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Impacts of climate change on forest restoration dynamics in the lower montane forest of Doi Suthep-Pui National Park, Northern Thailand

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Abstract. *Thinkampheang S, Nakashizuka T, Suksavate W, Kachina P, Hermhuk S, Asanok L, Phumphuang W, Chouibumroong B, Wu J, Kurokawa H, Marod D. 2024. Impacts of climate change on forest restoration dynamics in the lower montane forest of Doi Suthep-Pui National Park, Northern Thailand. Biodiversitas 25: 4829-4845.* Lower montane forest ecosystems are rich in biodiversity but prone to anthropogenic and natural disturbances. Forest restoration aims to emulate natural succession, particularly concerning tree species composition. In this study, we investigated the effects of different forest restoration practices on tree regeneration and the impact of climate change on forest dynamics following restoration in a natural lower montane forest. Three forest dynamics plots (100 m × 100 m) were established in the forest, and two artificial forest restoration approaches were simulated based on mono-species and multi-species plantations, respectively. All trees with a diameter at breast height > 2 cm were tagged, measured, identified, and monitored every 2 years from 2011 to 2021. Forest dynamics, including mortality and recruitment rates, were analyzed in relation to climate change. The results show that recovery varied among restoration practices. The mono-species plantation had the highest species diversity (114 species), followed by the natural forest and multi-species plantation. The mean mortality rate was higher than the net mean recruitment rate in mono-species and multi-species plantations, with significant variation in forest dynamics across species and periods. These fluctuations were closely linked to environmental changes, particularly droughts related to El Niño. Environmental factors exert a substantial influence on forest structure and species composition. In particular, the interplay between forest dynamics and ecological changes underscores the susceptibility of certain species to drought. Our findings imply that multi-species planting, with its potential to mirror the natural climax state of a forest closely, is a highly effective forest restoration strategy that can inspire and motivate further research and action in this field.

Keywords: Deforestation, disturbance regime, drought stress, mountain ecosystem, species niche

INTRODUCTION

Lower Montane Forest (LMF) ecosystems represent a majority of watershed areas and support a high concentration of biodiversity, including numerous endemic and rare species (Marod et al. 2014; Köhl et al. 2015; Yimyam et al. 2016). These ecosystems have a distinctive climatic regime characterized by persistent cloud cover and frequent mist, which fosters specialized habitats for a variety of plants and animals (Corlett 2014). In Thailand, mountain ecosystems are generally located in the northern part of the country, typically at elevations exceeding 1,000 meters above sea level; they are divided into upper and lower montane forests (Marod et al. 2022). These forests display high species diversity and are typically dominated by the families Fagaceae and Lauraceae (Zhu et al. 2015; Naidu and Kumar 2016; Li et al. 2023). However, their high sensitivity to environmental changes, such as changes in temperature, precipitation, and land use, makes montane ecosystems fragile and difficult to restore once disturbed. According to Crausbay and Martin (2016), both natural and anthropogenic disturbances affect these ecosystems, with forest conversion for agriculture having a particularly damaging impact (Kassa et al. 2017). The importance of biodiversity in forest ecosystems cannot be overstated, as it is essential for maintaining ecological balance and ensuring long-term sustainability (Sharma and Birman 2024).

Deforestation in Thailand during recent decades, mainly due to agricultural expansion into montane forests, has drastically altered the country's forest cover (Leinenkugel et al. 2015; Trisurat et al. 2019; Kruasilp et al. 2023). However, this situation is not irreversible. Forest restoration is not just a possibility but a necessity for rehabilitating deforested and degraded areas of Thailand.

