Sustainable Forestry

EnPress Publisher

SF

2021
Volume 4
Issue 1
ISSN: 2578-2002

https://systems.enpress-publisher.com/index.php/SF

ISSN 2578-2002

Volume 4 Issue 1 • 2021

Sustainable Forestry

EnPress Publisher

2021 | Volume 4 | Issue 1

Sustainable Forestry

https://systems.enpress-publisher.com/index.php/sf

Contents

Sustainable Forestry (2021) Volume 4 Issue 1
doi:10.24294/sf.v4i1.1597

Review Article

Biodiversity conservation evaluation and planning on the western slope of Cangshan Mountains

Zhi Ou*, Yan Qu

Landscape school, Southwest Forestry University, Kunming 650224, China. E-mail: bigbone@yeah.net

ABSTRACT

Taking the west slope of Cangshan Mountain in Yangbi County, Dali as the research site, on the basis of investigating the local natural geographical conditions, topography and biodiversity status of Cangshan Mountain, the CAP protection action planning method was adopted, and the priority protection objects were determined to be native forest vegetation, rare and endangered flora and fauna, alpine vertical ecosystems, hard-leaf evergreen broad-leaved forests and cold-tempered coniferous forests; The main threat factors were commercial collection, tourism development and overgrazing. Biodiversity conservation on the western slope of Cangshan Mountain should take species as "point", regional boundary as "line", ecosystem and landscape system as "plane", so as to realize the overall planning structure system combining "point—line—plane", which can be divided into conservation core area, buffer zone and experimental area. The results can provide a reference for biodiversity conservation on the western slope of Cangshan Mountain.

Keywords: CAP Protection Action; Protected Object; Threat Factor; Overall Planning Layout; Cangshan Mountain

ARTICLE INFO

Received: 5 January 2021
Accepted: 27 February 2021
Available online: 7 March 2021

1. Introduction

Yunnan is one of the world's top ten biodiversity hotspots and the core region of the Eastern Himalayas, with the largest number of species and endemic groups in China. And northwest Yunnan is one of the areas with the richest biodiversity in China and even in the world, and is also a vulnerable area in Yunnan's ecological environment[1-3]. Cangshan mountain is located at the junction of Dali, Eryuan and Yangbi, in northwest Yunnan province. It is an important part of Dali national key scenic spot and a famous mountain in the middle of Hengduan Mountain range. The western slope of Cangshan Mountain is located in Yangbi River, Yangbi County, and is an important part of Cangshan Erhai National Nature Reserve and scenic spot. Vegetation types are diverse and biodiversity is extremely rich and unique, which is mainly a land ecosystem of forest, grass and irrigation cade broad-leaved forest represented by rhododendron. Throughout the Dali Cangshan-Erhai Nature Reserves and biodiversity in Yunnan province, northwest Yunnan biodiversity conservation domain are in a very important position, but due to the regional economic development lag behind, fragile natural ecological system, to protect the biodiversity in this area, it's of great significance to implement the strategy of sustainable development of Dali prefecture, Yunnan province.

Experts from the United Nations Development Programme (UNDP) estimate that about 200 of the 2,300 species of wild plants on the

Cangshan Seed Plant List are no longer extant, and a large number of plants are endangered or threatened, with at least five species of plants lost each year in recent years[4]. Some studies have investigated the current status of biodiversity in Cangshan Mountain, and found that the area has the characteristics of rich animal and plant species, diverse vegetation types, ancient flora origin, complex geographical composition, and numerous endemic species, which put forward unreasonable economic activities, overtourism development, forest fire hazards and serious biological invasions, which constitute the main threat to biodiversity in the area, and put forward corresponding conservation measures[5,6]. Therefore, it is imperative to implement biodiversity conservation planning in the region, especially the determination of priority conservation objects and threat factors, and the overall planning and layout.

To sum up, in order to scientifically and reasonably protect the biodiversity of western slope of Cangshan mountain, this paper covers the scope of biodiversity conservation planning within the western slope of Cangshan Mountain, Yangbi County, Dali Bai Autonomous Prefecture, with a protected area of 20,610 hm^2. On the basis of fully understanding and controlling the present situation of biological diversity in Cangshan reserve, it is necessary to determine the priority protection objects, protected areas and main threat factors, and to put forward the reasonable layout structure system and zoning of biodiversity protection planning, so as to provide reference for further carrying out biodiversity protection in this area.

2. Research methods

There are two ways to prepare biodiversity conservation planning: species-centered protection planning approach; the former emphasizes the protection of endangered species itself, while the latter emphasizes the overall protection of scenic landscape system and natural land, and tries to realize the protection of biodiversity by protecting the landscape diversity of scenic spots[7]. Based on the two methods, scholars combine the spatial hierarchy and spatial location and pattern of biodiversity, and scientifically detail the spatial hierarchy relationship of biodiversity, and put forward the biological diversity protection approaches and measures suitable for Fuzhou Gushan scenic spots[8]. Some scholars have also carried out relevant protection of ecological diversity in Qilian Mountain from the perspective of landscape diversity, ecosystem diversity and species diversity research[9]. Other scholars have applied the principle of landscape ecology to urban biodiversity protection, pointing out that urban green space system structure and urban green space control planning are the main ways to realize urban biodiversity[10].

This plan adopts the Conservation Action Planning (CAP), which is a logical regional protection strategy developed by scientists of the Nature Conservancy (TNC) and other international environmental protection after long-term practice[11]. This method is widely used in the resource management of nature reserves and other types of protected areas. It is a protection planning method that follows the project management process and helps protect workers to focus on key protection objects and key threat factors. CAP mainly includes 4 processes: (1) determine priority protection objects according to the importance; (2) threat factor analysis of identified protection objects; (3) develop protection strategies for protection objects to improve their living conditions and reduce threats; (4) In the process of protection, a dynamic effect evaluation is carried out, and on the basis of the evaluation, each link of the whole process is adaptively adjusted. It's an adaptive management framework[12-14]. CAP was introduced into China by TNC around 2000. The principles and methods of CAP were adopted in its northwestern Yunnan project area (including Laojun Mountain in Lijiang), and in cooperation with relevant protected areas, the Conservation action plan for more than 10 protected areas or national parks, including Shangri-La Grand Canyon, Gaoligong Mountain, Beijing Songshan Mountain, Shanghai Chongming Dongtan Bird National Nature Reserve, and the Rare and Endemic Fish Reserve in the upper reaches of the Yangtze River[15,16]. At the same time, some scholars have used this method to protect wetland parks and site parks[17,18]. Among them, some scholars use the analysis and evaluation of threat factors to guide the planning of urban plant diversity construction[19].

The specific methods of this study are divided into two stages: early data collection (mainly including literature review) and field investigation. After sorting out the literature on the location of western slope, biodiversity distribution, endangered species habitat, and then conducting a field survey of western mountain, the investigation group was composed of 10 college students with biological diversity protection experience (mainly biological diversity protection, plant taxonomy, animal taxonomy, population ecology). After that, experts were rated by 15 experts (including 10 survey members, and 5 permanent work and management personnel in the protected areas, the same below) based on the information collected above and the annual work experience in biodiversity protection. It is mainly to sort out the threat degree of each protection object, filter out clear priority protection targets and main threat factors, so as to provide reference for the formulation of partition and protection strategies for further protection planning.

3. Results and analysis

3.1 Nature of the reserve

Western slope of Cangshan Mountain is a representative, typical and complete biological community and non-biological environment of central Asia, and has a distinct vertical distribution band spectrum (from the foothill to the top of the mountain has three vertical climatic zones, including subtropical, warm temperate and cold temperate). At the same time, there are many various vegetation types that transition from south Asian to alpine ice desert belt[20]. The special geographical location, geological geomorphology and climatic conditions make the western slope of Cangshan become the intersection and transition zone of north and south biology, and the distribution and differentiation center of various organisms is one of the regions with rich biodiversity. It can be used as the basic law and environmental monitoring site of natural processes in the north and south of the country, as well as the base of natural resources, genetic genes, and breeding of precious and rare plants and animals. Therefore, according to the location environment, natural environment and resource environment of the Cangshan Mountain Slope Nature Reserve, combined with the needs of the social environment, the nature of the nature is determined as: to protect the diverse ecosystems and rare animals and plants, and to maintain the naturalness, biodiversity, typicality, integrity and maintenance of ecological balance in the area for the purpose of the purpose, set species protection, water source protection, ecological protection, scientific research, science education, eco-tourism and other multi-functional integrated nature reserve.

3.2 Establishment of the object of protection

According to CAP's approach, the preservation of species representativeness and persistence is the two main objectives of protected areas[21]. Considering the importance and the process of being threatened, the principle of protecting the integrity and stability of the ecosystem in the region is to be taken as the criterion for selection, and the selection criteria are the severity of the degree of threat, whether it is the habitat or growth site of rare and endangered animals and plants, and the unique biodiversity landscape characteristics representing the western slope of Cangshan Mountain. The object of protection is established from three levels: species, ecosystem and vegetation.

3.2.1 Native forest vegetation

Focus on the protection of native forest vegetation, including secondary forests, shrubs and secondary bare land, which are the stages of vegetation succession and must be strictly protected. According to the survey, the natural vegetation below 2,500 m above sea level on the western slope has suffered from different degrees of damage. The semi-humid evergreen broad-leaved forest on the western slope is also preserved in small areas, and the large areas are distributed in *Pinus yunnanensis* forests that are closely related to its succession. Locally damaged areas, warm and warm shrublands or savanna slopes.

Native forest vegetation is the most precious natural heritage and gene bank of the Cangshan west slope biodiversity reserve, which contains rich germplasm resources and is also an indispensable material foundation for gardens, forestry, agriculture, etc. Effective protection of it is one of the important tasks of biodiversity conservation, and the use of local protection is an effective conservation measure, eliminating the bare land and barren mountains in the protected area, and gradually restoring the vegetation with zonal native forest belt structure, so that the forest ecosystem has entered a virtuous circle, and the biological reproduction and rest areas have been significantly improved. At the same time, special attention is paid to the construction of shrubs and ground cover plants, and the function of soil conservation and slope protection in the protected area is strengthened to avoid and reduce soil erosion.

3.2.2 Alpine vertical ecosystem

Alpine vertical ecosystem refers to alpine shrubland, alpine meadows, and alpine flowing rocky beaches distributed at an altitude of more than 4,000 m. It represents the unique biodiversity landscape characteristics of the western slope of Cangshan Mountain, and is the most unique area of the western slope of Cangshan Mountain. The western slope of Cangshan Mountain belongs to the alpine mountainous area, and several peaks in the main peak area are above 4,000 m above sea level, which has the conditions of the limit height of the altitude distribution of native forest tree species. Therefore, the vertical differentiation of mountain vegetation zones is obvious and relatively complete. With the continuous increase of altitude, the coniferous forest gradually transitions from warm and temperate coniferous forests in the horizontal zone (*Pinus yunnanensis*, *Pinus armandii*, Yunnan *Keteleeria fortunei* forest, etc.) to warm coniferous forest (hemlock coniferous and broad-leaved mixed forest) and cold temperate coniferous forest (*Abies* forest), and the broad-leaved forest gradually transitions from the semi-humid evergreen broad- leaved forest in the horizontal zone (*Castanopsis delavayi*, *Castanopsis orthacantha*, *Lithocarpus dealbatus*, *Cyclobalanopsis glaucoides*, *Cyclobalanopsis delavayi* forest, etc.) to the wet green broad-leaved forest in the Zhongshan zone (dominated by *Lithocarpus variolosus*) and the cold warm hard-leaved evergreen broad-leaved forest in the subalpine area (dominated by *Quercus guyavifolia*). Until the tree species disappears, native alpine rhododendron shrubs or alpine meadows appear.

3.2.3 Forest ecosystems

The destruction of ecosystem diversity will lead to the loss of species diversity and genetic diversity within the system, causing many of these organisms to lose their habitat and disappear, so they are listed as protected objects. The main vegetation on the western slope of Cangshan was initially divided into 9 vegetation types, 13 vegetation subtypes, and 21 biomes[22]. The vertical band spectrum of the western slope includes: semi-humid evergreen broad-leaved forest, Yunnan pine forest belt (altitude 2,000–2,900 m), the main vegetation types are *Cyclobalanopsis delavayi* forest, *Pinus yunnanensis* forest, *Rhododendron* shrubland, *Arundinella setosa* grass, *Fiddlehead* grass, etc.; *Tsuga* forest, montane moist evergreen broad-leaved forest (altitude 2,950–3,350 m), the main vegetation types are *Rhododendron* shrubland, *Tsuga* forest, *Fargesia* forest and *Lithocarpus variolosus* forest; *Abies* forest belt (altitude 3,350–3,560 m), the main vegetation types are *Abies*, Quercus semecarpifolia forest, *Rhododendron* shrubland, *Fargesia* forest, etc.; *Rhododendron* shrubland (altitude 3,560–3,700 m), the main vegetation types are *Rhododendron taliense* shrubland, *Rhododendron haematodes* shrubland; Alpine meadow belt (altitude 3,700–4,100 m), the main vegetation types are *Kobresia royleana* meadows and *Festuca* meadows[20,22].

Among the main vegetation types on the western slope of Cangshan Mountain, the hard-leaf evergreen broad-leaved forest in the subalpine zone has become a germinating shrub due to human destruction, and is distributed on the sunny slopes with poor standing conditions. The level of threat is the most severe. Cold temperate coniferous forests have a certain area distribution in the upper part of the mountain. Due to the influence of geographical location and topographic conditions, there is only one species of *Abies delavayi* in the *Abies* and *Picea*, and no positive tree species common after such deforestation in northwest Yunnan—*Larix potaninii* were found in the survey. Cold and temperate coniferous forests are concentrated in the main peak area of the eastern slope of Cangshan Mountain, because the peak wind is strong and the climate is harsh, which is replaced by *Sabina recurva* that are more drought-resistant and wind-resistant, forming a small sparse forest. *Abies delavay* after a long period of natural succession, at present, is in a state of apical community, forming a relatively stable forest type. Because it is located at a high altitude, the proportion of forest stand in the middle and over-mature forests is large, resulting in serious disease and rot, which poses a certain threat to the forest ecosystem[20]. Other vegetation on the western slope of Cangshan Mountain, such as the semi-humid evergreen broad-leaved forest belt, moist evergreen broad-leaved forest in Zhongshan belt, and mixed coniferous and broad-leaved forest of hemlock in the upper part of Zhongshan Mountain, have a certain area preserved on the west slope. Although there are also different degrees of man-made damage, the degree of threat and impact is relatively small compared to the former two.

Therefore, due to the great degree of threat of hard-leaved evergreen broad-leaved forests and cold-temperate coniferous forests, it directly affects the integrity of the forest ecosystem on the western slopes of Cangshan Mountain, and it is listed as a key protection object.

3.2.4 Rare and endangered protected plants

The flora of the western slope of Cangshan belongs to the pan-Arctic alpine flora and the Flora of the Chinese Himalayas, and belongs to one of the rich alpine floras in the world. The superior natural environment and special geographical location have bred a rich plant population and a variety of vegetation types, because it is a low-latitude subtropical region of the high mountains, the natural environment is complex, the species density is large, there are more rare and endangered plant species.

Cangshan is included in the List of Rare and Endangered Plants of China with 21 species[22,23], and in the survey, it was found that there are 12 species on the western slope of Cangshan Mountain, belonging to 11 families and 12 genera, accounting for 0% of the total number of floras in Cangshan 0.47%. The western slope of Cangshan Mountain is listed in the List of National Key Protected Wild Plants (First Batch)[24] with 1 species of fern and 7 species of angiosperms, and 1 species of national key protection and 7 species of second-level key protected plants (**Table 1**). According to the known plants list, there are 47 species of orchids on the western slopes of Cangshan, all of which are protected species under the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES)[25].

Table 1. Wild plants under national key protection on the western slopes of Cangshan Mountain and their distribution

Species name	Section	Distribution	Protection level
Taxus yunnanensis	Yew family	Altitude 2,000–3,500 m, wet evergreen broad-leaved forest in Zhongshan, hemlock forest	One
Tricholoma matsutake	White mushroom family	Evergreen broad-leaved forest above 2,500 m above sea level	Two
Toona ciliata	Meliaceae	West Slope	Two
Craigia yunnanensis	Linden family	The altitude is around 2,200 m	Two
Skapanthus oreophilus	Lamiaceae	Altitude 2,700–3,100 m, Yunnan pine forest	Two
Tetracentron sinense	Aquacarinaceae	Altitude 2,700–2,900 m, wet evergreen broad-leaved forest in Zhongshan	Two
Neocheiropteris palmatopedata	Hydrogalaceae	2,000 m above sea level, semi-humid evergreen broad-leaved forest	Two
Fagopyrum dibotrys	Polygonaceae	The altitude is 2,200 m, in the forest clearing	Two
Magnolia wilsonii	Magnoliaceae	2,700 m above sea level, evergreen broad-leaved forest	Two

Rare and endangered plants on the western slope of Cangshan mountain are characterized by a wide distribution, and in terms of vertical distribution, the survey found that there are rare and endangered plants in the range from the foot of the mountain to 4,000 m below 4,000 m. In terms of its number, because the western slope of Cangshan is listed as a national nature reserve, coupled with the high mountainous terrain of the western slope itself; Species above 3,000 m are well protected, and the population number has a certain scale, but the species with higher economic value in the left and right low-altitude areas of the western slope of 2,000 m, such as *Dendrobium candidum*, *Toona ciliata*, etc. have decreased sharply due to human over-mining, such as black knotweed has not been seen for many years, and some species distributed at low altitudes such as *Leucomeris decora*, collar spring wood (*Eupelea pleiosperum*) *et al.* are also decreasing year by year due to the serious destruction of habitats and the difficulty of population renewal.

There are about 433 species of animals in the West Slope Reserve, of which 285 are higher animals (82 species of mammals, 170 species of birds, 33 species of fish) and 148 species of lower animals. There are 26 species of national first- and second-level protected animals (5 species of first-class protected animals and 21 second-level protected animals), which belong to 16 families (**Table 2**). Amphibians, reptiles and insects were not surveyed. Due to the widespread indiscriminate hunting of animals in the past, tigers, leopards, red deer and other mammals that once inhabited the Cangshan Mountains are now difficult to find, and white-bellied golden pheasants, pangolins, and red pandas have been hunted and killed in large quantities in the past, and now the number is also rare.

Table 2. Cangshan national protected rare and endangered animal species

Species name	Section name	Protection level	Species name	Section name	Protection level
Panthera pardus	Feloidea Pantherinae	One	Pseudois nayaur	Bovidae	Two
Neofelis nebulosa	Feloidea Pantherinae	One	Capricornis sumat raensis	Bovidae	Two
Macaca assamensis	Cercopithecidae	One	Cervus unicolor	Cervoidea	Two
Budorcas taxicolor	Bovidae Caprinae	One	Manis pentadactyla	Lepidoptera	Two
Syrmaticus humiae	Phasianidae	One	Ratufa bicolor	Oriental giant squirrel	Two
Ailurus fulgens	Procyonidae	Two	Felis manul	Feloidea	Two
Felis temmincki	Feloidea Felinae	Two	Psittacula himalayana	Parrotidae	Two
Viverra zibetha	Viverridae	Two	Chrysolophus amherstiae	Phasianidae	Two
Viverricula indica	Viverridae	Two	Elanus caeruleus	Accipitridae Elaninae	Two
Felis chaus	Feloidea Felinae	Two	Falco tinnunculus	Falconidae	Two
Macaca mulatta	Cercopithecidae	Two	Psittacula alexandri	Parrotidae	Two
Ursus thibetanus	Ursidae	Two	Glaucidium brodiei	Strigidae	Two
Naemorhedus goral	Bovidae Caprinae	Two	Schizothorax taliensis	Cyprinidae	Two

3.3 Threat factor analysis

Based on the opinions of experts, protected area staff and management units involved in this planning field survey, the direct threat factors of each conservation object are first determined one by one in accordance with the specific methods of the Conservation Action Planning Manual. The threat factor rating is calculated based on the indicators of the severity, scope of impact, irreversibility, etc., with reference to the rating criteria for direct threat factors in the Protection Action Planning Manual[14].

It is divided into very high (level 4) and high (level 3), medium (level 2), low (level 1) 4 levels. Based on field surveys and years of experience in biodiversity conservation, 15 experts scored and summarized (**Table 3**, **Table 4**). Through the analysis of threat factors, a total of 10 threat factors were proposed, of which commercial collection, over-tourism development, overgrazing, and logging of construction timber were the most threatening.

Table 3. Protected objects key threat factors and levels of threat sorted table

Protect the object	Overall order of threat levels	Threat factor	Degree of compromise
Native forest vegetation	1	Overtourism development	4
		Overgrazing leads to degradation of grasslands	3
		Forest fires	2
		Mining activities	3
		Harvesting of building timber	2
		Illegal mining	2
		Farmers plant economic forests such as walnut forests	1
Alpine vertical ecosystems	3	Overtourism development	3
		Overgrazing	2
		Forest fires	1
		Commercial collection of medicinal plants	3
		Mining activities	1
		Economic forest planting	1
Cold temperate coniferous forests	5	The building consumes a lot of coniferous trees such as fir	4
		Commercial collection of fungi and medicinal plants	2
Sclerophyll evergreen broad-leaved forest	4	Firewood harvesting	4
		Harvesting of building timber	2
		Overgrazing	2
		Forest fires	1
		Commercial collection of non-timber forest products	1
Rare and endangered flora and fauna	2	Overtourism development	3
		Overgrazing	2
		Commercial collection of non-timber forest products	4
		Illegal poaching	3
		Biological invasions cause fragmentation or loss of habitat for species	2
		Farmers plant economic forests such as walnut forests	1
		Forest fires	1

Table 4. Comprehensive analysis of threat factors of protected objects

Threat factor	Native forest vegetation	Alpine vertical ecosystem	Cold temperate coniferous forest	Sclerophyll evergreen broadleaf forest	Rare and endangered flora and fauna	Worth	Threat sorting
Overtourism development	4	3	-	-	3	10	1
Overgrazing	3	2	-	2	2	9	2
Forest fires	2	1	-	1	1	5	4
Mining activities	3	1	-	-	-	4	5
Harvesting of building timber	2	-	4	2	-	8	3
Fuelwood collection		-	-	4	-	4	5
Illegal poaching and poaching	2	-	-	-	3	5	4
Economic forest planting	1	1	-	-	1	3	6
Biological invasion		-	-	-	2	2	7
Commercial collection		3	2	1	4	10	1

4. Functional zoning of biodiversity conservation on the western slope of Cangshan Mountain

According to the analysis of the protection objects and threat factors, with the goal of improving the living conditions of the protected objects and reducing the threat factors, the layout structure system of the overall plan is proposed, and the biodiversity of the Cangshan Mountain West Slope Protected Area is divided into the core area, buffer zone, experimental area and eco-tourism area of Cangshan Biodiversity Conservation, with species as the "point", regional boundary as the "line", ecosystem and scenic system as the "surface", and the comprehensive protection planning system combining "point-line-surface" is realized.

According to the main habitat types and ecosystem properties required by the biological species on the western slope of Cangshan Mountain, it is divided into three major functional areas (**Table 5**), and each functional area is divided into key protection areas and general control areas: Cangshan Mountain is set up as a core protection area above 3,000 m, and human circulation is prohibited in the area to protect species resources for the purpose of protecting the environment of germplasm resources. The core area is the most complete preservation of the native ecosystem, the protected objects and their native areas are concentrated, the rare and alpine dark coniferous forests are concentrated in the distribution area of natural vegetation, and the core of the protected glacier relics and species. The altitude of 2,600–3,000 m is the Cangshan Buffer Reserve, which is the outer periphery of the core area, in order to protect, prevent and mitigate the external influence and interference on the core area. The buffer zone can carry out a small or small capacity of tourism, popular science education, cultural activities, etc. The altitude of the eastern slope of Cangshan Nature Reserve is 2,200–2,600 m, and the altitude of the western slope is 2,000, 2,400–2,600 m for the experimental area. The main forests below 2,600 m in Cangshan Mountain are *Pinus armandii*, *Pinus yunnanensis*, *oak, Castanopsis fargesii, Rhododendron delavayi, Populus davidiana, Ericaceae, Camellia, Keteleeria evelyniana, Cinnamomum camphora, Schima superba*, handonggua, Shanyangmei, etc., and secondary oak bushes, ferns and other ornamental plants are mainly distributed in the area. In addition, in the buffer zone and experimental area, eco-tourism can be appropriately developed, which is mainly divided into Shimenguan Canyon Tour Area and West Slope Large Garden Natural Landscape Tour Area. The objects of protection are rare and endangered species resources and ecological natural landscapes.

On the basis of zoning protection, at the same time, the sustainable development of biodiversity planning, scientific research monitoring planning, infrastructure construction planning, ecotourism planning and publicity and education planning are implemented to curb threat factors and protect biodiversity systems.

Table 5. Biodiversity conservation zones on the western slope of Cangshan Mountain and the objects of conservation in each district

The partition type	Scope of protection	Area/hm^2	Conservation objects
Biodiversity conservation core zone	Altitude 3,000 m–pole	12,205.97	Remains of modern oceanic warm glaciers and ancient glaciers Complete alpine vertical with natural landscape Alpine meadow 3,700 ~ 4,100 m above sea level
Biodiversity conservation buffer zone	Altitude 2,600–3,000 m	7,232.45	Complete alpine vertical with natural landscape Colorful alpine vegetation types Cherish endangered animal resources
Biodiversity Conservation Experimental Zone	The eastern slope is 2,200–2,600 m above sea level, and the western slope is Altitude 2,000 m, 2,400–2,600 m 12,205. 97	1,171.58	Rare and endangered species Resources Existing vegetation and natural landscapes

5. Conclusion

The application of the Conservation Action Planning (CAP) method in the practice of biodiversity conservation on the western slope of Cangshan Mountain shows that the method provides a clear idea and logical analysis framework for the planning of biodiversity conservation in natural protected areas. Through meticulous preliminary investigation and analysis, the priority protection targets and primary threat factors are determined, which is the basis for guiding zoning planning and formulating protection strategies. The formulation of zoning scope and planning layout in the planning is the core content of the planning implementation. The results of this paper are yet to be done in step 4 of the CAP, which is a later effectiveness evaluation (including an assessment of the health status of species and ecosystems; assess the effectiveness of existing protection responses and protection actions; there is also an assessment of the extent to which the threat factor has decreased, the scale of investment in the protection project and the methods used) to examine and adjust the protection objectives, protection priorities and protection strategies through the feedback obtained, so as to determine new research directions.

Although CAP has been used to a certain extent in Nature Reserves in China, there are still some specific problems in practice, such as the preparation of conservation planning is mainly involved by relevant experts from scientific research institutions and universities, and the participation of protected area staff is not enough, which often leads to conservation planning often only staying in the preparation stage and unable to implement it in depth. Although the participation of the perennial staff of the protected area has been increased in this expert score, the participation of the staff of the reserve in the preparation of the conservation plan is still insufficient. Therefore, in the preparation stage of protection planning, more protected area staff should be involved in the formulation of the plan, so as to promote the real implementation of the protection plan.

Acknowledgements

This work was supported by Funded by Yunnan Provincial Department of Education (2012Y236); Yunnan Province (Southwest Forestry University Provincial and Ministerial-level Landscape Plants and Ornamental Horticulture Key Disciplines, Provincial University Key Laboratory and School Laboratory Sharing Platform).

Conflict of interest

The authors declared no conflict of interest.

References

1. Dali Branch of Yunnan Forestry Survey and Planning Institute. Survey report on forest resource planning and design in Dali City, Yunnan Province. Dali, 2005.
2. Yang Y, Wang J, Shi M. Biodiversity conservation and development in northwest Yunnan. Proceedings of the 2008 Yunnan Forum on Biodiversity and Social Harmonious Development and the First Youth Science and Technology Forum. Kunming: Yunnan People’s Publishing House; 2008.
3. Yang J. Status and solutions of biodiversity protection in northwest Yunnan. Forest Resources Management 2002; (3): 61–65.
4. Shen B. Studies on the resource exploitation and biodiversity conservation in Dali Cangshan-Erhai Nature Reserve. Chinese Biodiversity1998; 6(2): 151–156.
5. Mu J. Biodiversity of Cangshan Mountain in Dali and countermeasures for protection. Forest Inventory and Planning 2006; 31(1): 79–82.
6. Duan C. Scientific investigation of plants in Cangshan Mountain. Kunming: Yunnan Science and Technology Press; 1994.
7. Yu K, Li D, Duan T. Landscape approaches in biodiversity conservation. Chinese Biodiversity 1998; 6(3): 205–212.
8. Li L, Du P, Guo J. Study on biodiversity conservation planning in Gushan Scenic Area in Fuzhou. Jour of Fujian Forestry Science and Technology 2007; 34(3): 150–154.
9. Liu X, Wang J. Prospects and reviews on the studies of biodiversity in Qilian Mountains. Journal of Northwest Forestry University 2001; 16(Suppl.): 58–61.
10. Jiang T, Shen L, Yao P. Conservation of biodiversity in urban areas. Journal of Northwest Forestry College 2008; 23(2): 217–219.
11. Conserve Online. Conservation Action Planning: Introduction. 9 April 2012. Available from: http://conserveonline.org/work-spaces/cbdgateway/cap/practices/index_html.
12. Parrish JD, Braun DP, Unnasch RS. Are we conserving what we say we are measuring ecological integrity within protected areas. Bioscience 2003; 53(9): 851–860.
13. Salafsky N, Salzer D, Stattersfield AJ, *et al.* A standard lexicon for biodiversity conservation: Unified classifications of threats and actions. Conservation Biology 2008; 22(4): 897–911.
14. The Nature Conservancy. Conservation Action Planning Handbook (TNC Nature Conservation Series). Beijing: China Environmental Press; 2010.
15. Wang P, Zhou D, Liu D, *et al.* Application and perspective of Conservation Action Planning (CAP) in China. Forest Inventory and Planning 2012; 37(6): 95–99.
16. Ma J, Yang G, Han Y, *et al.* Study on approaches of biodiversity conservation and planning for Meili Snow Mountain National Park. Forest Inventory and Planning 2010; 35(3): 119–123.
17. Tang B, Liu X. Application of Conservation Action Planning software. Wetland Science & Management 2011; 7(1): 45–47.
18. Liu S. Baohu xingdong guihua CAP zai yizhi gongyuan baohu he guihua zhong de yingyong (Chinese) [Application of Conservation Action Planning (CAP) in Heritage Park conservation and planning] [Master’s thesis]. Kunming: Southwest Forestry University; 2008.
19. Cai J, Shu M, Lu W, *et al.* Study on plant biodiversity conservation and construction planning in Yu-yao city. Jour of Fujian Forestry Science and Technology 2008; 35(1): 155–159.
20. Dali Bai Autonomous Prefecture Cangshan Protection Administration. Cangshan Chronicle. Kunming: Yunnan Nationalities Publishing House; 2008.
21. Margules CR, Pressey RL. Systematic conservation planning. Nature 2000; 405(6783): 243–253.
22. Dali Branch of Yunnan Forestry Survey and Plan-

ning Institute, Dali Bai Autonomous Prefecture Cangshan Protection Administration. The overall plan for the protection and management of Cangshan Mountain in Dali Bai Autonomous Prefecture, Yunnan Province. Dali; 2003.
23. Environmental Protection Bureau of the People's Republic of China, Institute of Botany, Chinese Academy of Sciences. Information System of Chinese Rare and Endangered plants (ISCREP): Volume 1. Beijing: China Science Publishing & Meida Ltd. (CSPM); 1987.
24. The State Council Information Office of the People's Republic of China. List of National Key Protected Wild Plants (First Batch). Beijing; 1999.
25. United Nations. Convention on international trade in endangered species of wild fauna and flora (revision). Bangkok; 2010.
26. Dong X, Xu C, Li J. The geographic distribution and floristic featuers of the rare endangered in the cangshan. Journal of Chuxiong Normal University 2002; 17(6): 36–38.

Sustainable Forestry (2021) Volume 4 Issue 1
doi:10.24294/sf.v4i1.1598

REVIEW ARTICLE

Species diversity of typical forest communities in Taizhou Green Heart zone

Liqin Pan

Taizhou Vocational College of Science & Technology, Taizhou 318020, China. E-mail: lqpan@yzu.edu.cn

ABSTRACT

Taking six typical forest communities in Taizhou Green Heart (i: Liquidambar formosana + Ulmus pumila + Celtis sinensis; ii: Celtis sinensis + Pterocarya stenoptera + Pinus massoniana; iii: Sapindus mukorossi + Sapium sebiferum + Cupressus funebris; iv: Liquidambar formosana + Acer buergerianum + Cupressus funebris); v: Celtis sinensis + Ligustrum compactum + Pinus massoniana; vi: Machilus ichangensis + Sapindus mukorossi + Acer buergerianum) as the research objects, 5 indicators: Shannon-Wiener (H), Patrick richness (R1), Margalef species richness (R2), Pielou evenness (J) and ecological dominance (D) were used to analyze species diversity in forest communities. The results showed that: (1) the community was rich in plant resources, with a total of 50 species belonging to 40 genus and 31 families, including 19 species in tree layer, 22 species in shrub layer and only 9 species in herb layer, few plant species; (2) the species richness and diversity index of tree layer and shrub layer were significantly higher than that of herb layer, but there were differences among different communities in the same layer, and no significant difference was reached; (3) the species richness and community diversity of the six communities showed as follows: community VI > community I > community II > community IV > community V > community III.

Keywords: Forest Community; Species Diversity; Ecological Landscape Forest; Taizhou Green Heart

ARTICLE INFO

Received: 17 January 2021
Accepted: 9 March 2021
Available online: 14 March 2021

1. Introduction

Forest community structure and species diversity are one of the most important contents and hotspots in ecological research[1]. Species diversity is an important part of biodiversity[2]. Species diversity not only reflects the richness, variation and evenness of species in a community or habitat, but also reflects the type of community structure, organization level, development stage, degree of stability and habitat differences[3-5]. The investigation and research on forest community structure and determination of species richness and diversity index of community are conducive to better knowing the composition, structure, function and dynamics of the community, understanding the living conditions of the species, grasping the general rule of community succession and the formation mechanisms of the biological diversity, so as to provide theoretical basis for biodiversity protection[6].

The urban forest in the Green Heart zone of Taizhou has huge ecosystem service value and plays an important role in climate regulation, environmental protection, water conservation, soil and water conservation, windbreak and sand fixation, sightseeing and recreation, and beautification of the city, etc. It is an important part of the ecological environment construction of Taizhou[7,8]. This research takes planted forests around Jiufeng Mountain, Dayue mountain and Shizi Mountain, as investigation objects, investigating and analyzing the community characteristics and plant diversity, understanding the community structure and species diversity status and further analyzing the stability of community and ecological benefits, so as to provide basic information for the construction, planning and stand transformation of ecological landscape in the Green Heart zone, scientific basis for the conservation and sustainable utilization of forest vegetation species diversity as well as references for the protection and development of Taizhou Green Heart zone.

2. Overview of the study site

Taizhou is located in the central coastal area of Zhejiang Province. It is a "combined coastal city with green ring and heart shape". Green Heart is located in the center of urban area and is the core of urban spatial structure. The total area is about 6,333 hm^2, of which 3,466 hm^2 is hill. Mountains and hills occupy nearly 50% of the land, and mainly composed of Jiufeng Mountain, Dayue Mountain and Shizi Mountain. The highest peak in the area is Huangmao Shan of Jiufeng Mountain, with an altitude of 529.2 m, and there are four peaks above 500 m in height, all of which concentrated in Jiufeng Mountain in the west. The mountainous area with low hills and gentle slopes in Green Heart is about 10.31 km^2, in which the slope of 5°–15° are about 3.74 km^2 and that of 15°–25° are about 6.57 km^2. The study site is centered on Jiufeng Mountain, including some sample sites of Danyue Mountain and Shizi Mountain. The average annual temperature in the sample area is 16.6–17.5 °C, and the average annual precipitation is 1,480–1,530 mm, which belongs to the typical subtropical monsoon climate. The soil is mainly yellow soil, red soil and so on. The main vegetation types are subtropical evergreen broad-leaved forest and subtropical Zhejiang-Fujian hill forest.

3. Research methods

3.1 Survey methods

From December 2013 to June 2014, a comprehensive survey of Jiufeng Mountain, Dayue Mountain, Shizi Mountain and other mountains in Green Heart of Taizhou was conducted. It's found that there were few original evergreen broad-leaved forests and were mainly artificially cultivated landscape forests. Due to the influence of human activities, the species of understory shrubs and herbaceous plants are rare, and the forest community structure is relatively simple. According to the species and quantity of plants in the tree layer, the forest communities were divided into six typical community types: *Liquidambar formosana* + *Ulmus pumila* + *Celtis sinensis* (i); *Celtis sinensis* + *Pterocarya stenoptera* + *Pinus massoniana* (ii); *Sapindus mukorossi* + *Sapium sebiferum* + *Cupressus funebris* (iii); *Liquidambar formosana* + *Acer buergerianum* + *Cupressus funebris* (iv); *Celtis sinensis* + *Ligustrum compactum* + *Pinus massoniana* (v); *Machilus ichangensis* + *Sapindus mukorossi* + *Acer buergerianum* (vi). According to different community types, sampling method was used to investigate typical sample sites selected. Three tree quadrats of 20 m × 20 m were set up in each community, within which five 4 m × 4 m shrub quadrats and five 1 m × 1 m herbaceous quadrats were set, a total of 18 tree quadrats and 90 shrub and herbaceous quadrats were set. Each tree's name was recorded, and their diameter at breast height (DBH), tree height, crown width and other growth indexes were measured. Investigate and record the name, plant height, coverage, etc. of shrub and herb were investigated and recorded. In the sample site investigation, trees with DBH ≥4 cm were selected to measure the species diversity of the tree layer, while trees with DBH ≤2 cm were excluded from the calculation range of the diversity index of the tree layer, and young trees and seedlings were recorded in the shrub layer[9,10].

3.2 Data analysis

3.2.1 Species importance value

Significant value of trees (iv_{tree}) = (relative density (%) + relative significance (%) + relative frequency (%))/3; significant value of shrub and herb ($iv_{\text{shrub and herb}}$) = (relative density (%) + relative coverage (%) + relative frequency (%))/3. In the equation, relative density refers to the percentage of individuals of a certain species in the sum of individuals of all species; relative significance refers to the percentage of the chest height area of a species (1.3 m above the ground) in the total chest height area of all species; relative frequency refers to the percentage of the number of quadrats of a certain species in the total number of quadrats; coverage refers to the land area covered by the vertical projection of the above-ground part of plants[11,12].

3.2.2 Indicators of species diversity

In this study, several widely used measures methods were used[12-14]: (1) index of species diversity (Shannon-Wiener index), $H = -\sum P_i \ln P_i$; (2) Patrick richness index, $R_1 = S$; (3) Margalef species richness index, $R_2 = (S - 1)/\ln N$; (4) Pielou uniformity index, $J = H/\ln S$; (5) Ecological dominance index, $D = 1 - \sum P_i^2$. In the equation, P_i is the percentage of the number of individuals of a certain species in the community to the total number of individuals of all species in the community, that is, $Pi = N_i / N$, N_i is the number of the i^{th} species; N is the total number of individuals in the community; S is the number of species in the community.

3.2.3 Statistical analysis

For the calculated diversity indicators of different forest community types, the overall diversity differences of different community types and diversity differences at various vertical levels were compared through statistical analysis. One-way ANOVA and multiple comparison (Duncan) were used for analysis. Excel and SPSS 13.0 software were used for calculation.

4. Results and analysis

4.1 Species composition of forest communities

Species composition is one of the most basic characteristics of plant communities, and is the basis of community composition[15], affecting the biodiversity of forest communities. There were a total of 50 species of vascular plants in 40 genera and 31 families, including 5 species of ferns belonging to 4 families and 4 genera, 3 species of gymnosperms belonging to 3 families and 3 genera and 42 species of angiosperms belonging to 33 genera and 25 families. The dominant families were *Ulaceae*, *Lauraceae*, *Fagaceae*, *Jugaceae* and *Ilexaceae*. There were 19 species in the tree layer plants, 22 species of shrub layer plants and only 9 species in herbaceous layer. It shows that there are more plants in tree layer and shrub layer, whose species are more abundant.

