forest Plans[24]. Furthermore, the Forest Pest Control Act of 1950 focused on extermination of damaged trees and preventing damage,[25] but not utilization of damaged trees. One reason for this problem seems to be the different administrative systems for dealing with timber production and forest protection within the Forestry Agency and the Department of Forestry in prefectural governments. The problem, pointed out by GHQ and discussed in Section 4.1, is related to the lack of a coordinated program.

4.3 Forest practice

Recommendations by Furniss to GHQ in 1950 and 1951 included improvement of forest practice methods for dealing with forest pests. Legal systems, such as the Forest Pest Control Act of 1950 and the Act on Special Measures Concerning the Control of Pine Bark Beetle of 1977, did not include how to conduct forest practices. The basic contents of these legal systems were measures related to damage. As shown in Section 4.2, the post-war forest planning system in the Forest Act of 1951 did not consider the use of damaged trees, and the FMP system deals with normal forest management. The GHQ recommendations have not been implemented in postwar legislation relating to forest pest control.

4.3.1 Forest management plan

In the GHQ recommendations, the necessity of a better FMP was pointed out, but an FMP system for forest owners was not introduced in the Forest Act of 1951. An FMP system was introduced in the amendment to the Forest Act of 1939 [24], but it did not work well due to wartime timber production. In the first post-war forest planning system, a 5-year national plan was made by the Minister of the Ministry of Agriculture and Forestry and a 5-year prefectural plan was made by prefectural governors. Under this system, central and local governments had an

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use the damaged trees, expansion of damage by the pine bark beetle is not acceptable. It was pointed out that a decrease in fuelwood supplied from damaged trees is desirable.

[24] The great increase in timber demand after the Second World War demanded the utilization of trees damaged by pine bark beetles [6]. After the second peak, the transport of untreated damaged trees to sawmills, pulp factories, and building sites widened the spread of the damaged area [8]. Thus, the movement of damaged trees should be limited to smaller areas, such as the sub-prefecture scale area covered by the Regional Forest Plan, which appears to be appropriate for limiting beetle spread.

[25] Kobayashi [35] proposed using damaged trees as a preventative measure. Use of damaged trees is not included in the Forest Pest Control Act.

obligation to make forest plans.[26] Ultimately, the FMP referred to in the recommendation[27] by Furniss as an indirectmeasure for pest control was not introduced.

In the current forest administration and forest planning system, the increase in forests with unknown owners and unknown boundaries has become a problem[28]. The recommendation of the FMP as an indirect measure to control forest pests assumes that forest owners know the contents of their holdings. However, this is oftentimes not the case, and sometimes forest owners do not know that they have inherited forest. As a result, they cannot make FMPs by themselves.[29] One of the reasons forest owners do not have sufficient knowledge of their holdings was the abolition of the FMP system by forest owners under the Forest Act of 1939 amendment and the introduction of the forest planning system primarily by central and local governments in the Forest Act of 1951. Even in such an administration-led forest planning system, in the 1950s, the percentage of fuelwood in the total cutting volume was high (see **Figure 5**) and forest owners seemed to have a good understanding of their holdings. Such a relationship has now disappeared in most areas. Since most forest owners have no relationship to their holdings and rarely go into their forests, they cannot find beetle damage.

4.3.2 Cutting methods

Furniss' first recommendation was that damaged trees should be utilized as much as possible. Furniss [28] recommended the "Selective removal of infested trees only[30]". Almost at the same time, a recommendation on private coniferous forests by Kircher and Dexter pointed out the necessity of a better FMP, the elimination of clear-cutting on steep slopes, and the establishment of partial cutting methods.[31] The recommendation required the cessation of clear-cutting, especially on steep slopes. Furniss and Kircher and Dexter did not recommend clear-cutting. One reason for this was the existence of large areas of denuded forest (roughly 1.72 million hectares at the end of fiscal 1948) at the end of the Second World War [39]. This situation resulted in extensive damage resulting from natural disasters associated with the denudation, such as landslides.