The goals of this restoration are not just to restore richness and biodiversity (Naeem 2016) but also to mitigate climate change impacts and address issues such as biodiversity loss, soil erosion, and carbon sequestration loss (Kumar 2012; Bustamante et al. 2019; International Panel on Climate Change 2021). In addition to planting new trees, effective restoration involves reinstating essential ecological processes such as nutrient cycling to allow forests to recover naturally (Barton et al. 2013; Suding et al. 2015; Fischer et al. 2016). A fundamental mechanism of forest restoration is ecological succession, which involves a gradual change in plant and animal species composition over time. Initially, fast-growing and resilient species colonize disturbed areas, eventually facilitating the establishment of more stable, slow-growing species (Luisa 2012; Schulze et al. 2019). This process may facilitate the return of an ecosystem to a state of higher diversity and stability (Cadotte et al. 2012; Wang et al. 2016). The process of ecological succession from initiation to a stable or climax stage requires a long time and necessitates close monitoring and adaptive management plans (Campbell 2017). This potential for positive change can be effectively addressed by long-term ecological research based on forest dynamics plots (Hogan et al. 2018). Forest dynamics plots allow researchers to monitor species recruitment, survival, mortality, and growth over extended periods (Condit 1998; Hogan et al. 2018). Long-term ecological research has been practiced in the montane forests of Doi Inthanon and Doi Suthep-Pui National Park (DS-PNP), Northern Thailand, to elucidate natural forest dynamics (Sri-Ngernyuang et al. 2003; Marod et al. 2014). However, few studies have investigated plant succession and species adaptation to environmental changes (Riswan and Hartanti 1995; Brown et al. 2013; Joshi et al. 2015; Bustamante et al. 2016). Since 1981, Doi Suthep-Pui National Park has faced significant disturbances due to agricultural expansion and land development (Delang 2002; Elliott et al. 2013, et al. 2023; Chazdon 2015).

In 1985, the national park attempted to recover the degraded lower montane forest through both natural and artificial practices. In 1990, mono species planting (*Prunus cerasoides* Buch.-Ham. ex D.Don) and multispecies planting (*Castanopsis acuminatissima* (Blume) A.DC., *Castanopsis armata* (Roxb.) Spach, *Betula alnoides* Buch.-Ham. ex G.Don, *Cinnamomum iners* (Reinw. ex Nees & T.Nees) Blume, *Diospyros glandulosa* Lace and *Ternstroemia gymnanthera* (Wight & Arn.) Sprague was established (Asanok et al. 2013). These framework species included both rapid growing and native species to initiate succession in the lower montane forest (Elliott et al. 2013, et al. 2023; Chazdon 2015), with the aim of allowing natural processes to enhance plant biodiversity (Barton et al. 2013; Suding et al. 2015). However, the effects of environmental changes on these restoration plantations remain poorly understood. In this study, we investigated the effects of different restoration practices on forest regeneration and the impact of climate change on forest dynamics following the restoration of a lower montane forest. Further research is required in several critical areas to of comprehensively and effectively restore lower montane forests. These include the impacts of climate change, emphasizing the need to understand how rising temperatures and altered precipitation patterns affect plant successional processes and biodiversity. Additionally, forest restoration dynamics should be monitored over the long term, focusing on key parameters such as species recruitment, mortality, and growth within forest dynamics plots. Restoration strategies must also be evaluated, comparing the effectiveness of mono-species, multi-species, and natural regeneration approaches. Finally, adaptive management approaches should be developed to create management frameworks that respond dynamically to changing environmental conditions. Research in these areas is essential for advancing restoration efforts, enhancing ecosystem resilience, and ensuring the long-term sustainability and functionality of lower montane forests in the face of ongoing environmental challenges.

MATERIALS AND METHODS

Study site

The study site was Doi Suthep-Pui National Park, Chiang Mai Province, Northern Thailand (Figure 1). The park covers approximately 161 km^2, encompassing an elevation range of 300-1,650 masl (Hermhuk et al. 2020). Forests within the park generally include deciduous forests (mixed deciduous and deciduous dipterocarp forests) at lower elevations (<1,000 masl) and evergreen forests (lower montane and pine forests) at higher elevations (≥1,000 masl) (Hermhuk et al. 2020). The dominant species in the lower montane forest are mainly species of the family Fagaceae, such as *C. acuminatissima* and *C. armata*, *Castanopsis tribuloides* (Sm.) A.DC, *Lithocarpus truncatus* (King ex Hook.f.) Rehder, and family Lauraceae, such as *Litsea martabanica* (Kurz) Hook.fil., *C. iners*, *Persea gamblei* (Hook.f.) Kosterm., *Cinnamomum parthenoxylon* (Jack) Meisn., *Beilschmiedia gammieana* King ex Hook.fil., and *Litsea grandis* (Wall. ex Nees) Hook.fil. (Marod et al. 2014; Marod et al. 2022). The study site has a subtropical climate characterized by a long dry season alternating with a short-wet season; the annual rainfall typically surpasses 1,600 mm, concentrated during the rainy season (late May to October). The monthly mean temperature is approximately 20°C, reaching a maximum of 28°C in April and a minimum of 2°C in December. Geologically, the terrain consists of steep hills and valleys composed of high-grade metamorphic complex rocks and sandstone formations. The soil composition varies across the landscape; slope areas at 300-1,000 masl feature sandy soil and sandy clay loam, whereas regions above 1,000 masl primarily comprise sandy clay loam and clay soil (Marod et al. 2014, et al. 2022).