4.2 Characteristics of dominant species in community

Importance values can better reflect the position and role of different plants in the community, as well as the differences in the composition and structure of different plant communities[16,17]. In the investigation, it was found that the vertical layer of plants was obvious, and the tree layer and shrub layer were dominant, while the herb layer was weak. In the community, the dominant tree species in the tree layer were obvious, such as *Celtis sinensis*, *Ulmus parvifolia*, *Sapium sebiferum*, *Liquidambar formosana*, *Pterocarya stenoptera*, *Sapindus mukorossi* and *Cupressus funebris* etc. **Table 1** showed that the importance values of *Liquidambar formosana*, *Cupressus funebris* and *Pinus massoniana* were relatively high and played an important role in the community, serving as the building species and dominant species in the community; while the importance value of *Metasequoia glyptostroboides* and *Celtis julianae*, etc. were smaller. Du Ying and Chestnut are also occasionally seen in the tree layer, which are not listed due to their small importance value. The differences of dominant species in shrub layer were not obvious (**Table 2**), which mainly including young trees and seedlings in tree regeneration layer, such as *Celtis sinensis*, *Acer buergerianum*, *Ulmus pumila*. In addition, shrubs with higher important values include common under-forest shrubs such as *Castanea seguinii*, *Rhus chinensis*, and *Loropetalum chinense*, as well as *Trachelospermum jasminoides* and *Parthenocissus tricuspidata*. There are few plant species in the herb layer, but the dominance is more obvious (**Table 3**). In the sample sites with lower altitudes, the main species are *Pteris multifida* and *Mercurialis leiocarpa*, and their important values are 22.43 and 15.59, while in the quadrats with higher altitudes, ferns such as *Dicranopteris pedata* and *Dryopteridaceae* are dominant.

Table 1. Importance values of common plants in tree layer %

No.	Plant name	Relative density	Relative frequency	Relative significance	Importance value
1	*Liquidambar formosana*	9.09	10.1	13.62	10.94
2	*Cupressus funebris*	9.79	11.9	10.75	10.81
3	*Pinus massoniana*	7.69	9.8	13.40	10.30
4	*Celtis sinensis*	11.19	10.5	7.88	9.86
5	*Pterocarya stenoptera*	6.99	9.7	9.46	8.72
6	*Acer buergerianum*	9.12	10.2	5.92	8.41
7	*Pinus thunbergii*	6.99	8.5	7.39	7.63
8	*Machilus ichangensis*	5.59	8.4	7.41	7.13
9	*Sapindus mukorossi*	6.29	8.3	6.37	6.99
10	*Sapium sebiferum*	6.29	8.4	5.29	6.66
11	*Ulmus pumila*	5.59	6.4	4.87	5.65
12	*Celtis julianae*	4.90	5.2	4.84	4.98
13	*Metasequoia glyptostroboides*	2.80	4.1	3.27	3.39

Table 2. Importance values of plants in shrub layer %

No.	Plant name	Relative density	Relative frequency	Relative significance	Importance value
1	*Celtis sinensis*	7.01	10.30	9.29	8.87
2	*Acer buergerianum*	7.69	8.65	8.36	8.23
3	*Castanea seguinii*	5.95	9.89	8.46	8.10
4	*Rhus chinensis*	10.82	7.06	5.88	7.92
5	*Ulmus parvifolia*	6.86	10.10	6.77	7.91
6	*Trachelospermum jasminoides*	5.13	8.41	8.44	7.33
7	*Trachycarpus fortune*	5.85	5.90	8.03	6.59
8	*Loropetalum chinensis*	6.41	6.80	5.22	6.14
9	*Ligustrum compactum*	6.50	5.20	5.59	5.76
10	*Weigela florida*	6.63	5.90	4.65	5.73
11	*Lindera glauca*	3.56	6.01	5.06	4.88
12	*Parthenocissus tricuspidata*	2.89	5.78	4.82	4.50
13	*Elaeagnus pungens*	3.24	6.31	3.91	4.49
14	*Broussonetia papyrifera*	2.56	5.32	4.47	4.12
15	*Aralia chinensis*	3.85	4.72	3.76	4.11
16	*Quercus aliena*	2.23	4.06	4.47	3.59
17	*Aphananthe aspera*	2.35	5.20	2.60	3.38
18	*Ilex latifolia*	1.94	4.16	2.59	2.90
19	*Rosa multiflora*	2.19	3.83	2.93	2.98
20	*Osmanthus fragrans*	1.28	2.48	1.41	1.72
21	*Serissa japonica*	0.97	1.37	0.47	0.94

Table 3. Importance values of plants in herbaceous layer %

No.	Plant name	Relative density	Relative frequency	Relative significance	Importance value
1	Pteris multifida	20.11	35.12	12.07	22.43
2	Mercurialis leiocarpa	10.58	25.57	10.62	15.59
3	Dicranopteris dichotoma	8.47	18.39	15.52	14.12
4	Dryopteris decipiens	8.47	10.18	18.97	12.54
5	Ophiopogon bodinieri	8.56	8.64	8.34	8.51
6	Parathelypteris glanduligera	5.82	1.65	12.07	6.51
7	Cayratia japonica	4.41	3.48	6.90	4.93
8	Farfugium japonicum	5.42	2.88	5.63	4.64
9	Oxalis corniculata	2.23	1.83	3.90	2.65

4.3 Species diversity analysis

4.3.1 Comparison of overall diversity among different levels

Species diversity represents the basic characteristics of community organization level and function, and reflects the evenness of species number distribution[11]. In order to better understand the diversity of forest communities in Green Heart of Taizhou, Simpson ecological dominance index, Margalef richness index, Shannon diversity index and Pielou evenness index were used to describe the species diversity of different communities. The results showed that (**Table 4**): the variation trends of the five diversity indexes in the community were consistent, and the overall degree of diversity was shrub layer > tree layer > herb layer, which was consistent with the results of Xu *et al.*[18] on the forest community diversity of the ous broad-leaved forest community in Longwang Mountain. The shrub layer is rich in species and quantity, so its species richness and diversity are significantly higher than those in the tree layer and herb layer. The diversity index of the herb layer was lower than that of the tree layer and the shrub layer. This was because the canopy density of the community was relatively high and the understory light was insufficient, forming a shady, closed and humid niche. Therefore, the herb layer was sparse and had few species, so the species richness and diversity of the herb layer were significantly lower than that of the tree layer and the shrub layer.

Table 4. Comparison of the difference significance of the overall diversity index among different levels

Layers	R_1	R_2	H	D	J
Tree layer	10.833b	3.062a	2.273b	0.887a	0.958a
Shrub layer	15.330a	3.511a	2.564a	0.904a	0.957a
Herb layer	7.167c	1.423c	1.730c	0.796b	0.901b

*: Different letters in the same column are significant difference. Same as below.

The difference significance test and multiple comparison were conducted for the overall diversity of tree layer, shrub layer and herb layer (**Table 4**). The results showed that the species richness index (R_1) and diversity index (H) were consistent among different layers, and both were significantly different at different layers. Shrub layer and tree layer were significantly higher than herb layer. However, the differences of richness index (R_2), dominance index (D) and evenness index (J) of tree layer and shrub layer were not significant. From the Pielou evenness index, both tree layer and shrub layer were significantly higher than that of herb layer, indicating that the distribution of tree layer and shrub layer was basically uniform in the community, while the distribution of herb layer was not, but distributed in clusters and slices, which was determined by the life type of plant species in herb layer.

4.3.2 Analysis of species diversity difference in different communities

This paper compared five diversity index of six different communities, and the results were shown form **Figure 1** to **Figure 5**. As can be seen from **Figure 1** to **Figure 3**, the richness (R_1, R_2) and Shannon-Wiener (H) index of tree layer and shrub layer in community VI (*Machilus ichangensi* + *Sapindus mukorossi* + *Acer buergerianum*) were

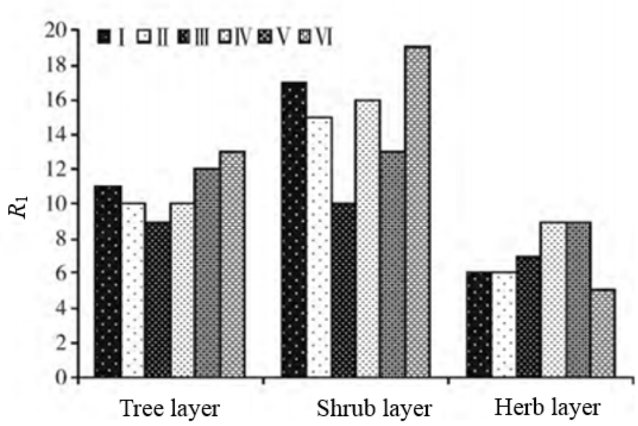


Figure 1. Patrick richness index.

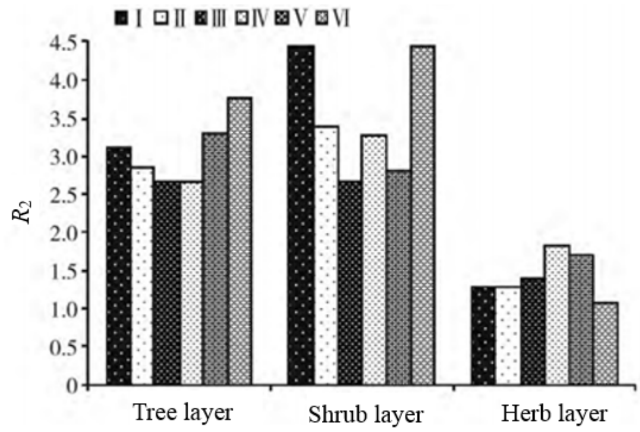


Figure 2. Margalef richness index

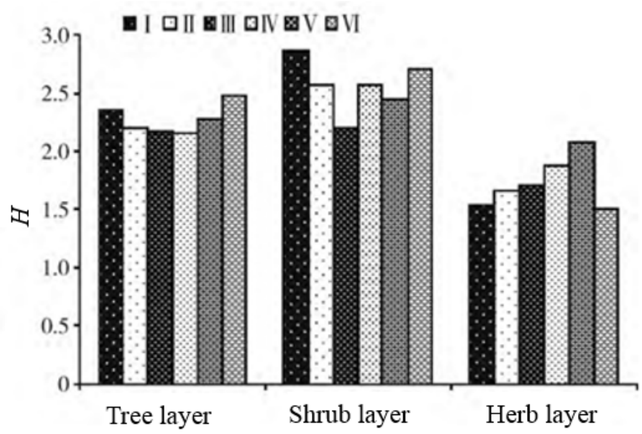


Figure 3. Shannon-Wiener index.

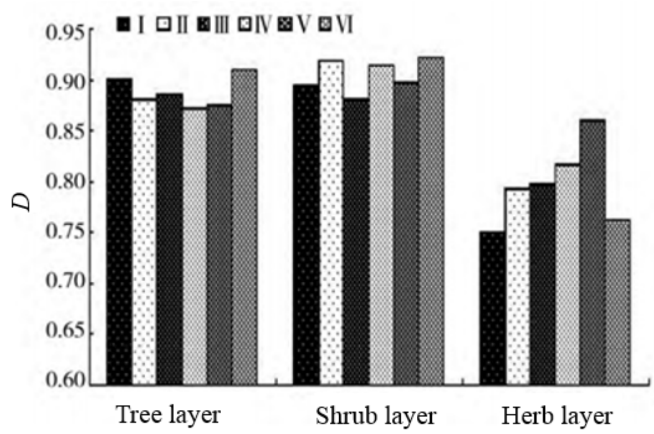


Figure 4. Ecological dominance index.

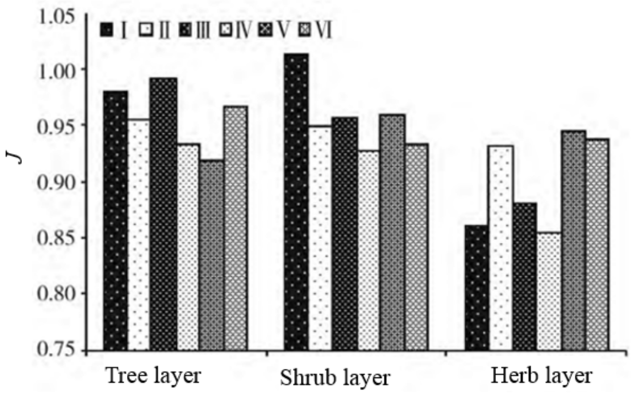


Figure 5. Pielou uniformity index.

higher, while the herbaceous layer was the lowest among the six communities. This may be related to the location of community. Community VI is located on the edge of road with a low altitude and serious human trampling phenomenon, which greatly reduces the species richness and diversity level of herbaceous layer of community. On the contrary, R_1, R_2 and H index of tree layer and shrub layer were the lowest in community III (*Sapindus mukorossi* + *Sapium sebiferum* + *Cupressus funebris*)), indicating that species richness and diversity of communities were low, while indexes of herb layer showed little difference from other communities.

The species richness and community diversity of the six communities showed as follows: community VI > community I > community II > community IV > community V > community III.

Pielou evenness index reflects the evenness of species distribution in a community. Even distribution of species in a community means that the degree of dominance is not concentrated or dominant phenomenon is not obvious. In **Figure 4** and **Figure 5** there was little difference in ecological dominance index and Pielou index among different communities. Only J index of shrub layer of community I showed a large value, which was significantly higher than that of other communities, indicating that shrub layer of this community had obvious population dominance and the community was in a stable development stage.

5. Conclusions and discussion

5.1 Forest community and the present situation of biological diversity in Taizhou Green Heart zone

Through the structure investigation of typical forest community in in Taizhou Green Heart zone, and the analysis of different species composition and species diversity index, the results show that there are have higher richness and diversity in tree layer and shrub layer of forest communities, and each index is significantly greater than the herb layer. In the tree layer, the canopy width and DBH of plants were larger, the species were less, the spacing of plants was larger, and the canopy density was higher. The shrub layer has the largest number of plants, and most of the species are similar to the tree layer. The plants are mainly grown under natural succession, without obvious dominance, but the density is relatively large, which can make full use of space, and has certain significance for the stability of community. Due to the poor light conditions under the forest, the number of herbaceous species is small, and most of them are ferns or some herbaceous climbing plants that can tolerate the shade and humidity. Due to the influence of altitude, geographical location and other factors, the phenomenon of artificial trampling of understory herbaceous layer is serious, which greatly reduces the species richness and diversity of understory herbaceous layer.

5.2 Influence of the present situation of forest community biodiversity on the construction of ecological landscape forest in Green Heart zone

Ecological landscape forest is an afforestation system and forest ecosystem mainly based on ecological public welfare forest, which plays an important role in climate regulation, environmental protection, water conservation, soil and water conservation, wind prevention and sand fixation, recreation and beautification of the city. The construction of ecological landscape forest is a new direction of modern urban forestry development and has become an important part of ecological environment construction in China.

Therefore, it's an urgent matter to build and construct ecological landscape forest in the Green Heart zone, and carry out protective development of Green Heart so as to improve the air quality and ecological environment of the main urban area in

Taizhou, and create a good leisure and health care space for the citizens. According to the survey, the forest vegetation in Green Heart area of Taizhou, centered on Jiufeng Mountain, is dominated by artificial forest, with less original evergreen broad-leaved forest, simple structure and poor community stability. Therefore, the effective protection of forest resources and the improvement of community structure in this area are of great significance to the ecological function of Green Heart.

According to the current situation of the forest vegetation of Green Heart mastered in this research, the community VI with better biodiversity, namely the Machilus ichangensis + *Sapindus mukorossi* + *Acer buergerianum*) community should be used as the typical community type, and under-forest shrubs and shade-tolerant grasses such as *Trachelospermum jasminoides*, *Hedera helix, Reineckia carnea*, Fatsia japonica and ferns, etc. should be appropriately replanted to further enrich the community structure. Due to the lack of tree species in most communities, it is suggested to replant tall trees such as *Schima superba*, *Lithocarpus*, *Cryptomeria, Quercus acutissima*, *Phoebe chekiangensis*, *Phoebe sheareri*, *Machilus thunbergii*, *Machilus leptophylla*, *Quercus glauca Thunb*, etc., deciduous trees such as *Metasequoia glyptostroboides*, *Taxodium ascendens*, etc. should also be planted, so as to enrich different seasonal landscapes. In general, broadleaf forest was the main ecological landscape, supplemented by bamboo and coniferous forest. From a functional point of view, ecological conservation should be the main type, which should be mainly distributed on the top of the mountain with Jiufeng Mountain as the main body. According to the gradual decrease of altitude, the forest structure mode with leisure health care mainly based on the leisure and health care type should be arranged respectively to create a good leisure environment for citizens. In the surrounding area of the Green Heart, forest land mainly protected by the Green Heart can be arranged, and ecological shelterbelt can be built to prevent the land around the Green Heart from being encroached by the surrounding land. In a word, the construction of Green Heart ecological landscape forest should be combined with the structural characteristics and habitat conditions of forest community, take the principle of Green Heart protective development as the principle, implement ecological management measures and means such as "building, replenishing, modifying, thinning and cultivating" according to local conditions, optimize the allocation of community plants, and gradually improve the forest community structure. From the perspective of ecosystem, we should create a stable and healthy forest ecosystem with aesthetic value, gradually improve the forest landscape value and ecological function, and further bring into play the ecological benefits of urban forest in Green Heart zone.

Acknowledgement

Key Project of Taizhou Philosophy and Social Science Planning Research (Ecological Protection and Development of Taizhou Green Heart zone 13GHZ01).

Conflict of interest

No conflict of interest was reported by the author.

Reference

1. Jiang M, Deng H, Tang T, *et al*. On spatial pattern of species richness in plant communities along riparian zone in Xiangxi River watershed. Acta Ecologica Sinica 2002; 2(5): 629–635.
2. Hu X, Yu M. Species diversity of evergreen broad-leaved forest dominated by Cyclobalanopsis glauca in the northern protection regions of Zhejiang Province. Guihaia 2003; 23(5): 399–403.
3. Chen Y. Evaluation on stand species diversity of broad-leaved trees in Nanping. Journal of Fujian College of Forestry 2012; 32(2): 136–140.
4. Zang R, Liu S, Jiang Y. Principles of forest biodiversity conservation. Scientia Silvae Sinicae 1999; 35(4): 71–79.
5. Li D, Li Y, Liang L, *et al*. Analysis of forest community types and species diversity in Majunshan Mount Forest Park. Journal of Southwest Forestry University 2013; 33(5):74–77.
6. Gao X, Ma K, Chen L. Species diversity of some deciduous broad-leaved forests in the warm-temperate zone and its relations to community stability. Acta Phytoecologica Sinica 2001; 25(3): 283–290.
7. Pan L, Xiang Y, Yang J. Evaluation of forest ecosystem service value of urban forest in Taizhou Green Heart. Journal of Chinese Urban Forestry 2014; 12(1): 16–18.
8. Xiang Y, Pan L. Ecological environment construc-

tion of urban “Green Heart” in Taizhou. Journal of Jiangsu Forestry Science and Technology 2013; 40(3): 31–33.
9. Pan L, Ji H, Fu Q, *et al*. Investigation of natural population and niche analysis for Adiantum reniforme var. sinense. Journal of Yangzhou University Agriculture and Life Sciences) 2005; 26(4): 74–78.
10. Fan W, Wang X, Guo H, *et al*. Species diversity of forest communities in Ziwuling Mountain of Shaanxi province. Journal of Shaanxi Normal University Natural Science Edition) 2014; 42(3): 59–66.
11. Chen M. Chengshi lvdi shengtai (Chinese) [Urban green space ecology]. Beijing: China Forestry Publishing House; 2011. p. 185–187.
12. Zhang J. Zhibei shuliang shengtaixue fangfa (Chinese) [Quantitative ecological methods of vegetation]. Beijing: Science and Technology of China Press; 1995.
13. Ma K, Huang J, Yu S, *et al*. Beijing donglinshan diqu zhiwu qunluo duoyangxing de yanjiu—fengfudu, junyundu he wuzhong duoyangxing zhishu (Chinese) [Study on community diversity in Dongling Mountain area of Beijing—Richness, evenness and species diversity index]. Acta Ecologica Sinica 1995; 15(3): 268–277.
14. Jiang L, Liu Z. Structure characteristics of natural birch community and species diversity in Great Xing’an Mountains. Forest Engineering 2014; 30(4): 12–17.
15. Qu Z, Wu Y, Wang H, *et al*. Shiwu shengtai xue (Chinese) [Plant ecology]. Beijing: Higher Education Press; 1983.
16. Ou Z, Li X, Su Z, *et al*. Tree species diversities of two forest communities in Yuanbao Mountain. Chinese Journal of Applied & Environmental Biology 2003; (6): 563–568.
17. Yue Y, Yu X, Niu L L, *et al*. Structural characteristics of plant communities and species diversity in Wuling Mountain, Beijing. Journal of Beijing Forestry University 2008; 30(Suppl.2): 165–167.
18. Xu J, Wei X, Wang J, *et al*. Species diversity and the community characteristics of deciduous broad-leaved forest in Longwang Mountain. Journal of Southwest Forestry University 2014; (3): 19–26.

Sustainable Forest (2021) Volume 4 Issue 1
doi:10.24294/sf.v4i1.1599

ORIGINAL RESEARCH ARTICLE

Analysis of the composition, structure and diversity of tree species in a temperate forest in northwestern Mexico

Gabriel Graciano-Ávila*, Oscar Alberto Aguirre Calderón, Eduardo Alanís-Rodríguez, José E. Lujan-Sotoju

Facultad de Ciencias Forestales, Universidad Autónoma de Nuevo León, Carretera Linares-Cd, Victoria km 145, Apartado Postal 41. CP. 67700. Linares, Nuevo León, México. E-mail: gabri.e.l@hotmail.com

ABSTRACT

The structure and diversity of tree species in a temperate forest in northwestern Mexico was characterized. Nine sampling sites of 50 × 50 m (2,500 m^2) were established, and a census of all tree species was carried out. Each individual was measured for total height and diameter at breast height. The importance value index (*IVI*) was obtained, calculated from the variable abundance, dominance and frequency. The diversity and richness indices were also calculated. A total of 12 species, four genera and four families were recorded. The forest has a density of 575.11 individuals and a basal area of 23.54/m^2. The species of *Pinus cooperi* had the highest *IVI* (79.05%), and the Shannon index of 1.74.

Keywords: Temperate Forest; Durango; Diversity Indices; *Pinus*; *Quercus*

ARTICLE INFO

Received: 4 February 2021
Accepted: 15 March 2021
Available online: 2 April 2021

1. Introduction

The Sierra Madre Occidental is a mountainous complex that extends from near the border with the United States to the north of Jalisco, covering more than 1,500 km from north to south, representing 30% of Mexico's territory[1]. In this region, there are extensive areas covered by conifer and oak forests, resulting in a flora rich in a diversity of pine, oak and arbutus associations[2]. 52 pine species exist in the country, and 20 are found in the state of Durango[3]. The importance of this type of vegetation is not only because of its high diversity, but also because pine and oak species are the trees of greatest economic interest[4].

Structural characterization is important to understand the functioning of ecosystems, which can provide decision elements to contribute to the adequate management of forests[5,6]. Structural indices and measure variables are taken into account[7]. Vertical and horizontal structure is considered a good forest management practice for biodiversity conservation in temperate ecosystems[8]. Structure, diversity and density are the main characteristics of forest stands; diversity is a concept that allows different interpretations, although in general, it is used as a synonym for species diversity[9]. On the other hand, tree structure is a key element to evaluate forest stability[10], which can be modified through the application of silvicultural treatments, changing the structure of forest stands and consequently the forest[11,6]. In Mexico, several studies have been conducted on the diversity of tree species in temperate climates[12,13]. Therefore, the composition, structure and diversity of tree species in a temperate forest in northwestern Mexico.

2. Materials and methods

The study was conducted in Victoria, located in the city of New Pueblo, located in southwestern Durango State. Geographically framed between 23°40’0’’ and 23°47’54’’N, and 105º21’31’’ and 105°29’52’’W (**Figure 1**). Orographically, the ejido Victoria is located in the physiographic province of the Sierra Madre Occidental. According to the INEGI (1988)[14] edaphological chart, the soils of the study area are cambisol, regosol and lithosol with coarse to medium texture. The vegetation consists of pine-oak forests, with different productivity conditions.

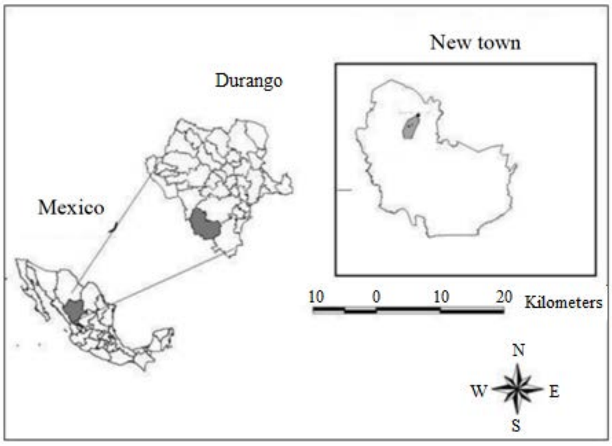


Figure 1. Location of the study area.

The measure data for the study were taken in nine permanent sample plots, quadrangular plots of 2,500 m^2 established in 2007. The database consisted of 1,294 trees whose normal diameter and total height were measured, and the species to which each individual belongs was recorded. To characterize the horizontal structure, abundance was determined according to the number of trees, dominance based on basal area, and frequency based on presence at the sampling sites. The relativized variables were used to obtain the importance value index (*IVI*), which acquires percentage values on a scale from zero to 300[15], the formula is:

$$A_i = \frac{N_i}{E} \;\; , \;\; AR_i = \left(\frac{A_i}{\sum_{i=1...n} A_i} \right) \times 100$$

Where: A_i is the absolute abundance, AR_i is the relative abundance of species i with respect to the total abundance, N_i is the number of individuals of species i, and E is the sampling area (ha).

Relative coverage was obtained with the formula:

$$D_i = \frac{Ab_i}{E(ha)} \;\; , \;\; DR_i = \left(\frac{D_i}{\sum_{i=1...n} D_i} \right) \times 100$$

Where: D_i is the absolute cover, DR_i is relative cover of species i with respect to the cover, Ab is the basal area of species i and E is the area (ha).

The relative frequency was obtained with the formula:

$$F_i = \frac{P_i}{NS} \;\; , \;\; FR_i = \left(\frac{F_i}{\sum_{i=1...n} F_i} \right) \times 100$$

Where: F_i is the absolute frequency, FR_i is the relative frequency of species i with respect to the sum of the frequencies, P_i is the number of sites where species i is present and NS is the total number of sampling sites.

The importance value index (*IVI*) is defined as[16,17]:

$$IVI = AR_i + DR_i + FR_i$$

To estimate species richness we used the Margalef index (D_{mg}) and for alpha diversity the Shannon-Weaver index (H) using the formulas[18,19]:

$$D_{mg} = \frac{(S-1)}{ln(N)} \; , \; H' = -\sum_{i=1}^{S} p_i \times ln(p_i) \; , \; P_i = \frac{n_i}{N}$$

Where: S is the number of species present, N is the total number of individuals, n_i is the number of individuals of species i and p_i is the proportion of individuals of species i with respect to the total number of individuals. For the characterization of the vertical structure of the species, the vertical distribution index of species (A) was used[11]. Where A has values between zero and a maximum value (A_{max}); when value A = 0 means that the stand is constituted by a single species occurring in a single stratum. A_{max} is reached when all species occur in the same proportion both in the stand and in the different strata[20]. For the estimation of vertical distribution of species, height zones were defined[21]: zone I: 80 to 100% of the maximum height of the area; zone II: 50 to 80% of the maximum height, and zone III: zero to 50% of the maximum height. The index was estimated with the following formula:

$$A = -\sum_{i=1}^{S}\sum_{j=1}^{Z} p_{ij} * ln(p_{ij}) \text{ , } Amax = ln(S * Z)$$

Where: S = number of species present; Z = number of height strata; $p_{i,j}$ = percentage of species in each zone, and is estimated by the following equation $p_{i,j} = n_{i,j}/N$; where $n_{i,j}$ = number of individuals of the same species (i) in zone (j) and N = total number of individuals. The value of A is standardized as follows:

$$A_{rel} = \frac{A}{ln(S * Z)} * 100$$

3. Results

A total of 12 tree species were recorded, distributed in four genera and four families (**Table 1**). The most representative family was Pinaceae with five species, followed by the families Fagaceae and Ericaceae with three species each. These three families included three genera and 11 species, which constitutes 91.7% of the vegetation recorded in the nine sampling sites.

Table 1. Scientific name and family of tree species recorded in the study area

Scientific name	**Family**
Arbutus bicolor S. González	Ericaceae
Arbutus madrensis S. González	Ericaceae
Arbutus xalapensis Kunth	Ericaceae
Juniperus deppeana Steud.	Cupressaceae
Pinus ayacahuite Ehrenb. ex Schltdl.	Pinaceae
Pinus cooperí C.E.Blanco	Pinaceae
Pinus durangensis Martinez	Pinaceae
Pinus leiophylla Schiede ex Schltdl. & Cham.	Pinaceae
Pinus teocote Schied. ex Schltdl. & Cham.	Pinaceae
Quercus crassifolia Bonpl.	Fagaceae
Quercus rugosa Née	Fagaceae
Quercus sideroxyla Bonpl.	Fagaceae

The plant community density in the study area was 575.11. The genus Pinus was the most abundant with a density of 369.89 trees, representing 69.01% of the total, followed by the genus Quercus with a density of 108 trees with 18.78% of the total (**Table 2**). The species with the highest density were *Pinus cooperi* (188.89 trees) with 32.84% of the total, *P. durangensis* (143.56 trees) with 24.94% of the total, *Quercus sideroxyla* (104.44 trees) with 18.16% of the total and *Junipers deppeana* (52.89 trees) with 9.2% of the total, which is equivalent to

Table 2. Abundance, dominance and frequency by genus of species recorded in the study area

Genre	**Abundance**		**Dominance**		**Frequency**	
	Absolute	**Relative**	**Absolute**	**Relative**	**Absolute**	**Relative**
	(number)	**(%)**	**(number/m^2)**	**(%)**	**-**	**(%)**
Pinus	396.89	69.01	16.71	70.98	100.00	30.00
Quercus	108.00	18.78	5.58	23.71	77.78	23.33
Juniperus	52.89	9.20	0.74	3.15	88.89	26.67
Arbutus	17.33	3.01	0.51	2.16	66.67	20.00
Total	575.11	100.00	23.54	100.00	333.33	100.00

Table 3. Estimated structural parameters for the species recorded in the study area

Genre	**Abundance**		**Dominance**		**Frequency**		***IVI***
	Absolute	**Relative**	**Absolute**	**Relative**	**Absolute**	**Relative**	
	(number)	**(%)**	**(number/m^2)**	**(%)**	**-**	**(%)**	
Pinus cooperi	188.89	32.84	6.96	29.55	100.00	16.67	79.05
Pinus durangensis	143.56	24.96	7.76	32.97	77.78	12.96	70.89
Quercus sideroxyla	104.44	18.16	3.62	15.40	77.78	12.96	46.53
Juniperus deppeana	52.89	9.20	0.74	3.15	88.89	14.81	27.18
Pinus ayacahuite	32.00	5.56	1.05	4.44	77.78	12.96	22.98
Pinus teocote	27.11	4.71	0.85	3.60	66.67	11.11	19.41
Arbutus bicolor	14.22	2.47	0.32	1.38	44.44	7.41	11.25
Quercus crassifolia	3.11	0.54	1.95	8.29	11.11	1.85	10.68
Pinus leiophylla	5.33	0.93	0.10	0.42	22.22	3.70	5.04
Arbutus xalapensis	2.22	0.39	0.16	0.70	11.11	1.85	2.94
Arbutus madrensis	0.89	0.15	0.02	0.08	11.11	1.85	2.1
Quercus rugosa	0.44	0.08	0.00	0.01	11.11	1.85	1.95
Total	575.11	100.00	23.54	100.00	600.00	100.00	

Notice: *IVI* = Importance value index. Species are ordered in descending order according to their *IVI*.

85.16% of the total species. While the least abundant species was *Quercus rugosa* (1 tree) with 0.08% of the total species (**Table 3**).

There was a basal area dominance of 23.54/m^2. The genus *Pinus* had the highest relative dominance (16.71/m^2) with 70.98% of the total, followed by the genus *Quercus* which had a relative dominance of 5.58/m^2, equivalent to 23.71% of the total (**Table 2**). The species with the largest basal area were *P. durangensis* (7.76/m^2) with 32.97%, *P. cooperi* (6.96/m^2) with 29.55%, *Q. sideroxyla* (3.63/m^2) with 15.40% and *Q. crassifolia* (1.95/m^2) with 8.29%, which together account for 86.22% of the total species.

The genus *Pinus* was present in all sampling sites, followed by the genus *Juniperus* which was found in eight of the nine sampling sites (**Table 2**). At the species level, *P. cooperi* was found in all nine sampling sites (100% absolute frequency) with 16.67% relative frequency, followed by *J. deppeana* which was found in eight sampling sites with an absolute frequency of 88%, representing 14.81% relative frequency. The species *Arbutus xalapensis*, *A. madrensis*, *Q. rugosa* and *Q. crassifolia* only occurred in one sampling site (**Table 3**).

The importance value index indicates that *P. cooperi* (79.05%), *P. durangensis* (70.89%), *Q. sideroxyla* (46.53%) and *J. deppeana* (27.18%) were the most outstanding species, having the highest *IVI* values. The rarest species were *A. xalapensis*, *A. madrensis* and *Q. rugosa* which had values less than three percent.

The specific richness of the plant community studied was 12 species, with a Margalef index of -1.53. In relation to the species diversity value, the Shannon index value was 1.74. By the vertical distribution index of the species, three height strata were defined, high (31.04–38.80 m), medium (19.40–31.03 m) and low (< 19–40 m). The high stratum is composed of *P. cooperi*, *P. durangensis* and *P. teocote* with 4.44 trees, equivalent to 0.77% of the zone, and the medium stratum is composed of *P. ayacahuite*, *P. cooperi*, *P. durangensis*, *P. teocote*, *Q. crassifolia* and *Q. sideroxyla* with 87.56 trees representing 15.22% of the zone. In the lower stratum, all the species of the study area were recorded with 483.11 trees, representing 84% of the area. The most abundant species was *P. cooperi* with 172.89 trees followed by *P. durangensis* with 104.89 trees and *Q. sideroxyla* with 100.44 trees (**Table 4**). The *A* index value was 2.07 with an A_{max} value of 3.58 and A_{rel} of 57%, which indicates that the evaluated area has average uniformity in height diversity. A_{rel} values close to 100% indicate that all species are equally distributed in the three height strata.

Table 4. Pretzch vertical index values for the study area

	Proportion (%)			
High stratum (38.80–31.04)	N	N/m^2	Total	In the area
Pinus cooperi	3.00	1.33	30.00	0.23
Pinus durangensis	6.00	2.67	60.00	0.46
Pinus teocote	1.00	0.44	10.00	0.08
Sum	10.00	4.44	100.00	0.77
Middle stratum (31.04–19.40)				
Pinus ayacahuite	67.00	29.78	34.01	5.18
Pinus cooperi	33.00	14.67	16.75	2.55
Pinus durangensis	81.00	36.00	41.12	6.26
Pinus teocote	4.00	1.78	2.03	0.31
Quercus crassifolia	3.00	1.33	1.52	0.23
Quercus sideroxyla	9.00	4.00	4.57	0.70
Sum	197.00	87.56	100.00	15.22
Low stratum (2.50–19.4)				
Arbutus bicolor	32.00	14.22	2.94	2.47
Arbutus madrensis	2.00	0.89	0.18	0.15
Arbutus xalapensis	5.00	2.22	0.46	0.39
Juniperus deppeana	119.00	52.89	10.95	9.20
Pinus ayacahuite	5.00	2.22	0.46	0.39
Pinus cooperi	389.00	172.89	35.79	30.06
Pinus durangensis	236.00	104.89	21.71	18.24
Pinus leiophylla	12.00	5.33	1.10	0.93
Pinus teocote	56.00	24.89	5.15	4.33
Quercus crassifolia	4.00	1.78	0.37	0.31
Quercus sideroxyla	226.00	100.44	20.79	17.47
Quercus rugosa	1.00	0.44	0.09	0.08
Sum	1087.00	483.11	100.00	84.00
Total	1294.00	575.11	300.00	100.00

4. Discussion

Due to its structure and composition, the study area evaluated corresponds to a typical forest of the Sierra Madre Occidental mountain massif[9]. The same dominant families reported for the Sierra Madre Occidental of the state of Durango were found, which are Pinaceae and Fagaceae[22]. The Pinaceae family was the most abundant, recording five species of the genus *Pinus*, which coincides with García and González[23], who point out that the distribution of the Pinaceae family and the genus *Pinus* is wide in all the mountain ranges of the country. These results coincide with those recorded for the state of Chihuahua, where species of the genus *Pinus* are reported to be dominant[24]. While López *et al.*[25] found that the genus *Pinus* also has higher abundance in temperate forests. Twelve species were recorded, a lower number than the 27 species recorded by Linares *et al*. (1999). The species *P. cooperi* was the most abundant, which is one of the species reported as the most abundant in temperate forests[12,26]. The species with the greatest importance for basal area occupying 86.22% were *P. durangensis*, *P. cooperi*, *Q. sideroxyla* and *Q. crassifolia*; these same species have been reported as those with the greatest basal area in oak-pine forests in the state of Durango with values greater than 65%[26]. The two species with the highest importance value index were *P. cooperi* and *P. durangensis* with 26.35 and 23.63%, respectively; in this regard *P. durangensis* has been reported as the species with the highest ecological value (57.05%) in temperate forests[24] report that the species *P. cooperi* has the highest value of ecological importance (19.92%); while Valenzuela and Granados[26] report *P. durangensis* and *P. cooperi* as the species with the highest value of ecological importance with 33.44 and 10.0%, respectively. In this regard, Margalef[27] mentions that the Shannon index normally varies from 1 to 5, with values lower than 2 being interpreted as low diversity, from 2 to 3.5 as medium diversity, and higher than 3.5 as high diversity. Therefore, the forest community studied has a low diversity (H' = 1.74). However, this value is higher than that recorded by Návar-Cháidez and González-Elizondo[12] for the temperate forests of the state of Durango. The value of the Margalef index (D_{Mg} = 1.53) is higher than those reported by Hernández-Salas *et al.*[24], who recorded values of D_{Mg} = 1.04 and D_{Mg} = 0.90 respectively. Which means that the study site presents higher diversity of tree species if compared to areas of the same region, however, this value is low if compared to values reported for other ecosystems[28,29]. The A index had values of 2.07 with A_{max} value of 3.58 and A_{rel} of 57%, indicating that the evaluated area presents average uniformity, in height diversity. Values of A_{rel} close to 100% indicate that all species are equally distributed in the three height strata. This coincides with that reported by Camacho *et al.*[30].

5. Conclusions

The temperate forest studied has a low specific richness and diversity of tree species. The most important families for their structural contribution to this forest are Pinaceae and Fabaceae, with the genera *Pinus* and *Quercus* being the most important. The most important species are *P. cooperi, P. durangensis* and *Q. sideroxyla*. For the vertical structure, the diversity of heights is medium, so the stage of development is latizal.

Conflict of interest

The authors declare no conflict of interest.