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[26] Please see [36] on the forest planning system. As regards the current system, please see the webpage (in Japanese) of the Forestry Agency: https://www.rinya.maff.go.jp/j/keikaku/sinrin_keikaku/pdf/taikeizu24.pdf, 2020/09/12

[27] Furniss [28] recommended the following: "Research should be undertaken to develop methods for the lasting control of bark beetles and other forest insects through improved forest management practices and thereby reduce the necessity for direct control expenditure. Early implementation of the forest management plan program is a logical first step towards attaining indirect control of bark beetles through improving the vigor of the host trees."

[28] This problem relates to the aging of forest owners [37].

[29] Under the current Forest Act, forest owners who own over 100 ha can make an FMP by themselves. However, forest owners who own less than 100 ha must band together to achieve the required area. Many forest owners cannot make a FMP by themselves.

[30] According to a summary of the first recommendation by Furniss [28], "The present practice of felling the infested trees and peeling and burning the bark of infested trees [should] be retained as the principal direct control measure, but the practice of clear-cutting entire stands for bark beetle control [should] be discontinued. Selective removal of infested trees only is recommended."

[31] Kircher and Dexter [38] recommended two systems for the management of coniferous forests: "(1) a shelterwood system or (2) a partial cutting system which will build up a several- or many-aged stand."

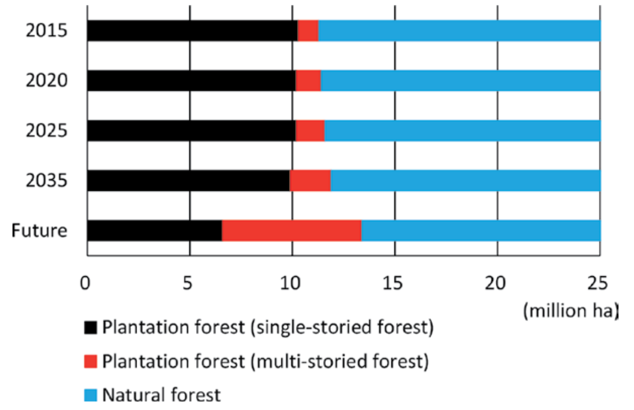

Figure 9.
Current and future forest areas based on the 2016 Basic Plan on Forest and Forestry. Source: [32]. Note: The total forest area for all years, including "future," is 25.1 million hectares. "Future" indicates goal-directed forest projections.

Nonetheless, selective cutting has developed only to a limited degree in specific areas since the end of the war, with clear-cutting remaining the dominant method in Japan. Thus, it follows that selective cutting has not been extensively adopted as a method for managing trees damaged by the pine bark beetle.

The Basic Plan for Forest and Forestry, based on Article 11 of the Forest and Forestry Basic Act (Act No. 161 of 1964), which is revised roughly every 5 years, determines the forest area classified by forest type since 2001. The current Basic Plan was published in May 2016, and the areas classified as single-storied plantation forest, multi-storied plantation forest, and natural forest are shown in **Figure 9**. The basic idea of future forest management is to increase multi-storied plantation forests. In "future," which shows the final ideal situation, the area of single-storied plantation forests is 6.6 million hectares and the area of multi-storied plantation forests is 6.8 million hectares. Methods other than clear-cutting will be necessary to change single-storied plantation forests and natural forests to multi-storied forests. An increase in multi-storied plantation forests has been delayed because of the long-standing use of clear-cutting. However, if the basic idea is realized and forest practice techniques other than clear-cutting are developed, there is the possibility that damaged trees can be cut selectively and removed for use to some degree.

Selective cutting is most viable in the early stages of damage, after which point widespread damage makes it difficult. In the act of 1982, tree species replacement was added to the list of measures against pine bark beetle damage. Considering that damaged areas influence surrounding areas over the long-term, the basic idea that damaged trees should be cut, removed, and used to the greatest extent possible is still important.

5. Conclusion

In light of the failure of the Japanese government to contain the spread of the pine bark beetle, this chapter explored the legal aspects related to its management. The first and second peaks in the volume of damaged trees were recorded in 1948 and 1978, with the second peak being larger than the first. Furniss' two recommendations on forest pest control to GHQ-occupied Japan at the end of the Second World War were followed by new legislation. In the Act on the Special Measures of

1982, the tree species replacement method was added to the list of countermeasures against the damage.