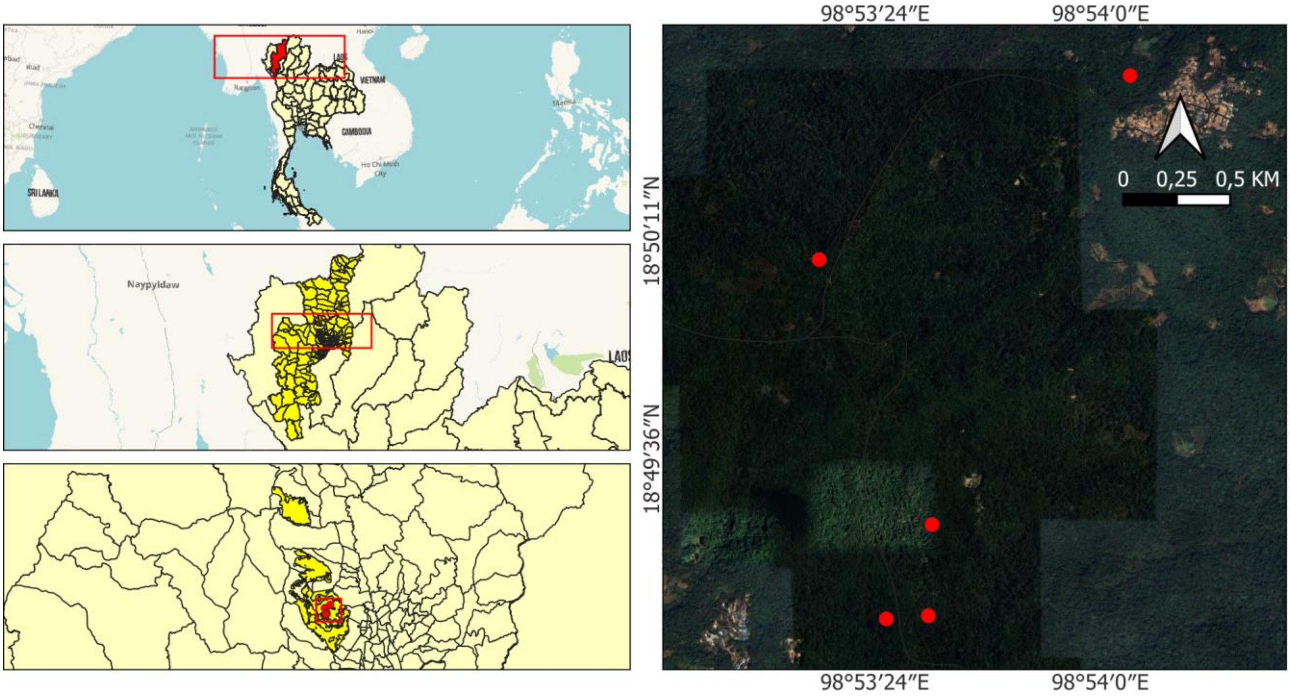


Figure 1. Forest type classification and forest dynamic plot location in Doi Suthep-Pui National Park, Northern Thailand, are shown. The abbreviations indicate the Lower Montane Forest (LMF), Mono-Species Planting Restoration (MOPR), and Multi-species Planting Restoration (MUPR)

We selected forest plots with homogenous environmental conditions, particularly in terms of topographic factors (Figure 1). The natural forest was selected as a control (18°4921'N, 98°53325'E), as well as two artificial restoration forests established in 1990 (Asanok et al. 2013): a mono species planting consisting of *P. cerasoides* (18°82'39" N, 98°89'254" E) and a multi-species plantation consisting of fast-growing and native species including *C. acuminatissima, C. armata, B. alnoides, C. iners, D. glandulosa*, and *T. gymnanthera* (18°83'5096" N, 98°88'848" E).