References

1. Descroix L, González Barrios JL, Ávalos JE. La sierra madre occidental: Una fuente de aguaamenazada (Spanish) [The western sierra madre: A threatened water source]. Gómez Palacio, Durango, Mexico: Instituto Nacional de Investigaciones Forestales, Agrícolas y Pecuarias-lnstitut de Recherche pour le Développement; 2004. p. 300.
2. González E, González M, Márquez M. Vegetación y ecorregiones de Durango (Spanish) [Vegetation and ecoregions of Durango]. Mexico: National Polytechnic Institute; 2007. p. 219.
3. Gernandt DS, Pérez JA. Biodiversidad de Pinophyta (coníferas) en México (Spanish) [Biodiversity of Pinophyta (coniferas) in Mexico]. Revista Mexicana de Biodiversidad 2014; 85: 126–133.
4. Sánchez O, Vega E, Peters E, *et al*. Conservación de ecosistemas templados de montaña en México (Spanish) [Conservation of temperate mountain ecosystems in Mexico]. Mexico: National Institute of Ecology-SEMARNAT; 2003. p. 315.
5. Aguirre O, Hui GY, Gadow KV, *et al*. An analysis of spatial forest structure using neighbourhood-based variables. Forest Ecology and Management 2003; 183(1-3): 137–145.
6. Castellanos-Bolaños JF, Treviño-Garza EJ, Aguirre-Calderón ÓA, *et al*. Estructura de bosques de Pino patula bajo manejo en Ixtlán de Juárez, Oaxaca, México (Spanish) [Structure of Patula pine forests under management in Ixtlán de Juárez, Oaxaca, Mexico]. Mexico: Madera y Bosques; 2008. p. 51–63.
7. Aguirre COA, Jiménez Pérez J, Kramer H, *et al*. Análisis estructural de ecosistemas forestales en el Cerro del Potosí (Spainish) [Structural analysis of forest ecosystems in Cerro del Potosí]. Ciencia UANL 2003; 6: 219–225.
8. Aguirre-Calderón OA. Manejo Forestal en el Siglo XXI (Spanish) [Forest management in the 21st century]. Madera y Bosques 2015; 21: 17–28.
9. Gadow KV, Sánchez OS, Álvarez JG. Estructura y crecimiento del bosque (Spanish) [Forest structure

and growth]. Germany: University of Goetingen; 2007. p. 87.
10. Lâhde E, Laiho O, Norokorpi Y, *et al.* Stand structure as the basis of diversity index. Forest Ecology and Management 1999; 115: 213–220.
11. Del Río M, Montes F, Cañellas I, *et al*. Índices de diversidad estructural en masas forestales (Spanish) [Indices of structural diversity in forest stands]. Agricultural Research: Forest Resources System 2003; 12: 159–176.
12. Návar-Cháidez JDJ, González-Elizondo S. Diversidad, estructura y productividad de bosques templados de durango (Spanish) [Diversity, structure and productivity of temperate forests of Durango]. Polibotánica 2009; 27: 71–87.
13. Leyva JC, Velazquez A, Angeles G. Patrones de diversidad de la regeneración natural en rodales mezclados de pino (Spanish) [Patterns of diversity of natural regeneration in mixed pine stands]. Revista Chapingo Serie Ciencias Forestales y del Ambiente 2010; 16(2): 227–239.
14. INEGI. Topographic map. Scale 1: 50,000. El Salto, Durango. F13-A18 spp [Accessed June 15, 2017]. Mexico: INEGI; 1988. Available from: http://internet.contenidos.inegi.org.mx/contenidos/Productos/prod_serv/contenidos/espanol/bvinegi/productos/geo-grafia/histor¡a/l_50_000/702825617394.pdf.
15. Mostacedo B, Fredericksen S. Manual de métodos básicos de muestreo y análisis en ecología vegetal (Spanish) [Manual of basic methods of sampling and analysis in plant ecology]. Bolfor; Santa Cruz, Bolivia; 2000. p. 87.
16. Whittaker RH. Evolution and measurement of species diversity. Taxon 1972; 21: 213–251.
17. Moreno CE. Métodos para medir la biodiversidad (Spanish) [Methods for measuring biodiversity]. Vol. 1. Pachuca, Hidalgo, Mexico: Cyted, Orcyt/Unesco, SEA; 2001. p. 83.
18. Shannon C. The mathematical theory of communication. In: Shannon CEJ, Weaver W (editors). Illinois, USA: Urbana Press, University of Illinois; 1948. p. 29–125.
19. Magurran AE. Measuring biological diversity. Cambridge, USA: Blackwell; 2004. p. 256.
20. Corral J, Aguirre OA, Jiménez J, *et al.* Un análisis del efecto del aprovechamiento forestal sobre la diversidad estructural en el Bosque Mesófilo de Montaña “El Cielo”, Tamaulipas, Mexico (Spanish) [An analysis of the effect of forest exploitation on structural diversity in the “El Cielo” Mountain Mesophyll Forest, Tamaulipas, Mexico]. Sistema de Recursos Forestales 2005; 14(2): 217–228.
21. Jiménez J, Aguirre O, Kramer H. Análisis de la estructura horizontal y vertical en un ecosistema multicohortal de pino-encino en el norte de México (Spanish) [Analysis of horizontal and vertical structure in a pine-oak multihortal ecosystem in northern Mexico]. Agricultural Research: Forest Resources System 2001; 10: 355–366.
22. Aragón-Piña EE, Garza-Herrera A, González-Elizondo MS, *et al*. Composición y estructura de las comunidades vegetales del rancho El Durangueño, en la Sierra Madre Occidental, Durango, Méxic (Spanish) [Composition and structure of the plant communities of the El Durangueño ranch, in the Sierra Madre Occidental, Durango, Mexico]. Revista Mexicana de Biodiversidad 2010; 81(3): 771–787.
23. García AA and González EMS. Pinaceae of Durango. Durango, Dgo. Mexico: CIIDIR-IPN, Instituto de Ecología A.C.j SIVILLA-Gobierno del Estado de Durango; 1998. p. 179.
24. Hernández-Salas J, Aguirre-Calderón ÓA, Alanís-Rodríguez E, *et al*. Efecto del manejo forestal en la diversidad y composición arbórea de un bosque templado del noroeste de México (Spanish) [Effect of forest management on tree diversity and composition in a temperate forest of northwestern Mexico]. Revista Chapingo. Serie Ciencias Forestales y del Ambiente 2013; 19(2): 189–200.
25. López JA, Aguirre-Calderón OA, Alanís-Rodríguez E, *et al.* Composición y diversidad de especies forestales en bosques templados de puebla (Spanish) [Forest species composition and diversity in temperate forests of puebla, Mexico]. Madera y Bosques 2017; 23: 39–51.
26. Valenzuela NLM, Granados SD. Caracterización fisonómica y ordenación de la vegetación en el área de influencia de El Salto, Durango, México (Spanish) [Physiognomic characterization and vegetation management in the area of influence of El Salto, Durango, Mexico]. Revista Chapingo. Serie Ciencias Forestales y del Ambiente 2009; 15(1): 29–41.
27. Margalef R. Homage to E. Hutchison, or why is there an upper limit to diversity. Transactions of the Connecticut Academy of Arts and Sciences 1972; 44: 211–235.
28. Agis J, Castro NDH, Martínez JL, *et al*. Diversidad y estructura de la selva mediana subperennifolia de Acapulco, Gro., México (Spanish) [Diversity and structure of the medium sub evergreen rainforest of Acapulco, Gro.]. Revista Iberoamericana de las Ciencias Biológicas y Agropecuarias: CIBA 2016; 5: 1–20.
29. Alanis E, Jiménez J, Aguirre OA, *et al.* Efecto del uso del suelo en la fitodiversidad del matorral espinoso tamaulipeco (Spanish) [Effect of land use on the phytodiversity of the Tamaulipan thornscrub]. Ciencia UANL 2008; 11: 56–62.
30. Camacho R, Alonso E, González-Tagle MA, *et al.* Diversidad y distribución vertical de especies vegetales mediante el índice de Pretzsch (Spanish) [Diversity and vertical distribution of plant species using the Pretzsch index]. Ciencia UANL 2014; 17: 34–41.

Sustainable Forestry (2021) Volume 4 Issue 1
doi:10.24294/sf.v4i1.1600

ORIGINAL RESEARCH ARTICLE

Progressive fragmentation and loss of natural forest habitat in one of the world's biodiversity hotspots

Samuel Otavo[1*], Cristian Echeverría[1,2]

[1] *Laboratorio de Ecología del Paisaje, Facultad de Ciencias Forestales, Universidad de Concepción, Casilla 160-C-Correo 3, Concepción, Chile. E-amil: samuelotavo@udec.cl*

[2] *Núcleo Milenio Centro para el Impacto Socioeconómico de las Políticas Ambientales (CESIEP), Santiago, Chile.*

ABSTRACT

Knowledge of the state of fragmentation and transformation of a forested landscape is crucial for proper planning and biodiversity conservation. Chile is one of the world's biodiversity hotspots; within it is the Nahuelbuta mountain range, which is considered an area of high biodiversity value and intense anthropic pressure. Despite this, there is no precise information on the degree of transformation of its landscape and its conservation status. The objective of this work was to evaluate the state of the landscape and the spatio-temporal changes of the native forests in this mountain range. Using Landsat images from 1986 and 2011, thematic maps of land use were generated. A 33% loss of native forest in 25 years was observed, mainly associated to the substitution by forest plantations. Changes in the spatial patterns of land cover and land use reveal a profound transformation of the landscape and advanced fragmentation of forests. We discuss how these patterns of change threaten the persistence of several endemic species at high risk of extinction. If these anthropogenic processes continue, these species could face an increased risk of extinction.

Keywords: Deforestation; Nahuelbuta Mountain Range; Diversity; Endemism; Landscape-scale Conservation; Effect of Exotic Plantations

ARTICLE INFO

Received: 11 March 2021
Accepted: 6 April 2021
Available online: 15 April 2021

1. Introduction

The state of transformation of a forested landscape is characterized by the degree of habitat destruction and modification, the rate of deforestation, patterns of natural forest fragmentation and changes in the matrix[1,2]. In this sense, one can find little modified landscapes, with low deforestation rates, and extensive and well-connected areas of native forests[1], or highly fragmented and modified landscapes, where forest cover occupies a reduced area of the landscape, with scattered, isolated and non-connected habitat fragments, and a matrix composed of anthropogenic uses[3,4]. Knowledge of the state of transformation of a landscape is a crucial antecedent for proper landscape planning, management and restoration, as it indicates the degree of integrity or functionality of the landscape and the main attributes that need to be maintained or improved for the conservation of biodiversity and human well-being[5]. These antecedents are especially important in landscapes with high diversity and endemism of flora and fauna species, but with substantial losses of forested habitats[4].

Fragmentation of natural forests is one of the main causes of biodiversity loss in the world[6]. The main effects of fragmentation at the landscape level are reduction of forest habitat size and quality, increase in edge and number of patches, and loss of connectivity[7,8]. The edge of a patch in the landscape is defined as the transition zone between habitats, and its perception varies depending on the observer, organism or variable studied[9]. Forest fragments can have smooth or abrupt edges, depending on the degree of contrast between habitats, which plays a critical role in the ability of species to adapt and move within disturbed habitats, the resilience of the forest and the penetration of edge effects that originate a deterioration in habitat quality in regression[8,10,11]. The reduction in forest fragment size generates changes in composition and structure at the community level[12,13], modifying population dynamics and various ecological processes[14,15] that, together with isolation and decreased functional connectivity, have negative effects on the persistence of species populations[8,16]. It has been found that the continued fragmentation and loss of tropical and temperate forests have affected forest richness and structure, bird abundance and diversity, insect community assemblages, and the persistence of mammal populations, among others[12,17-20].

Hotspots are regions with global priority for conservation due to their high degree of endemism and strong anthropogenic impact; alarming figures indicate that more than 85% of the original habitat present in hotspots has been destroyed[21]. Currently, 35 hotspots have been defined, among which the hotspot "Chilean winter rainfall-Valdivian forests" is characterized by hosting a total of 3,893 native vascular plant species, 50% of them endemic to the hotspot itself and containing more than half of the temperate forests of the southern hemisphere[21]. One of the most unique, least protected and most altered landscapes of the Chilean hotspot is that present in the Nahuelbuta mountain range ("cordillera", hereafter)[22]. This area is considered by some scientists as an area of high conservation value and world biodiversity reserve, due to its high levels of species diversity and endemism, and physical characteristics that favored the persistence of some species, even during the last glaciation[22,23]. The high species richness and endemism of this area is attributed to the fact that the deciduous forests and scrublands characteristic of the Mediterranean zone of northern Chile converge with the evergreen Valdivian vegetation of southern Chile, forming a unique eco-tonal ecosystem and for having remnant species from the Mesozoic of Gondwanic origin, and Tertiary species of tropical origin[22].

Despite its high value for biodiversity conservation, various anthropic processes, such as the replacement of native forest by plantations of exotic species, forest fires, firewood extraction, intensive agriculture and overexploitation of native species, have been associated with the degradation and loss of the natural forests of this "cordillera"[22,23]; which has affected different species of flora and fauna endemic to Chile and present in the "cordillera", which are threatened and at risk of extinction, such as the trees, queule (Gomortega keule (Mol.) Baillon) and pitao (Pitavia punctata (R. et P.) Mol.), amphibians, Darwin's frog (Rhinoderma darwini) and Contulmo toad (Eupsohus contulmoensis), among mammals, Darwin's fox (Lycalopex fulvipes) and the marsupial (Dromiciops gliroides)[23,24], among many others. Although there are punctual studies of the transformation of Chile's coastal landscapes[25], there are no spatio-temporal studies on the state of transformation of the "cordillera", and in particular, of the changes in the spatial patterns of its native forests. Because of this, there is no basic information to plan conservation and restoration actions or strategies at a landscape scale.

The objective of this work is to evaluate the state of the landscape and the spatio-temporal changes of native forests in the "cordillera", and from these results, to discuss the implications that such changes have on the persistence of several endangered species of flora and fauna of the "cordillera", dependent on native forest habitats. It is possible to expect that due to the constant loss and substitution of native forest by forest plantations of exotic species and habilitation of areas for agriculture, the "cordillera" is in an advanced state of transformation and fragmentation of natural forests.

2. Materials and methods

The "cordillera" is located between the Biobío River and the Imperial River, in the administrative regions of Biobío and La Araucanía, respectively, and extends for 200 km in a north-south direction, reaching a maximum altitude of 1,530 m asl (**Fig-**

ure 1). Precipitation is 80% concentrated in the autumn and winter months, increasing in 2 gradients, one latitudinal (north-south) and the other longitudinal (west-east), with frequent snowfalls above 1,000 m asl in the winter months[26].

Since there is no study that precisely defines the limits of the “cordillera”, in the present work, limits were established based on geological, pedological, vegetational and altimetric criteria. The geological criterion considers the formation of the “cordillera” in the upper Paleozoic, being much older than the Andes, which dates from the end of the Tertiary[27]. This delimitation excluded marine sedimentary platform sequences and pyroclastic deposits associated with collapse calderas[27,28]. The pedology of the “cordillera” corresponds mostly to ultisols of the Nahuelbuta Association series generated from metamorphic rocks, of clayey, silty-clay loam texture, deep and generally of steep topography[29]. Redelimitation from vegetational formations consisted of smoothing and closing the edges of the product obtained from the previous redelimitations, for which the classification of vegetation floors proposed by Luebert and Pliscoff was used[30]. The altimetric limit was based on the elevation of 200 m asl, due to its spatial congruence with the other criteria analyzed.

To generate thematic maps of land use, Landsat satellite images from spring and summer 1986 (TM) and 2011 (ETM+) were used, with a cloud cover of less than 10%. To facilitate their processing and carry out quantitative comparison of land use coverages, the images were projected at a spatial resolution of 30 × 30 m/pixel and subsequently geometrically, atmospherically and topographically corrected[31]. By means of the C-factor methodology, shadows cast by site topography were removed[32]. To increase the accuracy of the classification, the vegetation indices NDVI, SR, SAVI and LSWI were used[33-35].

A supervised classification was performed for each image using the maximum likelihood statistical method and training points, which represent the patterns of land cover types. A total of 300 training points were taken in different field bells for the classification of the most recent images. While for the 1986 images, land cover maps generated by previous studies[36,37] and local consultation of land

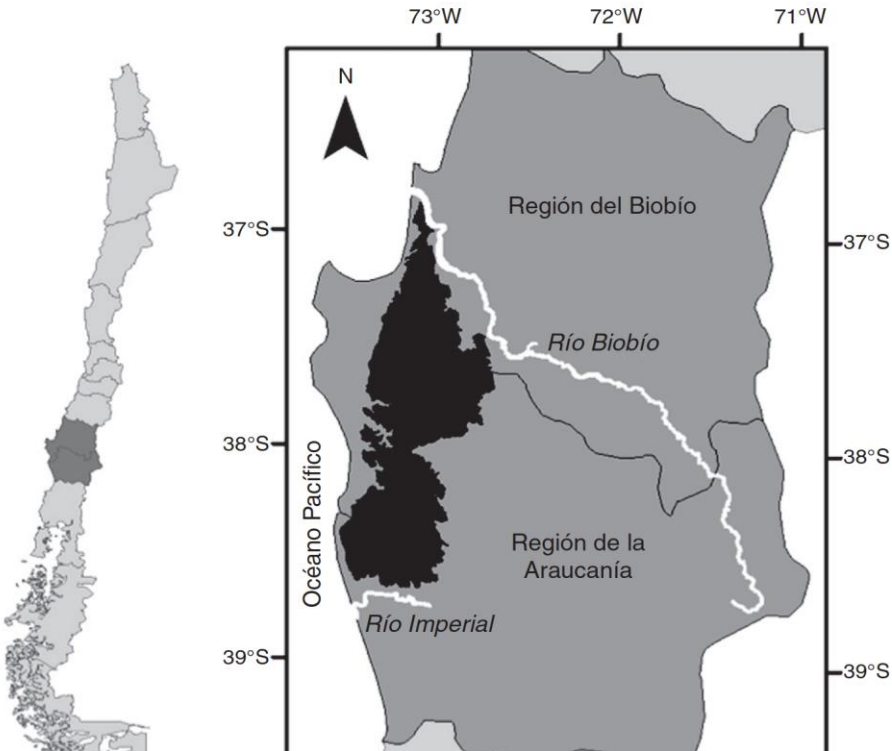


Figure 1. Location of the Nahuelbuta mountain range between the Biob ó River and the Imperial River in the administrative regions of Biob ó and La Araucan á, Chile.

Table 1. Landscape metrics used for the analysis of spatial patterns of native forests in the Nahuelbuta mountain range[38]

Class	Metrics	Description	Unit of measure	Range of variation
Area, density and border	Patch area	Area of each patch of native forest in the landscape	Hectares	0 and no limit
	Larger patch index	Percentage of the area of the largest native forest patch in relation to the total area of the landscape	Percentage	0 < LPI ≤ 100
	Patch density	Number of native forest patches per 100 hectares	Numeric	PD > 0
	Edge density	Density of edge length of native forest patches in the landscape	Meters per hectare	ED ≥ 0, no limit
Form	Average perimeter-area ratio	Relationship between perimeter and area	None	FOR > 0 and without limit
Core area	Average core area	Core area of the native forest, specified by the depth of the edge according to the proximity of the patch to other land uses	Hectares	CORE ≥ 0 and no limit
Isolation proximity	Average distance to nearest neighbor	Proximity of native forest patches, based on weighted average distance	Meters	ENN > 0 and no limit
Contrast	Area-weighted mean edge contrast	Degree of contrast of native forest with respect to its density. The contrast represents the magnitude of the difference between native forest and other land uses	Percentage	0 ≤ ECON ≤ 100
Contagion interspersion	Aggregation index	Proportional deviation of adjacencies involving native forest class from that expected for a spatially random distribution	Percentage	-1_CLUMPY_1

use covers that showed no change of use in the last 20 years were used. The classification accuracy of the images was calculated by means of confusion matrices, from a range of between 60 and 90 validation points taken in the field and recent aerial photographs, for each type of cover, of each satellite image. A classification filter was applied to each of the cover classes present in order to exclude patches with less than 4 pixels.

The land cover or land use classes of the "cordillera" landscape identified from each image were: 1) primary native forest (old-growth forest originating from natural succession); 2) secondary native forest (forest regeneration following disturbance); 3) stunted forest; 4) arborescent scrub; 5) shrub-land; 6) exotic forest plantations (commercial plantations); 7) agricultural land (agricultural crops, grasslands and livestock use areas); and 8) other uses and cover (water bodies, urban, cloud and shade).

The transition from native forest cover to other land uses was analyzed using the Change Analysis module of the Land Change Modeler extension of the IDRISI software[39]. In the case of changes in native forests, the following formula was used to determine the annual deforestation rate:

$$P = \left[\frac{A_2^{\left(\frac{1}{(t_2 - t_1)}\right)}}{A_1} - 1 \right] \times 100$$

where A_1 and A_2 are the area of native forest at time t_1 and t_2, respectively, and P is the percentage loss per year[40].

The current state of the landscape was analyzed based on the landscape change models proposed by McIntyre and Hobbs and Echeverría *et al.*[1,2]. Specifically, the following variables were used: loss and fragmentation of native forests, deforestation rate, dominant landscape process, and changes in the matrix during the period studied. On the other hand, landscape transformation was measured as percentage of remaining habitat in terms of natural forests[2].

For the analysis of spatial patterns of native forest cover between 1986 and 2011, FRAGSTATS software[38] was used. The choice of the set of metrics to use was based on the review of several studies on landscape metrics that best expressed the spatial configuration of real landscapes, and were also representative of essential components of landscape structure such as: patch quality, patch edge, patch context in the landscape, and patch connectivity (**Table 1**)[5,41-44].

For the calculation of the contrast index, contrast weights were assigned between the edges of native forest patches and the other cover types present in the study area. The contrast weight of the edges was determined based on variables of vegetation composition and structure, measured in 13 sampling plots of 20 × 10 m. For the calculation of the core area, perpendicular distances were used from the edge to the center of the patch, which corresponded to the edge effect zone between the native forest and the other cover types. The distance assigned was based on the work of Laurance *et al*.[14], López-Barrera[9] and Lindenmayer and Fischer[8], considering that the degree of contrast between habitats (soft or abrupt edges) expresses the magnitude and distance of primary and secondary responses of the structure, composition and processes of the habitat to edge effects.

3. Results

Image classification. The accuracy of the 1986 classification was 87%, with the exotic plantation class being the least accurate, being confused with primary native forest cover. Primary native forests for 1986 had an accuracy of 88%, while secondary and stunted native forests had an accuracy of 86 and 87%, respectively. For 2011, the classification accuracy was estimated at 85%, with the arborescent shrubland class being the least accurate (77%). Plantation forests had 78% accuracy, while primary native forests obtained 86%, secondary native forests 85% and stunted forests 88%.

Loss of forest cover. The “cordillera” has an area of approximately 620,000 ha, of which 206,130 ha ±26.8 ha were native forests in 1986, decreasing to 137,700 ha ±19.3 ha in 2011 (**Figure 2**). In other words, in 25 years the forest cover decreased by 33.2% with a deforestation rate of 1.6% per year. The net loss was more intense in primary native forests than in secondary native forests, while stunted forests showed no loss. In 1986, primary native forests occupied only 8% of the total landscape (52,019 ha), reducing their area by half in 2011, with a deforestation rate of 2.6% per year (**Figure 2**). In comparison, secondary native forests presented a net loss of 43,000 ha, representing 28% of the original area in 1986 (152,200 ha), with a deforestation rate of 1.3% per year (**Figure 2**).

Regarding transitions from primary native forests, 68% of the net loss was due to substitution to plantations of exotic species and, to a lesser extent, degradation to secondary native forests (11%) and arborescent shrublands (12%). On the other hand, 95% of the changes in secondary native forest cover corresponded to substitution by plantations of exotic species. In this same study period (1986-2011), plantations of exotic species increased by almost 150% from 90,750 ha in 1986 to 251,250 ha in 2011, with an afforestation rate of 4.2% per year, being the predominant cover in 2011 (**Figure 2**).

Analysis of spatial patterns of native forest cover. Considerable changes were observed in the patch size distribution of forest cover during the period 1986–2011 (**Figure 3**). In 1986, 61% of the primary native forest area was in patches smaller than 50 ha, 22% in patches of 50 to 500 ha and 17% distributed in 7 patches larger than 500 ha (**Figure 3**). Subsequently, in 2011, of the 26,600 ha of existing primary native forest, 72% was distributed in patches smaller than 50 ha, 18% in patches of 50 to 500 ha and only 10% was distributed in 3 patches larger than 500 ha (**Figure 3**). In 2011, a single large patch of primary native forest of 1,120 ha was observed. Of the 25,420 ha of primary native forest deforested, 49% occurred in patches of less than 50 ha in area.

In relation to the secondary native forest, approximately 50% was in patches of less than 50 ha in area, with only one large fragment persisting over time, representing 4.3% of the total area of the landscape in 1986 and 4.5% in 2011 (**Figure 3**). In the study period, the secondary native forest presented an approximate regeneration of 5,680 ha, of which 82% occurred in patches of 2,000 to 5,000 ha. However, of the 43,000 ha of secondary native forest deforested, 53% of the loss occurred in patches of less than 50 ha in size.

From 1986 to 2011, there was an increase in the complexity of the shape of primary and secondary native forest fragments (**Table 2**). The degree of habitat contrast was greater in primary native forest

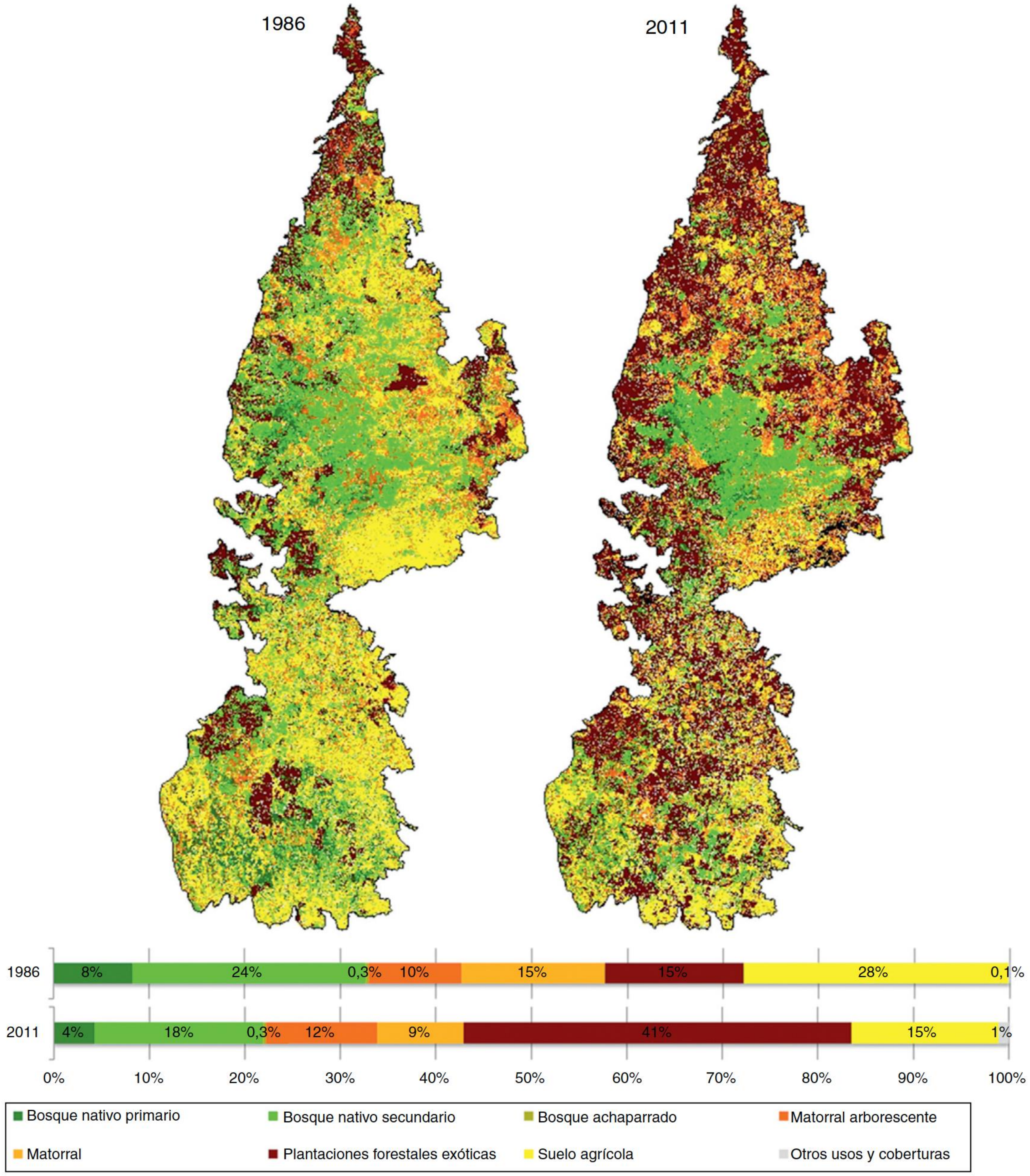


Figure 2. Spatio-temporal variation and percentage of the area occupied by land cover types in 1986 and 2011 in the Nahuelbuta mountain range.

than in secondary forest, with a slight tendency to decrease over time (**Table 2**). Stunted forests presented the lowest degree of contrast in the study period (1986 = 38%, 2011 = 29%) (**Table 2**).

In 1986, the density of primary and secondary native forest patches was 3.73 and 5.58, respectively (**Table 2**). In 2011, this index decreased to 2.51 fragments for primary native forest and 4.62 fragments for secondary native forest (**Table 2**). Similarly, edge density decreased from 1986 to 2011 for all 3 forest types, with the greatest edge reduction occurring in secondary native forest (**Table 2**).

An increase in the distance between primary and secondary native forest patches was observed (**Table 2)**. The aggregation index of primary native forest decreased, while that of secondary native forest

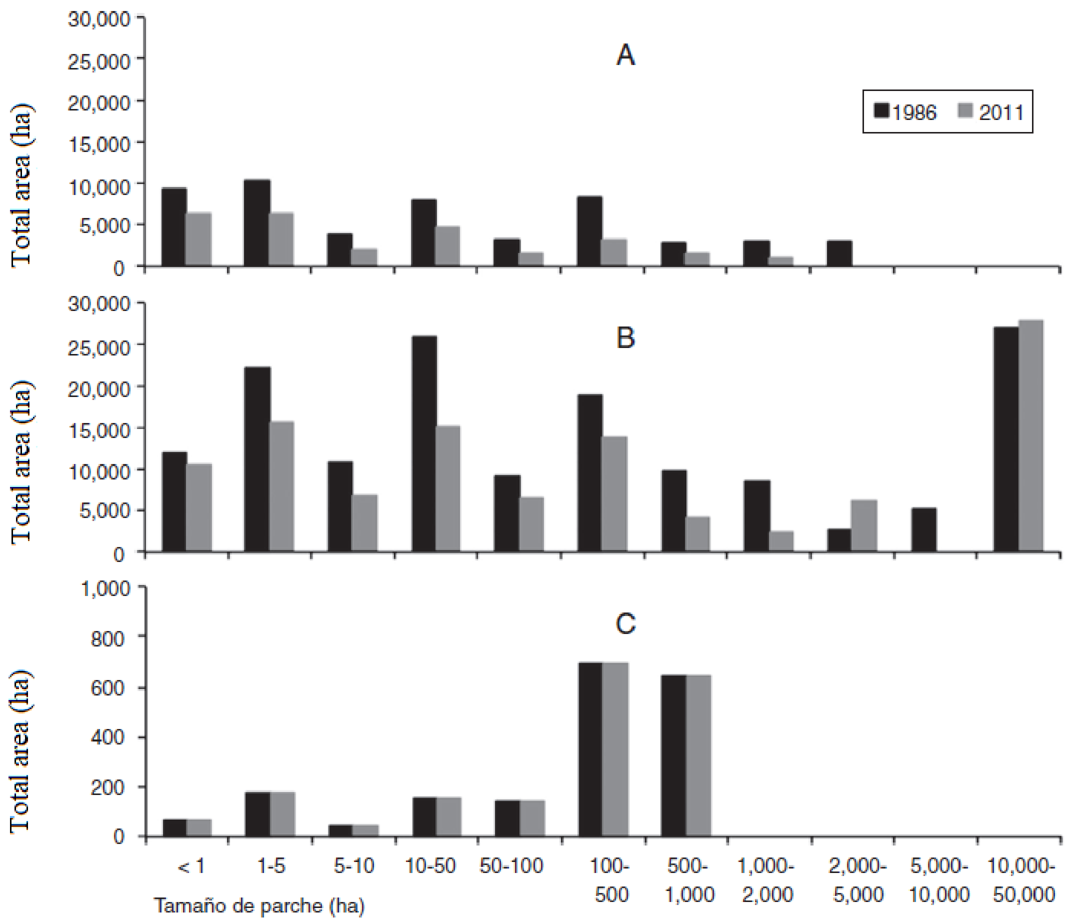


Figure 3. Temporal variation of patch size (ha) of primary **(A)**, secondary **(B)** and stunted **(C)** native forests in the Nahuelbuta mountain range.

increased (**Table 2**). However, there was no considerable variation in the aggregation index of forest fragments in the landscape. On the other hand, stunted forests did not show changes in the mean distance between fragments and in their aggregation index (**Table 2**).

The core area of primary native forest had a 4% loss, which is consistent with a 63% decrease in the area of the largest patch for this forest type (**Table 2**). In contrast, secondary native forest and stunted forest increased by 20% and 12% in core area, respectively (**Table 2**).

Table 2. Changes in landscape pattern indices in the Nahuelbuta mountain range at the class level between 1986 and 2011

Type of forest cover	Class	Contrast	Form	Area, density and border			Isolation proximity	Contagion interspersion	Core area
	Metrics	Area-weighted mean edge contrast	Average perimeter-area ratio	Patch density	Edge density	Larger patch index	Average distance to nearest neighbor	Aggregation index	Average core area
Primary native forest	1986	44.72	776.46	3.73	30.14	0.49	133.99	0.63	0.79
	2011	42.16	817.77	2.51	19.82	0.18	154.67	0.57	0.76
Secondary native forest	1986	60.17	731.04	5.58	76.45	4.33	93.01	0.59	0.85
	2011	56.70	785.56	4.62	55.54	4.52	105.09	0.62	1.06
Stunted forest	1986	38.20	654.97	0.04	0.65	0.10	113.14	0.79	4.09
	2011	29.03	654.97	0.04	0.61	0.10	113.14	0.79	4.64

4. Discussion

Loss of forest cover. The present study revealed a total loss of native forest of 33.2% at a deforestation rate of 1.6% per year between 1986 and 2011 in a landscape of high conservation value, located in a biodiversity hotspot. This loss was more intense in primary native forests (49%) than in secondary native forests (28%), and occurred mainly due to substitution to exotic forest plantations. A loss of native forest similar to that of the present study was also observed in the Chilean hotspot, between the regions of Biobío and La Araucanía in the period 1979–2000, where the net forest loss was 28.2% at a deforestation rate of 1.6% per year[45]. Of this percentage of loss, 71.7% was due to forest substitution by exotic forest plantations[45]. On the other hand, in the Chilean coastal landscape of the Maule region, between 1975 and 2000, a loss of 67% of native forests was estimated, double that recorded in the present study, at a deforestation rate of 4.5% per year, mainly as a consequence of substitution by exotic forest plantations[3]. This reflects the constant loss of native forest by substitution to exotic forest plantations and the current absence of large extensions of native coastal forests in Chilean hotspot landscapes.

Loss of forest habitat in biodiversity hotspots has also been reported in other areas of the world in recent decades. For example, in the hotspot of Limón province in Costa Rica, deforestation in 1997 was 54,830 ha, equivalent to 8.4% of the forest cover existing in 1986[46]. Higher percentages of deforestation than estimated in the “mountain range” were observed in the Eastern Arc mountain hotspot of Tanzania, where natural evergreen forests and wooded savannas disappeared by 48% between 1975 and 2000[47], and in the Ecuador hotspot, located in the provinces of Loja and Zamora Chinchipe, where forest cover loss was 46% between 1976 and 2008[48].

Landscape condition. The landscape exhibited a deforestation rate of 2.6% per year for primary native forest and 1.3% per year for secondary native forest, with 22% of the forest cover persisting in 2011. Likewise, there was a decrease in the density of patches and edges of primary and secondary native forest, an increase in the distance between patches, and an increase in the matrix dominated by plantations of exotic species. According to these results, the “cordillera” corresponds to an advanced stage of landscape transformation[1,2]. In this landscape state, the loss of forest fragments dominates over forest partitioning and usually occurs subsequent to intense forest fragmentation processes[1,2]. Very different was observed in India between 1975 and 2005[49], and in the biodiversity hotspot in southern Ecuador[48], between 1976 and 2008, where there was a reduction and division of large forest patches, while there was an increase in the number of patches, length and edge density, a pattern characteristic of initial phases of fragmentation[40].

If the trajectory of the studied landscape continues, and without planning and management measures for the conservation, protection and restoration of forest ecosystems, a considerable loss of forest cover is expected in the future, with a decrease in the rate of deforestation. Likewise, if the current rate of afforestation with exotic species remains constant at the estimated rate (4.2% per year), it is possible to expect a greater homogenization of the landscape with one or two species of commercial interest, turning the “cordillera” into a relictual landscape, severely deforested, with small patches of highly modified native forest, isolated and without connectivity, surrounded by highly contrasting land uses[1,2]. In this state, the slight increase in secondary native forests and changes in trajectories would indicate regeneration in abandoned agricultural areas, as has been reported in the commune of Ancud, Los Lagos region, Chile[58].

4.1 Changes in the spatial pattern of forest cover and implications for biodiversity conservation

Impacts of the decrease in the size of habitat patches. The rapid process of deforestation in the “cordillera” caused the loss of large fragments of native primary and secondary forest in just 25 years, with few patches larger than 1,000 ha and 50% of the forest area in patches smaller than 50 ha. In this sense, in the coastal landscape of the Maule-Cobquecura river in Chile (north of the “cordillera”), it was found that during the period 1975–2000 in the first year of study, 44% of the forest area was concentrated in a large patch of between 20,000 and 100,000 ha. By 2000, 69% of the forests were in patches of less than 100 ha and only 3% of the forest area was in patches larger than 1,000 ha[3]. This reduction in the size of native forest patches in Chilean landscapes may affect the availability of habitat for various species that require large areas to persist[14,19,59]. In this sense, it has been reported that carnivore species present in the “cordillera”, such as the puma (*Puma concolor*)

Table 3. Linkage between spatial patterns reported in the present study and ecological processes of threatened species in the Nahuelbuta mountain range

Spatial pattern of native forests in the Nahuelbuta mountain range reported by the present study	**Ecological impacts reported by other studies for endangered species in the Nahuelbuta mountain range**				
	Species	**Current category**	**Source**	**Impacts**	**References**
Loss of surface area. Reduction of the number of patches. Few patches with area greater than 1,000 ha. 50% of forest area in patches < 50 ha. Decrease of the area of the largest patch of primary native forest	Lycalopex culpaeus Lycalopex griseus Lycalopex fulvipes Leopardus guigna Puma concolor Galictis cuja Conepatus chinga	Vulnerable Minor concern In danger Vulnerable Near threatened Vulnerable Rare	RCE RCE RCE RCE RCE CAZA CAZA	Loss of forest cover reduces habitat for native carnivore species that prefer large patches of mature forest. Increased levels of spatial overlap, competition for prey and territory are expected.	Moreira-Arce *et al.*[50]
	Campephilus magellanicus	In danger	RCE	The lack of patches of primary native forest with large areas of interior habitat in the landscape could affect the survival of the species. To form family groups, it requires patches with an area greater than 100 ha of mature forests of the genus Nothofagus. Between 1986 and 2011, a 56.7% reduction of the species' habitat in Nahuelbuta was estimated.	Llabrés[51]
	Birds	-	-	The composition of avifauna is significantly different between forest habitat types, which could be attributed to differences in composition and structure between habitats, as a result of landscape fragmentation processes.	Fontúrbel and Jiménez[52]
	Scelorchilus rubecula Scytalopus magellanicus	Minor concern Minor concern	UICN UICN	The composition and structure of the understory is a key component of species habitat. Reduction in understory complexity and diversity affects site availability for species.	Moreno-García *et al.* [53]
	L. culpaeus *L.* griseus *L.* fulvipes *L.* guigna	Vulnerable Minor concern In danger Vulnerable	RCE RCE RCE RCE	The probability of site use by specialist carnivores increases as structural as the structural diversity of the habitat diversity of the habitat increases. Therefore, changes in vegetation structure, as a result of fragmentation processes, decrease the availability and use of habitats for the species.	Moreira-Arce *et al.*[54]
High rate of contrast between primary and secondary native forests. Increase in the complexity of the shape of the fragments.	Alsodes vanzolinii	Endangered-rare	RCE	Their populations are associated with streams in small remnant patches of native forest surrounded by forest plantations of exotic species. This affects the movement of amphibians in the landscape, which depend on connected habitats for the development of their life cycle.	Rabanal and Alarcón[55]
	Telmatobufo bullocki	Vulnerable-rare	RCE	Erosion processes related to harvesting and management of forest plantations sediment water bodies associated with their habitat and biophysical changes, which limits food resources and eliminates adequate conditions for their reproduction.	Soto-Azat *et al.*[56]
Increased isolation between primary and secondary native forest fragments	Pitavia punctata	In danger	RCE	Degraded habitats, isolated and surrounded by plantations of exotic species. Low genetic diversity, high degree of genetic differentiation between localities and a high level of isolation, which makes the species sensitive to fragmentation and increases its risk of extinction.	Venegas[57]

and Darwin's fox, show preferences for mature native forest habitats and large patches, which would concentrate their populations in a few patches in the "cordillera", increasing the levels of spatial overlap between carnivore species, including the feral dog, and competition for prey and territory[50] (**Table 3**). For their part, threatened bird species present in the "cordillera" and associated with primary inland native forests, such as the black woodpecker (*Campephilus magellanicus*), have experienced a decline in their habitats, along with a decrease in the size and distribution range of their populations[51] (**Table 3**).