In analyzing the legal measures taken to combat pine bark beetle damage from 1897 to date, several important points emerge.

The first is the lack of a coordinated program. This was pointed out by GHQ not only for pest control policies but also for other forest and forest-related policies. Cooperation between national forests, administrated by the Forestry Agency, and non-national forests, administrated by the Departments of Forestry for each prefectural government, is lacking. Cooperation between local governments and forest owners is also lacking. Accurate statistics[32] and reporting systems resulting from coordination amongst administrative sectors, FOAs, and forest owners are needed.

The second point is that the use of damaged trees, as recommended by GHQ, has not increased. Utilization of damaged trees as fuelwood for woody biomass powerplants and as chips for paper companies has been expected to increase, but these projected increases have not yet been realized, partly due to cheaper prices for fuelwood from damaged forests.

The third point is that countermeasures against pine bark beetle damage do not include FMPs. There is no linkage between pest control and the forest planning system. Despite GHQ pointing out the necessity of making better FMPs and the cessation of clear-cutting, both have gone unrealized.

The failure of the Japanese government to heed these recommendations has resulted in pine no longer being the dominant plantation species in Japan. Young trees are almost non-existent. Not only is pine important for the scenic beauty of forests and Japanese gardens, it is also a necessary material in traditional Japanese construction.

The volume of damage has decreased by almost 80% compared to the volume of damage at the time of the second peak, however, it remains large and the damaged area is expanding every year. Both legal and technical developments are needed to stop this expansion.

Japan recently implemented new forest policies. In fiscal year 2013, the financial system for national forest management changed from a special accounting system to a general accounting system. In 2019, the Forest Environmental Tax began generating new forestry funds that are distributed from the central government to municipal governments. The recent Basic Plan for Forest and Forestry showed a shift towards multi-storied plantation forests for future forest management. These recent changes, however, have not influenced legal policy changes with respect to pine bark beetle damage. Post-war GHQ recommendations are still valuable today and could play an important role in constructing forestry policies to combat pests.

Acknowledgements

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[32] Accurate statistics on forest damage are necessary. In the forest resource database for non-national forests managed by the Department of Forestry in each prefectural government, called "*shinrinbo*," volume has generally been calculated using the fixed yield table. In the areas classified by species, damaged and non-damaged forests are not distinguished. As a result, the area of pine forest has not decreased in spite of the spread of damage. For example, Futai [40] pointed out that the areas of pine forest in 1980 and 1990 in Kyoto Prefecture were almost equal, despite the large volume damaged.

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References

[1] Forestry Agency, editor. Shinrin ringyo tokei yoran, 2019 (Annual statistics on forests and forestry, 2019). Tokyo: Nihon Shinrin Ringy Shinko Kyokai; 2019. 262 p. (in Japanese)

[2] Forestry Agency, editor. Shinrin Ringyo Hakusyo, 2020 (Annual report on forests and forestry, 2020). Tokyo: Zenkoku Ringyo Kairyo Fukyu Kyokai; 2020. 279 p. (in Japanese)

[3] Shinrin Byogaichu Jugai Bojo Kyokai, editor. Shinrin boeki seidoshi (History of the system on forest pest control). Tokyo: Zenkoku Shinrin Byogaichu Jugai Bojo Kyokai; 1978. 277 p. (in Japanese)

[4] Yano M. Nagasaki-kenka matsu-jyu koshi gennin tyosa (Survey on the cause of death of pine trees in Nagasaki Prefecture). Sanrin Koho. 1913;4: Appendix 1-14. (in Japanese)

[5] Forestry Agency, supervising editor. Matsukuimushi higai taisaku seido no kaisetsu (System on the measures for the damage by pine bark beetle). Tokyo: Zenkoku Shinrin Byochujyugai Boujyo Kyokai; 1988. 391 p. (in Japanese)

[6] Ministry of Agriculture and Forestry, editor. Norin Gyoseishi, Vol. 8 (Administrative history of Agriculture and Forestry, Vol. 8). Tokyo: Ministry of Agriculture and Forestry; 1972. 1120 p. (in Japanese)

[7] Shinrin Byogaichutou Boujyo Kenkyukai, editor. Matsukuimushi higai taisaku no tebiki (Companion on the measures for the damage by pine bark beetle). Tokyo: Zenkoku Shinrin Byochujyugai Boujyo Kyokai; 1989. 391 p. (in Japanese)

[8] Yamane A. History of coping with pine beetles and its damage. Forest Economy. 1982;35(6):1-12. DOI: 10.19013/rinrin.35.6_1 (in Japanese)

[9] GHQ/SCAP (General Headquarters of the Allied Forces, Supreme Commander for the Allied Powers). Bark beetle epidemic in Japan. Natural Resources Section, Report, No. 90. Tokyo: GHQ/SCAP; 1947.