Plot establishment and tree species

One permanent plot (1 ha, 100 m × 100 m) was established in each of the mono-species and multi-species plantations in 2011 and the natural forest in 2021. These plots were divided into subplots (10 m × 10 m) total 100 subplots, and all trees with Diameter at Breast Height (DBH, i.e., 1.30 m) ≥ 2 cm were marked with aluminum tags, and their position (x, y), height, and DBH were recorded. The DBH measurement point was marked with paint, and the tree was identified to species; samples were collected from unidentified species for comparison with identified specimens at the Bangkok Forest Herbarium, Department of National Parks, Wildlife and Plant Conservation, Thailand. Nomenclature followed that of Smitinand (2014). Trees in the mono species and multi-species planting were monitored in five censuses conducted in 2013 (Y2), 2015 (Y4), 2017 (Y6), 2019 (Y8), and 2021 (Y10). A tree census was conducted in the natural forest plot in 2021, and these data were compared with the plantation data.

Climatic data

Daily maximum temperature, minimum temperature, and total daily rainfall data were obtained from the nearest weather station (18°48'21" N, 98°55'21" E; elevation, 1,040 masl), located approximately 1,500 m from the study area (Upper Northern Region Irrigation Hydrology Center 2021). We used 12 years of data (2010-2022) for climate change analysis.

Data analysis

Forest structure and tree regeneration

Tree species composition was analyzed based on data collected for trees with DBH ≥ 2 cm in the latest survey (2021). Forest structure was described in terms of the number of tree stems and the basal area of dominant species (Berhanu et al. 2017), which was determined according to the Importance Value Index (IVI), which was calculated by summing relative density, relative frequency, and relative dominance scores (Ismail et al. 2021).

Tree regeneration was evaluated according to the DBH size class distributions of all tree species, with at least 20 individuals having DBH ≥ 2 cm, following the method described by Shumi et al. (2019). Species diversity was evaluated according to the Shannon–Wiener Index (H') (Song et al. 2016), calculated as follows:

$$H' = -\sum_{i=1}^{S} pi \ln pi$$

Where: S is the number of species and π is the proportion of individuals or the abundance of the ith species.

The Sørensen Similarity Index (SI) was used to evaluate the similarity of species between two areas; the range of values is 0-1, with a value closer to 1 indicating a higher degree of similarity between the two regions (Christopher 2020). We calculated SI as follows:

$$SI = 2C/(A+B)$$

Where C is the number of species found in both areas, A is the number of species found only in the first area, and B is the number of species found only in the second area.

Climate change

Dry and wet conditions were extracted from air temperature and precipitation data for 2010-2021 according to the Standardized Precipitation–Evapotranspiration Index (SPEI), which is a multi-scalar drought index based on climate data (Liu et al. 2021). The SPEI was calculated for 12-month periods for drought area analysis as previously described (Yu et al. 2014) using the SPEI package (Beguería et al. 2014) in R v3.6.2 (R Core Team, Vienna, Austria). Potential Evapotranspiration (PET) was estimated as previously described (Gharbia et al. 2018).

To compare climatic conditions between local and global scales, we used the Multivariate El Niño Southern Oscillation (ENSO) Index v2 (MEI.v2) data product compiled by the National Oceanic and Atmospheric Administration Physical Sciences Laboratory (https://psl.noaa.gov/enso/mei/). MEI.v2 index values ≥ 0.5 and ≤ -0.5 are classified as warm and cold phases, respectively (Zhang et al. 2019).

Forest dynamics

Forest dynamics were considered only for woody plants with DBH ≥ 2 cm and characterized using the mortality rate (M) and recruitment rate (R) based on Condit et al. (1999) as follows:

$$R\,(\%) = \frac{[\ln(N_t) - \ln(S_t)]}{t} x100 \quad , \quad M\,(\%) = \frac{[\ln(N_0) - \ln(N_t)]}{t}\ x100$$

Where N_0 and N_t are the population size at time 0 and time t, respectively; S_t is the number of survivors at time t; and t is the census interval.