Different studies conducted in diverse ecosystems show that fragmentation and deforestation alter a set of variables related to forest structure; likewise, fragment size is significantly related to species composition and community structure[12,13,18,59,60]. Structural differences in vegetation have been associated with changes in bird composition between habitats in the "cordillera"[52,53] (**Table 3**), possibly responding to variables such as increased irradiance and temperature[61]. Likewise, it has been shown that the probability of site use by carnivores at risk of extinction, such as the guigna (*Leopardus guigna*), Darwin's fox and puma, increases as the structural diversity of the habitat increases[54] (**Table 3**).

Edge effects. In the present study, primary and secondary native forests exhibited a high contrast due to the interface between anthropogenic land uses and secondary native vegetation. Previous studies in Chile and southern Portugal report that eucalyptus (*Eucalyptus globulus* Labill) plantations present high contrast edges with the natural habitat due to their monospecific composition and simple structure[62,63], which can directly affect the movement of organisms[11]. In the "cordillera" populations of amphibians at risk of extinction, such as Vanzolini's spiny-breasted toad and Bullock's toad (*Telmatobufo bullocki*), have been found in small remnants of native forest surrounded by plantations of exotic species. These plantations affect the mobility of amphibians and the development of their life cycle[55,56] (**Table 3**). Likewise, erosion processes related to harvesting and forest plantation management generate bio- physical changes and sedimentation of water bodies associated with their habitat, eliminating suitable conditions for their reproduction[56] (**Table 3**).

Isolation effects between habitats. A discontinuous spatial pattern of habitats, such as the one described for the study area, can lead to a decrease in functional connectivity between suitable sites for species and to an alteration of dispersal capacity depending on the conditions of the matrix that separates them[10,59]. This in turn induces an increased risk of local extinctions by making species more vulnerable to stochastic processes, natural catastrophes, human threats and loss of genetic variability[8,10,64]. In this sense, it has been shown that populations of the tree species pitao (*P. punctata*) in the "cordillera", and of restricted distribution in the coastal mountain range in Chile, present low genetic variability within each locality, and a high degree of genetic differentiation between localities due to the effect of isolation between populations[57] (**Table 3**), compared to other tree species of south-central Chile, such as queuleyelhualo (*Lophozonia glauca*) Heenan and Smissen[65,66]. Accordingly, it can be inferred that *P. punctata* is in a genetic bottleneck, even more so if one considers that its area of occupancy is less than 1,000 ha and that less than 1,000 mature individuals persist in the wild[24].

The Nahuelbuta mountain range should be considered a premium conservation landscape, as it still concentrates high levels of biodiversity and endemism, but at the same time an advanced state of transformation and progressive fragmentation of native forests. This premium condition within the biodiversity hotspot justifies the urgency of safeguarding the different species of flora and fauna whose risk of extinction may increase due to the loss and modification of their habitats. This condition is also based on their low degree of protection (±10.4%), with 10 areas of high conservation value with an approximate surface area of 57,500 ha, with Caramávida Creek being the largest conservation area in the landscape (37,000 ha). The only two areas protected by the National System of State Protected Wildlife Areas are Nahuelbuta National Park (6,832 ha) and Contulmo Natural Monument (82 ha), whose size is difficult to sustain the area's high biodiversity[67,68].

We suggest the urgent implementation of strategies for conservation planning in the Nahuelbuta mountain range that involve: restoration of degraded ecosystems, connectivity of forest fragments in the landscape, conservation of threatened species in the landscape, updating of priority sites for biodiversity conservation, implementation of new protected areas, and analysis of the ecological impacts of habitat loss on species and communities of flora and fauna.

Finally, it is suggested that future research should address the causes of landscape transformation and the loss of native forest in the "cordillera", in order to be included in actions for biodiversity conservation and territorial planning.

Conflict of interest

The authors declared no conflict of interest.

References

1. Echeverría C, Newton A, Nahuelhual L, *et al*. How landscapes change: Integration of spatial patterns and human processes in temperate landscapes of southern Chile. Applied Geography 2012; 32: 822–831.
2. McIntyre S, Hobbs R. A framework for conceptualizing human effects on landscapes and its relevance to management and research models. Conservation Biology 1999; 13: 1282–1292.
3. Echeverría C, Coomes D, Salas J, *et al*. Rapid deforestation and fragmentation of Chilean Temperate Forests. Biological Conservation 2006; 130(4): 481–494.
4. Steininger M, Tucker CJ, Ersts P, *et al*. Clearance and fragmentation of tropical deciduous forest in the Tierras Bajas, Santa Cruz, Bolivia. Conservation Biology 2001; 15(4): 856–866.
5. Liu J, Taylor WW. Integrating landscape ecology into natural resource management. Cambridge, UK: Cambridge University Press; 2002.
6. Turner IM. Species loss in fragments of tropical rain forest: A review of the evidence. Journal of Applied Ecology 1996; 33(2): 200–209.
7. Forman RT, Godron M. Landscape ecology. New York: Wiley; 1986.
8. Lindenmayer DB, Fischer J. Habitat fragmentation and landscape change: An ecological and conservation synthesis. Washington: Island Press; 2006. p. 1–329.
9. López-Barrera F. Estructura y función en bordes de bosques (Spanish) [Structure and function in forest edges]. Revista Ecosistemas 2004; 13(1): 55–68.
10. Bennett AF. Linkages in the landscape: The role of corridors and connectivity in wildlife conservation. Gland, Switzerland and Cambridge, UK: IUCN; 1999. 1–241.
11. Peyras M, Vespa NI, Bellocq MI, *et al*. Quantifying edge effects: The role of habitat contrast and species specialization. Journal of Insect Conservation 2013; 17(4): 807–820.
12. Echeverría C, Newton A, Lara A, *et al*. Impacts of forest fragmentation on species composition and forest structure in the temperate landscape of southern Chile. Global Ecology and Biogeography 2007; 16(4): 426–439.
13. Tabarelli M, Mantovani W, Peres CA. Effects of habitat fragmentation on plant guild structure in the montane Atlantic forest of southeastern Brazil. Biological Conservation 1999; 91(2-3): 119–127.
14. Laurance WF, Lovejoy TE, Vasconcelos HL, *et al*. Ecosystem decay of Amazonian forest fragments: A 22-year investigation. Conservation Biology 2002; 16(3): 605–618.
15. Wiegand T, Revilla E, Moloney KA. Effects of habitat loss and fragmentation on population dynamics. Conservation Biology 2005; 19(1): 108–121.
16. Crooks KR, Burdett CL, Theobald DM, *et al*. Global patterns of fragmentation and connectivity of mammalian carnivore habitat. Philosophical Transactions of the Royal Society B: Biological Sciences 2011; 366(1578): 2642–2651.
17. Barbosa O, Marquet PA. Effects of forest fragmentation on the beetle assemblage at the relict forest of Fray Jorge, Chile. Oecologia 2002; 132(2): 296–306.
18. Castelletta M, Thiollay JM, Sodhi NS. The effects of extreme forest fragmentation on the bird community of Singapore Island. Biological Conservation 2005; 121(1): 135–155.
19. De Angelo C, Paviolo A, di Bitetti M. Differential impact of landscape transformation on pumas (*Puma concolor*) and jaguars (*Panthera onca*) in the Upper Paraná Atlantic Forest. Diversity and Distributions 2011; 17(3): 422–436.
20. Stratford JA, Stouffer PC. Forest fragmentation alters microhabitat availability for Neotropical terrestrial insectivorous birds. Biological Conservation 2015; 188: 109–115.
21. Mittermeier RA, Turner WR, Larsen FW, *et al*. Global biodiversity conservation: The critical role of hotspots. In: Zachos FE, Habel JC (editors). Biodiversity hotspots: Distribution and protection of conservation priority areas. Berlin, Heidelberg: Springer; 2011. p. 3–23.
22. Smith-Ramírez C. The Chilean coastal range: A vanishing center of biodiversity and endemism in South American temperate rainforests. Biodiversity & Conservation 2004; 13(2): 373–393.
23. Wolodarsky-Franke A, Díaz S. Cordillera de Nahuelbuta Reserva mundial de biodiversidad (Spanish) [Nahuelbuta Mountain Range World Biodiversity Reserve]. Valdivia, Chile: World Wildlife Fund; 2011. p. 1–56.
24. Hechenleitner V, Gardner MF, Thomas PI, *et al*.

Plantas amenazadas del centro-sur de Chile. Distribución, conservación y propagación (Spanish) [Threatened plants of south-central Chile. Distribution, conservation and propagation]. Valdivia, Chile: Trama Impresores S.A.; 2005.
25. Miranda A, Altamirano A, Cayuela L, *et al*. Native forest loss in the Chilean biodiversity hotspot: Revealing the evidence. Regional Environmental Change 2017; 17(1): 285–297.
26. Di Castri F, Hajek ER. Bioclimatología de Chile (Spanish) [Bioclimatology of Chile]. Santiago: Editorial Universidad Católica de Chile; 1976.
27. Mardones M. La cordillera de la Costa: caracterización físico-ambiental y regiones morfoestructurales (Spanish) [The Coastal Cordillera: Physical-environmental characterization and morphostructural regions]. In: Smith-Ramírez C, Armesto JJ, Valdovinos C (editors). Historia, biodiversidad y ecología de los bosques costeros de Chile. Santiago, Chile: Editorial Universitaria; 2005. p. 39–59.
28. SERNAGEOMIN (Servicio Nacional de Geología y Minería). Mapa geológico de Chile: Version digital. Publicación Geológica Digital, Núm. 4 (CD-ROM, versión1.0) (Spanish) [Geological map of Chile: Digital version. Digital Geological Publication, No. 4 (CD-ROM, version 1.0)]. Santiago: Servicio Nacional de Geología y Minería; 2003. p. 1–25.
29. CIREN (Centro de Información de Recursos Naturales). Descripciones de suelos. Materiales y símbolos. Estudio agrológico VII y IX Región (Spanish) [Soil descriptions. Materials and symbols. Agrological study VII and IX Region]. Santiago: Centro de Information de Recursos Naturales (CIREN); 1999.
30. Luebert F, Pliscoff P. Sinopsis bioclimática y vegetacional de Chile (Spanish) [Bioclimatic and vegetational synopsis of Chile]. Santiago: Editorial Universitaria; 2006.
31. Chander G, Markham BL, Helder DL. Summary of current radiometric calibration coefficients for Landsat MSS, TM, ETM+, and EO-1 ALI sensors. Remote Sensing of Environment 2009; 113(5): 893–903.
32. Reese H, Olsson H. C-correction of optical satellite data over alpine vegetation areas: A comparison of sampling strategies for determining the empirical c-parameter. Remote Sensing of Environment 2011; 115(6): 1387–1400.
33. Huete AR. A soil-adjusted vegetation index (SAVI). Remote Sensing of Environment 1988; 25(3): 295–309.
34. Rouse Jr JW, Haas RH, Schell JA, *et al*. Monitoring vegetation systems in the Great Plains with ERTS-1. In: Freden SC, Mercanti EP, Becker MA (editors). Third Earth Resources Technology Satellite-1 Symposium. Volume 1: Technical Presentations, section A. The Proceedings of a Symposium Held by Goddard Space Flight Center at Washington, D.C.; 1974 Dec 10-14; Washington. Washington: NASA; 1974. p. 309–317.
35. Tucker CJ. Red and photographic infrared linear combinations for monitoring vegetation. Remote Sensing of Environment 1979; 8(2): 127–150.
36. Corporación Nacional Forestal (Conaf), Comisión Nacional de Medio Ambiente (Conama), International Bank for Reconstruction and Development (IBRD), Universidad Austral de Chile, Pontificia Católica de Chile and Universidad Católica de Temuco. Catastro y evaluación de los recursos vegetacionales nativos de Chile: monitoreo de cambios. Informe nacional con variables ambientales (Spanish) [Cadastre and evaluation of native vegetational resources of Chile: Change monitoring. National report with environmental variables]. Santiago: National Government of Chile; 1999.
37. Lara A, Araya L, Capella J, *et al*. Evaluación de la destrucción y disponibilidad de los recursos forestales nativos en la VII y VIII Región (Spanish) [Evaluation of the destruction and availability of native forest resources in the VII and VIII Region]. Santiago: Informe Técnico, Comité Pro Defensa Fauna y Flora; 1989.
38. McGarigal K, Cushman S, Neel M, *et al*. FRAGSTATS: Spatial pattern analysis program for categorical maps [Internet]. Amherst, MA: University of Massachusetts; (Landscape Ecology Program); 2002 [accessed 2013 Aug 7]. Available from: http://www.umass.edu/landeco/research/fragstats/fragstats.html
39. IDRISI (software). Selva help system. Clark University: Clark Labs; 2012.
40. Newton AC. Biodiversity loss and conservation in fragmented forest landscapes: The forests of montane Mexico and temperate South America. Wallingford, Oxford UK: CABI; 2007. p. 1–416.
41. Cushman SA, McGarigal K, Neel MC. Parsimony in landscape metrics: strength, universality, and consistency. Ecological Indicators 2008; 8(5): 691–703.
42. Mas JF, Gao Y, Pacheco JAN. Sensitivity of landscape pattern metrics to classification approaches. Forest Ecology and Management 2010; 259(7): 1215–1224.
43. Peng J, Wang Y, Zhang Y, *et al*. Evaluating the effectiveness of landscape metrics in quantifying spatial patterns. Ecological Indicators 2010; 10(2): 217–223.
44. Zeng H, Wu X. Utilities of edge-based metrics for studying landscape fragmentation. Computers, Environment and Urban Systems 2005; 29(2): 159–178.
45. Aguayo M, Pauchard A, Azócar G, *et al*. Land use change in the south central Chile at the end of the 20th century. Understanding the spatio-temporal dynamics of the landscape. Revista Chilena de Historia Natural 2009; 82(3): 361–374.
46. Van Laake PE, Sánchez-Azofeifa GA. Focus on deforestation: Zooming in on hot spots in highly fragmented ecosystems in Costa Rica. Agriculture, Ecosystems & Environment 2004; 102: 3–15.
47. Green JMH, Larrosa C, Burgess ND, *et al*. Deforestation in an African biodiversity hotspot: Extent,

variation and the effectiveness of protected areas. Biological Conservation 2013; 164: 62–72.

48. Tapia-Armijos MF, Homeier J, Espinosa CI, *et al*. Deforestation and forest fragmentation in South Ecuador since the 1970s–Losing a hotspot of biodiversity. PloS one 2015; 10(9): 1–18.
49. Reddy CS, Sreelekshmi S, Jha C, *et al*. National assessment of forest fragmentation in India: Landscape indices as measures of the effects of fragmentation and forest cover change. Ecological Engineering 2013; 60: 453–464.
50. Moreira-Arce D, Vergara PM, Boutin S. Diurnal human activity and introduced species affect occurrence of carnivores in a human-dominated landscape. PloS one 2015; 10(9): 1–19.
51. Llabrés VJ. Variación de los patrones espaciales del bosque native adulto de Nothofagus de la cordillera de Nahuelbuta y su relación con la abundancia de Campephilus magallanicus (King, 1828) (Spanish) [Variation in the spatial patterns of the adult native Nothofagus forest of the Nahuelbuta mountain range and its relationship with the abundance of Campephilus magallanicus] [Unpublished thesis]. Concepción, Chile: Facultad de Ciencias Forestales, Universidad de Concepción; 2015.
52. Fontúrbel FE, Jiménez JE. Does bird species diversity vary among forest types? A local-scale test in Southern Chile. Naturwissenschaften 2014; 101(10): 855–859.
53. Moreno-García RA, Zamora R, Herrera MA. Habitat selection of endemic birds in temperate forests in a biodiversity “Hotspot”. Forest Systems 2014; 23(2): 216–224.
54. Moreira-Arce D, Vergara PM, Boutin S, *et al*. Mesocarnivores respond to fine-grain habitat structure in a mosaic landscape comprised by commercial forest plantations in southern Chile. Forest Ecology and Management 2016; 369: 135–143.
55. Rabanal FE, Alarcón D. Amphibia, Anura, Cycloramphidae, *Alsodes vanzolinii* (Donoso-Barros, 1974): Rediscovery in nature, latitudinal and altitudinal extension in Nahuelbuta Range, southern Chile. Check List 2010; 6(3): 362–363.
56. Soto-Azat C, Cuevas C, Flores E, *et al*. Conservación de Telmatobufo bullocki (sapo de Bullock) y su hábitat en los bosques degradados de Nahuelbuta (Spanish) [Conservation of Telmatobufo bullocki (Bullock’s toad) and its habitat in degraded forests of Nahuelbuta]. In: Soto-Azat C, Valenzuela-Sánches A (editors). Conservación de anfibios de Chile: Memorias del taller de conservación de anfibios para organismos públicos (Spanish) [Amphibian conservation in Chile: Proceedings of the amphibian conservation workshop for public agencies]. Santiago: Universidad Nacional Andrés Bello; 2012. p. 70–75.
57. Venegas CF. Variabilidad genética en subpoblaciones de Pitavia punctata Mol. Especie endémica amenazada del sur de Chile (Spanish) [Genetic variability in subpopulations of Pitavia punctata Mol. Threatened endemic species of southern Chile] [PhD thesis]. Concepción, Chile: Faculty of Forestry Sciences, Universidad de Concepción; 2015.
58. Carmona A, Nahuelhual L, Echeverría C, *et al*. Linking farming systems to landscape change: An empirical and spatially explicit study in southern Chile. Agriculture, Ecosystems & Environment 2010; 139(1-2): 40–50.
59. Uezu A, Metzger JP. Vanishing bird species in the Atlantic Forest: Relative importance of landscape configuration, forest structure and species characteristics. Biodiversity and Conservation 2011; 20(14): 3627–3643.
60. Cadotte MW, Franck R, Reza L, *et al*. Tree and shrub diversity and abundance in fragmented littoral forest of southeastern Madagascar. Biodiversity & Conservation 2002; 11(8): 1417–1436.
61. Patten MA, Smith-Patten BD. Testing the microclimate hypothesis: Light environment and population trends of Neotropical birds. Biological Conservation 2012; 155: 85–93.
62. Echeverría C, Gatica P, Fuentes R. Habitat edge contrast as an indicator to prioritize sites for ecological restoration at the landscape scale. Natureza & Conservação 2013; 11: 170–175.
63. Reino L, Beja P, Osborne PE, *et al*. Distance to edges, edge contrast and landscape fragmentation: Interactions affecting farmland birds around forest plantations. Biological Conservation 2009; 142(4): 824–838.
64. Hooftman DA, Billeter RC, Schmid B, *et al*. Genetic effects of habitat fragmentation on common species of Swiss fen meadows. Conservation Biology 2004; 18(4): 1043–1051.
65. Delaveau C, Fuentes-Arce G, Ruiz E, *et al*. Variabilidad genética mediante AFLP en tres relictos de Gomortega keule (Molina) Baillon: Especie endémica chilena en peligro de extinción (Spanish) [Genetic variability by AFLP in three relicts of Gomortega keule (Molina) Baillon: Endangered Chilean endemic species]. Gayana Botánica 2013; 70(2): 188–194.
66. Vergara R, Gitzendanner MA, Soltis DE, *et al*. Population genetic structure, genetic diversity, and natural history of the South American species of *Nothofagus* subgenus *Lophozonia* (Nothofagaceae) inferred from nuclear microsatellite data. Ecology and Evolution 2014; 4(12): 2450–2471.
67. Armesto JJ, Rozzi R, Smith-Ramírez C, *et al*. Conservation targets in South American temperate forests. Science 1998; 282(5392): 1271–1272.
68. Simonetti J, Mella JE. Park size and the conservation of Chilean mammals. Revista Chilena de Historia Natural 1997; 70: 213–220.

Sustainable Forestry (2021) Volume 4 Issue 1
doi:10.24294/sf.v4i1.1601

ORIGINAL RESEARCH ARTICLE

Light habitat, structure, diversity and dynamics of the tropical dry forests of the upper Magdalena river

Omar Melo[1]*, Fernando Fernandez-Mendez[1,2], Boris Villanueva[1]

[1] *Universidad del Tolima, Ibagué, Colombia. E-mail: omelo@ut.edu.co*

[2] *Laboratorio de Ecológia e Evolucão de Plantas, Universidad Federal de Viçosa, Minas Gerais, Brazil.*

ABSTRACT

Tropical dry forests are complex and fragile ecosystems with high anthropogenic intervention and restricted reproductive cycles. They harbor unique richness, structural, physiological and phenological diversity. This research was carried out in the upper Magdalena valley, in four forest fragments with different successional stages. In each fragment, four permanent plots of 0.25 ha were established and the light habitat associated with species richness, relative abundance and rarity was evaluated, as well as the forest dynamics that included mortality, recruitment and diameter growth for a period of 5.25 years. In mature riparian forest, species richness was found to be higher than that reported in other studies for similar areas in the Cauca Valley and the Atlantic coast. Values of species richness, heterogeneity and rarity are higher than those found in drier areas of Tolima. Forest structure, diversity and dynamics were correlated with light habitat, showing differences in canopy architecture and its role in the capture and absorption of radiation. The utilization rate of photosynthetic effective radiation in the forest underlayer with high canopy density is low, which is related to the low species richness, while the underlayer under light is more abundant and heterogeneous.

Keywords: Light Extinction Coefficient; Floristic Diversity; Structure; Plant Area Index; Mortality; Photosynthetically Active Radiation, Recruitment

ARTICLE INFO

Received: 12 March 2021
Accepted: 17 April 2021
Available online: 26 April 2021

1. Introduction

The tropical dry forest is characterized by at least one drought season per year, which generates water deficit in the soil with the consequent alterations in the functionality of the vegetation. Thus, defoliation is generated in a large number of species that have adapted to drought under this strategy. However, another group of species remain evergreen, for which the adaptive strategy follows effective and controlled stomatal conductance[1–3]. The diversity of tropical dry forests is simpler than that of tropical rainforests and Andean forests. However, its value lies in the great number of endemic species that can reach between 43% and 73%, which, together with its low growth rates, classify it as a highly fragile ecosystem. Also typical of this ecosystem is the physiological complexity of the species and the spatial distribution patterns of the species and their populations[4–7]. The structure and pattern of diversity in tropical dry forests is contrary to that of rainforests, which increase species richness with their proximity to the equator. The highest species density in tropical dry forests is located more in the northern than equatorial areas[8–10]. The limiting factor in tropical dry forests is the availability of water in the soil, which restricts nutrient leaching processes. This has led to a worldwide change of land use in these ecosystems towards agriculture and livestock production. To this end, agriculture and animal husbandry production strategies based on irrigation systems have been formulated, which has increased food production, and greatly affected the flow of ecosystem services by destroying natural mulch and replacing it with man-made landscapes[11–13].

Colombia's tropical dry forests are located especially in two large regions that correspond to the Caribbean plain and the inter-Andean valleys of the Cauca and Magdalena rivers. In these areas, precipitation is less than 2,000 mm per year. However, the distribution of precipitation determines the particular characteristics of the vegetation. The Caribbean plain has a monomodal climate, while the inter-Andean valleys of the Cauca and Magdalena rivers have a bimodal climate, with the addition of the shadow effect of the mountain ranges[14,15]. Current natural cover in the Upper Magdalena is located in the north and south of the department of Tolima, as well as in the north and south of the department of Huila. The sub-region with the lowest frequency of natural fragments of tropical dry forest and the smallest size corresponds to the south of Tolima-north of Huila, where natural cover does not exceed 2%. Fragments larger than 200 ha in size and with a higher degree of conservation are found in the north of the department of Tolima, with cover 7% greater of the original area[7,16,17]. The results of research on the structure and diversity of tropical dry forests are scarce, and it is even more critical for the values of their dynamics. Only today, people begin to understand the functions of this kind of forest, making it one of the most degraded ecosystems, with a high level of vulnerability and with gaps in knowledge that would allow true conservation, restoration and sustainable use of its ecosystem services[18,19].

Therefore, the present study was conducted in the tropical dry forest areas located in the north of the department of Tolima, which are part of the ecoregion of Alto Magdalena. The objective was to determine the relationship between the light habitat generated by the forest canopy (supply of photosynthetically active radiation in the growth environment) and the structure, floristic diversity and cover dynamics in terms of growth, mortality and reforestation, four secondary forests with different successional stages located on the eastern flank of the foothills of the Central Cordillera are part of the tropical dry forest complex of the geographic valley of the Magdalena river.

2. Materials and methods

2.1 Study area

This research was carried out on four natural fragments of tropical dry forest (bs-T) with different successional stages located in the upper part of the geographic valley of the Magdalena River, lands belonging to the Centro Universitario Regional del Norte (Curn) of the University of Tolima in the municipality of Armero-Guayabal in the north of the department of Tolima. The selected forests are part of the slope of the Magdalena River and are located in the foothills of the eastern flank of the Central Cordillera and the Alluvial plain. The flat coordinates of the area are: 4°59'53.48" N and 74°55'38.87".

Regarding climatic characteristics, the study area has an average annual temperature of 27 °C, an average rainfall of 1,750 mm, and a relative humidity of 71%. The altitude above sea level ranges from 475 to 580 m. The climatic assessment shows a bimodal behavior, with a first rainy period between the months of March to May and a second stronger period between September and November[20].

2.2 Sampling

The assessed forest coverage corresponds to forests in four different succession states. The first is an early secondary forest (BST) that is 10 years old, originated from anthropogenic fires and dominated by the Curatella-Xilopia association. The second forest type corresponds to a 20-year-old secondary forest in recovery (BSR), generated by the abandonment of extensive cattle ranching activities and dominated by Cordia alliodora (Ruiz & Pav.) Oken. The third corresponds to a succession in a state of advanced recovery of more than 40 years, which for the purposes of this study will be called mature secondary forest (BSM), with a heterogeneous floristic composition. The fourth forest type corresponds to a mature riparian forest (BRM) dominated by Anacardium and Ceiba whose recovery time exceeds 60 years. In each of the forests, four permanent monitoring units of 0.25 ha (50 × 50 m) were established, with subplots of 10 x 10 m, for a total sampling area per cover type of 1 ha. This sectioning of the sampling unit was due to the size and shape limitations of the remaining fragments.

For all sampling units, all individuals with a normal diameter greater than or equal to 5 cm were recorded, marked, measured and collected. Measurements of normal diameter were made with a diametric tape to the nearest millimeter[21]. The collection of plant materials was carried out in cooperation with the herbarium of Medellin botanical garden and the dendrology Laboratory of the University of Tolima. The first inventory was conducted in June 2009 with subsequent annual monitoring until September 2014, for a time interval of 63 months (5.25 years). New individuals that exceeded 5 cm normal diameter were recorded as incoming or recruited and dead trees were recorded as mortality status[22].

2.3 Structure and diversity

For the assessment of floristic diversity and conventional structural parameters, the Stimate-S 9.1.0 program was used[23]. Measures of species abundance at the intra-community level (species richness, relative heterogeneity and rarity) were selected and measures of similarity and dissimilarity at the inter-community level were used[24–26].

2.4 Forest dynamics

The evaluation of the dynamics for the four forest types included the calculation of mortality, recruitment and diameter growth for the evaluation period of 5.25 years. The mortality rate was determined from the model proposed by Castro *et al.*[27] and recruitment according to the models presented in Melo and Vargas[21]. The mortality pattern contemplated the types of death: broken trunk (TP), fallen root (CR), missing individuals (DE) and cut (COR)[28]. It's the application of traditional deterministic growth model[29].

2.5 Light habitat

To evaluate the light habitat, which indicates the amount of energy used by the forest canopy to carry out the functional processes of assimilation and productivity, direct measurement of the leaf area index (LAI) was used with a LI-2200TC canopy analyzer. The different levels of light absorbed by the canopy of the forests under evaluation were expressed as relative values of photosynthetically active radiation (PAR), which was measured as the unabsorbed radiation on the forest floor surface expressed as the photon flux density of photosynthetic photons ($\mu mol \cdot m^{-2} \cdot s^{-1}$) between 400 and 700 nm of the light spectrum. For this purpose, the LI-191SA-Line Quantum Sensor and LI-190SA-Quantum Sensor and an LI 1400 data collector (LI-COR Inc., Lincoln, NE. USA) are used. Likewise, the light extinction coefficient (K) was derived by applying Beer & Lambert's law[3,30]. This process was performed below the canopy above the soil surface in the center of each 10 × 10 m subplot, and four cardinal readings were recorded. Measurements were taken between 10 am and 2 pm.

3. Results

Table 1 shows the results for the response variables by evaluation levels (structure, floristic diversity, dynamics and light habitat), for the four types of secondary forest in the upper Magdalena (BST: early secondary forest, BSR: recovering secondary forest, BSM: mature secondary forest, BRM: mature riparian forest).

3.1 Structure

Regarding forest structure, the number of trees (NA) in absolute values and basal area (G) in $m^2 \cdot ha^{-1}$ and maximum stem diameter (dmax.) in cm are shown. In general, the diametric structure of the forests showed inverted J or L trends, which is typical of dynamic canopies. The stage of development of each forest is manifested in the decrease of the diametric range, so that in the early successional coverages the range is short (27.5 cm) in comparison with the coverages of more advanced ages that exceed 60 cm. For the BRM type, there is a bimodal trend that corresponds to the presence of two populations of trees, the first with diametric ranges up to 35 cm and a second population whose diametric range varies between 45 and 110 cm. The accumu-

Table 1. Behavioral structure, diversity, dynamics and light habitat in four types of tropical dry forest in the upper Magdalena

Evaluation level	Variables	Forests			
		BST	BSR	BSM	BRM
Structure	NA	1774	928	388	672
	S	18	27	36	48
	G	21.9	18.7	17.7	34.2
	Dmax.	27.5	55	65	110
	DMg	2.27	3.81	5.87	7.22
	DMn	0.43	0.89	1.83	1.85
	H’	1.38	2.34	2.76	2.98
	E	0.61	0.69	0.83	0.81
	L/D	4.78	5.30	11.24	10.98
Diversity	L/d	2.73	2.97	5.11	8.84
	CH_1	22.25	21.37	36.28	42.79
	Ab_1	3	5	11	14
	Ab_2	2	4	7	5
	U	2	3	5	5
	Uab.	0	1	4	4
	Alpha	4.45	9.21	17.67	18.21
	TMC	2.37	1.37	0.63	0.45
Dynamics	M%	5.78	3.45	2.31	1.87
	R%	3.8	3.2	2.1	1.2
	LAI	8.3	5.1	4.5	3.1
Light habitat	K	0.75	0.63	0.53	0.49
	RFA	16.3	18.2	22.2	25.7

BST: early secondary forest. BSR: recovering secondary forest. BSM: mature secondary forest. BRM: mature riparian forest. NA: number of trees. G: basal area in $m^2 \cdot ha^{-1}$. Dmax: maximum stem diameter in cm. S: number of species. DMg: Margalef species richness. DMn: Menhinick species density. H’: Shannon diversity. E: Shannon’s evenness. L/D: Simpson’s heterogeneity. L/d: Berger Parker dominance. CH_1: species rarity. Ab_1: species with one individual. Ab_2: species with two individuals. U: species represented in a single plot. UAb: unique species in a plot. Alpha: species diversity. LAI: leaf area index. K: light extinction coefficient. PAR: photosynthetically active radiation. TMC: mean growth rate. M%: annual mortality rate. R%: annual recruitment rate.

lation of basal area reaches the highest value in BRM. However, the effect of population size (1,774) on G (21.99) for BST is clear. The trend of basal area accumulation (G) for secondary forests is typical of a chrono-sequence. Therefore, with the progress of forest succession, the population size decreases and the individual reaches a larger size (**Figure 1a**). The BRM manifests a smoother distribution indicating uniformity of populations in the size range. In the accumulation of basal area (**Figure 1b**), the trends are differential and contrary as the succession progresses, so that in the BST the greatest accumulation occurs between the diameter categories of 12.5 and 17.5 cm, while in advanced successions (BSM) the greatest accumulation of basal area is represented in the diameter categories greater than 65 cm, showing opposite patterns.

3.2 Diversity

Regarding floristic diversity, values for Margalef’s species richness index (DMg), Menhinick’s species density (DMn), Shannon’s diversity (H’), Shannon’s evenness (E), Simpson’s reciprocal heterogeneity (1/D) were determined for the four forest types, Berger Parker reciprocal dominance (1/d), species rarity (CH_1), species with only one individual (Ab_1), species with two individuals (Ab_2), species represented in only one plot (U), species only in one plot and only one individual (UAb) and

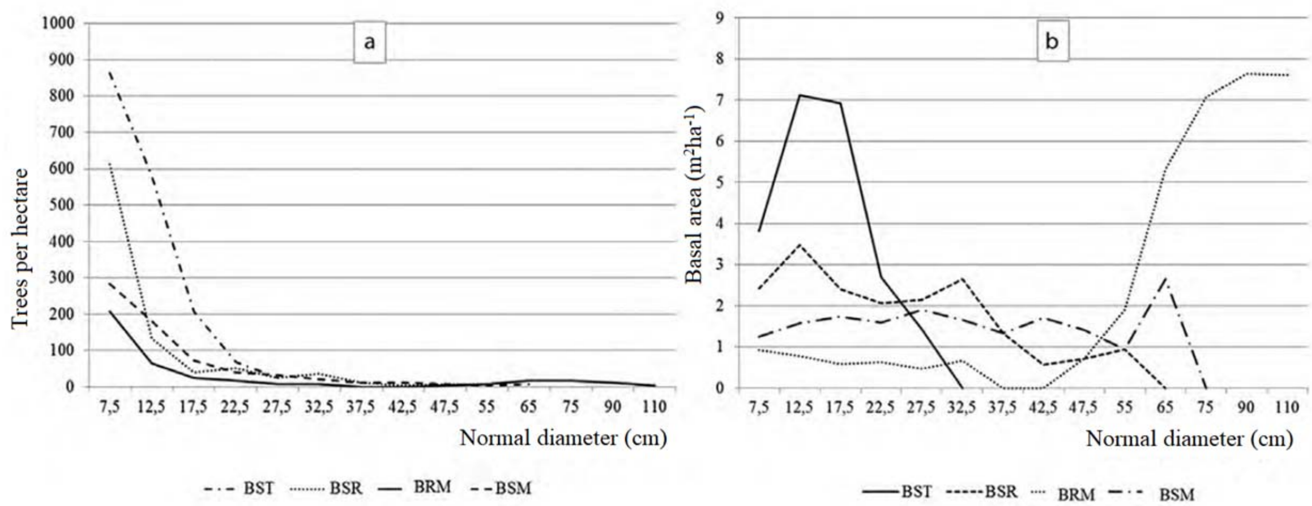


Figure 1. Trends in population distribution (NA) and basal area $m^2 \cdot ha^{-1}$ (G) classified by size, for four secondary forest types in the upper Magdalena. BST: early secondary forest. BSR: recovering secondary forest. BRM: mature riparian forest. BSM: mature secondary forest.

the log series parameter indicating species diversity (Alpha). The BRM and BSM have the highest species richness (S, DMg) and similar densities (DMn). An increase in diversity (H’, E, 1/D, 1/d) with successional development is evident. In terms of species rarity (CH_1), the BSM and BRM present the highest values. However, despite the homogeneity (E) of the early successional covers (BST and BSR), they contain rare species (Ab_1 and Ab_2), which increases their importance for diversity. According to the results of the alpha index of the logarithmic series, which is considered the most biologically representative measure of diversity, the BSM and BRM have equivalent values (17.67 and 18.21) despite their structural difference. Intercommunity complementarity (**Table 2**) for the evaluated forests, is a measure of biodiversity expressed from Sorensen's similarity index (SR), Jaccard's similarity (JC), percentage of dissimilarity (PD), percentage of remoteness (PR) and euclidean distance (DE). High similarity values (SR, JC) were observed between BST and BSR, BSR and BSM, showing a gradient of species accumulation with successional advancement. There are large differences between the diversity (PD, PR and DE) of BRM and the other secondary forests both in shared species and in the distribution of their abundances.

3.3 Forest dynamics

In the evaluation level corresponding to the dynamic processes of the forest (**Table 1**), the values of the mean growth rate (TMC), annual mortality rate (M%) and annual recruitment rate (R%) were determined. The results show an inverse relationship of TMC with successional advancement, such that the maximum values (2.37 $cm \cdot year^{-1}$) are achieved at BST. Likewise, the same trend is observed for both M% and R%. The highest mortality and recruitment rates (5.78 and 3.8) are found in BST as a result of high competition for resources in contrast to BRM (1.87 and 1.2), which shows a more stable habitat.

3.4 Light habitat

The high leaf area index (LAI, 8.3 m^2 leaf area/m^2 ground area) of the BST canopy allows the optimization of energy, which generates high values in growth rates and increases competition, mortality and recruitment by freeing growth spaces. On the contrary, in the BRM, which has only reached an LAI of 3.1 (m^2 leaf area/m^2 ground area), the functional behavior is inverse, the mature trees generate a canopy with low leaf area, there is less use of radiation and the growth rate, mortality and recruitment have lower values compared to the early successions. The highest LAI in BST (8.3) and BSR (5.1) leads to the lowest intensities in PAR (16 to 18 µmoles photons. $m^2 \cdot s^{-1}$) as a consequence of self-shading, which obeys the planophilic architecture that is defined as the arrangement of leaves and branches inserted in angles fluctuating between 0° and 30° of those two communities, expressed by K = 0.75 and 0.63 respectively. For forests that are in a more advanced successional stage (BRM and BSM), which present LAI of 3.1 and 4.5, additional to the K = 0.49 and 0.53, plagiophilous architecture (arrangements of leaves and branches inserted at angles fluctuating between 30° and 60°), self-shading is lower, leading to higher PAR intensities of 22 to 25 µmoles photons. $m^2 \cdot s^{-1}$ (**Table 1**).

Table 3 characterizes the light habitat and links it to forest structure, diversity and dynamics based on a correlation matrix, for the parameters leaf area index (LAI), light extinction coefficient (K), photosynthetically active radiation (PAR), log series diversity index (Alpha), basal area in m^2 ha^{-1} (G), Margalef species richness (DMg), mean growth rate (MT), annual mortality rate (M) and and annual recruitment rate (R). High correlation values were observed between LAI, which expresses the magnitude of canopy cover and the role in radiation capture and absorption, versus floristic diversity, mean growth rates, mortality and recruitment. Negative values in the correlation with diversity indicate that in forests with canopies that have large leaf area (LAI) to capture the low availability of PAR. Species richness is low. More illuminated understories allow the establishment of more species per unit area and

Table 2. Intercommunity diversity values for four secondary forest types in the Alto Magdalena

Comparison	SR	JC	PD	PR	DE
BST-BSR	0.42	0.48	37.26	48.43	22.67
BST-BRM	0.18	0.25	59.37	75.33	36.77
BST-BSM	0.33	0.37	45.93	52.57	23.92
BSR-BRM	0.21	0.29	51.47	63.71	30.73
BSR-BSM	0.53	0.59	29.82	37.39	19.32
BRM-BSM	0.23	0.27	53.31	75.05	31.32

BST: early secondary forest. BSR: secondary forest in recovery. BRM: mature riparian forest. BSM: mature secondary forest. SR: Sorensen similarity. JC: Jaccard similarity. PD: dissimilarity percentage. PR: percentage of remoteness. DE: Euclidean distance.