[10] GHQ/SCAP (General Headquarters of the Allied Forces, Supreme Commander for the Allied Powers). Forestry in Japan 1945-51. Natural Resources Section, Report, No. 153. Tokyo: GHQ/SCAP; 1951.

[11] Forestry Agency, editor. Ringyo tokei yoran, 1953 (Annual statistics on forestry, 1953). Tokyo: Rinya Kyosaikai; 1953. 132 p. (in Japanese)

[12] Forestry Agency, editor. Ringyo tokei yoran, jikeiretsuban (Annual statistics on forestry, time-series edition). Tokyo: Rinya Kosaikai; 1982. 133 p. (in Japanese)

[13] Forestry Agency, editor. Shinrin ringyo tokei yoran, jikeiretsuban (Annual statistics on forests and forestry, time-series edition). Tokyo: Rinya Kosaikai; 2005. 148 p. (in Japanese)

[14] Forestry Agency, editor. Shinrin ringyo tokei yoran, 2003-2018 (Annual statistics on forests and forestry, every yearly edition, 2003-2018), Tokyo: Nihon Shinrin Ringy Shinko Kyokai; 2003-2018. (in Japanese)

[15] GHQ/SCAP (General Headquarters of the Allied Forces, Supreme Commander for the Allied Powers). Forestry. History of the nonmilitary activities of the occupation of Japan. September 1945-January 1951. Tokyo: GHQ/SCAP; N.D.

[16] Forestry Agency, editor. Ringyo tokei yoran, ruinenban (Annual statistics on forestry, time-series edition). Tokyo: Rinya Kyosaikai; 1964. 139 p. (in Japanese)

[17] Forestry Agency. Current state of forest resources (as of March 31, 2017). Available from: https://www.rinya.maff.go.jp/j/keikaku/genkyou/h29/ [Accessed: 2020-08-25] (in Japanese)

[18] Hagino T. Kindai nihon rinsei no kiso kouzou (Basic Structure of the forest and forestry policy in modern Japan). Tokyo: Nihon Ringyo Chosakai; 1984. 225 p. (in Japanese)

[19] Shimada K. Rinseigaku Gaiyo (Forest Policy). Tokyo: Chikyu Shuppan; 1948. 326 p. (in Japanese)

[20] Zenkoku Shinrin Kumiai Rengokai, editor. Shinrin kumiai seidoshi, Volume III (History of the system of the forest owners' association, Volume III). Tokyo: Zenkoku Shinrin Kumiai Rengokai; 1973. 961 p. (in Japanese)

[21] Shioya T. Rinseigaku (Forest and Forestry Policy). 2nd ed. Tokyo: Chikyusha; 1973. 370 p. (in Japanese)

[22] Kanpo (Official Gazette), 2 July 1908 (in Japanese)

[23] Zenkoku Shinrin Kumiai Rengokai, editor. Shinrin kumiai seidoshi, Volume I (History of the system of the forest owners' association, Volume I). Tokyo: Zenkoku Shinrin Kumiai Rengokai; 1973. 723 p. (in Japanese)

[24] Yamamoto N. The origin of the forest planning system in Japan. Journal of the Japanese Forest Society. 2020;102(1):24-30. DOI: 10.4005/jjfs.102.24 (in Japanese with English summary)

[25] Akiyama T. Kokuyurin keieishi ron (History of national forest management). Tokyo: Nihon Ringyo Chosakai; 1960. 410 p. (in Japanese)

[26] Forestry Agency, editor. Ringyo tokei yoran, 1955 (Annual statistics on forestry, 1955). Tokyo: Rinya Kyosaikai; 1955. 190 p. (in Japanese)

[27] Sakamura K. Matsukuimushi-tou kujo yobou houan nitsuite (On the bill of the Extermination and Prevention Pine Bark Beetle and Other Similar Borers and Other Destructive Forest Pests and Diseases). Nourin Jihou. 1950; 9(4):13-16. (in Japanese)

[28] Furniss R L. Forestry insect control in Japan. Natural Resources Section, Preliminary Study, No. 45. Tokyo: GHQ/SCAP; 1951.