Dominant tree species with more than 15 individuals at each census were used to evaluate the relationship between recruitment and mortality rates. Tree density and basal area differences among census years were evaluated using Analysis of Variance (ANOVA).

RESULTS AND DISCUSSION

Forest structure and species composition

In the natural LMF, focusing on trees with DBH ≥ 2 cm in the year 2021, a total of 105 species, 76 genera, and 48 families were found (Table 1). The tree density and basal area were found to be 2,436 individual ha^{-1} and 35.59 m^2 ha^{-1}, respectively. The most dominant tree species based on IVI (%) was found in *C. acuminatissima* (31.84 %) and *C. Armata* (24.34 %), followed by *L. martabanica, Schima wallichii* (DC.) Korth., *B. gammieana, Quercus oidocarpa* Korth, *P. gamblei., Wendlandia paniculata* (Roxb.) DC., *C. tribuloides, Engelhardtia spicata* Lechen ex Blume, *Helicia nilagirica* Bedd., *Prunus arborea* (Blume) Kalkman, *Styrax benzoides* Craib, *Eurya acuminata* DC., *Sloanea sigun* (Blume) K.Schum., *Vernonia volkameriiolia* Bedd., *Anneslea fragrans* Wall., *Symplocos macrophylla* Wall. ex DC. and *L. grandis,* respectively (Table 1).

In the Mono-species Planting Restoration (MOPR), focusing on trees with DBH ≥ 2 cm in the year 2021, a total of 114 species, 83 genera, and 47 families (Table 2). The tree density and basal area were found to be 1,749 individual ha^{-1} and 34.06 m^2 ha^{-1}, respectively. The most dominant tree species based on IVI (%) was found in *V. volkameriiolia* (30.20%) and *C. armata* (26.79%), followed by *P. cerasoides, Pinus kesiya* Royle ex Gordon, *Saurauia roxburghii* Wall., *Erythrina subumbrans* (Hassk.) Merr., *S. wallichii, C. iners, D. glandulosa, Calophyllum polyanthum* Wall. ex Choisy, *Macaranga indica* Wight, *Turpinia pomifera* (Roxb.) DC., *C. parthenoxylon, Q. oidocarpa, S. benzoides, P. gamblei, Maesa ramentacea* (Roxb.) A.DC., *L. grandis, S. sigun,* and *C. acuminatissima,* respectively.

In the Multi-species Planting Restoration (MUPR), focusing on trees with DBH ≥ 2 cm in the year 2021, a total of 89 species, 54 genera, and 38 families (Table 3). The tree density and basal area were found to be 1,640 individual ha^{-1} and 30.96 m^2 ha^{-1}, respectively. The most dominant tree species based on IVI (%) was found in *C. acuminatissima* (31.02%), *C. armata* (30.93%), *S. roxburghii*, *B. gammieana*, *E. spicata*, *V. volkameriiolia*, *S. wallichii*, *B. alnoides*, *P. declinata*, *W. paniculata*, *H. nilagirica*, *C. iners*, *M. ramentacea*, *Lithocarpus auriculatus* (Hickel & A.Camus) Barnett, *P. gamblei*, *D. glandulosa*, *P. cerasoides*, *E. acuminata*, *L. grandis* and *E. subumbrans*, respectively.