Table 3. Correlation matrix between light habitat, structure, diversity and dynamics of tropical dry forests

	LAI	K	RFA	Alpha	G	DMg	TC	M%	R%
LAI		0.9714	-0.9002	-0.8918	-0.4534	-0.8085	0.9696	0.9838	0.9121
		100	100	100	100	100	100	100	100
		0.0286	0.0998	0.1082	0.5466	0.1915	0.0304	0.0162	0.0879
K	0.9714		-0.9568	-0.9702	-0.4562	-0.9041	0.9955	0.9885	0.9664
	100		100	100	100	100	100	100	100
	0.0286		0.0432	0.0298	0.5438	0.0959	0.0045	0.0115	0.0336
RFA	-0.9002	-0.9568		0.9242	0.6651	0.8197	-0.9262	-0.9034	-0.9994
	100	100		100	100	100	100	100	100
	0.0998	0.0432		0.0758	0.3349	0.1803	0.0738	0.0966	0.0006
Alpha	-0.8918	-0.9702	0.9242		0.3307	0.9756	-0.9744	-0.951	-0.9349
	100	100	100		100	100	100	100	100
	0.1082	0.0298	0.0758		0.6693	0.0244	0.0256	0.049	0.0651
G	-0.4534	-0.4562	0.6651	0.3307		0.1242	-0.3706	-0.3536	-0.6413
	100	100	100	100		100	100	100	100
	0.5466	0.5438	0.3349	0.6693		0.8758	0.6294	0.6464	0.3587
DMg	-0.8085	-0.9041	0.8197	0.9756	0.1242		-0.9261	-0.8999	-0.8353
	100	100	100	100	100		100	100	100
	0.1915	0.0959	0.1803	0.0244	0.8758		0.0739	0.1001	0.1647
TC	0.9696	0.9955	-0.9262	-0.9744	-0.3706	-0.9261		0.9959	0.939
	100	100	100	100	100	100		100	100
	0.0304	0.0045	0.0738	0.0256	0.6294	0.0739		0.0041	0.061
M%	0.9838	0.9885	-0.9034	-0.951	-0.3536	-0.8999	0.9959		0.9177
	100	100	100	100	100	100	100		100
	0.0162	0.0115	0.0966	0.049	0.6464	0.1001	0.0041		0.0823
R%	0.9121	0.9664	-0.9994	-0.9349	-0.6413	-0.8353	0.939	0.9177	
	100	100	100	100	100	100	100	100	
	0.0879	0.0336	0.0006	0.0651	0.3587	0.1647	0.061	0.0823	

LAI: Leaf area index. K: light extinction coefficient. PAR: photosynthetically active radiation. Alpha: log series diversity index. G: basal area in $m^2 \cdot ha^{-1}$. DMg: Margalef species richness. TC: mean growth rate. M%: annual mortality rate. R%: annual recruitment rate.

there is more heterogeneity of growth niches. Likewise, a negative correlation was detected between the LAI and the survival of the tree community. Basal area has a low correlation with light habitat, mainly due to the influence of the large trees of the BRM that share the same habitat as smaller trees of the secondary forests. Mortality correlates positively with LAI and K, otherwise with RFA. Recruitment has a similar trend.

4. Discussion

The evaluated forests in general show a chrono-sequence that allows the increase of the values of diversity and structural complexity offered by the succession of the natural cover of the tropical dry forest of the Upper Magdalena as expressed by Mendoza[31] in preliminary studies in fragments of tropical dry forest of the Caribbean coast and Magdalena valley. The values of structural parameters and species richness, heterogeneity and species rarity (**Table 1**) for BST, BSR and BSM are relatively higher than those found by Fernandez *et al.*[20] for the south of the department of Tolima, an area with a drier climate (1,350 mm of mean annual precipitation). This is in agreement with Gentry[9,32], who states that species richness is associated with the availability of moisture in the environment. Regarding the BRM, the species richness is the highest reported for this type of cover in tropical dry forest areas, since studies by Linares & Fandiño[15], Cabrera & Galindo[33] and Etter[16] recorded a lower density of species in similar study areas in the Cauca Valley and the Atlantic coast.

In terms of intercommunity diversity (**Table 2**), the successional complex of secondary forests shows gradual changes in floristic composition, which is associated with successional development after the interruption of anthropic activity. This increase in species in correlation with the recovery of ecosystem services and forest functionality has been studied by Kalascka *et al.*[10] in Mesoamerican dry forests.

There are great differences found in growth (TC), mortality (M) and recruitment (R) rates for the four tropical dry forest types shown in **Table 1** compared to similar studies conducted in tropical dry forests in Nicaragua[34], which may be associated with both orography and moisture availability, which for the eastern flank of the central mountain range are related to the mountain range shade effect that increases the relative humidity value, generating a better growth environment.

The growth of the individual tree and of the forest as a whole depends to a large extent on its functionality, that is, on how it obtains the resources offered by the environment and how it uses them. The main factors are the light and water in the soil[35], which is expressed in the distribution and quality of the canopy. The canopy directly influences the accumulation of biomass, whose differential distribution in its structural components varies according to the competition relationships generated by neighboring trees located in its living space[36]. Regarding the values of the variables describing light habitat and their correlation with forest dynamics (**Table 3**), for the four evaluated canopies (BST, BSR, BSM, BRM), the direct effects of IAF, K, and RFA on growth, mortality, and reforestation are clear. Thus, the highest mortality and recruitment rates are generated under canopies with high LAI and maximum K values. That is, there is dependence on self-shading due to the maximum leaf area of the crown and the structure described by the crown affect the capture of radiation and the consequent productivity of the forest community, which directly affects both the structure of plant communities and their floristic diversity[37], similar relationships between functionality and cover types have been found by Sterck *et al.*[38], Craine and Dybzinski[39], which supports the present results for the tropical dry forests of the Upper Magdalena.

5. Conclusions

There is a direct relationship between the availability of light resources and the dynamics of the tropical dry forest. The highest growth rates of BST are associated with habitats with high availability of PAR, which is absorbed by a canopy with high LAI, resulting in high growth that generates an increase in mortality rates. The freed spaces are occupied by new individuals waiting for the opportunity to grow. For more advanced successions such as BSM, the canopy that characterizes the forest structure has a lower LAI, which allows greater availability of resources in the understory allowing a greater diversity of habitats that are occupied by various types of species, hence the greater heterogeneity and structural complexity. The consequence is lower growth and greater stability between mortality and recruitment, which owe their dynamics to factors endogenous to the biotic community.

References

1. Murphy P, Lugo A. Dry forests of Central America and the Caribbean. In: Mooney H, Bullock S (editors). Seasonally tropical forests. Cambridge: University of Cambridge Press; 1996. p. 9–34.
2. Lobo J, Quesada M, Stoner K, *et al*. Factors affecting phenological patterns of bombacaceous trees in seasonal forests in Costa Rica and Mexico. American Journal of Botany 2003; 90(7): 1054–1063.
3. Lüttge U. Physiological ecology of tropical plants. 2nd ed. Berlin: Springer-Verlag; 2008. p. 458.
4. Janzen D. Tropical dry forests: The most endangered major tropical ecosystem. In: Wilson EO (editor). Biodiversity. Washington D.C.: National Academy Press; 1986. p. 130–137.
5. Mooney H, Bullock S, Medina E. Introduction to tropical dry forest. In: Bullock S, Mooney H, Medina E (editors). Seasonally dry tropical forests. Cambridge: Cambridge University Press; 1996. p. 1–6.
6. Burnham K. Distributional results for special cases of the Jolly-Seber model. Communications in Statistics 1997; 26: 1395–1409.
7. Instituto Alexander von Humboldt Institute (IAVH). El bosque seco tropical (bs-t) en Colombia, programa de inventario de la biodiversidad (Spanish) [The tropical dry forest (bs-t) in Colombia, biodiversity inventory program]. Bogotá: Alexander von Humboldt Institute; 2002. p. 1–24.
8. Chazdon R, Denslow J. Floristic composition and species richness. In: Chazdon R, Whitmore T (editors). Foundations of tropical forest biology. Chicago: University of Chicago Press; 2002. p. 513–522.
9. Gentry A. Diversity and floristic composition of neotropical dry forest. In: Bullock S, Medina E, Mooney H (editors). Tropical deciduous forest ecosystems. Cambridge: Cambridge University Press; 1995. p. 116–194.
10. Kalascka M, Sanchez-Asofeifa G, Calvo-Alvarado J, *et al*. Species composition, similarity and diversity in three successional stages of a seasonally dry tropical forest. Forest Ecology and Management 2004; 200: 227–247.
11. Bazzaz F, Pickett S. Physiological ecology of tropical succession: A comparative review. Annual Review of Ecology, Evolution, and Systematics 1980; 11: 287–310.
12. Quesada M, Stoner K. Threats to the conservation of the tropical dry forest in Costa Rica. In: Frankie G, Mata A, Vinson S (editors). Biodiversity conserva-

tion in Costa Rica: Learning the lessons in a seasonal dry forest. Berkeley (CA, U.S.A.): University of California Press; 2004. p. 266–280.
13. Montagnini F, Jordan C. Tropical forest ecology: The basis for conservation and management. Berlin: Springer Verlag; 2005. p. 295.
14. Repizzo A, Devia C. Árboles y arbustos del valle seco del río Magdalena y la región Caribe colombiana: su ecología y usos (Spanish) [Trees and shrubs in the dry valley of the Magdalena River and the Caribbean region of Colombia: Ecology and uses]. Bogotá D.C.: Faculty of Environmental and Rural Studies, Pontificia Universidad Javeriana; 2008. p. 120.
15. Linares R, Fandiño M. Estado del bosque seco tropical e importancia relativa de su flora leñosa, islas de la Vieja Providencia y Santa Catalina, Colombia, Caribe suroccidental (Spanish) [State of the tropical dry forest and relative importance of its woody flora, islands of old Providence and Santa Catalina, Colombia, southwestern Caribbean]. Revista de la Academia Colombiana de Ciencias 2009; 33(126): 1–12.
16. Etter A. Ecosystem diversity in Colombia today. In: Nuestra diversidad biótica (Spanish) [Our biodiversity]. Bogotá: CEREC y Fundación Alejandro Ángel Escobar; 1993. p. 43–61.
17. University of Tolima. Informe técnico proyecto "Caracterización biofísica de la Eco-región Estratégica del desierto de la Tatacoa y su área de influencia" (Spanish) [Technical report of the project "biophysical characteristics of the ecological strategic area and its affected areas in the Tatacoa desert"]. Ibagué: Universidad del Tolima, Universidad Surcolombiana, Universidad de Cundinamarca; 2002. p. 250.
18. Sánchez-Azofeifa A, Kalacska M, Quesada M, *et al*. Need for integrated research for a sustainable future in tropical dry forests. Conservation Biology 2005; 19(2): 1–2.
19. Prance W. Tropical savannas and seasonally dry forests: An introduction. Journal of Biogeography 2006; 33: 385–386.
20. Fernández F, Melo O, Álvarez E, *et al*. Status of lnowledge, conservation and management of tropical dry forest in the Magdalena river Valley, Colombia. In: Sanchez-Asofeifa A, Power J, Fernandes G, *et al*. (editors). Tropical dry forest in the Americas. Florida: CRC Press; 2014. p. 35–68.
21. Melo O, Vargas R. Evaluación ecológica y silvicultural de ecosistemas boscosos (Spanish) [Ecological and silvicultural evaluation of forest ecosystems]. Ibagué: Universidad del Tolima; 2003. p. 235.
22. Phillips O, Vasquez R, Monteagudo A, *et al*. Large lianas as hyperdynamic elements of the tropical forest canopy. Ecology 2005; 86(5): 1250–1258.
23. Colwell R, Chao A, Gotelli N, *et al*. Models and estimators linking individual-based and sample-based rarefaction, extrapolation, and comparison of assemblages. Journal of Plant Ecology 2012; 5: 3–21.
24. Magurran A. Ecological diversity and its measurement. New York: Princeton University; 1988. p. 179.
25. Magurran A. Measuring biological diversity. Oxford: Blackwell Publishing; 2004. p. 256.
26. Magurran A, McHill B. Biological diversity: Frontiers in measurement and assessment. New York: Oxford University Press; 2011. p. 345.
27. Castro G, Nigard R, Gonzalez B, *et al*. Stand dynamics and basal area change in a tropical dry forest reserve in Nicaragua. Forest Ecology and Management 2005; 208: 63–75.
28. Carson W, Schnitzer S. Tropical forest community ecology. Hoboken: Wiley-Blacwell; 2008. p. 517.
29. Weiskittel A, Hann D, Kershaw J, *et al*. Forest growth and yield modeling. Hoboken: Wiley-Blacwell; 2011. p.415.
30. Pallardy S. Physiology of woody plants. 3rd ed. San Diego: Academic Press, Elsevier; 2008. p. 454.
31. Mendoza C. Estructura y riqueza florística del bosque seco tropical en la región Caribe y el valle del río Magdalena, Colombia (Spanish) [Structure and floristic richness of the tropical dry forest in the Caribbean region and the Magdalena River valley, Colombia]. Caldasia 1999; 21(1): 70–94.
32. Gentry A. Changes in plant community diversity and floristic composition on environmental and geographical gradients. Annals of the Missouri Botanical Garden 1988; 75: 1–34.
33. Cabrera E, Galindo G. Aproximación metodológica para la delimitación de ecosistemas de enclaves secos: caso piloto, cañones del río Dagua y del río Tuluá, Valle del Cauca-Colombia (Spanish) [Methodological approach for the delimitation of dry enclave ecosystems: A pilot case of the Dagua River and the Tulua River Gorge in the Cauca Valley, Colombia]. Bogotá, D.C.: Instituto de Investigación de Recursos Biológicos Alexander von Humboldt; 2006. p. 105.
34. Flynn B, Mirotchnick N, Jain M, *et al*. Functional and phylogenetic diversity as predictors of biodiversity-ecosystem-function relationships. Ecology 2011; 92(8): 1573–1581.
35. Burkhart H, Tomé M. Modeling forest trees and stand. Berlin: Springer-Verlag; 2012. p. 457.
36. Trinder C, Brooker R, Robinson D. Plant ecology's guilty little secret: Understanding the dynamics of plant competition. Functional Ecology 2013; 27: 918–929.
37. Pretzch H. Canopy space filling and tree crown morphology in mixed-species stands compared with monocultures. Forest Ecology and Management 2014; 327: 251–264.
38. Sterck F, Schieving F, Lemmens A, *et al*. Performance of trees in forest canopies: Explorations with a bottom-up functional-structural plant growth model. New Phytologist 2005; 166(3): 827–843.
39. Craine J, Dybzinski R. Mechanisms of plant competition for nutrients, water and light. Functional Ecology 2013; 27: 833–840.

Sustainable Forestry (2021) Volume 4 Issue 1
doi:10.24294/sf.v4i1.1602

ORIGINAL RESEARCH ARTICLE

Temporal variation of tree diversity of main forest vegetation in Xishuangbanna

Jianbo Yang[1,2,3], Youxin Ma[1,2*], Yang Bai[1,2], Hui Cao[1,2,3]

[1] *Xishuangbanna Tropical Botanical Garden, Chinese Academy of Sciences, Xishuangbanna 666303, Yunnan, China. E-mail: may@xtbg.ac.cn*

[2] *Center for Integrative Conservation, Xishuangbanna Tropical Botanical Garden, Chinese Academy of Sciences, Xishuangbanna 666303, Yunnan, China.*

[3] *University of Chinese Academy of Sciences, Beijing 100049, China.*

ABSTRACT

In order to evaluate the temporal changes in tree diversity of forest vegetation in Xishuangbanna, Yunnan Province, the study collected tree diversity data from four main forest vegetation in the region through a quadrat survey including tropical rainforest (TRF), tropical coniferous forest (COF), tropical lower mountain evergreen broad-leaved forest (TEBF), tropical seasonal moist forest (TSMF). We extracted the distribution of four forest vegetation in the region in four periods of 1992, 2000, 2009, and 2016 in combination with remote sensing images, using simp son Shannon Wiener and scaling species diversity indexes compare to the differences of tree evenness of four forest vegetation and use the scaling ecological diversity index and grey correlation evaluation model to evaluate the temporal changes of forest tree diversity in the region in four periods. The results show that: (1) The proportion of forest area has a trend of decreasing first and then increasing, which is shown by the reduction from 65.5% in 1992 to 53.42% in 2000, to 52.49% in 2009, and then to 54.73% in 2016. However, the tropical rainforest shows a continuous decreasing trend. (2) There are obvious differences in the contributions of the four kinds of forest vegetation to tree diversity. The order of evenness is tropical rainforest > tropical mountain (low mountain) evergreen broad-leaved forest > warm coniferous forest > tropical seasonal humid forest, and the order of richness is tropical rainforest > tropical mountain (low mountain) evergreen broad-leaved forest > tropical seasonal humid forest > warm coniferous forest, The order of contribution to tree diversity in tropical rainforest > tropical mountain (low mountain) evergreen broad-leaved forest > tropical seasonal humid forest > warm tropical coniferous forest. (3) The tree diversity of tropical rainforests and tropical seasonal humid forests showed a continuous decreasing trend. The tree diversity of forest vegetation in Xishuangbanna in four periods was 1992 > 2009 > 2016 > 2000. The above results show that economic activities are an important factor affecting the biodivesity of Xishuangbanna, and the protection of tropical rainforest is of great significance to maintain the biodiversity of the region.

Keywords: Xishuangbanna; Tree Diversity; Forest Vegetation; Remote Sensing; Scaling Ecological Diversity Index; Grey Correlation Evaluation Model

ARTICLE INFO

Received: 25 March 2021
Accepted: 29 April 2021
Available online: 18 May 2021

1. Introduction

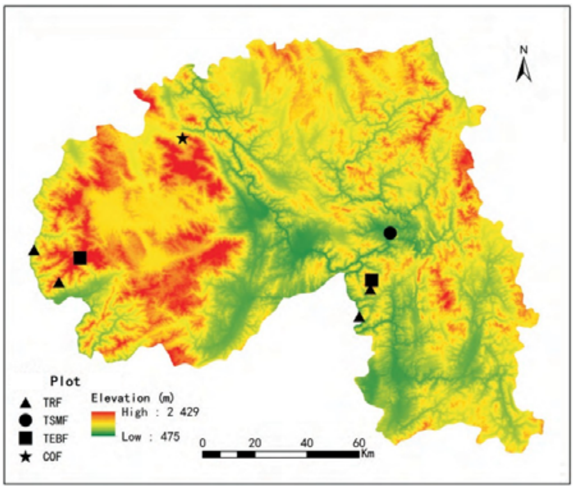
Biodiversity assessment can not only understand the current situation and change trends of regional biodiversity but also provide a scientific basis for the government's decision-making on regional biodiversity protection and sustainable development[1]. At present, the evaluation methods of regional biodiversity can be divided into the frequency analysis method and remote sensing inversion method. The frequency analysis method uses an expert consultation method to set weights by screening practical indicators, such as plant species, animal species, ecosystem types, and alien species, so as to compare the biodiversity of different regions. This method is not only highly subjective and lacks further scientific quantification, but also the regional biodiversity evaluation system and method are not unified, and the evaluation results of different regions are not well comparable[2]. Remote sensing technology has the characteristics of multi-space-time and multi-spectrum, which provides a new way for the monitoring and evaluation of regional biodiversity[3]. At present, remote sensing data can be used to map species or habitats, establish the relationship model between surface species diversity and the remote sensing spectrum, analyze the landscape index of the land use map interpreted by remote sensing, and evaluate and monitor regional biodiversity[4,5]. Although remote sensing can provide information such as area, structure, and species types for the evaluation of regional biodiversity, the evaluation of regional biodiversity also relies on geographic information technology and mathematical models[6]. Therefore, the scaling ecological diversity index[7] is proposed. The scaling ecological diversity index is composed of species' evenness and area. It is a comprehensive reflection of species evenness, species richness, and area in the region. The larger the index, the higher the ecological diversity[8].

Xishuangbanna, located in the south of Yunnan Province, is a hot biodiversity distribution area[9]. On the land whose land area is only 0.2% of the national area, there are nearly 20% of China's mammals and birds, as well as more than 5000 kinds of plants[10]. However, since 1976, natural forests in Xishuangbanna have been decreasing, while rubber forests, tea gardens, and agricultural land have been increasing[13,14], of which rubber forests are the direct cause of the loss of biodiversity in the tropical rainforest in the region[11,13]. The fragmentation of tropical rainforests caused by land use change is more serious than that of other forest vegetation[14], resulting in changes in the composition and structure of tropical rainforest communities[15] and the loss of tree species diversity[16]. Although many scholars have proved the continuous loss of biodiversity in Xishuangbanna by various means, it is difficult to explain the differences between vegetation only by remote sensing[13,14]. Community survey[15-17] is difficult to reflect the biodiversity of the whole region. At present, there are few studies on the temporal changes of biodiversity in Xishuangbanna.

Taking Xishuangbanna as the study area, based on the quadrat survey of tree data of different forest vegetation, combined with Landsat TM and oli image interpretation of Xishuangbanna vegetation spatial distribution data in 1992, 2000, 2009, and 2016, using the scaling ecological diversity index[8], calculate the changes of tree diversity of different forest vegetation in four periods, The grey correlation evaluation model was used to evaluate the current situation and changes of forest vegetation arbor diversity in this area, so as to provide decision-making basis for biodiversity protection in Xishuangbanna.

2. Overview of the research site

Xishuangbanna (99°58'–102°00'E, 21°09'–22°30'N) is located in the south of Yunnan Province, connected with Pu'er City in the north and Myanmar and Laos in the South (**Figure 1**). The terrain is high in the East and West, and the Lancang River Valley in the middle is low above sea level. The West and East belong to the rest of the Nu mountain range and the Wuliang Mountain range. This area is deeply affected by the Indian Ocean monsoon, belonging to the tropical monsoon climate, with an annual average temperature of 18–21.7 °C; the seasonal variation of precipitation is obvious, with obvious dry and wet seasons, and the precipitation is 1,193–2,491 mm. Xishuangbanna has 32 typical formations, belonging to 7 main vegetation[18]. Under the influence of climate and vegetation, the tropical rain forest and seasonal rain forest at lower altitudes in Xishuangbanna are distributed with zonal latosol, while the monsoon evergreen broad-leaved forest is distributed with lateritic soil, and the limestone area is distributed with calcareous soil.



Note: TRF: Tropical rainforest; TSMF: Tropical seasonal moist forest; TEBF: Tropical lower montane evergreen broad-leaved forest; COF: Tropical coniferous forest. The same is below.

Figure 1. Locations of Xishuangbanna and sample plots.

3. Research methods

3.1 Sample plot setting and investigation methods

In order to compare the tree diversity of different vegetation and calculate the scaling ecological diversity index, according to the principles and basis of forest vegetation classification in Xishuangbanna[18], the well-developed forests that are less affected by human activities are selected in Xishuangbanna, and four tropical rainforests, one tropical seasonal humid forest, two tropical mountain (low mountain) evergreen broad-leaved forests, and An investigation sample plot of warm coniferous forest with four different forest vegetation (**Figure 1**), each sample plot has an area of 2,500 m^2. Identify and record all tree individuals with DBH ≥3 cm and height ≥3 m in the sample plot, and measure the DBH, height, coverage, and other information of tree individuals.

3.2 Vegetation distribution and area extraction

In order to obtain different vegetation areas in four periods and calculate the scaling ecological diversity index, this study uses Landsat TM and OLI images with a spatial resolution of 30 m in Xishuangbanna and Gdemv2 topographic data. With the support of Envi5.3 software, perform flash atmospheric correction on each image, select short wave infrared, near-infrared, red band, and green band data, and use Teillet module to perform terrain correction on the image, so as to weaken the impact of atmosphere and mountain shadow on vegetation extraction; then, the image is visually interpreted to extract broad-leaved forests, warm coniferous forests, and other vegetation (including bamboo forests, shrubs, grasses, cultivated and aquatic vegetation); finally, Arcgis10.3 software is used to superimpose the classified broad-leaved forest with altitude, extract the broad-leaved forest below 1,000 m as the distribution area of Tropical Rainforest[14], extract the limestone mountain distribution area[19] and superimpose the broad-leaved forest to obtain the distribution area of tropical seasonal humid forest. The kappa accuracy of 2016, 2009, 2000 and 1992 is 0.95, 0.92, 0.88 and 0.85, respectively.

3.3 Diversity measurement method

In this paper, Simpson (HS), Shannon Wiener (H), and scaling species diversity index (d') are used to calculate the tree diversity of tropical rainforest, tropical seasonal moist forest, tropical mountain evergreen broad-leaved forest, and warm coniferous forest, and scaling ecological diversity index (d) is used to compare the changes of tree diversity of different forest vegetation in Xishuangbanna[7,8]. The calculation formula is as follows:

$$HS = 1 - \sum_{i=1}^{m} P_i^2 \tag{1}$$

$$H = -\sum_{i=1}^{m} P_i \ln P_i \tag{2}$$

$$D' = \ln\left(\sum_{i=1}^{m} P_i^{\frac{1}{2}}\right)^2 \tag{3}$$

$$D = -\frac{\ln\left(\sum_{i=1}^{m} P_i^{\frac{1}{2}}\right)^2}{\ln \varepsilon} \tag{4}$$

Where, P_i is the proportion of the i-th tree individual to all tree individuals in the sample plot; $\varepsilon = (e + A)^{-1}$, e = 2.71828, a represents the area of the survey object (hm^2).

3.4 Evaluation of forest tree diversity in Xishuangbanna

After calculating the scaling ecological diversity index of different vegetation in four periods, in order to compare the changes of biodiversity in Xishuangbanna in four periods, this study introduces the grey correlation evaluation model into the evaluation and comparison of tree diversity in Xishuangbanna between different periods. The grey correlation evaluation model infers the correlation of various indicators by reflecting the similarity between curves[20]. Its advantage is that it can infer the correlation with a small number of indicators with different dimensions, which is widely used in various fields. The steps of grey correlation degree are as follows[21,22].

First, set the reference sequence,

$$X_0 = \{x_0(1), x_0(2), \cdots, x_0(n)\}$$

and the compared sequence,

$$X_i = \{x_i(1), x_i(2), \cdots, x_i(n)\}$$

where X_0 takes the maximum value of scaling ecological diversity of each vegetation in four periods as the optimal state, and X_i represents the scaling ecological diversity index of each vegetation in the i year. Secondly, the scaling ecological diversity index of each vegetation is dimensionless by using the averaging method, and the expression is:

$$x_i'(k) = \frac{x_i(k)}{\overline{X_i}}$$

Where:

$$\overline{X_i} = \frac{1}{n}\sum_{k=1}^{n} x_i(k)$$

k represents different forest vegetation; then calculate the grey correlation index of ecological diversity of each vegetation, and the expression is:

$$\gamma_{0i}(k) = \frac{\min_i \min_k |x_0'(k) - x_i'(k)| + \varepsilon \max_i \max_k |x_0'(k) - x_i'(k)|}{|x_0'(k) - x_i'(k)| + \varepsilon \max_i \max_k |x_0'(k) - x_i'(k)|}$$

$\varepsilon = 0.5$. Finally, the grey correlation evaluation index of tree diversity in Xishuangbanna in each period is calculated, and the expression is:

$$\gamma_{0i} = \sum_{k=i}^{n} \omega(k)\gamma_{0i}(k)$$

where $\omega(k)$ represents the weight obtained from the scaling ecological diversity index of 4 plantations, and the calculation formula is as follows:

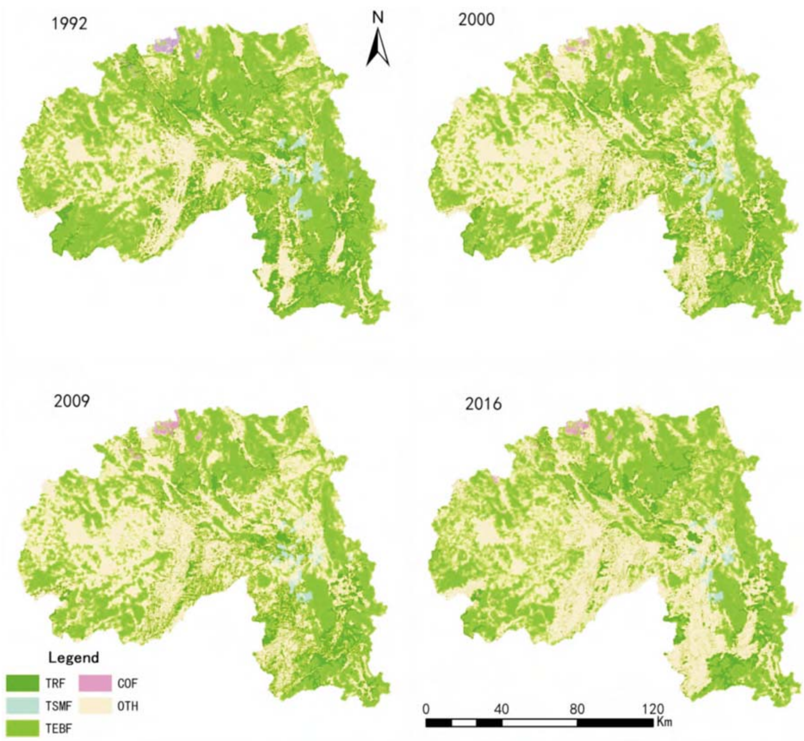
$$\omega(k) = \frac{\overline{x_k}}{\sum_{k=1}^{4} \overline{x_k}} \tag{5}$$

4. Results and analysis

4.1 Changes in forest vegetation area

According to the vegetation distribution in the four periods of remote sensing image interpretation (**Figure 2**), from 1992 to 2016, the forest area of Xishuangbanna generally showed a trend of decline first and then increase. The area of tropical rainforest decreased from 22.77% in 1992 to 15.1% in 2016; as the largest tropical mountain evergreen broad-leaved forest, it decreased from 41.6% in 1992 to 33.61% in 2000 and 33.35% in 2008, and increased to 38.4% in 2016; the distribution area of tropical seasonal moist forest and warm tropical coniferous forest in Xishuangbanna is small, both fluctuating below 1.2%; however, the area of other vegetation increased from 34.5% in 1992 to 45.27% in 2016.

It can also be seen from **Figure 2** that tropical rain forests are mainly distributed in the lower altitude areas in the middle and east of Xishuangbanna, tropical seasonal moist forests are distributed in the middle and East, and tropical mountain monsoon evergreen broad-leaved forests are distributed in Xishuangbanna, other vegetation is mainly distributed in the gentle terrain areas, and warm coniferous forests are distributed in the north of Pu'er City. From the comparison of time and space, the vegetation in Xishuangbanna has changed dramatically, especially the low-altitude tropical rainforest vegetation in Jinghong City and Mengla County has changed into other vegetation; the evergreen broad-leaved forest in tropical mountainous areas in the Midwest, northwest, and northeast of Xishuangbanna has been transformed into other vegetation; the low altitude tropical seasonal moist forest in the middle of Mengla County has been transformed into other vegetation.



Note: OTH. Other vegetations.

Figure 2. Map of main forest vegetation types in Xishuangbanna in 1992, 2000, 2009 and 2016.

4.2 Comparison of tree diversity of forest vegetation in the sample plot

From the number of arbor species in each sample area (**Table 1**), the average number of species in the tropical rainforest is 55, which is the vegetation with the largest number of arbor species; the second is 35 species of evergreen broad-leaved forests in tropical mountainous areas, higher than 29 species of tropical seasonal humid forests and 27 species of warm coniferous forests. In terms of the number of arbor plants, the number of trees in the tropical seasonal humid forest is 252, higher than that in tropical rainforest, 220, Simao Pine forest is 185, and the lowest number of trees in tropical mountain evergreen broad-leaved forest is 151. The evenness of the three species reflects that the evenness of different forest vegetation in Xishuangbanna is ranked as tropical rainforest > tropical mountain evergreen broad-leaved forest > warm coniferous forest > tropical seasonal humid forest.

Table 1. Tree diversity indices of main forest vegetation types in Xishuangbanna

Diversity index	TRF	TSMF	TEBF	COF
Area of each plot (m^2)	2,500	2,500	2,500	2,500
Plot number	4	1	2	1
Average species number	55 ± 11.34	29	35	27
Average plant number	220 ± 12.4	252	151	185
HS	0.95 ± 0.01	0.46	0.85	0.53
H	3.45 ± 0.28	1.35	2.70	1.54
D	3.72 ± 0.24	2.39	3.14	2.47

Note: Value = $\bar{x} \pm s$; TRF. Tropical rainforest; TSMF. Tropical seasonal moist forest; TEBF. Tropical lower montane evergreen broad-leaved forest; COF. Tropical coniferous forest. The same is below.

4.3 changes in tree diversity of forest vegetation in the whole region

Based on the survey data of each vegetation sample plot and the vegetation data interpreted by remote sensing images, the changes in tree scaling ecological diversity of each vegetation in different periods in Xishuangbanna are calculated (**Table 2**).

Table 2 Tree Scaling diversity index change and weight [ω(k)] of main forest vegetation types from 1992 to 2016

Vegetation	Scaling diversity index				Ω (k)
	1992	2000	2008	2016	
TRF	48.31	47.53	47.41	46.78	0.36
TSMF	23.90	23.47	22.84	22.81	0.17
TEBF	40.86	40.26	40.24	40.66	0.30
COF	22.87	21.64	22.84	22.73	0.17

It can be seen from **Table 2** that the scaling ecological diversity index of tropical rainforest is above 46, which is the most important vegetation to maintain the forest tree diversity in Xishuangbanna. The second is tropical mountain evergreen broad-leaved forest and tropical seasonal moist forest, with the scaling ecological diversity index of 40–41 and 22–24, respectively. The scaling ecological diversity index of warm tropical coniferous forest with Pinus Simao as the dominant species is lower than 23. The contribution of tree diversity of Xishuangbanna vegetation to tree diversity is in the order of tropical rainforest > tropical mountain evergreen broad-leaved forest > tropical seasonal humid forest > warm coniferous forest.

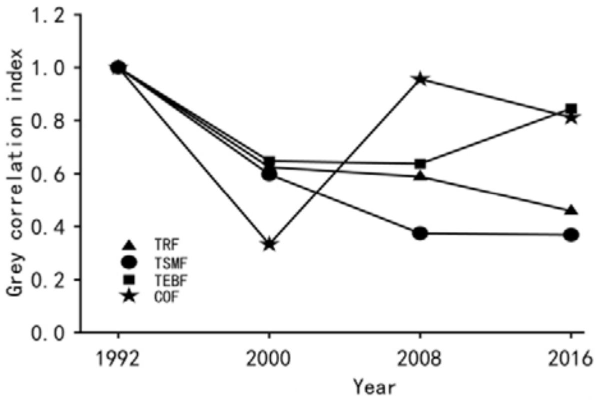


Figure 3. Change in the grey correlational index of tree diversity of main forest vegetation types in Xishuangbanna.

The grey correlation evaluation model was used to calculate the interannual changes of scaling ecological diversity of four forest vegetation trees in Xishuangbanna in four periods. From the changing trend of the grey correlation index (**Figure 3**), it is found that the tree diversity of tropical rainforest and tropical seasonal humid forest has a continuous decreasing trend during the study period, while the tree diversity of tropical mountainous evergreen broad-leaved forest shows a decreasing trend first and then increasing trend, and the warm coniferous forest shows an upward and downward fluctuation trend. The grey correlation evaluation indexes of forest tree diversity in Xishuangbanna in 1992, 2000, 2009, and 2016 were 1, 0.58, 0.63, and 0.62, respectively, so the order of forest tree diversity in Xishuangbanna in the four periods was 1992 > 2009 > 2016 > 2000.

5. Discussion

5.1 Cause analysis of tree diversity change

Economic activities are the main reason for the loss of biodiversity in Xishuangbanna. The results of this study show that the tree diversity of forest vegetation in Xishuangbanna has been losing continuously from 1992 to 2016, and the planting of economic crops such as rubber forests and tea gardens is the main reason for the loss of biodiversity[12,13]. Since the reform and opening up, with the increasing demand for rubber in production and life, a larger area of tropical rainforest has been destroyed to form rubber forests. Moreover, since 1992, the altitude of rubber forests has reached 1400 m[13], and some low-altitude evergreen broad-leaved forests in tropical mountains have also been destroyed to form rubber forests. Since the evergreen broad-leaved forest in the tropical mountains with higher altitudes in Xishuangbanna is not suitable for the planting of rubber forests, since 1990, the evergreen broad-leaved forest in the tropical mountains has been destroyed and formed tea gardens[12]. Xishuangbanna implemented the natural forest protection project[23] and the conversion of farmland to Forests Project[24] in 1998 and 2002, respectively. From 2000 to 2009, the biodiversity loss of tropical rain forests and tropical mountain evergreen broad-leaved forests in Xishuangbanna was not as severe as before, while the warm coniferous forests increased. However, after 2009, due to the impact of economic activities, the continuous rise of rubber prices led to the destruction of a large area of tropical rainforest in Xishuangbanna and the formation of rubber forests[12], which was contrary to the goal of tropical rainforest area restoration and development in the natural conservation project[25].

5.2 Significance of tropical rainforest in maintaining regional biodiversity

Tropical rainforest is the most important vegetation to maintain the forest tree diversity in Xishuangbanna. This study shows that the contribution of tropical rainforest to forest tree diversity in Xishuangbanna has always been the first in four periods. Because the area of tropical rainforest is second only to the evergreen broad-leaved forest in tropical mountains, and it is the second largest forest vegetation in Xishuangbanna. At the same time, in the same area quadrat, the evenness and richness of trees in the tropical rainforest are much higher than that of other forest vegetation, and the abundance is also maintained at a high level, so the loss caused by tropical rainforest per unit area is higher than that of other forest vegetation[26]. In addition, as an important indicator of ecosystem services, the value of biodiversity maintenance (biological regulation and genetic support) created by tropical rainforests per hectare per year is $41, while that of other forests is $4[27]. Therefore, from the perspective of biodiversity value assessment, the loss of tropical rainforests has a more serious impact on the maintenance function of regional biodiversity.

The loss of rare species in forests will lead to the loss of forest biodiversity[27], especially in tropical rain forests and tropical mountain evergreen broad-leaved forests. In the warm and hot coniferous forest and tropical seasonal humid forest with Simao Pine and closed flower trees as the dominant species, the number of other tree species does not account for a high proportion of the number of species in the whole community, so the reduction of the number of other tree species does not significantly reduce the evenness of this vegetation. However, in the tropical rainforest where the dominant species are not significant, the proportion of rare species in the number of community species is high, and the reduction of rare species will affect the loss of biodiversity of the whole community[28]. Although Fagaceae and Lauraceae trees are the dominant species in the evergreen broad-leaved forest in the tropical mountains, rare species also account for a high proportion. In conclusion, the loss of rare species in the forest has a greater impact on the tropical rainforest and tropical mountain evergreen broad-leaved forest.

5.3 Applicability analysis of scaling ecological diversity index

Scaling ecological diversity index combined with remote sensing technology can be applied to regional biodiversity assessment. Scaling, Simpson and Shannon Wiener indexes can measure community evenness. As the number of species increases with the increase of area, the scaling ecological diversity index also uses area index to measure species richness in the study area[8]. In addition, the combination of ground survey and remote sensing technology can also extract ecosystem types and monitor the integrity of vegetation. The traditional regional biodiversity research and the scaling ecological diversity index in this study combined with remote sensing technology coincide with the indicators with higher weight in species richness, ecosystem type and vegetation integrity[29,30], indicating that the scaling ecological diversity measurement indicators combined with remote sensing technology can be applied to the evaluation of regional biodiversity. However, different areas and different sampling standards will lead to no contrast in the calculation results of diversity. In order to make the calculation results of diversity among different vegetation comparative, it is necessary to unify the sampling standards of different vegetation sample plots in community investigation.