[29] Ozawa Y. Matsugare taisaku to chiiki no shigen junkan: Iwateken gyosei no torikumi yori (Measures for dead pine trees and resource cycle: efforts by Iwate Prefecture). In: Nakamura K, Otsuka I, editors. Shirnin hogo to ringyo no bijinesuka (Forest protection and making a business of forestry). Tokyo: Nihon Ringyo Chosakai; 2019. p. 179-188. (in Japanese)

[30] Nakano H. Iwayuru Furniss kankoku (Recommendation by Furniss). Ringyo Gijutsu. 1950;104:4-11. (in Japanese)

[31] Matsuyama S. GHQ jidai no Matsukuimushi boujo (Control of pine bark beetle during the GHQ occupation). Shokubutsu Boeki. 1966; 20(1):45. (in Japanese)

[32] MAFF (Ministry of Agriculture, Forestry, and Fisheries). Shinrin ringyo kihon keikaku (Basic Plan on Forest and Forestry). 2016. 37 p. Available from: https://www.rinya.maff.go.jp/j/kikaku/plan/attach/pdf/index-2.pdf [Accessed: 2020-11-7] (in Japanese)

[33] Otsuka I. Matsu higaiboku no nenryou riyou to chiiki no ringyo saisei (Usage of the damaged pine trees for fuelwood and renewal of the local forestry). In: Nakamura K, Otsuka I, editors. Shirnin hogo to ringyo no bijinesuka (Forest protection and making a business of forestry). Tokyo: Nihon Ringyo Chosakai; 2019. p. 151-164. (in Japanese)

[34] Hayashi M, Kobayashi S. Matsu higaiboku no chipu muke riyou: yamagataken syounai chihou wo jirei toshite (Use of damaged pine trees for chips: the case of Shounai, Yamagata Prefecture). In: Nakamura K, Otsuka I, editors. Shirnin hogo to ringyo no bijinesuka (Forest protection and making a business of forestry). Tokyo: Nihon Ringyo Chosakai; 2019. p. 165-177. (in Japanese)

[35] Kobayashi F. Matsugare wo fusegu tameni (To prevent the death of pine trees). In: Matsugare Mondai Kenkyukai, editor. Matsu ga kareteiku (Pine trees are dying). Tokyo: Dai-ichi Planning Center; 1981. p. 196-211. (in Japanese)

[36] Matsushita K. Forest planning. In: Iwai Y. Forestry and the forest industry in Japan. Vancouver: UBC Press; 2002. p. 118-144.

[37] Matsushita K, Yoshida Y, Senda T. Household composition and aging of forest owners in Japan. Austrian Journal of Forest Science. 2017;134(S1):101-130. (https://www.forestscience.at/artikel/2017/1a/ueberalterung-der-waldbesitzer-in-japan.html)

[38] Kircher J C, Dexter A K. Management of private coniferous forests of Japan. Natural Resources Section, Preliminary studies, No. 43. Tokyo: GHQ/SCAP; 1951.

[39] Forestry Agency editor. Chisan chisui dokuhon (Readings on water conservation and erosion control). Tokyo: Nihonshuhosya; 1952. 230 p. (in Japanese)

[40] Futai K. Matsugare wa mori no kansenshou (Death of pine trees is a forest infection). Tokyo: Bun-ichi Sougou Shuppan; 2003. 222 p. (in Japanese)

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Silviculture is integral for the perpetuity and sustainability of forest stands and their yields. It encompasses several methods and techniques that make the bridge between individual trees and the stand. This book focuses on sustainable forest management with chapters on such topics as afforestation, thinning, pest control, and mitigation of climate change, among others.

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