Forest structure and species composition among LMF, MOPR, and MUPR revealed significant differences in species diversity, tree density, and basal area, which help us understand the impacts of different forest restoration practices (Condé et al. 2022; Larrieu et al. 2023). For example, certain species, such as *C. acuminatissima, C. armata, B. gammieana, Q. oidocarpa, P. declinata, W. paniculata, C. tribuloides, C. iners,* and *D. glandulosa*, were dominant in both the natural forest and the multi-species plantation, highlighting their importance in species regeneration and distribution (Lopez-Sampson et al. 2021; Han et al. 2023). In contrast, the dominant species in the MOPR, included *S. wallichii T. angustifloia, C. polyanthum, E. subumbrans, M. indica, M. ramentacea, S. roxburghii,* and *V. volkameriiolia,* which are often referred to as pioneer species because they can facilitate the establishment of native species such as *C. iners, D. glandulosa, Q. oidocarpa, S. benzoides,* and *C. acuminatissima* etc; planting these species in MOPR can enhance species diversity during ecosystem restoration (Klopf et al. 2017; Wu et al. 2021). Such facilitator species may have influenced the high species diversity observed in the MOPR, consistent with reports that multi-species planting can rapidly increase species diversity in forest restoration (Lopez-Sampson et al. 2021; Shi et al. 2021). In this study, the natural forest had the highest tree density, at 2,436 individuals ha^{-1}, indicating that effective species conservation and forest management practices can result in a more densely distributed tree population (Lindenmayer and Laurance 2017; Lin et al. 2023) reflecting the impact of forest management on promoting tree density and maintaining biodiversity. The natural forest also had a higher basal area (35.59 m^{2} ha^{-1}) than both the MOPR (33.96 m^{2} ha^{-1}) and the MUPR (30.96 m^{2} ha^{-1}), indicating higher growth and biomass accumulation in the natural forest, which is consistent with previous findings that basal area as a key indicator of forest health (Lindenmayer and Laurance 2017).

Table 1. Dominant tree species with DBH ≤ 2.0 cm in the LMF at Doi Suthep-Pui National Park, Northern Thailand, in 2021

Species	**Family**	**BA)m.ha^{-1}(**	**Density individual ha^{-1}**	**IVI)%(**	**DBH)cm(Average ± sd**	**Habit**
Castanopsis acuminatissima)Blume(A.DC.	Fagaceae	6.36	224.00	31.66	12.02 ± 10.16	T
Castanopsis armata)Roxb.(Spach	Fagaceae	6.19	115.00	27.27	17.17 ± 19.29	T
Litsea martabanica)Kurz(Hook.f.	Lauraceae	1.59	329.00	24.33	5.85 ± 3.20	ST
Schima wallichii)DC.(Korth.	Theaceae	4.72	96.00	21.72	17.42 ± 17.51	T
Beilschmiedia gammieana King ex Hook.f.	Lauraceae	0.70	175.00	14.97	4.24 ± 6.99	T
Quercus oidocarpa Korth	Fagaceae	1.38	104.00	12.27	6.47 ± 10.84	T
Persea gamblei)Hook.f.(Kosterm.	Lauraceae	1.25	93.00	11.53	7.15 ± 8.85	T
Wendlandia paniculata)Roxb.(DC.	Rubiaceae	0.83	114.00	11.13	7.97 ± 5.10	ST
Castanopsis tribuloides)Sm.(A. DC.	Fagaceae	2.67	14.00	8.81	36.29 ± 36.00	T
Engelhardtia spicata Lechen ex Blume	Juglangdaceae	0.88	51.00	7.62	9.29 ± 12.37	T
Helicia nilagirica Bedd.	Proteaceae	0.49	57.00	6.21	6.68 ± 7.03	T
Prunus arborea)Blume(Kalkman	Rosaceae	0.13	59.00	6.10	4.35 ± 2.91	T
Styrax benzoides Craib	Styracaceae	0.60	36.00	5.26	10.86 ± 9.55	T
Eurya acuminata DC.	Theaceae	0.34	41.00	4.57	8.12 ± 5.64	ST
Sloanea sigun)Blume(K.Schum.	Elaeocarpaceae	0.11	46.00	4.53	3.99 ± 1.93	ST
Vernonia volkameriiolia Bedd.	Asteraceae	0.10	51.00	4.47	4.48 ± 1.77	S
Anneslea fragrans Wall.	Theaceae	0.22	45.00	4.39	5.15 ± 4.22	T
Symplocos macrophylla Wall. ex DC.	Euphorbiaceae	0.10	53.00	4.30	4.21 ± 1.96	ST
Litsea grandis)Wall. ex Nees(Hook.f.	Lauraceae	0.21	38.00	4.15	4.83 ± 4.26	T
Olea rosea Craib	Oleaceae	0.06	46.00	4.07	3.64 ± 1.54	ST
Other species (85)		6.68	649.00	80.65		
Total		35.59	2436	300	11.38 ± 21.13	

Table 2. Dominant tree species with DBH ≤ 2.0 cm in the MOPR at Doi Suthep-Pui National Park, Northern Thailand, in 2021