The scaling ecological diversity index also has problems in regional biodiversity assessment. Although scholars have concluded that the scaling ecological diversity index is not affected by the resolution within the spatial scale range of 30 m to 150 m resolution[31], at the same time, the species-area relationship can be applied to the scale transformation of biodiversity[32]. However, species diversity is retrieved from the community scale to the regional scale. The difficulty of scale conversion is affected by the sampling effect and habitat heterogeneity[32] The difficulty of scale conversion also increases with the increase of the area of the study area. Therefore, in the study area with a large area or high heterogeneity, the reliability of scaling ecological diversity index needs further research. In addition, how to apply the index to the comparison of different vegetation in different regions also needs more in-depth research. The grey correlation evaluation model has some problems, such as the lack of order-preserving effect in dimensionless processing, and the dissatisfaction with the standardization of correlation degree[20]. In future work, it is also necessary to develop or use the corresponding mathematical model to improve the evaluation system of forest vegetation arbor diversity in the study area.

Conflict of interest

The authors declare that they have no conflict of interest.

References

1. Office of Biodiversity Protection in Ministry of Environmental Protection. Progress of the ty biodiversity evaluation. Environment Protect 2010; (10): 23–26.
2. Fu B, Yu D, Lv N. An indicator system for biodiversity ecosystem services evaluation in China. Acta Ecologica Sinica 2017; 37(2): 341–348.
3. Li W, Zhang S. Research progress on GIS and remote sensing's application in ecological security assessment and biodiversity conservation. Acta Ecologica Sinica 2010; 30(23): 6674–6681.
4. Xu W, Wu B. Progress on measuring forest biodiversity with remote sensing technique. Acta Ecologica Sinica 2005; 25(5): 1199–1204.
5. Hu H, Li X, Du Y, *et al*. Research advances in biodiversity remote sensing monitoring. China Journal of Ecology 2012; 31(6): 1591–1596.
6. Yue T. Discussion on studying biodiversity by remote-sensing. Biodiversity Science 2000; 8(3): 343–346.
7. Yue T, Liu J, Li Z, *et al*. Considerable effects of diversity indices and spatial scales on conclusions relating to ecological diversity. Ecological Modelling 2005; 188: 418–431.
8. Yue T, Ma S, Wu S, *et al*. Comparative analyses of the scaling diversity index and its applicability. International Journal of Remote Sensing 2007; 28(7): 1611–1623.
9. Myers N, Mittermeier RA, Mittermeier CG, *et al*. Biodiversity hotspots for conservation priorities. Nature 2000; 403: 853–858.
10. Liu H, Xu Z, Duan Q. An approach to conserve plant diversity through the Dai's traditional beliefs in Xishuangbanna. Guihaia 2001; 21(2): 173–176.
11. Li H, Mitch AT, Ma Y, *et al*. Demand for rubber is causing the loss of high diversity rain forest in SW China. Biodiversity and Conservation 2007; 6(16): 1731–1745.
12. Liu X, Feng Z, Jiang L, *et al*. Spatial-temporal pattern analysis of land use and land cover change in Xishuangbanna. Resource Science 2014; 36(2): 233–244.
13. Li Z, Ma Y, Li H, *et al*. Relation of land use and cover change to topography in Xishuangbanna, Southwest China. Chinese Journal of Plant Ecology 2008; 32(5): 1091–1103.
14. Li H, Ma Y, Liu W, *et al*. Clearance and frag-fragmentation tropical rain forest in Xishuangbanna, SW, China. Biodiversity and Conservation 2009, 18(13): 3421–3440.
15. Zhu H, Xu Z, Wang H, *et al*. Over 30-year changes of floristic composition and population structure from an isolated fragment of tropical rain forest in Xishuangbanna. Acta Botanica Yunnanica 2001; 23(4): 415–427.
16. Liu J, Slik JW. Forest fragment spatial distribution matters for tropical tree conservation. Biological Conservation 2014; 171: 99–106.
17. Zhu H, Xu Z, Wang H, *et al*. Floristic composition and change of rain forest fragments in Xishuangbanna, southern Yunnan. Biodiversity Science 2000; 8(2): 139–145.
18. Zhu H, Wang H, Li BG, *et al*. Studies on the forest vegetation of Xishuangbanna. Plant Science Journal 2015; 33(5): 641–726.
19. Wang H, Zhu H, Li B. Vegetation on limestone in Xishuangbanna Southwest China. Guihaia 1997; 17(2): 101–117.
20. Zhang S, Zhang G. Comparison between computations of Grey Interconnect Degree and analysis on their shortages. System Engineer 1996; 14(3): 45–49.
21. Zou S, Chen Y, Xu W, *et al*. Impact factors of the total flavonoid content in Guizhou Miao medicine *Laportea bulbifera* based on grey correlation analysis. Guihaia 2017; 37(4): 461–469.
22. Sun X, Wang X. Study on soil fertility of forest type in Malan forest Region on the Loess P. Guihaia 2006; 26(4): 418–423.
23. Chen Z. The economic and social benefits generated after the implementation of Natural Forest Protection Program in Jinghong. Journal of Green Science and Technol 2015; 1: 138–139.
24. Li B, Huang J, Liu B. Impacts of sloping land conversion program on households' sustainable livelihood. Biotechnol World 2015; 3: 32–32.
25. Xiang Y. Some understanding of the implementation of "Natural Forest Protection" Program in Xishuangbanna. Yunnan For Invest Plan Des 2001; 26(1): 10–14.
26. Cao M, Zhang J. Tree species diversity of tropical forest vegetation in Xishuangbanna, SW China. Biodiverstiy Conservation 1997; 6(7): 995–1006.
27. Costanza R, Ralph A, Groot R, *et al*. The value of the world's ecosystem services and natural capital. Nature 1997; 387: 253–260.
28. Zhu H, Xu Z, Li B, *et al*. A discussion on the loss of biodiversity of the tropical rain forest by Amomum planting under-neath in South Yunnan. Guihaia 2002; 22(1): 55–60.
29. Wan B, Xu H, Ding H, *et al*. Methodology of com-prehensive biodiversity assessment. Biodiversity Science 2007; 15(1): 97–106.
30. Zhu W, Fan J, Wang Y, *et al*. Assessment of biodiversity conservation important in the upper reaches of the Yangtze River: By taking county area as the basic assessment. Acta Ecologica Sinica 2009; 29(5): 2603–2611.
31. Yue T, Ma S, Wu S, *et al*. Theoretical analysis of ecological diversity models and their application in Fukang of Xinjiang Uygur Autonomous Region. Chinese Journal of Applied Ecology 2006; 17(5): 867–872.
32. Tang Z, Qiao X, Fang J. Species-area relationship in biological communities. Biodiversity Science 2009; 17(6): 549–559.

Sustainable Forestry (2021) Volume 4 Issue 1
doi:10.24294/sf.v4i1.1603

ORIGINAL RESEARCH ARTICLE

Time of tree diversity recomposition along plant succession in the forests of the Chanchamayo Valley, Junín, Peru

Flavio Quintero Cardozo[1], Belvi Margui Cáceres Rodríguez[2], Carlos Augusto Reynel Rodríguez[1*], Robin Fernandez-Hilario[1,3], Akira Armando Wong Sato[1,3], Jorge Mario Chávez Salas[1], Sonia Cesarina Palacios Ramos[1]

[1] *Herbario Forestal MOL, Facultad de Ciencias Forestales, Universidad Nacional Agraria La Molina. Av. La Molina s/n, La Molina, Lima, Perú. E-mail: reynel@lamolina.edu.pe*

[2] *Servicio Forestal y de Fauna Silvestre (SERFOR). Av. Javier Prado Oeste N°2442, Urb. Orrantia, Magdalena del Mar, Lima, Perú.*

[3] *División de Ecología Vegetal - CORBIDI, Calle Santa Rita 105 Of. 2, Urb. Huertos de San Antonio Monterrico, Surco, Lima, Perú.*

ABSTRACT

A topic of current interest in forestry science concerns the regeneration of degraded forests and areas. Within this topic, an important aspect refers to the time that different forests take to recover their original levels of diversity and other characteristics that are key to resume their functioning as ecosystems. The present work focuses on the premontane rainforests of the central Peruvian rainforest, in the Chanchamayo valley, Junín, between 1,000 and 1,500 masl. A total of 19 Gentry Transects of 2 × 500 m, including all woody plants ≥2.5 cm diameter at breast height were established in areas of mature forests, and forests of different ages after clear-cutting without burning. Five forest ages were considered, 5–10, 20, 30, 40 and ≥50 years. The alpha-diversity and composition of the tree flora under each of these conditions was compared and analyzed. It was observed that, from 40 years of age, Fisher's alpha-diversity index becomes quite similar to that characterizing mature forests; from 30 years of age, the taxonomic composition by species reached a similarity of 69–73%, like those occurring in mature forests. The characteristic botanical families, genera and species at each of the ages were compared, specifying that as the age of the forest increases, there are fewer shared species with a high number of individuals. Early forests, up to 20 years of age, are characterized by the presence of Piperaceae; after 30 years of age, they are characterized by the Moraceae family.

Keywords: Premontane Forests; Secondary Forests; Forest Dynamics; Tree diversity; Gentry Transects

ARTICLE INFO

Received: 26 March 2021
Accepted: 10 May 2021
Available online: 21 May 2021

1. Introduction

In recent years, work has been carried out to better understand the floristic composition of the central Peruvian rainforest and how it characterizes the different forest types existing in the area[1-3]. Currently, the initiative of establishing permanent plots under standard methodologies in tropical forest areas has opened the door to successive remeasurements over time, facilitating a vision of forest dynamics. Research teams such as the Amazonian Forest Inventory Network (RAINFOR) are currently leading works that cover geographically wide areas, allowing a better understanding of flora composition, taxa distribution patterns and forest dynamics.

In the Department of Junín, most of the floristic studies have been carried out in the Chanchamayo Province[4-9], but despite these efforts, this is still one of the Amazonian departments with the lowest levels of botanical collections and knowledge[10]. In addition, in Chanchamayo and surrounding areas, more than 80% of the territory has been deforested[11], a scenario generated by the conversion of protected land use and forest use to agriculture or cattle ranching.

Following RAINFOR methodologies[12], it has been possible to generate relevant information on the composition of the different forest types existing in the Chanchamayo valley, through the installation of Permanent Monitoring Plots that include the marking of trees with more than 10 cm diameter at breast height (DBH)[4,6,8,13]. These studies have allowed us to better understand the floristic structure and composition of montane and premontane forests as a function of their successional stage.

To date, studies conducted in the pre-montane rainforests of the Chanchamayo valley reveal that pioneer forests established from a burn are characterized by presenting in their initial stage woody species such as *Vernonanthura patens* (Kunth) H. Rob. Rob. (Asteraceae) and *Acalypha* spp. (Euphorbiaceae) among the most important; after 25 years, the composition tends to be dominated by species such as *Piper aduncum* L. and *Allophylus foribundus* (Poepp.) Radlk. (Sapindaceae)[14]. On the other hand, the late secondary forests present species such as *Trophis caucana* (Pittier) C.C. Berg (Moraceae), *Cupania cinerea* Poepp. (Sapindaceae), *Inga edulis* Mart. (Fabaceae) and *Mauria heterophylla* Kunth (Anacardiaceae)[15]. In primary forests, the most representative species with DBH <40 cm are *Batocarpus costaricensis* Standl. & L.O. Williams (Moraceae), *Inga cinnamomea* Spruce ex Benth. (Fabaceae) and *Trophis caucana* (Pittier) C.C. Berg (Moraceae), and with DBH >40 cm, *Clarisia racemosa* Ruiz & Pav. (Moraceae) and *Pseudolmedia laevis* (Ruiz & Pav.) J.F. Macbr. (Moraceae)[15].

Currently, in the Chanchamayo valley, most of the remaining primary forests, or those where anthropic intervention has been minimal, are located in areas with steep slopes and, therefore, difficult to access; while secondary forests, from areas previously dedicated to coffee and citrus cultivation, present a great variety of ages and cover a large part of the valley. Although the aforementioned studies are allowing us to understand the composition and structure of these forests, there are still many gaps in knowledge to be studied. Proof of this is that, in the last 30 years, other works with a taxonomic focus have generated the discovery of new endemic tree species for the central rainforest[17-19]. Likewise, very few studies have focused on knowing the diversity of woody plants smaller than 10 cm DBH[20,21], and even fewer, those that have analyzed the times of forest recomposition generated from anthropogenic disturbances[14].

There is a clear need for further research to provide more information on the stages of floristic recovery and tree diversity associated with different forest ages. Therefore, in this research the following objectives were proposed: (i) to document the floristic composition and tree diversity at different forest ages, (ii) to analyze the floristic similarities between the different forest ages, and (iii) to determine the recovery time of the premontane rainforests in the study area.

2. Materials and Methods

2.1 Scope of study

The study area is the premontane forest, between 1,000–1,500 masl, in the Chanchamayo valley, district of San Ramón, Junín, Peru; there are tropical rainforest areas of different ages where the sample units were established. The study area and the ecological classification by Holdridge life zones are shown in **Figure 1**. The central sampling area is located in the Instituto Regional de Desarrollo (IRD), Fundo La Génova, of the Universidad Nacional Agraria La Molina, which has areas of intact forest in its innermost parts, and a mosaic of forests of different ages in the outermost parts, towards the Chanchamayo River. It covers an area of 300 hm^2, of which approximately 40% is covered by premontane tropical rainforest and the rest by tropical crops, mainly citrus and pineapple.

2.2 Survey of sample units in forests of different ages.

The sample units used are the so-called Gentry Transects (TG), consisting of a 2 × 500 m evaluation grid along which all plants ≥2.5 cm DBH were recorded and collected. The TG were randomly established in humid forest areas of different ages within the Fundo Santa Rosa and the IRD Fundo La

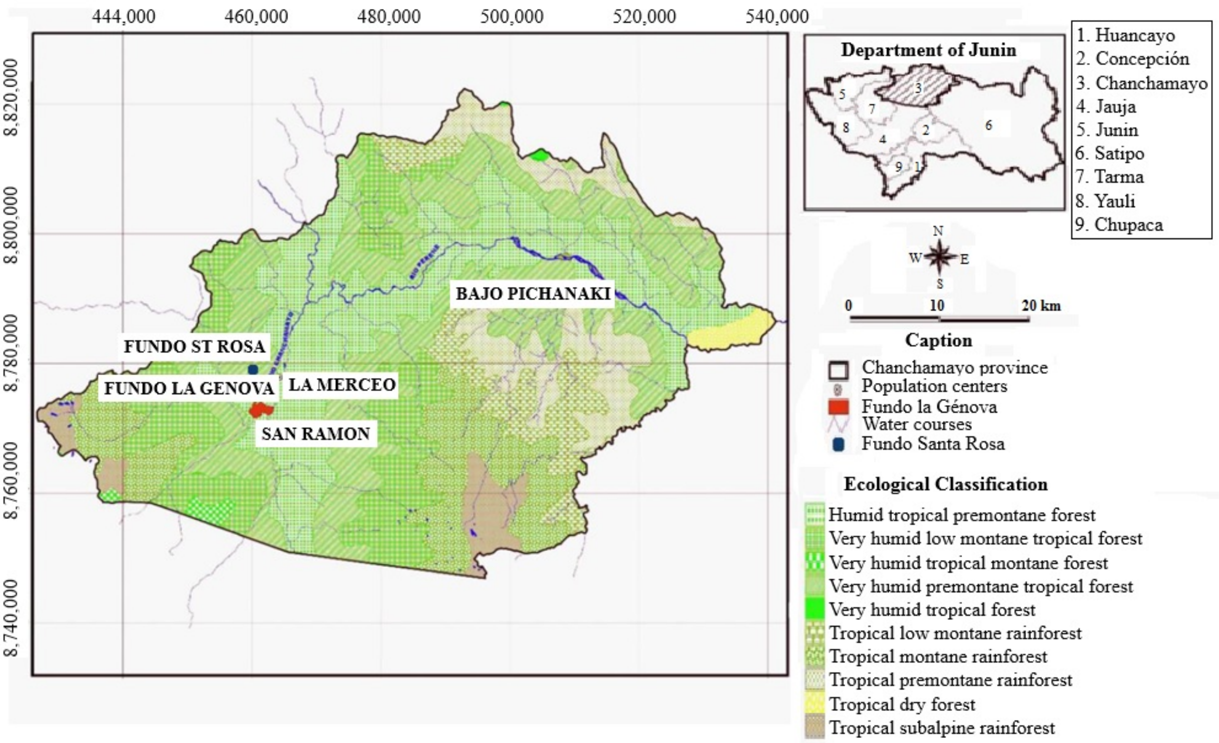


Figure 1. Stratification of the study area based on the Holdridge ecological classification system by life zones.

Table 1. Location of 0.1 ha Gentry Transects established at different approximate forest ages

Approximate ages	Transects	Coordinates			Altitude	Location
		Zone	East	South	(masl)	
<10 years	T1	18 L	461463	8776420	1,150	Santa Rosa
	T2	18 L	461529	8776525	1,150	Santa Rosa
	T3	18 L	461419	8776802	1,150	Santa Rosa
	T4	18 L	461479	8776781	1,150	Santa Rosa
20 years	T5	18 L	460753	8773466	1,190	IRD La Génova
	T6	18 L	460870	8773519	1,170	IRD La Génova
	T7	18 L	460986	8773493	1,130	IRD La Génova
30 years	T8	18 L	460284	8773083	1,280	IRD La Génova
	T9	18 L	460367	8773166	1,240	IRD La Génova
	T10	18 L	460492	8773232	1,230	IRD La Génova
40 years	T11	18 L	460437	8773458	1,260	IRD La Génova
	T12	18 L	460465	8773384	1,230	IRD La Génova
	T13	18 L	460613	8773484	1,290	IRD La Génova
>50 years	T14	18 L	460823	8772416	1,116	IRD La Génova
	T15	18 L	460841	8772586	1,152	IRD La Génova
	T16	18 L	460881	8772617	1,136	IRD La Génova

Génova UNALM. The main source of information on the age of the forest areas studied was the reference of workers of the farms, several of whom have lived in the interior of the farm since their childhood. Mature forests were considered to be those areas that have not suffered human intervention in the last 50 years; and their state of maturity was also reflected in their flora assemblage. For forest areas of different ages along the plant succession, only those resulting from anthropogenic intervention without burning were considered (plant succession in forests generated from burning can be consulted in Echía *et al.*[14]). All plants included were collected, properly preserved and conditioned to be deposited in the Herbarium and identified by taxonomists. At least three TGs were collected for each approximate age of the forests in the field, for a total of 16 TGs. The codes of the collected TG, with indication of their position and corresponding ages, are shown in **Table 1**. The botanical specimens collected were deposited in the Herbarium of the Faculty of Forestry Sciences of the UNALM (collection of woody plants of the MOL Herbarium), where they were botanically determined by C. Reynel and R. Fernandez-Hilario.

Table 2. Diametric and altimetric classes established for the analysis of Alpha Diversity in the Gentry Transects

Classes		Intervals (cm)
Diametric	1	2.5–4.99
	2	5–9.99
	3	10–14.99
	4	15–19.99
	5	20–24.99
	6	25–29.99
	7	30–34.99
	8	35–39.99
	9	>40
Altimetric	1	1–4.99
	2	5–9.99
	3	10–14.99
	4	15–19.99
	5	20–24.99
	6	25–29.99
	7	>30

2.3 Analysis performed

2.3.1 Alpha diversity

Diameter classes of 5 cm were established to analyze the number of species (alpha diversity) recorded in each class. Likewise, for the altitudinal classes, 5 m classes were established (**Table 2**). The transects were analyzed as a whole for each of the ages studied.

2.3.2 Recomposition of alpha diversity (Dα) in forests of different ages

In the comparison of alpha diversity (number of species per unit area) between sample units of different ages, the absolute values of diversity in each case were taken into account, as well as the values obtained with Fisher's Alpha index, which corrects for the differences caused by different numbers of individuals in each Gentry Transect[22]. In tree diversity studies, this index shows a high stability with respect to sample size, since it depends on the number of individuals sampled[23,24]. Likewise, this index takes into account the positive effect that abundance has on diversity, allowing it to estimate the diversity of large geographic areas using samples from small areas and to compare sampling units[24,25]. Fisher's alpha index was calculated using Formula 1.

$$S = \alpha \ln[1 + {}^{N}/_{\alpha}]$$

Where:

α = Fisher's alpha index;

S = number of species in the sample;

N = number of individuals in the sample.

Fisher's alpha index was calculated with the PAST program, version 1.91[26].

2.3.3 Comparison of floristic composition in forests of different ages

The comparison was made by means of a similarity analysis using the Dice Index[27] and a cluster analysis. For both cases, the PAST program version 1.91 was used[26].

3. Results

3.1 Alpha diversity recomposition (Dα)

Regarding the recomposition of the Dα in forests of different ages (**Table 3**), it can be seen that the values of this parameter are very modest between the ages of <10 years, becoming considerably higher from the age range 20 years onwards. When the number of individuals recorded using Fisher's alpha diversity index is taken into consideration, it is also observed that from 40 years onwards there is a substantial recovery of the diversity of tree species present.

Observation of the data from the TGs collected shows that, in all cases, the first three diameter classes, 2.5–14.99 cm in diameter, register the highest abundances (**Table 4**). The values found suggest that, at the level of species and individuals, there is a great difference between pioneer forests (less than 10 years old) and forests older than 20 years. Also, in **Figure 2** it can be seen (except for the forest <10 years old) that the curves reflect an "inverted J" distribution where the largest number of individuals are found in the lower diameter classes. This indicates a clear tendency towards forest replenishment. With respect to the results obtained for the altitudinal classes (**Table 5**), we can see that both the greatest number of species and individuals are found in the second class (5–9.99 m), and although we would expect a "normal curve" trend in the altitudinal classes (**Figure 3**), this tends to the left, with the greatest number of individuals grouped in the first three classes.

Regarding the composition by families at different ages (**Table 6**), some interesting trends can be observed. It is only after the age of 40 that the Piperaceae family ceases to be a markedly abundant component in the forest. Similarly, after20 years of age, the Moraceae family begins to present a greater number of individuals, until it becomes the most important in mature forests (>50 years). With respect to diversity, in all forest ages the Fabaceae and Lauraceae families, in a fairly stable manner, present the greatest number of species, although the species are not necessarily the same at all ages. We also observed that in mature forests (>50 years), additional families appear that had not been recorded at previous ages (Connaraceae, Nyctagynaceae), and Annonaceae emerges as one of the most diverse families.

3.2 Fisher's alpha index

Although the highest number of individuals and species is recorded at the oldest forest ages (40 and >50 years), these do not necessarily represent the highest Fisher's alpha values. The maximum value is recorded for 40 years old forests. It is important to highlight that forests <10 years old present a high Fisher's alpha variability (5.78–14.66) and at least two of the corresponding transects show the lowest diversity of the entire evaluation (**Table 3**).

Table 3. Values of species, families and individuals in the different forest ages evaluated. Fisher's alpha is the average value found for the transects of each age

Approximate ages	Transects	Species		Families	Individuals		Fisher's Alpha	
<10 years	T1	11	48	19	33	188	5.78	10.2
	T2	16			29		14.66	
	T3	18			62		8.51	
	T4	22			64		11.85	
20 years	T5	58	95	32	491	1378	17.1	17.9
	T6	55			471		16.14	
	T7	63			416		20.64	
30 years	T8	56	87	28	601	1841	15.1	15.4
	T9	55			600		14.74	
	T10	60			640		16.21	
40 years	T11	80	108	33	683	1897	23.51	18.7
	T12	63			578		18	
	T13	56			636		14.8	
>50 years	T14	63	97	33	709	2002	16.7	18.4
	T15	75			684		21.48	
	T16	61			609		16.88	

Table 4. Number of species and individuals (in parentheses) by diameter class in the evaluated forest ages

Diameter class	<10 years	20 years	30 years	40 years	>50 years
1	24 (45)	63 (520)	68 (793)	82 (905)	71 (906)
2	31 (82)	68 (434)	59 (576)	71 (518)	59 (497)
3	14 (32)	39 (178)	46 (204)	48 (182)	39 (252)
4	9 (15)	34 (92)	27 (90)	33 (94)	42 (151)
5	3 (3)	19 (47)	21 (60)	30 (74)	26 (61)
6	4 (4)	12 (36)	15 (40)	17 (51)	25 (47)
7	2 (2)	10 (21)	16 (34)	12 (25)	20 (33)
8	2 (2)	4 (21)	4 (6)	7 (13)	14 (20)
9	3 (3)	5 (29)	11 (38)	15 (35)	10 (35)

Table 5. Number of species and individuals (in parentheses) by altitudinal class in the evaluated forest ages

Height class	<10 years	20 years	30 years	40 years	>50 years
1	16 (32)	50 (225)	51 (329)	55 (292)	46 (264)
2	35 (114)	77 (849)	72 (1 040)	85 (1 075)	71 (1 017)
3	13 (25)	37 (107)	51 (198)	48 (203)	39 (232)
4	7 (11)	29 (114)	33 (124)	47 (158)	37 (189)
5	4 (5)	13 (49)	13 (29)	25 (61)	33 (103)
6	1 (1)	5 (26)	15 (60)	23 (66)	30 (82)
7	0 (0)	1 (8)	12 (61)	13 (42)	29 (115)

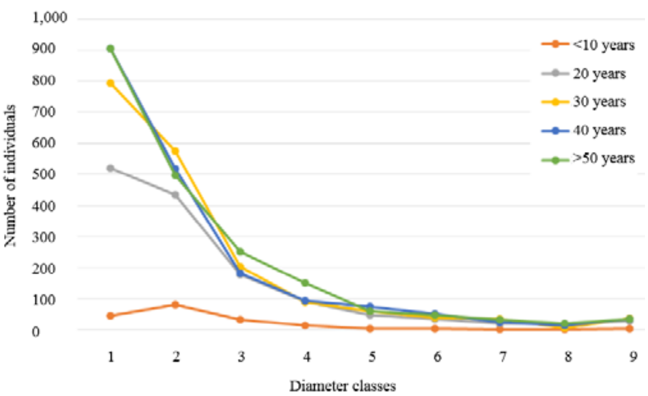


Figure 2. Diameter classes in the five forest ages evaluated.

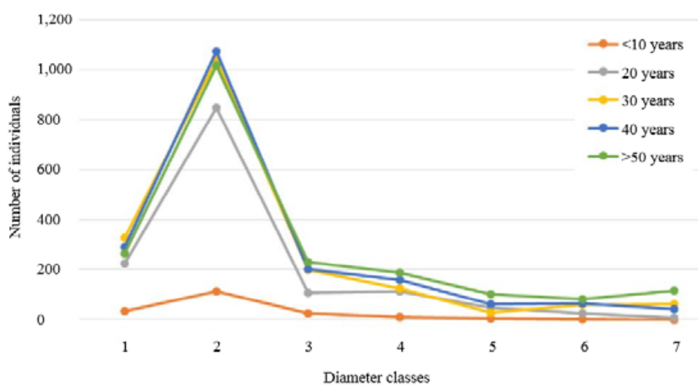


Figure 3. Altimetric classes in the five forest classes evaluated.

Table 6. The ten most diverse and abundant families for each forest age evaluated

N°	<10 years Family	Esp.	Family	Ind.	20 years Family	Esp.	Family	Ind.
1	Fabaceae	9	Fabaceae	31	Fabaceae	12	Piperaceae	303
2	Lauraceae	5	Lauraceae	25	Lauraceae	9	Urticaceae	230
3	Euphorbiaceae	4	Piperaceae	25	Melastomataceae	8	Moraceae	167
4	Solanaceae	4	Juglandaceae	24	Rubiaceae	7	Fabaceae	106
5	Malvaceae	3	Euphorbiaceae	20	Urticaceae	7	Euphorbiaceae	84
6	Piperaceae	3	Malvaceae	12	Euphorbiaceae	6	Melastomataceae	78
7	Urticaceae	3	Urticaceae	11	Moraceae	5	Arecaceae	69
8	Anacardiaceae	2	Rutaceae	8	Piperaceae	5	Rubiaceae	64
9	Melastomataceae	2	Myrtaceae	6	Malvaceae	4	Lauraceae	53
10	Moraceae	2	Anacardiaceae	5	Anacardiaceae	3	Malvaceae	53
N°	**30 years Family**	**Esp.**	**Family**	**Ind.**	**40 years Family**	**Esp.**	**Family**	**Ind.**
1	Fabaceae	11	Moraceae	360	Lauraceae	14	Moraceae	236
2	Lauraceae	10	Urticaceae	282	Fabaceae	11	Urticaceae	222
3	Urticaceae	9	Piperaceae	281	Piperaceae	9	Lauraceae	161
4	Moraceae	6	Fabaceae	275	Melastomataceae	8	Arecaceae	158
5	Malvaceae	5	Meliaceae	115	Urticaceae	8	Meliaceae	149
6	Melastomataceae	4	Lauraceae	106	Moraceae	7	Fabaceae	121
7	Meliaceae	4	Malvaceae	84	Malvaceae	5	Piperaceae	120
8	Myrtaceae	4	Myrtaceae	64	Arecaceae	4	Ochnaceae	101
9	Rutaceae	4	Arecaceae	55	Meliaceae	4	Rubiaceae	89
10	Arecaceae	3	Solanaceae	31	Rubiaceae	4	Myrtaceae	65
N°	**>50 years Family**	**Esp.**	**Family**	**Ind.**				
1	Lauraceae	12	Moraceae	633				
2	Fabaceae	9	Lauraceae	316				
3	Moraceae	8	Meliaceae	170				
4	Piperaceae	8	Nyctaginaceae	135				
5	Melastomataceae	7	Rubiaceae	87				
6	Urticaceae	6	Fabaceae	76				
7	Malvaceae	4	Urticaceae	64				
8	Annonaceae	3	Connaraceae	63				
9	Arecaceae	3	Piperaceae	60				
10	Euphorbiaceae	3	Myrtaceae	58				

(Esp. = Number of species; Ind. = Number of individuals).

3.3 Re-composition of the species assemblage (floristic composition)

The similarity analysis (Dice index; **Table 7**) reveals that the floristic composition of early ages (<10 years) is not very compatible with that of forests older than 20 years, with a similarity of less than 24%. The similarity of the species assemblages is noticeable in forests from 20 years and older, and very noticeable in forests 30 years and older, at which age 69–74% of the forest species are the same as at older ages. Forests between 30 and 40 years of age are quite similar to each other in that sense; and forests older than 50 years have a slightly different species assemblage.

The clustering analysis (**Figure 4**) shows that the <10 years old forest forms a solitary group, with only 20% similarity with the rest of the ages. on the other hand, the 30, 40 and >50 years old forests present a similarity of approximately 70%. Likewise, the two ages with the highest similarity (about 70%) are the 30 and 40-year-old forests. This suggests that, in the first stage, when the forest is a pioneer, there are very few species shared with mature forests, and only after 20 years of age do the species that characterize mature forests begin to proliferate.

Along the diagonal (from left to right and from top to bottom) is the number of species by forest age. Below the diagonal is the number of species in common among forest ages, and above is the Dice index. The minimum and maximum values recorded are highlighted in orange and green, respectively.

Table 7. Similarity between age groups according to the Dice Index.

Approximate ages	<10 years	20 years	30 years	40 years	>50 years
<10 years	48	0.238	0.222	0.192	0.152
20 years	17	95	0.659	0.640	0.594
30 years	15	60	87	0.738	0.685
40 years	15	65	72	108	0.693
>50 years	11	57	63	71	97

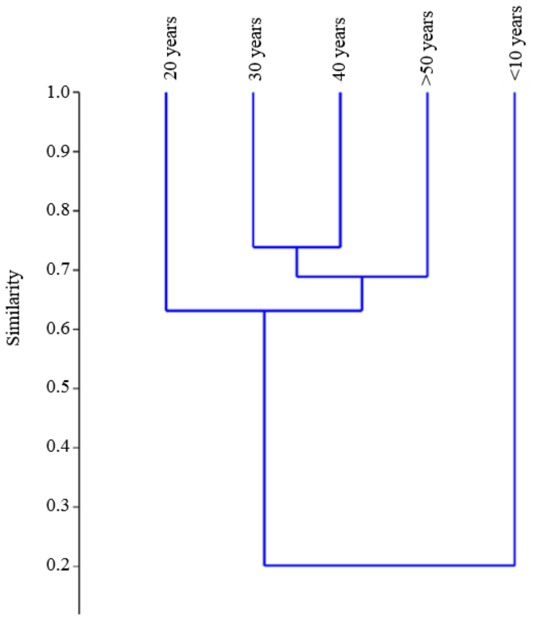


Figure 4. Dendrogram of similarity using the Dice Index for the forest ages evaluated.

Finally, we observed that few of the species shared between different ages show high abundance (**Table 8**). Most of the shared species present few individuals, the extreme case being the species in common between forests <10 years old and >50 years old, where only one presents more than 10 individuals at both ages. As the age of the forests increases, there are fewer shared species with a high number of individuals. Seven species with more than 50 individuals are shared in forests of 40 years and >50 years. From 20 years on, the species *Trophis caucana* is an abundant component in common with mature forests (>50 years).

4. Discussion

The premontane rainforests of the Amazon are one of the natural formations with the greatest anthropogenic impact and alteration in Peruvian territory. Productivity in montane forests is lower than in lowlands, consequently, recovery rates are relatively slower, in an environment marked by heterogeneity, where local factors such as landslides are common, and where species with divergent evolutionary histories interact[28]. An influential factor in the recomposition of tropical Andean forests is the edge effect of the surrounding vegetation[29].

Table 8. Species in common between mature forest (>50 years) and the other forest ages evaluated and based on a minimum number of individuals recorded

Approximate age of forest	>50 years Species in common				
	In total	With >10 individuals at both ages	With >20 individuals at both ages	With >30 individuals at both ages	With >50 individuals at both ages
<10 years	11	1	-	-	-
20 years	57	10	3	2	1 (*Trophis caucana*)
30 years	63	20	10	5	3 (*Myrcia splendens*; *Guarea guidonia*; *Trophis caucana*)
40 years	71	23	13	8	7 (*Prunus debilis*; *Ruagea glabra*; *Myrcia splendens*; *Ocotea cernua*; *Clarisia racemosa*; *Guarea guidonia*; *Trophis caucana*)

It has been postulated that a disturbed area embedded in a surrounding forest matrix with an intact structure and composition has significantly better chances of recovery than one surrounded by highly degraded forests[30]. To understand the recovery process of these forests, it is important to clarify the time it takes for them to recompose, from the point of view of their diversity and also of the species assemblages that characterize them. Based on such information it would be possible, for example, to find technical alternatives to shorten these times.

The data obtained show that, for secondary forests that have not been subjected to burning, the early stages of vegetation, also known as pioneers (up to 10 years of age), are very different in species abundance and diversity compared to the corresponding mature forests, containing considerably less diversity. In contrast to lowland Amazon forests, where at 20 years there is a recovery of 80% of species richness[31], the most important moments in the recomposition of these forests are shown above all from 30 years of age, time in which a similarity close to 70% with mature forests is reached. An open question for future research is what properties are achieved at these ages that allow the rapid recovery of diversity and species assemblage characteristic of mature forest formations. The rate of recovery could be limited by soil fertility and soil type[30]; in this sense, speculatively, it could be assumed that important properties related to soil fertility and carrying capacity, as well as the increase of biotic interveners such as seed dispersers, and the microclimate produced in the forest, could show substantial changes from these times onwards.

On the other hand, in the case of the premontane rainforest studied, the presence of some taxa could be interpreted as an indicator of the age and state of recovery of the forest; Piperaceae is a family characteristically present at early ages, remaining abundant until 30–40 years old, represented mainly by the species *Piper aduncum*; *Moraceae*, represented by the species *Trophis caucana*, is perceived as a family indicator of the mature condition of the forest in the studied forest. This coincides with the observations of Gentry & Ortiz[32]. If we compare these data with those obtained in forests regenerating after burning[14], we observe that in the latter the recovery is apparently slower. Considering the age range between 25–30 years, in the latter 57% of the diversity is recovered in that period, vs. 69–74% in forests that have not been subjected to burning, analyzed in the present study.

5. Conclusions

The floristic composition at different ages in the premontane moist forests studied is recognizable; additionally, the presence and abundance of some taxa can be interpreted as an indicator of the age and state of recovery of the forest; for example, early secondary forests, up to 20 years old, are characterized by the abundance of the Piperaceae family, with *Piper aduncum* as the most abundant species, while mature forests, after 30 years of age, are characterized by the Moraceae family, with the species *Trophis caucana*.

From the point of view of their diversity and flora assemblage, early forests, less than 20 years old, are significantly less diverse and differentiated in their floristic composition with forests older than 30 years (less than 24% similarity); after this age, species diversity increases significantly.

In the humid premontane forests of the study area, the most important moments in the recomposition of diversity and species assemblage occur after 20 years of age, and especially after 30 years of age, at which time a similarity of nearly 70% is reached with the composition of the flora of mature forests.

Conflict of interest

The authors declare that they have no conflict of interest

References

1. Vásquez R, Rojas R, Monteagudo A, *et al*. Flora Vascular de la selva central del Perú: Una aproximación de la composición florística de tres Áreas Naturales Protegidas (Spanish) [Vascular of the central jungle of Peru: An approximation of the floristic composition of three Protected Natural Areas]. Arnaldoa 2005; 12(1-2): 112–125.
2. Monteagudo A, Huamán M. Catalog of woody plants trees in the Selva Central of Peru. Arnaldoa 2010; 17(2): 203–242.
3. Marcelo J, Reynel C. Diversity patterns and floristic composition of permanent evaluative plots in the peruvian central forest. Rodriguesia 2014; 65(1): 35–47.
4. Antón D, Reynel C (editors). Relictos de bosques de excepcional diversidad en los andes centrales del Perú (Spanish) [Exceptionally diverse forest relicts in the central Andes of Peru]. Lima, Perú: Herbario de la Facultad de Ciencias Forestales, Universidad Nacional Agraria La Molina; 2004.
5. Cáceres P, Reynel C. Los árboles de *Ficus* (“Ojé”) del valle de Chanchamayo, Dpto. Junín, Perú (800–2500 msnm) (Spanish) [The *Ficus* (“Ojé”) trees of the Chanchamayo Valley, Department of Junín, Peru (800 – 2,500 masl)]. Lima, Perú: Asociación Peruana para la Promoción del Desarrollo Sostenible (APRODES); 2010.
6. Aguilar M, Reynel C. Dinámica forestal y regeneración en un bosque montano nublado de la selva central del Perú (Spanish) [Forest dynamics and regeneration in a montane cloud forest of the central jungle of Peru]. Lima, Perú: Herbario de la Facultad de Ciencias Forestales, Universidad Nacional Agraria La Molina; 2011.
7. Palacios S, Reynel C. Una formación vegetal subxerófila en el valle de Chanchamayo, Dp. de Junín (Spanish) [A sub xerophytic plant formation in the Chanchamayo valley, Department of Junín]. Lima, Perú: Herbario de la Facultad de Ciencias Forestales, Universidad Nacional Agraria La Molina; 2011.

8. Buttgenbach H, Vargas C, Reynel C. Dinámica forestal en un bosque premontano del valle de Chanchamayo (Dp. de Junín, 1,200 msnm) (Spanish) [Forest dynamics in a premontane forest of the Chanchamayo valley (Department of Junín, 1,200 masl)]. Lima, Perú: Herbario de la Facultad de Ciencias Forestales, Universidad Nacional Agraria La Molina; 2012.
9. Reynel C. Flora y fauna del bosque montano nublado Puyu Sacha, valle de Chanchamayo, Dp. Junín (1,800–3,200 msnm) (Spanish) [Flora and fauna of the cloudy montane forest Puyu Sacha, Chanchamayo valley, Department of Junín (1,800 – 3,200 masl)]. Lima, Perú: Asociación Peruana para la Promoción del Desarrollo Sostenible (APRODES); 2012.
10. Honorio E, Reynel C. Vacíos en la colección de la flora de los Bosques Húmedos del Perú (Spanish) [Gaps in the collection of the flora of the Humid Forests of Peru]. Lima, Perú: Herbario de la Facultad de Ciencias Forestales, Universidad Nacional Agraria La Molina; 2003.
11. Reynel C, León J. Especies forestales comunes de los bosques secundarios de Chanchamayo, Perú (Spanish) [Common forest species of the secondary forests of Chanchamayo, Peru]. Lima, Perú: Universidad Nacional Agraria La Molina, Facultad de Ciencias Forestales Proyecto Utilización de Bosques Secundarios en el Trópico Húmedo Peruano; 1989.
12. Phillips O, Baker T, Feldpausch T, *et al*. RAINFOR Manual de campo para la remedición y establecimiento de parcelas permanentes (Spanish) [RAINFOR Field manual for the remediation and establishment of permanent plots]. Brazil: FORESTPLOTS; 2009.
13. Giacomotti J, Reynel C. Tree mortality and recruitment in a late secondary forest of the Chanchamayo valley, Peru. Revista Forestal del Perú 2018; 33(1): 42–51.
14. Echía E, Reynel C, Manta I. 2019. The woody flora established after burning in the Chanchamayo Valley–Rainforest of central Peru. Revista Forestal del Perú 2019; 34(1): 83–101.
15. Almeyda A. Diversidad y composición de la flora arbórea en un área de bosque secundario tardío: Fundo Génova-UNALM, valle de Chanchamayo, 1,000–1,500 msnm (Spanish) [Diversity and composition of the arboreal flora in a late secondary forest area: Fundo Génova-UNALM, Chanchamayo valley, 1,000–1,500 masl]. In: Antón D, Reynel C (editors). Relictos de bosques de excepcional diversidad en los andes centrales del Perú. Lima, Perú: Herbario de la Facultad de Ciencias Forestales, Universidad Nacional Agraria La Molina; 2004. p. 263–302.
16. Reynel C, Antón D. Diversidad y composición de la flora arbórea en un área de cumbre de colinas en bosque premontano: Fundo Génova-UNALM, valle de Chanchamayo, 1,000–1,500 msnm (Spanish) [Diversity and composition of tree flora in a hilltop area in premontane forest: Fundo Génova-UNALM, Chanchamayo valley, 1,000–1,500 masl]. In: Antón D, Reynel C. (editors). Lima, Perú: Relictos de bosques de excepcional diversidad en los andes centrales del Perú; 2004. p. 143–186.
17. Pipoly III JJ. New species of Geissanthus (Myrsinaceae) from the Hylaea/Andean interface of Ecuador and Peru. SIDA, Contributions of Botany; 1996; 17(2): 459–470.
18. Arroyo F. A new species of Magnolia (Magnoliaceae) from central Peru. Phytotaxa 2014; 167(2): 220–222.
19. Berg CC, Ulloa C. Two new species of Coussapoa (Urticaceae, Cecropieae). Novon: A Journal for Botanical Nomenclature 2014; 23(1): 14–17.
20. Phillips O, Miller J. Global patterns of plant diversity: Alwyn H. Gentry's forest transect data set. MSB (Monographs in Systematic Botany). Missouri Botanical Garden Press 2002; 89: 1–319.
21. Palacios-Ramos S, Montenegro R., Linares-Palomino RL, *et al*. Forest dynamics of a sub-xeophylous vegetation formation in central Peru–Chanchamayo valley, Peru. Revista Árvore 2018; 42(6): e420603.
22. Fisher RA, Corbet AS, Williams CB. The relation between the number of species and the number of individuals in a random sample of an animal population. The Journal of Animal Ecology 1943; 12(1): 42–58.
23. Rosenzweig ML. Species diversity in space and time. Cambridge, UK: Cambridge University Press; 1995.
24. Condit R, Hubbell SP, Lafrankie JV, *et al*. Species-area and species-individual relationships for tropical trees: A comparison of three 50-ha plots. Journal of Ecology 1996; 84(4): 549–562.
25. Berry PE, Guariguata MR, Kattan GH. Diversidad y endemismo en los bosques neotropicales de bajura (Spanish) [Diversity and endemism in neotropical lowland forests]. In: Guariguata MR, Kattan GH (editors). Ecología y conservación de bosques neotropicales. Cartago, Costa Rica: Editorial Tecnológica; 2002. p. 83–96.
26. Hammer Ø, Harper DAT, Ryan PD. PAST: Paleontological statistics software package for education and data analysis. Paleontología Electrónica 2001; 4(1): 1–9.
27. Dice LR. Measures of the amount of ecologic association between species. Ecology 1945; 26(3): 297–302.
28. Crausbay SH, Martin P. Natural disturbance, vegetation patterns and ecological dynamics in tropical montane forests. Journal of Tropical Ecology 2016; 32(5): 384–403.
29. Bueno A, Llambí L. Facilitation and edge effects influence vegetation regeneration in old-fields at the tropical Andean forest line. Applied Vegetation Science 2015; 18(4): 613–623.
30. Chadzon R. Tropical forest recovery: Legacies of human impact and natural disturbances. Perspectives in Plant Ecology, Evolution and Systematics

2003; 6(1–2): 51–71.
31. Rozendaal DM, Bongers F, Mitchell Aide T, *et al*. Biodiversity recovery of Neotropical secondary forests. Science Advances 2019; 5(3): eaau3114.
32. Gentry A, Ortiz R. Patrones de composición florística en la Amazonia peruana (Spanish) [Patterns of floristic composition in the Peruvian Amazon]. In: Kalliola R, Puhakka M, Danjoy W (editors). Amazonia peruana, vegetación húmeda tropical en el llano subandino. Lima, Perú: Proyecto Amazonia, Universidad de Tu (PAUT) and Oficina Nacional de Evaluación de Recursos Naturales (ONERN); 1993. p. 155–166.

Sustainable Forestry (2021) Volume 4 Issue 1
doi:10.24294/sf.v4i1.1604

ORIGINAL RESEARCH ARTICLE

The relationship between species diversity and tree growth in natural secondary forests in Northeast China

Ye Luo[1,2], Huaijiang He[1,2], Zhonghui Zhang[1,2*], Ting Liu[1,2], Jingqi Yuan[1,2], Zhongliang Yu[1,2], Guangdao Bao[1,2], Dawei Zhang[1,2]

[1] *Jilin Provincie Academy of Forestry Science, Chang-chun 130033, Jilin, China. E-mail: zzhgiszzh@163.com*

[2] *Interregional Cooperation Science and Technology Innovation Center for Restoration and Reconstruction of Degraded Forest Ecosystem in Jilin Province, Changchun 130033, Jilin, China.*

ABSTRACT

[Objective] To understand the relationship between species diversity and tree growth in natural secondary forests in Northeast China, to determine the reasonable size of species diversity, and to carry out appropriate nurturing harvesting and artificial replanting, so as to provide a scientific and theoretical basis for secondary forest management and management. **[Methods]** A total of 123 sample plots were set up in the Xiaoxinganling (XXAL), Zhangguangcailing (ZGCL), Laojialing (LYL), Changbai Mountain (CBS), Hadaling (HDL) and Longgang Mountain (LGS) areas in Northeast China, they were used to investigate the species composition, importance value, diversity and tree growth in each area. **[Results]** A total of 48 species belonging to 17 families and 31 genera were investigated in all the sample plots, among which the sample plots in Longgang Mountain contained the largest number of families, genera and species, followed by Hada Ling, Changbai Mountain, Laoyaling, Zhangguangcai Mountain and Xiaoxinganling. The α-diversity index of species in the sample sites was the largest in Changbai Mountain and the smallest in Xiaoxinganling, and the difference between them was significant ($P < 0.05$), while the richness index was the largest in Longgang Mountain and the smallest in Xiaoxinganling. The difference between them was significant ($P < 0.05$), while the greater the difference in latitude between the regions, the more obvious the difference in β-diversity index of species in the sample sites, and the fewer species shared between the two regions. The higher the rate of community succession, the higher the average diameter at breast height and the average tree height in each region were CBS > LYL > LGS > ZGCL > HDL > XXAL. The largest breast tree species in each region was Mongolian oak in Changbai Mountain with a diameter at breast height of 64.8 cm, and the smallest breast tree species in each region was Tyrannus sylvestris in Longgang Mountain with a diameter at breast height of 4.0 cm. The highest tree species in each region was Liriodendron sylvestris in Longgang Mountain with a height of 28.9 m, and the smallest species is yellow pineapple with a height of 1.3 m in Longgang Mountain. **[Conclusion]** Within a certain range, species diversity has a facilitating effect on the average diameter at breast height and average tree height of species within a stand. Therefore, during the management of secondary forests, appropriate nurturing harvesting and artificial replanting should be adopted to ensure reasonable species diversity in the stands and provide optimal space for the growth of natural secondary forests.

Keywords: Natural Secondary Forest; Species Composition; Importance Value; Species Diversity; Tree Height; Diameter at Breast Height

ARTICLE INFO

Received: 2 April 2021
Accepted: 20 May 2021
Available online: 4 June 2021

1. Introduction

Since the 20th century, with anthropogenic activity disturbance and destruction of primary forests, natural secondary forests in China have accounted for half of the total forest area[1], especially in northeast China, natural secondary forests have become the main forest stand type, so conducting experiments related to natural secondary forests has become an important research content today. Although most scholars have researched natural secondary forest community composition, structure, diversity, biomass, and ecosystem function[2-6], fewer studies have been conducted on the relationship between species diversity and tree growth in natural secondary forests on a large regional scale in Northeast China.

The study of the relationship between species diversity and tree growth is a fundamental issue in ecology[7], and the relationship between the two is one of the more studied issues in ecology, including the relationship between species diversity and diameter at breast height, tree height, biomass and productivity[8]. Studying the interrelationships is essential for understanding the mechanisms of global species diversity[9].

Species diversity research is currently focused on the relationships between species diversity and biomass and productivity[10-13], the effects of environmental factors on species diversity at different altitudes and latitudes[14,15], the effects of anthropogenic disturbances on species diversity and functional diversity[16,17], and the spatial pattern distribution of species diversity[18]. Related studies have been relatively systematic and extensive. However, there are relatively few studies on the relationship between species diversity and tree growth in forest ecosystems[19]. And they mainly focus on small-scale and homogeneous habitats, which ignore the influence of large-scale and spatial heterogeneity on the relationship between species diversity and tree growth. Meanwhile, the ecosystems in forest stands are affected by many factors, and it is impossible to limit all experimental factors at the same time, resulting in the current conclusions on the relationship between species diversity and tree growth in forest stands. The relationship between species diversity and tree growth in forest stands is not uniformly concluded[20]. Therefore, it is important to focus on the relationship between species diversity and tree growth in a large region of Northeast China.

There are three main spatial scales of diversity indices: α diversity, β diversity, and γ diversity[21]. Each spatial scale has different environments and different measured data. A diversity is currently the most studied biodiversity[4,13,22]. A diversity is mainly concerned with the number of species in a local homogeneous habitat, and is therefore also referred to as intra-habitat diversity. B diversity is mainly the variation along environmental gradients, the similarity of species composition between different habitat communities or the turnover rate of species along environmental gradients and is also It is also known as interhabitat diversity, and this study is one of the current research hotspots[23,24]. Γ diversity describes the diversity at regional or continental scales, and refers to the number of species at regional or continental scales, and is also known as regional diversity, and because it is relatively difficult to study under continental scale regions, current studies are conducted at small scales, so γ diversity studies are relatively rare.

In this study, in order to investigate the relationship between species diversity and tree growth in natural secondary forests in northeast China, we applied α-diversity and β-diversity to analyze the species composition, structural characteristics, and latitudinal gradient, which can provide scientific basic information for natural secondary forest management and management in northeast China. At the same time, by determining a reasonable size of species' diversity and conducting appropriate nurture harvesting, we can improve the growth conditions of the forest through appropriate felling and harvesting or artificial replanting.

2. Overview of the study area

The northeastern region has a temperate continental monsoon climate from south to north.

It has four distinct seasons and a relatively high latitude, with cold and long winters and warm and short summers, and is mainly a humid and semi-humid region. The vegetation types in the three northeastern provinces belong to the Changbai Mountain flora, and the main tree species are *Liriodendron juglans mandshurica, Fraxinus mandshurica, Phellodendron amurense, Quercus mongolica, Tilia mongolica, Tilia mandshurica, Fraxinus rhynchophylla, Ulmus pumila, Ulmus laciniata, Tilia mandshurica, Ulmus rhynchophylla, Ulmus pumila, Ulmus laciniata, Tilia mandshurica. Amurensis, Tilia mandshurica, Fraxinus rhynchophylla, Ulmus pumila, Ulmus laciniata, Carpinus cordata, Acer mono, Acer pseudosieboldi pseudosieboldianum, Syringa reticulata, Prunus padus, Rhamnus davurica, Betula platyphylla, Betula costata, Betula davurica, Populus Populus davidiana, Populus ussuriensis, Albizia kalkora, Maackia* *amurensis, Acer mandshuricum, Acer triflorum, Alnus sibirica, Pinus koraiensis, Pinus sibirica, Pinus sibirica, Pinus sibirica, Pinus koraiensis. Pinus koraiensis, sand pine Abies holophylla, camphor pine Pinus sylvestris, bristlecone pine Abies nephrolepis,* and *larch Larix gmelinii*.

This research studied a comprehensive survey of natural secondary forest tree species in the areas of Xiaoxinganling (I), Zhangguangcailing (II), Laojialing (III), Changbai Mountain (IV), Hadaling (V) and Longgang Mountain (VI) (123°58'13"–130°24'10"E, 40°52'25"–46°48'50"N) from 2017 to 2019 according to the distribution of mountain systems and latitudes (**Figure 1**).

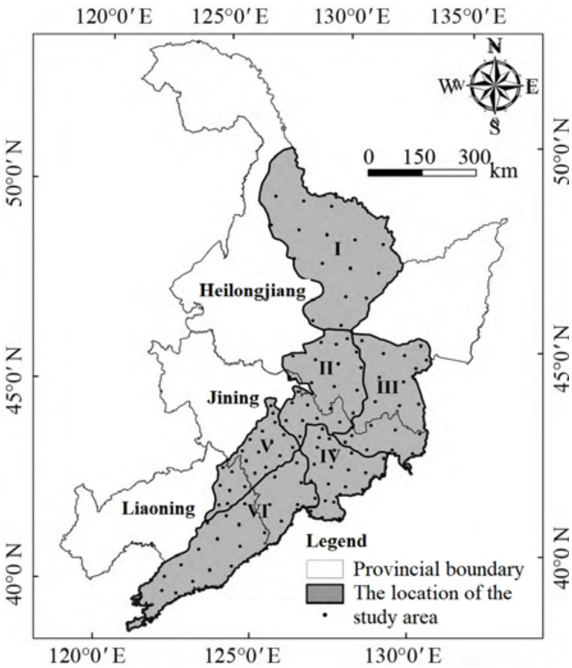


Figure 1. Plot distribution.

3. Experimental methods

3.1 Sample plot setting

In the six areas, sample points were evenly distributed in each area according to the geographic grid, and sample plots were set up by the sample circle method at the sample points already laid out. That is at a distance of more than 20 m from the forest edge, a circular sample plot was established with a point as the center and 17.85 m as the radius without crossing rivers, roads or logging lines. All trees with diameter at breast height of about 5 cm and above within the sample circle were surveyed and registered respectively. The main survey contents included species name, diameter at breast height, tree height, crown width (S–N, W–E), angle and distance from the center of the circle, etc. The longitude, latitude, elevation, slope, slope direction, slope position, soil depth and humus layer depth of each sample plot were also recorded.

3.2 Species diversity measurement

3.2.1 Importance value (IV)

$$IV = (RD_i + RF_i + RP_i)/3 \tag{1}$$

Where, RD_i is the relative density of species *i*, RF_i is the relative frequency of species i, and RP is the relative significance of species *i*.

3.2.2 Species diversity

Species diversity is an important measure of the functional complexity and stability of communities and mainly includes α diversity index, *β* diversity index and γ diversity index[21]. A diversity index (Shannon-Wiener diversity index H', Simpson diversity index D and Pielou evenness index J_{sw}), *β* diversity index (Sφrensen β diversity index βs and Cody diversity index βc) and species richness (Margalef richness index R) were evaluated together.

$$H' = -\sum_{i=1}^{n} P_i \ln P_i \tag{2}$$

$$D = 1 - \sum_{i=1}^{n} P_i^2 \tag{3}$$

$$J_{sw} = \left(-\sum_{i=1}^{n} P_i \ln P_i\right) / \ln S \tag{4}$$

$$\beta s = 1 - 2c/(a+b) \tag{5}$$

$$\beta c = [g(H) + l(H)]/2 \tag{6}$$

$$R = (S-1)/\ln N \tag{7}$$

Where: $P_i = N_i/N$, P_i is the relative importance values of species, N is the sum of the importance values of each species in the sample site where species *i* is located. N_i is the importance value of the i^{th} species. S is the total number of species in the sample site. *a* and *b* are the number of all species in the two communities, *c* is the number of species shared between communities. g(H) is the number of species increasing along the habitat gradient H. l(H) is the number of species lost along the habitat gradient H, i.e., the number of species present in the previous gradient but absent in the next gradient. N is the total number of individuals in the sample plot.

3.3 Data analysis

Experimental data and table creation were organized using Excel 2007 software, while data were analyzed and compared using SPSS 19.0 software and plotted using sigmaplot 12.0 software.

4. Results and analysis

4.1 Species composition and diversity

The species composition of natural secondary forests in different areas was investigated (**Table 2**), and 48 species belonging to 17 families and 31 genera were investigated in all sample sites, including 19 species belonging to 13 families and 17 genera in the Xiaoxinganling area. The importance values were 22.88%, 18.34%, 16.72%, 8.37% and 7.95%, and the total importance values were more than half, which were the dominant species. Thirty species were investigated in the sample site of Zhang Guangcai Ridge, belonging to 13 families and 19 genera, and 13 species with importance values ≥2. The top five species were Maple, Elm, Liriodendron, Ash and Red Pine, with importance values of 29.98%, 11.07%, 10.70%, 4.49% and 4.38%, respectively, which were more than half of the total importance values and were the dominant species. There were 28 species belonging to 14 families and 20 genera, and 14 species with importance value ≥2. Red pine, linden, larch, lupine and elm ranked in the top five with importance value of 23.16%, 15.52%, 7.66%, 6.31% and 6.22% respectively, and the total importance value was more than half, which were the dominant species. The sample site in Changbai Mountain area surveyed 28 species belonging to 15 families and 22 genera, 14 species with species importance value ≥2. Tyrannus sylvestris, Acer sylvestris, Elm, Acacia sylvestris and Linden ranking in the top five with importance values of 22.81%, 11.70%, 8.69%, 8.05% and 6.73% respectively, with a total importance value of more than 50%, which are the dominant tree species. There were 29 species belonging to 15 families and 22 genera in the Haddaling area, and 14 species with importance value ≥2. The top five species were Maple, Elm, Mongolian oak, Quercus alba and Liriodendron, with importance values of 37.16%, 9.56%, 8.10%, 5.12% and 4.28% respectively, and more than half of them were dominant. There are 39 species in the Longgang Mountain sample area, belonging to 15 families and 27 genera, with 13 species with importance value ≥2. The top five species in importance value are Maple, Elm, Mongolian oak, Quercus alba and Liriodendron, with importance value size of 27.04%, 9.21%, 7.73%, 6.46% and 4.39%, respectively, and more than half of the total importance value is dominant. Overall, the dominant species in each region were mainly elm, colored maple and water willow.

The α-diversity (Shannon-Wiener diversity index H', Simpson diversity index D and Pielou evenness index Jsw) and richness (Margalef richness index R) of species in natural secondary forest sample plots in different regions were determined (**Table 3**).

The Shannon-Wiener index (H') was the largest in Changbai Mountain and the smallest in Xiaoxinganling. The difference between them was significant ($P < 0.05$). The Shannon-Wiener index

Table 1. Basic situation of plots

Variables	XXAL	ZGCL	LYL	CBS	HDL	LGS
Number of sample	16	22	20	22	21	22
Number of family	13	13	14	15	15	15
Number of genus	17	19	20	22	22	27
Number of species	19	30	28	28	29	39
Stand density/(plan·hm^{-2})	619	590	551	605	582	610
Latitude/(°)	46°16’15”–46°48’50”	43°2’10”–46°03’03”	42°48’56”–45°59’41”	41°19’36”–43°32’29”	41°57’11”–43°08’01”	41°02’10”–42°56’51”
Altitude/m	308–401	230–495	312–577	437–834	397–723	257–815
Soil depth/cm	15.6–47.0	7.5–56.0	11.5–46.8	5.2–32.0	7.6–31.3	15.6–34.2
Humus thickness/cm	8–16	4–35	5–14	7–23	6–21	8–18

Table 2. Tree species of important values ≥2 in different areas

XXAL		ZGCL		LYL		CBS		HDL		LGS	
Species	Important value /%	Species	Important value /%	Species	Important value /%	Species	Important value /%	Species	Important value /%	Species	Important value /% Species Important value /%
U. Pumila	22.88	*A. Mono*	29.98	P. Koraiensis	23.16	S. Reticulata	22.81	*A. Mono*	37.16	*A. Mono*	27.04
P. Koraiensis	18.34	*U. Pumila*	11.07	Z awwre/w/s	15.52	*A.mono*	11.70	*U.pumila*	9.56	*U.pumila*	9.21
F. Mandshurica	16.72	J. Mandshurica	10.70	L. Gmelinii	7.66	U. Pumila	8.69	Q. Mongolica	8.10	Q. Mongolica	7.73
S. Reticulata	8.37	F. Mandshurica	4.49	S. Reticulata	6.31	A. Kalkora	8.05	F. Mandshurica	5.12	K mandshurica	6.46
J. Mandshurica	7.95	P. Koraiensis	4.38	U. Pumila	6.22	T. Amurensis	6.73	J. Mandshurica	4.28	A. Triflorum	4.39
A. Holophylla	4.04	S. Reticulata	4.14	J. Mandshurica	5.60	J. Mandshurica	5.58	U. Laciniata	3.84	J. Mandshurica	4.35
L. Gmelinii	3.66	P- amurense	3.76	T. Mandshurica	4.65	*F. Mandshurica*	5.51	Lime*t. Amurensis*	3.30	*T. Amurensis*	4.22
P. Padus	3.09	T. Amurensis	3.38	A. Mono	4.55	U. Laciniata	4.27	L. Gmelinii	3.08	F. Rhynchophylla	3.94
P. Amurense	2.95	A. Mandshuricum	3.29	F. Mandshurica	4.21	A. Triflorum	3.21	A. Mandshuricum	2.95	A. Mandshuricum	2.94
A. Mono	2.57	*Q. Mongolica*	2.94	P. Amurense	2.65	P. Amurense	2.90	P. Amurense	2.37	P. Amurense	2.88
T. Amurensis	2.57	U. Laciniata	2.64	P. Padus	2.50	A. Sibirica	2.69	P. Padus	2.33	A. Pseudo-sieboldianum	2.85
R. Davurica	2.15	T. Mandshurica	2.49	A. Mandshuricum	2.26	A. Mandshuricum	2.61	P. Sylvestris	2.03	C. Cordata	2.64
-C. Cordata 2.64	-C. Cor data	C. Cor data	2.27	Fp A. Holophylla	2.12	*Q. Mongolica*	2.09	*F. Rhynchophylla*	2.03	*L. Gmelinii*	2.23
-	-	- -	- -		2.08	P. Koraiensis	2.03	P. Ussuriensis	- -	-P. Ussuriensis	-Total
Total	95.30	- - - - Total 95.30	85.50	-Total	89.46	-	88.90	-	86.18	-	80.89

(H') of species in Laozi Mountain, Longgang Mountain, Zhangguangcai Mountain and Hada Mountain was located between that of Changbai Mountain and Xiaoxinganling, and it was not significantly different from species in Changbai Mountain but was significantly different from species in Xiaoxinganling ($P < 0.05$). Simpson's diversity index (D) was the largest in Changbai-Mountain and the smallest in Xiaoxinganling, and the difference between them was significant ($P < 0.05$), followed by Laoyanling, Longgang Mountain, Zhangguangcai Mountain and Hada Mountain, and the species diversity index in Longgang Mountain and Zhangguangcai Mountain area was in the middle region, and it was not significantly different from other areas. The Pielou evenness index (Jsw) was the largest in Changbai Mountain, followed by Laozi Mountain, Longgang Mountain, Zhangguangcai Mountain and Hada Mountain, and the smallest in Xiaoxingan Mountain, and there was no significant difference between the regions. The Margalef richness index (R) was the largest in Longgang Mountain and the smallest in Xiaoxinganling, and the difference between them was highly significant ($P < 0.05$), and the species diversity was not significantly different between the rest of the regions, but all of them were significantly different from Longgang Mountain and Xiaoxinganling ($P < 0.05$).

Table 4 statistical analyzed the species β-diversity (Sφrensen diversity and Cody diversity) within natural secondary forest sample plots in different regions. It revealed that there were mainly latitudinal differences between regions. Sφrensen diversity indices range from 0.1724 to 0.2681 in the range from high latitude areas in the

Table 3. α diversity of species in different areas

Index	XXAL	ZGCL	LYL	CBS	HDL	LGS
Shannon-Wiener (H')	2.3197 ± 0.08^{b}	2.6413 ± 0.06^{a}	2.6817 ± 0.08^{a}	2.6950 ± 0.07^{a}	2.5026 ± 0.05^{a}	2.6657 ± 0.06^{a}
Simpson (D)	0.7378 ± 0.04^{b}	0.7766 ± 0.03^{ab}	0.8048 ± 0.01^{a}	0.8373 ± 0.03^{a}	0.7432 ± 0.03^{b}	0.7822 ± 0.02^{ab}
Pielou (J_{sw})	0.8457 ± 0.02^{a}	0.8728 ± 0.01^{a}	0.8957 ± 0.03^{a}	0.9021 ± 0.01^{a}	0.8638 ± 0.03^{a}	0.8753 ± 0.03^{a}
Margalef (R)	2.8002 ± 0.06^{c}	3.8289 ± 0.05^{b}	3.9458 ± 0.07^{b}	4.0698 ± 0.04^{b}	4.2750 ± 0.05^{b}	5.1511 ± 0.03^{a}

†Different lowercase letters in the same row indicate significant differences in species diversity in different areas (P<0.05).

Xiaoxinganling to low latitude areas in the Longgang Mountains. Sφrensen diversity indices ranging from 0.1724 to 0.2681 between species in the Xiaoxinganling. On the contrary, the Sφrensen diversity index was the smallest between Laozi Ridge and Zhangguangcai Ridge, and the most species were shared between the two areas. In addition, the Cody diversity index increased with decreasing latitude gradient, and the lower the latitude, the greater the Cody diversity, indicating the higher the community succession rate.

4.2 Species community characteristics

Table 5 analyzed species diameter at breast height and tree height in the natural secondary forest sample plots of different regions. We found that the average diameter at breast height of species in Changbaishan region was the largest with 17.1 cm, followed by Laoyan Ridge, Longgang Mountain Zhangguangcai Ridge and Hada Ridge. The average diameter at breast height in Xiaoxingan Ridge was the smallest with 15.0 cm, and the largest species at breast height in each region was Mongolian oak in Changbaishan region with a diameter at breast height of 64.8 cm. The species with the smallest diameter at breast height was Mongolian oak in the Longgang Mountains with a diameter at breast height of 4.0 cm. In terms of tree height, the largest average tree height was in Changbai Mountain with a mean height of 15.5 m, followed by Laojialing, Longgang Mountain, Zhangguangcai Mountain, and Hada Mountain, and the smallest average tree height was in Xiaoxinganling with 11.1 m. The largest tree height species in each region was Liriodendron hirsutum in Longgang Mountain with a height value of 28.9 m, and the smallest tree height species was yellow pineapple in Longgang Mountain with a height value of 1.3.

Table 6 analyzed the diameter at breast height and height of the top 5 tree species in different regions. It was found that elm was distributed in large

Table 4. β diversity of species in different areas

Area	XXAL	ZGCL	LYL	CBS	HDL	LGS
XXAL	-XXAL	0.1853	0.1936	0.2156	0.2384	0.2681
ZGCL	6.0	-ZGCL	0.1724	0.2414	0.2358	0.2634
LYL	6.0	7.5	-0.2453	0.2453	0.2498	0.2583
CBS	7.0	8.0	7.5	-7.5	0.1963	0.2318
HDL	7.5	8.0	8.5	7.0	-0.2059	0.2059
LGS	7.5	8.5	8.5	7.5	8.0	-

†The upper right corner is the Sφrensenβdiversity index, and the lower left corner is the Codyβdiversity index.

Table 5. DBH and tree height of tree species in different areas

Area	Mean	Min	Max	Mean	Min	Mean Min
XXAL	15.0	4.8	63.6	11.1	1.8	26.8
ZGCL	16.8	4.3	62.2	14.3	2.2	27.8
LYL	17.0	5.0	63.7	15.4	2.6	27.7
CBS	17.1	4.6	64.8	15.5	2.4	27.1
HDL	16.4	5.0	61.4	13.0	2.2	28.1
LGS	16.9	4.0	628	14.8	1.3	28.9

Table 6. DBH and tree height of the top 5 important values trees species in different areas

XXAL			ZGCL			LYL		
Species	Mean DBH/cm	Mean Height/m	Species	Mean DBH/cm	Mean Height/m	Species	Mean DBH/cm	Mean Height/m
U. Pumila	13.28	10.15	A. Mono	15.10	11.09	P. Koraiensis	19.11	12.38
P. Koraiensis	10.73	7.78	U. Pumila	14.18	11.08	T. Amurensis	14.51	10.80
F. Mandshurica	16.76	15.44	J. Mandshurica	12.35	10.78	L. Gmelinii	20.33	17.25
S. Reticulata	6.31	5.59	F. Mandshurica	16.45	14.64	S. Reticulata	7.89	6.41
J. Mandshurica	10.46	9.56	P. Koraiensis	10.81	7.28	U. Pumila	13.29	9.98
	CBS			HDL			LGS	
Species	Mean DBH/cm	Mean Height/m	Species	Mean DBH/cm	Mean Height/m	Species	Mean DBH/cm	Mean Height/m
S. Reticulata	8.93	6.63	A. Mono	11.77	10.65	A. Mono	11.59	9.73
A. Mono	12.41	8.39	U. Pumila	11.89	9.05	U. Pumila	14.24	11.28
U. Pumila	11.79	7.53	Q. Mongolica	25.33	17.85	Q. Mongolica	18.18	13.5
A. Kalkora	12.83	9.14	F. Mandshurica	19.65	18.58	F. Mandshurica	18.90	15.97
T. Amurensis	13.94	10.81	J. Mandshurica	14.89	12.61	A. Triflorum	14.34	12.11

Table 7. Relationship between species diversity (x) and mean DBH (y) in each region

Area	Shannon-Wiener (H')	Simpson (D)	Pielou (Jsw)	Margalef (R)
XXAL	$y=-29.0352+38.7745x$ $-7.9027x^2$ $R^2=0.9753$	$y=-31.1462+68.5300x$ $-42.3019x^2$ $R^2=0.5443$	$y=-54.2821+111.1314x$ $-62.6278x^2$ $R^2=0.8924$	$y=-2.9016+16.0765x$ $-2.0805x^2$ $R^2=0.9705$
ZGCL	$y=-64.589+86.2756x$ $-16.5986x^2$ $R^2=0.9625$	$y=-89.3564+146.8567x$ $-90.6528x^2$ $R^2=0.8523$	$y=-88.6529+158.3649x$ $-88.6498x^2$ $R^2=0.9152$	$y=-78.5631+115.6425x$ $-14.2389x^2$ $R^2=0.8946$
LYL	$y=25.8964+59.8748x$ $-10.6919x^2$ $R^2=0.8735$	$y=116.3528+258.6492x-$ $159.6658x^2$ $R^2=0.8527$	$y=96.5849+186.5467x$ $-104.8743x^2$ $R^2=0.9028$	$y=6.3916+18.6942x$ $-2.5951x^2$ $R^2=0.7437$
CBS	$y=108.5679+235.2854x$ $-40.5664x^2$ $R^2=0.9134$	$y=98.6431+198.6428x$ $-122.6856x^2$ $R^2=0.7394$	$y=48.6792+123.8216x$ $-65.8127x^2$ $R^2=0.8582$	$y=35.4689+72.5691x$ $-8.8627x^2$ $R^2=0.9226$
HDL	$y=56.4628-105.6482x$ $+27.8022x^2$ $R^2=0.8546$	$y=62.8564+98.4567x-$ $60.7759x^2$ $R^2=0.7945$	$y=59.6243+110.6943x$ $-62.1886x^2$ $R^2=0.9213$	$y=28.6913+68.9643x$ $-8.4106x^2$ $R^2=0.8533$
LGS	$y=82.3492+162.5492x$ $-30.1017x^2$ $R^2=0.8992$	$y=105.8694+186.5943x-$ $115.4698x^2$ $R^2=0.9053$	$y=137.4961+187.3694x$ $-105.2679x^2$ $R^2=0.9017$	$y=92.4682+173.3346x$ $-21.1382x^2$ $R^2=0.8941$

numbers in all regions and was the main tree species, and the average diameter at breast height and height of elm, color wood maple, water willow and hoodia were basically the same in all regions with no significant differences. While red pine showed some differences in each region, with larger values of diameter at breast height and height of red pine in Laozi Ridge, and smaller values in Xiaoxinganling and the average diameter at breast height and the average height of red pine in each region are small, and the red pine is in the stage of natural regeneration.

4.3 Relationship between species diversity and diameter at breast height and tree height

4.3.1 Relationship between species diversity and diameter at breast height and tree height in a single area

Tables 7 and 8 compared the species diversity with mean diameter at breast height and mean height of tree species in the sample plots of individual areas. It showed that the relationship between α diversity and richness of each species and mean diameter at breast height of tree species in the remaining areas showed a single-peaked curve except for the Shannon-Wiener diversity index in the Haddaling area, i.e., with increasing α diversity and richness, the mean diameter at breast height of tree species in the sample plots. In addition, the R^2 values of α diversity and richness were above 0.70 in all regions except for the smaller R^2 value of Simpson's diversity index in the Xiaoxinganling region, which was a good fit. In general, the alpha diversity of species within the sample plots in each area had a positive effect on the growth of tree species at breast height within the stand.

Table 8 showed that the relationship between α diversity and richness of species and the average tree height of tree species in all regions showed a single-peaked curve except for the Shannon-Wiener diversity index in Xiaoxinganling and Hadaling. The average height of tree species in the sample site showed a trend of first increasing and then decreasing with the increasing α diversity and richness, and there was a maximum value at a specific value. The R^2 values of α diversity and richness in each area were above 0.69, which was a good fit.

4.3.2 Relationship between species diversity and diameter at breast height and tree height in all areas

Figure 2 and **3** found the trends of species diversity in relation to the mean diameter at breast height and mean tree height of tree species within the sample plots in all areas. The mean diameter at breast height of tree species within the sample

Table 8. Relationship between species diversity (x) and average tree height (y) in each region

Area	Shannon-Wiener (*H'*)	Simpson (*D*)	Pielou (*Jsw*)	Margalef (*R*)
XXAL	$y = 48.6972 - 91.2576x$ $+17.2564x^2$ $R^2 = 0.8943$	$y = 111.6943 + 289.6143x$ $-176.5946x^2$ $R^2 = 0.9191$	$y = 195.2618 + 364.8954x$ $-200.5461x^2$ $R^2 = 0.8463$	$y = 26.5916 + 68.5317x$ $-8.1564x^2$ $R^2 = 0.8229$
ZGCL	$y = -69.3162 + 123.1218x$ $-23.8128x^2$ $R^2 = 0.9152$	$y = -88.9134 + 176.8259x$ $-107.8216x^2$ $R^2 = 0.8391$	$y = -134.2256 + 302.2561x$ $-166.0745x^2$ $R^2 = 0.7539$	$y = -55.9127 + 87.3189x$ $-10.3954x^2$ $R^2 = 0.6939$
LYL	$y = 105.2141 + 203.2581x$ $-38.5529x^2$ $R^2 = 0.9088$	$y = 216.5943 + 346.1678x$ $-211.0825x^2$ $R^2 = 0.8716$	$y = 456.3258 + 756.3215x$ $-415.2364x^2$ $R^2 = 0.9341$	$y = 62.8916 + 99.3387x$ $-11.8207x^2$ $R^2 = 0.7988$
CBS	$y = 168.8829 + 259.3349x$ $-48.5961x^2$ $R^2 = 0.8164$	$y = 208.9437 + 425.6179x$ $-259.2456x^2$ $R^2 = 0.7966$	$y = 391.2546 + 660.2518x$ $-362.7756x^2$ $R^2 = 0.9022$	$y = 69.8520 + 125.3394x$ $-14.9213x^2$ $R^2 = 0.9006$
HDL	$y = -125.9438 - 208.5691x$ $+39.6489x^2$ $R^2 = 0.9561$	$y = -188.2679 + 312.1649x$ $-189.4628x^2$ $R^2 = 0.9316$	$y = -221.3566 + 450.4169x$ $-247.9648x^2$ $R^2 = 0.8989$	$y = -81.2953 + 144.7749x$ $-17.2569x^2$ $R^2 = 0.8564$
LGS	$y = 99.6482 + 196.5219x$ $-37.1259x^2$ $R^2 = 08886$	$y = 113.3125 + 166.3314x -$ $101.4215x^2$ $R^2 = 0.8006$	$y = -334.8567 + 524.4893x$ $-288.1802x^2$ $R^2 = 0.7781$	$y = 72.1438 + 119.6874x$ $-14.2485x^2$ $R^2 = 0.8317$

plots all showed an overall trend of increasing and then decreasing with increasing species diversity and species richness. It is that the mean diameter at breast height of tree species within the sample plots showed a single-peaked curve in relation to α diversity index and species richness. It is that the average diameter at breast height of tree species in the stand increased and then decreased with the increase of α diversity index and species richness within a certain range. The average diameter at breast height of tree species showed a quadratic relationship with α-diversity index and species richness with R^2 of 0.9894, 0.7808, 0.9973 and 0.8454 respectively, which were well fitted. The average diameter at breast height of tree species reached the maximum value at 2.7985, 0.8137, 0.8917 and 4.5587 for the Shannon-Wiener diversity index, Simpson diversity index, Pielou evenness index and Margalef richness index, respectively, through the fitting calculation. Overall, the species diversity has a positive effect on the diameter at breast height of tree species in the stand within a certain range.

As shown in **Figure 3**, the mean tree height of tree species increased with the increase of species diversity and species richness, and then decreased, i.e., the mean tree height of tree species in the sample area showed a single-peaked curve relationship with α diversity index and species richness. The R^2 was 0.9867, 0.9261, 0.9770 and 0.646, respectively, which were all good fits. From the fitting calculations, it can be seen that, except for the Shannon-Wiener diversity index, the Simpson diversity index, the Pielou evenness index and the Margalef richness index can reach the maximum value for the average tree height of tree species at 0.8206, 0.9023 and 4.5830, respectively. As a whole, within a certain range, species diversity has a contributing effect on tree height of tree species within the stand, and the Shannon-Wiener diversity index has the highest degree of influence on it.

5. Conclusion and discussion

Species composition and population structure affect the growth of trees to some extent[25]. In this study, through the survey of species composition in natural secondary forest sample plots in different regions, it was found that 48 species belonging to 17 families and 31 genera investigated in all sample plots. Among them, Longgang Mountain contained the largest number of families, genera, and species, followed by Hada Ling, Changbai Mountain, Laozi Ling, Zhang Guangcai Ling, and Xiao Xing'an Ling, and the overall number of species decreased with latitude. The overall number of species decreases with the increasing latitude, which is similar to the results of studies on American forest communities[26] and subtropical plant communities[27]. Overall, elm, stained maple, and buffalo willow were the dominant species in each region, and studies have shown that the number of dominant species within a stand affects the tree growth within a stand to some extent[28], and the differences in dominant species among regions in this study were small and therefore neglected.