Species	**Family**	**BA)m.ha^{-1}(**	**Density individual ha^{-1}**	**IVI)%(**	**DBH)cm(Average ± sd**	**Habit**
Vernonia volkameriiolia Bedd.	Asteraceae	1.10	337.00	30.20	5.60 ± 1.65	S
Castanopsis armata)Roxb.(Spach	Fagaceae	5.54	86.00	26.79	17.42 ± 1.66	T
Prunus cerasoides D. Don	Rosaceae	2.41	137.00	21.19	11.09 ± 1.07	T
Pinus kesiya Royle ex Gordon	Pinaceae	4.92	20.00	17.16	52.73 ± 1.99	T
Saurauia roxburghii Wall.	Actinidiaceae	0.98	175.00	16.91	7.16 ± 0.36	ST
Erythrina subumbrans)Hassk.(Merr.	Fabaceae	2.61	28.00	11.64	28.76 ± 1.22	T
Schima wallichii)DC.(Korth.	Theaceae	2.74	21.00	11.27	32.22 ± 2.58	T
Cinnamomum iners Reinw. ex Blume	Lauraceae	0.15	78.00	9.27	3.95 ± 0.54	T
Diospyros glandulosa Lace	Ebenaceae	0.72	49.00	8.34	10.87 ± 0.89	T
Calophyllum polyanthum Wall. ex Choisy	Guttiferae	0.07	68.00	8.23	3.56 ± 0.78	S
Macaranga indica Wight	Euphorbiaceae	1.55	25.00	7.99	19.00 ± 4.23	T
Turpinia pomifera)Roxb.(DC.	Staphyleaceae	0.27	50.00	7.08	7.22 ± 0.47	T
Cinnamomum parthenoxylon)Jack(Meisn.	Lauraceae	1.50	12.00	6.05	24.98 ± 3.09	T
Quercus oidocarpa Korth	Fagaceae	0.09	44.00	5.74	4.30 ± 0.39	T
Styrax benzoides Craib	Styracaceae	0.90	25.00	5.61	17.66 ± 0.91	T
Persea gamblei)Hook.f.(Kosterm.	Lauraceae	0.38	36.00	5.05	8.99 ± 0.46	T
Maesa ramentacea)Roxb.(A.DC.	Myrsinaceae	0.07	49.00	4.88	3.86 ± 0.39	S
Litsea grandis)Wall. ex Nees(Hook.f.	Lauraceae	0.61	23.00	4.63	11.37 ± 2.85	T
Sloanea sigun)Blume(K.Schum.	Elaeocarpaceae	0.09	29.00	4.28	4.17 ± 1.01	ST
Castanopsis acuminatissima)Blume(A.DC.	Fagaceae	0.69	18.00	4.11	13.35 ± 1.27	T
Other species)94(		6.58	438.00	83.57		
Total		33.96	1748	300	9.35 ± 9.39	

Table 3. Dominant tree species with DBH ≤ 2.0 cm in the MUPR at Doi Suthep-Pui National Park, Northern Thailand, in 2021