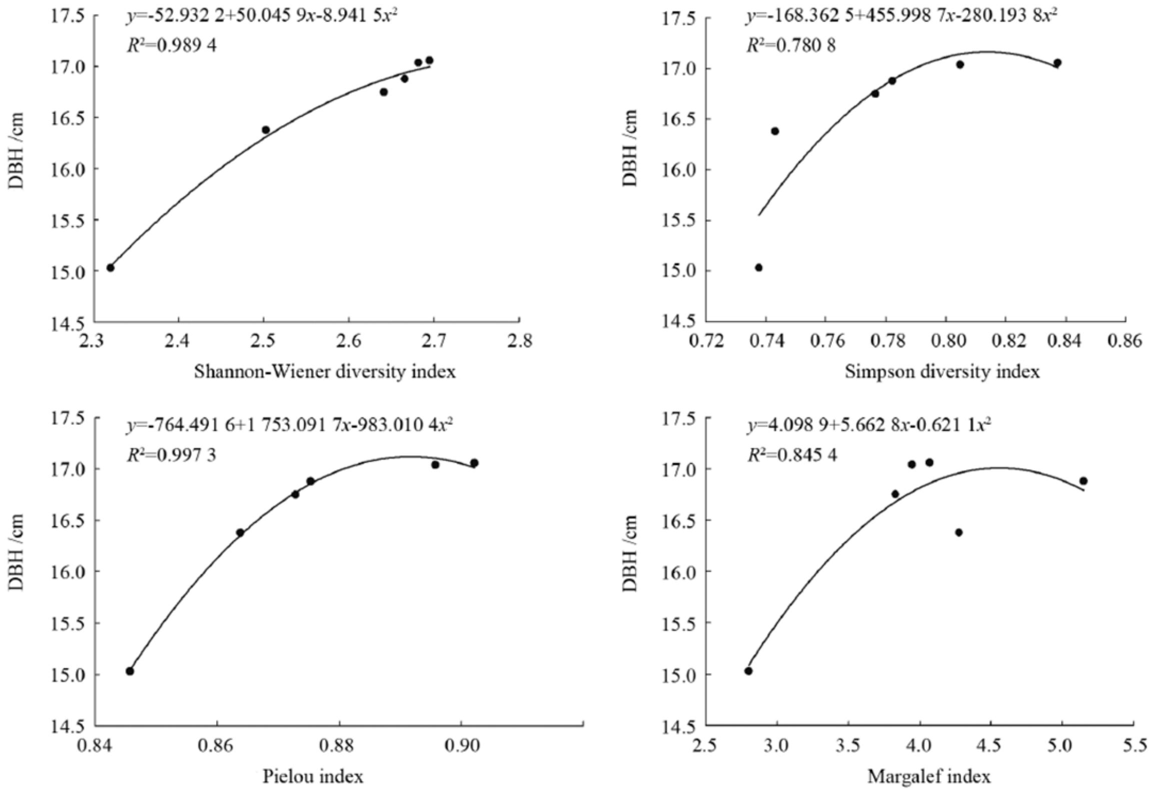


Figure 2. Relationship between biodiversity and DBH in different areas.

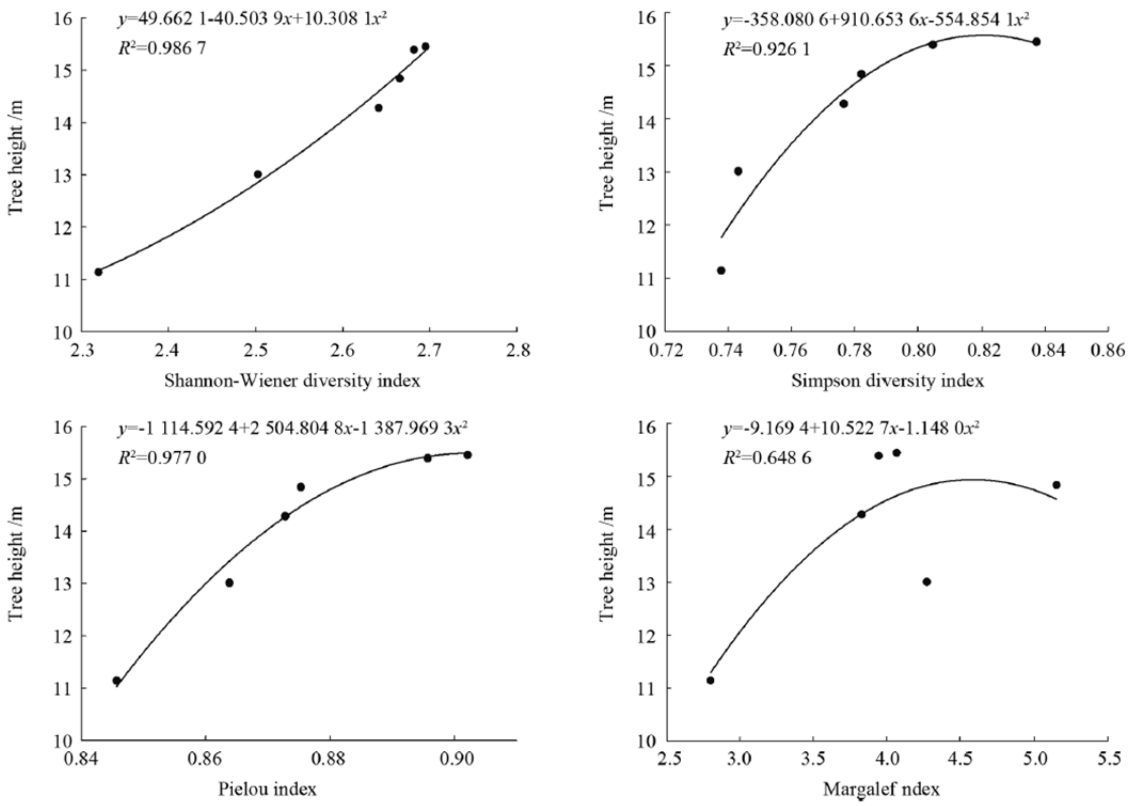


Figure 3. Relationship between biodiversity and tree height in different areas.

In this study, α-diversity index and β-diversity index[21] were used for analysis aiming to understand the community species composition and tree growth between local homogeneous habitats and different habitats in the forest stand. It was found that the species α-diversity in natural secondary forest samples in different regions was the largest in Changbai Mountain and the smallest in Xiaoxinganling. While the richness was the largest in Longgang Mountain and the smallest in Xiaoxinganling. A result indicates that species richness gradually decreases with increasing latitude, and the higher the latitude, the smaller the number of species. The diversity index is influenced by latitude, as well as species composition, community structure, dominant tree species and environmental conditions[25]. The Sφrensen diversity index ranged from 0.1724 to 0.2681. The largest Sφrensen diversity index was found between the Xiaoxinganling and Longgang Mountain areas, and it indicated that the two areas had the fewest species in common, while the smallest Sφrensen diversity index was found between the Laojialing and Zhangguangcai Ridge areas. On the contrary, the Sφrensen diversity index between Laozi Ridge and Zhangguangcai Ridge was the smallest, and the number of species shared between them was the largest. Therefore, it can be found that the greater the difference in latitude between regions, the more obvious the difference in diversity indices between them. In addition, the Cody diversity index increased with decreasing latitudinal gradient, and the lower the latitude, the greater the Cody diversity, and it indicated a higher rate of community succession[29].

By analyzing the diameter at breast height and tree height of tree species in natural secondary forest sample plots in different regions, we found that the average diameter at breast height and average tree height in each region shown CBS > LYL > LGS > ZGCL > HDL > XXL. The largest tree species at diameter at breast height in each region was Mongolian oak in Changbaishan region, and the smallest diameter at breast height was Tyrannus sylvestris in Longgang Mountain. The species with the largest height in each region was Liriodendron spp, the smallest tree height is yellow pineapple in Longgang Mountain area. This indicates that the species grows better in Changbai Mountain area, and the other areas are relatively inferior, and the larger the latitude, the smaller the average diameter at breast height and the average height of trees.

In this study, the relationship between species diversity and mean diameter at breast height and mean height of tree species was found to increase and then decrease with species diversity and species richness in both cases. It is that the mean diameter at breast height and mean height of tree species in the sample area showed a single-peaked curve relationship with α-diversity index and species richness, and the fitting effect was good in both cases. The mean diameter at breast height and the mean height of tree species in the sample site showed a single-peaked curve relationship with the α diversity index and species richness, and the fit was good with large R^2 values. Therefore, it can be seen that within a certain range, species diversity has a certain promotion effect on the diameter at breast height and height of tree species in a stand, but it should not be too high. This is consistent with the results of Zhang *et al.*[30] and Symstad *et al.*[31] regarding the relationship between species diversity and tree diameter at breast height, tree height and biomass. But there are also different findings, for example, Montserrat *et al.*[32] and Thompson *et al.*[33] found a negative relationship between species diversity and tree growth, while Kahmen *et al.*[25] and Grace *et al.*[34] showed no relationship between the two. The main reason for the inconsistent results is that the current studies focus only on the relationship between species diversity and tree growth, thus ignore the possible effects of regional environmental conditions, climatic conditions, species composition, soil fertility, and intra- or interspecific relationships.

In summary, the species survey and analysis of the natural secondary forest sample plots in different regions revealed that all the sample plots contained 48 species belonging to 17 families and 31 genera. Among them, Longgang Mountain contained the largest number of families, genera and species, followed by Hada Ling, Changbai Mountain, Laozi Mountain, Zhangguangcai Mountain and Xiaoxing'an Mountain; the α-diversity index of species in the sample plots was the largest in Changbai Mountain and the smallest in Xiaoxing'an Mountain. The greater the difference in tude between regions, the more obvious the difference in species β-diversity index and the higher the rate of community succession. The ship between mean diameter at breast height and mean tree height in each region is CBS > LYL > LGS > ZGCL > HDL > XXL; within a certain range, species diversity has a significant effect on the mean diameter at breast height and mean tree height of species in a stand. Within a certain range, species diversity has a positive effect on the mean diameter at breast height and mean height of species within a stand. The present study explored the relationship between species diversity and tree growth in a stand under large-scale conditions based on previous studies, which is representative in terms of geographical space.

Conflict of interest

The authors declare that they have no conflict of interest.

References

1. Zhang X, Zhang H, Lu J, *et al*. Early effects of crop tree release tending on growth of natural secondary Quercus mongolica forest. Scientia Silvae Sinicae 2020; 56(10): 83–92.
2. Zhu J, Li X, Liu Z, *et al*. Factors affecting the snow and wind induced damage of a montane secondary forest in northeastern China. Silva Fennica 2006; 40(1): 37–51.
3. Bonino E, Araujo P. Structural differences between a primary and a secondary forest in the Argentine Dry Chaco and management implications. Forest Ecology and Management 2005; 206(1): 407–412.
4. Ma S, Wang Q, Cui D, *et al*. Effect of thinning on stand structure and plant species diversity in natural Fraxinus mandschurica forest stands. Journal of Northeast Forestry University 2019; 47(2): 1–7.
5. Wang Y, Li S, Miao N, *et al*. Analysis of plant diversity of five typical forests in mountain areas around Sichuan basin. Journal of Central South University of Forestry & Technology 2020; 40(7): 89–98.
6. Liang S, Peng S, Chen Y. Responses of productivity of typical natural secondary forests and plantations to climate change in Shaanxi province, China. Chinese Journal of Applied Ecology 2019; 30(9): 2892–2902.
7. Whittaker RJ. Meta-analyses and mega-mistakes: Calling time on meta-analysis of the species richness-productivity relationship. Ecology 2010; 91(9): 2522–2533.
8. Gillman LN, Wright SD. The influence of productivity on the species richness of plants: A critical assessmentc. Ecology 2006; 87(5): 1234–1243.
9. Cusens J, Wright SD, Mcbride PD, *et al*. What is the form of the productivity-animal-species-richness relationship? A critical review and meta-analysis. Ecology 2012; 93(10): 2241–2252.
10. Liu X, Shi Z, Yang D, *et al*. Advances in study on changes of biodiversity and productivity along elevational gradient in mountainous plant communities. World Forestry Research 2005; 18(4): 27–34.
11. Zhang L, Qi J, Li T, *et al*. Effects of stand density on understory plant diversity and biomass in a Pinus massoniana plantation in Wenfeng mountain, Xinjin county. Acta Ecologica Sinica 2019; 39(15): 5709–5717.
12. Fridley JD, Grime JP, Huston MA, *et al*. Comment on "Productivity is a poor predictor of plant species richness". Science 2012; 335(6075): 1441.
13. Adler PB, Seabloom EW, Borer ET, *et al*. Productivity is a poor predictor of plant species richness. Science 2011; 333(6050): 1750–1753.
14. Jin H, Zhao Y, Zhao W, *et al*. Elevational changes in species diversity and similarity analysis of a Rhododendron chrysanthum community on Changbai mountain. Acta Ecologica Sinica 2015; 35(1): 125–133.
15. Liu Z, Li Q, Chen D, *et al*. Patterns of plant species diversity along an altitudinal gradient and its effect on above-ground biomass in alpine meadows in Qinghai-Tibet plateau. Biodiversity Science 2015; 23(4): 451–462.
16. Kong B, Wei X, Du J, *et al*. Effects of clipping and fertilization on the temporal dynamics of species diversity and functional diversity and their relationships in an alpine meadow. Chinese Journal of Plant Ecology 2016; 40(3): 187–199.
17. Hao J, Li Y, Qi J, *et al*. Effects of anthropogenic disturbances on the species diversity and niche of the dominant populations in a Castanopsis fargesii secondary forest community in Bifengxia, Sichuan. Acta Ecologica Sinica 2016; 36(23): 7678–7688.
18. Wu A, Deng X, Ren X, *et al*. Biogeographic patterns and influencing factors of the species diversity of tree layer community in typical forest ecosystems in China. Acta Ecologica Sinica 2018; 38(21): 7727–7738.
19. Zhang Y, Chen H. Individual size inequality links forest diversity and above-ground biomass. Journal of Ecology 2015; 103(5): 1245–1252.
20. Nadrowski K, Wirth C, Scherer-lorenzen M. Is forest diversity driving ecosystem function and service? Current Opinion in Environmental Sustainability 2010; 2(1): 75–79.
21. Ma KP. Measuring method of biocommunity diversityI: measuring method ofαdiversity (Part 1). Chinese Biodiversity 199; (3): 162–168.
22. Qu Y, Li Q, Duan R, *et al*. Effects of grazing methods on community characteristics and alpha diver-

sity of Stipa breviflora in warm-temperate steppe. Chinese Journal of Grassland 2019; 41(4): 87–93.
23. Weng C, Zhang T, Wu D, *et al*. Drivers and patterns of α- and β-diversity in ten main forest community types in Gutianshan, eastern China. Biodiversity Science 2019; 27(1): 33–41.
24. Zheng Y, Ji N, Wu B, *et al*. Climatic factors have unexpectedly strong impacts on soil bacterial β-diversity in 12 forest ecosystems. Soil Biology and Biochemistry 2020; 142: 1–9.
25. Kahmen A, Jorg P, Audorff V, *et al*. Effects of plant diversity, community composition and environmental parameters on productivity in montane European grasslands. Oecologia 2005; 142(4): 606–615.
26. Li L, Wei S, Lian J, *et al*. Distributional regularity of species diversity in plant communities at different latitudes in subtropics. Acta Ecologica Sinica 2020; 40(4): 1–9.
27. Chen S, Ouyang Z, Zheng H, *et al*. Latitudinal gradient in beta diversity of forest communities in America. Acta Ecologica Sinica 2011; 31(5): 1334–1340.
28. Liu X, Yang F, Zhang M, *et al*. Plant diversity and productivity of grassland communities at different altitudes of Jiucaiping mountain, Guizhou. Chinese Journal of Applied & Environmental Biology 2018; 24(2): 207–213.
29. Sklenar P, Balslev H. Superpáramo plant species diversity and phytogeography in Ecuador. Flora 2004; 200(5): 416–433.
30. Zhang Y, Chen HYH, Reich PB. Forest productivity increases with evenness, species richness and trait variation: A global meta- analysis. Journal of Ecology 2012; 100(3): 742–749.
31. Symstad AJ, Stuart CF, Wall DH, *et al*. Long-term and large-scale perspectives on the ship between biodiversity and ecosystem functioning. Bioscience 2003; 53(1): 89–98.
32. Vila M, Carrillo-Gavilan A, Vayreda J, *et al*. Disentangling biodiversity and climatic determinants of wood production. Plos ONE 2013; 8(2): 1–9.
33. Thompson K, Askew AP, Grime JP, *et al*. Biodiversity, ecosystem function and plant traits in mature and immature plant communities. Functional Ecology 2005; 19(2): 355–358.
34. Grace JB, Anderson TM, Smith MD, *et al*. Does species diversity limit productivity in natural grassland communities? Ecology Letters 2007; 10(8): 680–689.

Sustainable Forestry (2021) Volume 4 Issue 1
doi:10.24294/sf.v4i1.1605

CASE REPORT

Comparison of community structures of *Phoebe chekiangensis* pure forests and heterogeneous mixed forests

Liangyan Wang[1], Zhen Ying[1]*, Qi Zhao[2], Chunfeng Yan[2], Fujie Zhang[2]

[1] *Zhejiang Tiantong National Forest Park, Ningbo 315114, China. E-mail: yingzhen0908@foxmail.com*

[2] *Ningbo Yinzhou District Forestry Technology Management Service Station, Ningbo 315114, China.*

ABSTRACT

Taking the 13 years pure artificial forest *Phoebe chekiangensis* and heterogeneous mixed forests in Tiantong mountain, Zhejiang Province as the research object, the characteristics of stand development, tree competition differentiation, tree height/breast diameter ratio and dominant wood growth were compared and analyzed from the perspective of ecology. The results show that compared with pure forests, the growth advantages of heterogeneous mixed-age forests were significant. Average breast diameter growth of stand increased 1.8%; the growth of single plant wood accumulation increased 7.4%. The relationship between tree height and diameter showed that the high growth of *Phoebe chekiangensis* individuals in the heterogeneous mixed forest was significantly promoted, and the high growth of the tree was 8.4% higher than that of pure forest. 1–5 grade wood scale sizes *Phoebe chekiangensis* in heterogeneous mixed forests and pure forests are ranked grade 3 (43.7%) > grade 2 (26.5%) > grade 4 (15.7%) > grade 1 (12.9%) > grade 5 (1.2%); grade 3 (34.7%) > level 2 (25.6%) > level 4 (20.0%) > level 1 (18.2%) > level 5 (1.2%); the straight-diameter structure shows a normal distribution, and the degree of differentiation of pure forests is greater than that of heterogeneous forests. The dominant trees of *Phoebe chekiangensis* pure forest and heterogeneous forest accounted for 18.2% and 12.9% of the total number of plants respectively, providing a reserve of 51.1% and 35.4% respectively, reflecting the contribution of dominant trees caused by the self-thinning effect.

Keywords: *Phoebe Chekiangensis*; Artificial Pure Forest; Heterogeneous Mixed Forests; Stand Structure

ARTICLE INFO

Received: 12 April 2021
Accepted: 29 May 2021
Available online: 7 June 2021

1. Introduction

The structure is the basis of function, and the diameter structure, tree species composition structure and spatial structure of the stand determine the function of the stand. Although the structure of forest stands has been studied for nearly a hundred years, in different periods, it has changed with the purpose of forest management, and has gradually developed from the past benefits of timber production and economy to the goal of attaching equal importance to multiple benefits. In order to overcome the problems of single structure, inefficient function and low productivity of plantation forests, German forester Gayer put forward the management theory of "near-naturalization of planted forests", advocated the use of indigenous tree species, and advocated that the establishment, cultivation and logging methods of forest stands should be as close as possible to "potential natural vegetation", and its business objectives were: mixed forests—heterogeneous forests—multi-layer forests. The core of the cultivation process is the "single tree target tree" management system, which is an effective method for the management and cultivation of plantations. *Phoebe chekiangensis* is a precious timber tree species endemic to China, with a small natural distribution area limited to Zhejiang, Fujian, Jiangxi and southern Anhui. Due to its good material, medium growth rate, stress resistance, and strong germination ability[1-7], it was introduced to the Yinzhou Tiantong Forest Farm after 2002, becoming one of the most important afforestation tree species recovered after the cleaning of the damaged Masson pine forest. This paper compares and analyzes the productivity, diameter structure and growth characteristics of pure and mixed forests from the perspective of ecology, discusses the structural differences between the two, and provides a reference for the cultivation of plantations and sustainable forest management of precious timber species.

2. Overview of the test area

The test site is located in Tiantong Forest Farm (29°48'N, 121°47'E) in Yinzhou District, Ningbo City, which belongs to the Subtropical humid monsoon climate. The average annual temperature is 16.2 °C; the annual accumulated temperature ≥10 °C is 5166.2 °C, the temperature index of Kira is 135 °C; the frost-free period is 237.8 d; the average annual precipitation is 1,375 mm, the evaporation is 1,320.1 mm, the relative humidity is 82%, the annual sunshine time is 2,057.9 h, and the sunshine rate is 47%. The natural vegetation of the survey area is the subtropical evergreen broad-leaved forest, and the dominant tree species in the hillside standing type are *Schima superba*, *Castanopsis fargesii* and *Castanopsis carlesii*) *et al.*; the dominant species in the gully stand type are *Choerospondias axillaris*, *Liquidambar formosana*, and *Machilus leptophylla*, *Machilus thunbergii*, etc., the composition of tree species is complex and the degree of diversity is high.

3. Research methods

The 13-year-old *Phoebe chekiangensis* pure forest and the heterogeneous mixed forest (I Acid II Phoebe, i.e. the main forest layer is *Choerospondias axillaris* and *Liquidambar formosana*, and the renewal layer is *Phoebe chekiangensis*) was selected as the research object. *Phoebe chekiangensis* pure forest was originally a citrus reticulata forest, which was harvested in the winter of 2002 and reclaimed with a specification of length 0.5 m × width 0.5 m × depth 0.35 m, 2 years old grade I and II seedlings (playing small mud balls) were used in March 2003 for afforestation; the initial planting density is 3,500/hm^{-2}, the heterogeneous mixed forest was originally a mixed forest of *Pinus massoniana* and *Choerospondias axillaris*, and all the pine trees died due to the damage of pine wood nematode disease. After the sick wood is cleared, the near-mature mother trees of *Choerospondias axillaris* and *Liquidambar formosana* on the forest land are retained. In the spring of 2003, the land was prepared and afforested with the above methods, and the initial planting density was 3,150/hm^{-2}. The locations of the two test sites are similar, the standing conditions are all valleys, and the terrain is relatively consistent. The status of the site is shown in **Table 1**.

Table 1. Growth of stand in the sample plot

Project	Sample plot	
	P_1	P_2
Stand type	Pure forest	Mixed forest of different ages
Area/hm^2	0.433	0.2
Age/a	13	40/13*
Altitude/m	110	120
Slope position	Low	Low
Slope direction	SE	SW
Gradient	10–15	10–16
Mother rock	Igneous rock	Igneous rock
Soil type	Red earth	Red earth
Soil thickness	90–120	90–120

The plots is set as 0.433/hm^{-2} and 0.2 hm^{-2} in the pure stand heterogeneous forest respectively. Each tree was surveyed in December 2013 to investigate plant names, quantities, breast diameter, crown and its renewal and health condition, and to calculate individual tree volume and stand accumulation volume. The single plant volume is calculated with the formula of Zhejiang Forestry Survey and Design Institute: $V = G\ (H + 3)f$, where V is the accumulation of a single plant, G is the area of the DBH, H is the height of the tree, and f is calculated by the figured number of 0.38. Accumulation = average wood volume per plant × number of stand-reserved plants.

Stand grading method uses each wood gauge data, the formula $r = d/D$ is adopted for the calculation, where: d is the DBH of the forest (cm) and D is the average breast diameter of the forest stand (cm). It is divided into 5 levels based on the size of the r value[5], grade 1 wood: $r \geq 1.336$; grade 2 wood: $1.026 \leq r < 1.336$; grade 3 wood: $0.712 \leq r < 1.026$; grade 4 wood: $0.383 \leq r < 0.712$, grade 5 wood: $r < 0.383$.

Table 2. The growth comparison between pure forest and mixed forest of different ages

Stand type	**Tree species**	**Age/a**	**Average ABH/cm**	**Height /m**	**Density**		**Breast height sectional area**		**Volume**	
					Quantity/(a tree/hm^2)	**Proportion /%**	**Quantity /(m^2/hm^2)**	**Proportion /%**	**Quantity /(m^3/hm^2)**	**Proportion /%**
Pure forest	*Phoebe chekiangensis*	13	5. 7 ± 2. 04	3.80	3,460	100.0	8.83	100.0	22.87	100.0
mixed forest of different ages	*Choerospondias axillaris*	45	26	13.50	125	3.8	6.64	33.6	41.61	44.4
	Liquidambar formosana	45	25	13.00	100	3.0	4.91	24.9	29.85	31.9
	Phoebe chekiangensis	13	5. 8 ± 1. 74	4.12	3,100	93.2	8.19	41.5	22.16	23.7
	Total				3,325	100.0	19.74	100.0	93.62	100.0

4. Results and analysis

4.1 Comparison of the growth of Different Stands in *Phoebe chekiangensis*

From **Table 1** and **Table 2**, it can be seen that the standing conditions of *Phoebe chekiangensis*'s heterogeneous mixed forest and pure forest plot are about the same, but the growth of *Phoebe chekiangensis* in heterogeneous forest is significant. The average annual growth rates of *Phoebe chekiangensis*'s DBH and tree height in the heterogeneous forest were 0.45 cm and 0.32 m respectively, which were higher than 1.8% and 8.4% in pure forests respectively, and the high growth pole of *Phoebe chekiangensis* in the heterogeneous forest is significantly higher than pure forest ($P > 0.01$). The breast height sectional area of *Phoebe chekiangensis* in the heterogeneous forest is 8.19 m^2/hm^{-2}, which is slightly smaller than pure forest, mainly due to density. Judging from the amount of stand stock, the heterogeneous forest is as high as 93.62 m^3/hm^{-2}, which is 3 times higher than that of pure forests, of which *Phoebe chekiangensis* is 22.16 m^3/hm^{-2}, and the growth effect of *Phoebe chekiangensis* in heterogeneous forests is remarkable.

4.2 Differences in the height/DBH ratio of *Phoebe chekiangensis* in different stands.

The correlation of growth relationships that are prevalent between individuals in the forest indicates that different organs in the individual or the same organ with different directions increase in a certain geometric way. Changes in environmental conditions can make this geometric growth change to accommodate the changed growing environment[8]. The frequency distribution of the tree height/breast diameter ratio of individuals in different stands is shown in **Figure 1**. In pure forests, the tree height/breast diameter ratio of *Phoebe chekiangensis* is distributed at 0.35–1.25, presenting a normal distribution, where the majority have the ratio of 0.5–0.95, accounting for 84.4%. The tree height/breast diameter ratio of *Phoebe chekiangensis* in the heterogeneous forest is also distributed at 0.35–1.25, where majority have the ratio of 0.5–1.1, accounting for 89.5%. The tree height/breast diameter ratio of *Phoebe chekiangensis* in heterogeneous forests that greater than 0.8 accounts for 31.1%, while pure forests account for only 20.7%, and the difference between them was significant through the analysis of variance ($P < 0.01$). It can be seen that the heterogeneous mixed forest has a significant promoting effect on *Phoebe chekiangensis* in the growth of the trunk in the vertical direction (tree height), which is conducive to the robust growth of young trees of nanmu, and the development of individual trees are well-balanced, which alleviates intraspecific differentiation.

4.3 Differences in the structure of the diameter class in *Phoebe chekiangensis* of different stands

The diameter structure can correctly reflect the competitive relationship between stand growth and forest trees. With the growth of forest trees, the forest stands gradually closed, and the competition within and between tree species became fierce, and the forest began to differentiate to form hierarchical trees. The competitive state of stand is related to its distribution of diameter class and number of plants[9]. The distribution of diameter class reflects the existence and degree of differentiation. The wider the distribution range, the greater the possibility of the existence of the bipolar level, the higher the probability of differentiation; the distribution of trees of different diameter levels reflects the intensity of differentiation, and the greater the number of plants at the polar level, the greater the intensity of differentiation. As can be seen from **Figure 2**, the proportion of grade 1–5 wood in heterogeneous mixed forests, from largest to smallest, is grade 3 (43.7%) > grade 2 (26.5%) > grade 4 (15.7%) > grade 1 (12.9%) > grade 5 (1.2%), and the proportion of grade 1–5 wood in the pure forest from largest to smallest were grade 3 (34.7%) > grade 2 (25.6%) > grade 4 (20.0%) > grade 1 (18.2%) > grade 5 (1.2%). The distribution trend of the two vertical wood diameter grades is about the same, but the proportion is significantly different, and the proportions of grade 1 and grade 5 of heterogeneous mixed forests are significantly lower than those of pure forests; the proportion of grade 2–4 is about 85.9%, 5.6% higher than that of pure forest, indicating that the heterogeneous mixed forest is relatively balanced. At the same time, the proportion of grade 1 and grade 5 trees in heterogeneous mixed forests is 14.1%, which is 5.6% less than that of pure forests, indicating that the forest differentiation degree of pure forests is stronger, and the competition of trees is fierce, resulting in natural sparseness earlier than that of heterogeneous forests.

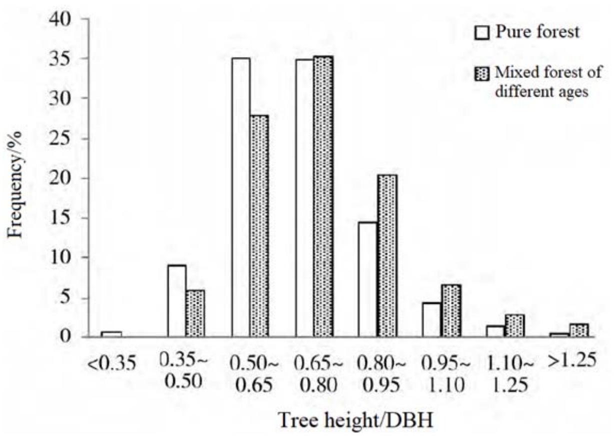


Figure 1. Frequency distribution of individual tree height/breast diameter ratio and chest diameter grade in different forest stands.

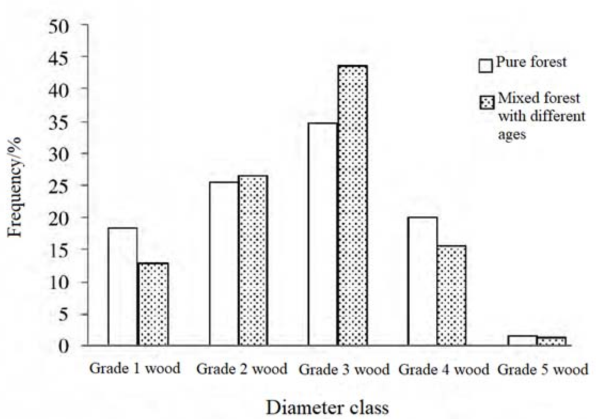


Figure 2. Frequency distribution of tree grading in different forest stands of *Phoebe chekiangensis*.

From **Figure 3**, it can be seen that the diameter structure distribution of *Phoebe chekiangensis* pure forest and heterogeneous forest is monomodal, and the diameter level distribution is continuous, indicating the lack of seedlings and young trees in the plantation.

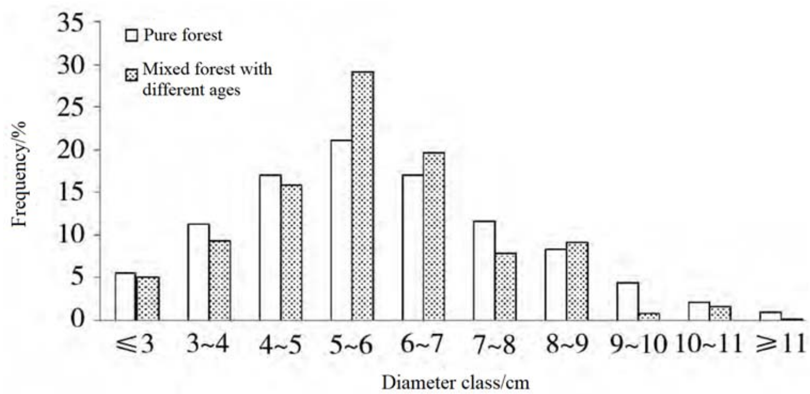


Figure 3. Diameter class distribution of the *Phoebe chekiangensis* of different stands.

4.4 The growth of dominant wood of *Phoebe chekiangensis* in different stands

Whether a large number of advantageous target wood can be cultivated and evenly distributed is a prerequisite for obtaining forest stand with high quality and high yield, and it is also a prerequisite for the cultivation of large-diameter timber. From **Table 3**, it can be seen that the numbers of dominant trees of *Phoebe chekiangensis* plantation pure forest and heterogeneous mixed forest are 630 and 401 respectively, accounting for 18.2% and 12.9% of the total number of plants, i.e. providing 51.1% and 35.4% of the forest stand accumulation, indicating that the main part that can provide harvest of forest management materials is completed by a small number of dominant trees. This provides valuable information for the cultivation of high-quality large-diameter materials in the future.

Table 3. Analysis of the effect of the number and diameter of dominant trees in *Phoebe chekiangensis* pure forest and mixed-age forest on the accumulation and growth of stand

Type	Total number of plants/(plant/hm^{-2})	Average diameter/cm	Average height/m	Growing stock/(m^3/hm^{-2})	Dominant trees that $D \geq 8$						
					Number of plants/(plant/hm^{-2})	Proportion/%	Average diameter/cm	Range/cm	Height/m	Volume/(m^3/hm^{-2})	Proportion/%
Pure forest	3,460	5.79	3.80	24.87	630	18.2	9.16	8–13	5.06 ± 0.51	12.72	51.1
Mixed forest of different ages	3,100	5.80	4.12	22.16	401	12.9	8.86	8–13	5.34 ± 0.81	7.84	35.4

5. Conclusions and discussions

Comparative analysis of stand growth and structure of artificial pure forests and heterogeneous mixed forests of *Phoebe chekiangensis* showed that compared with pure forests, the growth advantages of heterogeneous mixed forests were significant. The average thoracic diameter growth of the stand increased 1.8%, and the growth of single plant wood accumulation increased 7.4%. The analysis results of the relationship between tree height and breast diameter relationship analysis showed that the high growth of individuals in *Phoebe chekiangensis* was significantly promoted in the heterogeneous forest, and the high growth of trees was 8.4% higher than that of pure forests. The proportion of grade 1–5 wood in *Phoebe chekiangensis* of different ages ordered from the largest to the smallest are grade 3 (43.7%), > grade 2 (26.5%), > grade 4 (15.7%), > grade 1 (12.9%) > grade 5 (1.2%), and proportion of grade 1–5 wood in pure forest from the largest to the smallest are grade 3 (34.7%) > grade 2 (25.6%) > grade 4 (20.0%) > grade 1(18.2%) > grade 5 (1.2%); heterogeneous mixed forests have a small degree of differentiation and maintain high growth rates.

In this study, the height and breast diameter growth of *Phoebe chekiangensis* in the heterogeneous mixed forest was better than that in the pure forest, which was caused by the relatively low density of heterogeneous forests and the weak competition between tree species. Generally speaking, there is a density restriction effect among individuals in the forest stand, and the competition between individuals for space resources, material resources and light resources occurs due to the increase in density, resulting in limited resources that can only be allocated to those individuals with strong competitiveness. In this process, it is inevitable to eliminate some trees with weak competitiveness, and also put some trees in a unfavorable position, which will lead to a decline in the overall level of competitiveness, and ultimately make the growth of all individuals in the forest stand poor. On the contrary, at the same level of resource availability, if the number of competing individuals is small, the resource allocation is sufficient, which can ensure high growth of all individuals in the low-density stand. This phenomenon can be further demonstrated in the ratio of tree height and breast diameter of individuals in different stands in this study. The analysis of the tree high breast diameter ratio relationship showed that the tree height growth of *Phoebe chekiangensis* in the heterogeneous forest was significantly promoted, and the tree height growth was 8.4% higher than that of pure forest, which fully showed that the heterogeneous mixed forest was conducive to the growth of nanmu, the development of individuals are well-balanced, and the intraspecific differentiation was moderated, providing a favorable growth environment for the stand. In the study, 18.2% and 12.9% of dominant (grade 1) woods were produced in the 13-year-old *Phoebe chekiangensis* forest, accounting for 51.1% and 35.4% of the accumulation respectively, indicating that the self-thinning effect of the forest stand developed according to a positive allometric growth relationship, that is, along with the process of self-thinning, the development process of the stand will inevitably be accompanied by the elimination of most individuals with weak competitiveness, and the total accumulation of forest stands must be contributed by a small number of competitive individual dominant trees. Therefore, in the current breeding process, the cultivation of the dominant target trees should be strengthened, which is of great significance for forest land productivity and forest renewal. Since *Phoebe chekiangensis* is a long-term cultivar, and the observation time of this study is limited, it is necessary to conduct long-term follow-up observation of its growth characteristics and the self-thinning effect of forest stands.

In view of the growth of two kinds of *Phoebe chekiangensis* forest stand in Tiantong Mountain, in order to promote the diameter and high growth of the dry timber stage as soon as possible, give play to the precious timber tree species as the main ecological benefits, and take into account other functional benefits, the traditional forestry cultivation mode should be changed and the near-natural management and cultivation should be adopted. In pure forests, due to the competitive differentiation of the stand, it can be tended once in the near future, removing the shrubs in the forest, and ing broad-leaved young trees such as Liquidambar formosana, Cinnamomum camphora and red Phoebe. Among the dominant (grade 1) trees, healthy, vigorous, dry and smooth young trees were selected as target trees, and numbered with sign, and the interfering trees within 4 m of them were thinned or transplanted to promote the rapid growth and even distribution of the target trees. For the mother trees that have borne fruits, it is necessary to strengthen the upper layer of light and the cleaning and care of woodland, promoting natural regeneration, and promoting the formation of single-layer forests as soon as possible. Biomass bamboo charcoal can be applied to plots with thin woodland and poor growth of Phoebe to improve the soil. In heterogeneous mixed forests, due to the low initial planting density and low degree of forest stand differentiation, it is also necessary to carry out the tending of removing the indocalamus and shrub vines once, and retain the top broadleaf saplings like *Choerospondias axillaris*, *Machilus leptophylla*, and *red Phoebe*, thereby increasing the hybridization and hierarchy of forest stands, improving the structure of forest stands, and realizing the transformation from plantation forest stand to zonal green broad-leaved forests.

Reference

1. Editorial Board of Chinese Dendrology. Zhongguo shumuzhi: Di 1 juan (Chinese) [Dendrography of China: Volume 1]. Beijing: China Forestry Publishing House; 1983.
2. Xiang Q. Zhennan shu yi xinzhong—Phoebe chekiangensis (Chinese) [Phoebe zhennan belongs to a new species of Phoebe chekiangensis]. Journal of Plant Taxonomy 1974; 12(3): 295–297.
3. Editorial Board of Zhejiang Flora. Zhejiang zhiwuzhi: Di 2 juan (Chinese) [Flora of Zhejiang: Volume 2]. Hangzhou: Zhejiang Science and Technology Press; 1992.
4. Li D, Jin Y, Xiang Q. Zhenxi shuzhong Zhejiangnan de zaipei liyong yanjiu (Chinese) [Study on the cultivation and utilization of rare tree species Phoebe chekiangensis]. Jiangsu Forestry Science and Technology 2004; 31(1): 23–25.
5. Ou B, Wang B, Liu R, *et al*. Zhejiangnan mianmu wuhou ji shengzhang guilv yanjiu (Chinese) [Study on the property and growth law of Phoebe chekiangensis]. Jiangxi Forestry Science and Technology 2002; (4): 1–2.
6. Ji D, Xiang Q. Effects of light condition on the growth and photosynthetic characters of Phoebe chekiangensis seedings. Journal of Nanjing Forestry University 2004; 28(5): 27–31.
7. Shu J, Jiang B, Cheng Xi, *et al*. Effect of underplanted trees in pinus massoniana plantation on growth and soil nutrients. Chinese Agricultural Science Bulletin 2013; 29(1): 82–86.
8. Fang J, Liu G, Zhang S. Fenbuqu xiyuan yousong zhongqun de shengzhang texing (Chinese) [Growth characteristics of pinus tabulaeformis population in the western margin of the distribution area]. Journal of Plant Ecology and Geobotany 1993; 17(4): 305–316.
9. Ding Y, Zhang S, Zhang S. Study on the classification of man-made larch forest. Journal of Northeast

Forestry University 1980; (2): 12–28.
10. Wang B, Zhang J, Wang L. Comparative study on characteristics of stand structures in larix gmelinii forests of different origin. Journal of Northeast Forestry University 2013; 41(2): 18–21.

EnPress Publisher, LLC
Add: 9650 Telstar Avenue, Unit A, Suit 121, El Monte, CA 91731
Tel: +1 (949) 299 0192
Email: contact@enpress-publisher.com
Web: https://enpress-publisher.com