Species	Family	BA)m.ha^{-1}(	Density)individual ha^{-1}(	IVI)%(	DBH)cm(Average ± sd	Habit
Castanopsis acuminatissima)Blume(A.DC.	Fagaceae	3.96	178.00	31.02	12.87 ± 8.08	T
Castanopsis armata)Roxb.(Spach	Fagaceae	6.01	84.00	30.93	22.13 ± 21.36	T
Saurauia roxburghii Wall.	Actinidiaceae	0.99	225.00	22.45	6.57 ± 4.34	S
Beilschmiedia gammieana King ex Hook.f.	Lauraceae	1.94	147.00	22.25	5.8 ± 12.46	T
Engelhardtia spicata Lechen ex Blume	Juglangdaceae	2.39	58.00	16.07	14.80 ± 16.01	T
Vernonia volkameriiolia Bedd.	Asteraceae	0.24	134.00	13.87	4.18 ± 1.428	ST
Schima wallichii)DC.(Korth.	Theaceae	2.00	44.00	12.84	17.18 ± 17.89	T
Betula alnoides Buch.-Ham. ex G.Don	Betulaceae	2.64	26.00	11.22	34.56 ± 6.626	T
Persea declinata)Blume(Kosterm.	Lauraceae	0.65	54.00	9.22	9.17 ± 6.74	T
Wendlandia paniculata)Roxb.(DC.	Rubiaceae	0.53	59.00	8.99	8.08 ± 6.76	ST
Helicia nilagirica Bedd.	Proteaceae	1.15	31.00	8.43	15.36 ± 17.00	ST
Cinnamomum iners Reinw. ex Blume	Lauraceae	0.18	56.00	8.17	5.19 ± 2.12	T
Maesa ramentacea)Roxb.(A.DC.	Myrsinaceae	0.12	68.00	7.85	4.20 ± 1.86	S
Lithocarpus auriculatus (Hickel & A.Camus) Barnett	Fagaceae	1.01	35.00	7.75	9.70 ± 16.63	T
Persea gamblei)Hook.f.(Kosterm.	Lauraceae	0.60	40.00	7.44	7.76 ± 8.14	T
Diospyros glandulosa Lace	Ebenaceae	0.54	27.00	5.60	11.20 ± 11.80	T
Prunus cerasoides D. Don	Rosaceae	0.73	18.00	4.58	14.41 ± 6.29	T
Eurya acuminata DC.	Theaceae	0.33	21.00	3.83	11.35 ± 7.51	ST
Litsea grandis)Wall. ex Nees(Hook.f.	Lauraceae	0.32	18.00	3.60	11.80 ± 7.84	T
Erythrina subumbrans)Hassk.(Merr.	Fabaceae	0.72	8.00	3.55	29.95 ± 14.27	T
Other species)69(		3.90	309.00	60.35		
Total		30.96	1640	300	9.80 ± 10.27	

Forest regeneration

In all three forest dynamics plots, tree diameter (DBH ≥ 2 cm) in 2021 showed a negative exponential or L-shaped distribution, as illustrated in Figure 2. Notably, smaller, i.e., younger, trees were dominant in all plots. This dominance of younger trees is a key indicator of a sustainable forest structure, as it suggests that these trees can continually replace older ones, thereby maintaining a dynamic balance (Adams et al. 2016; McDowell et al. 2020). This regeneration process is typical of montane and tropical forests, facilitated by light gaps and a supply of saplings and juvenile trees, ensuring the replacement of crown canopy species (Mazdi et al. 2021). As older trees die, younger ones fill the gaps, supporting biodiversity and natural succession (Montgomery and Frelich 2015). The natural regeneration observed in these plots points to ecological resilience, vital for forest conservation in the face of human disturbance and climate change (Chazdon and Guariguata 2016).

Among the five selected dominant tree species in the multi-species plantation (*C. acuminatissima, C. armata, D. glandulosa, C. iners*, and *B. alnoides*) and the mono species (*P. cerasoides*), only *B. alnoides* deviated from the negative exponential growth form (Chazdon 2014; Chen et al. 2018) instead displaying a normal diameter class distribution with large populations of intermediate-sized trees and small populations of small and large trees (Figure 3). Thus, *B. alnoides* does not have an ample supply of young trees to replace mature trees over time. Chen et al. (2018) reported that Betula species regeneration requires highlight conditions for establishment, particularly forest gaps so this finding highlights the importance of understanding the diverse regeneration strategies among tree species. It also presents a challenge to create and maintain these specific conditions for successful regeneration. Although some species can gradually integrate into the forest structure through natural succession, others, such as *B. alnoides*, may require more specific conditions for successful regeneration (Fukushima et al. 2008). However, *C. iners* is a dominant species that was selected for reforestation in multi-species plantations. However, this species exhibits traits characteristic of a Shade-Tolerant (ST) species and serves as a secondary layer tree in lower montane forests. Observations reveal that *C. iners* contributes relatively low basal area (Table 3), indicative of smaller individual tree sizes, but demonstrates a high stem density.

The diameter class distribution of selected native species in the multi-species plantation (*S. wallichii*, *B. gammieana*, and *P. declinata*) and mono-species plantation (*C. acuminatissima*, *C. armata*, and *D. glandulosa*) also exhibited a negative exponential growth trend (Figure 4), which implies that these species will sustainably regenerate over the long term. These six native species were well established in both plantation types, indicating successful forest restoration through regeneration (Adams et al. 2016) and implying that these species are highly suitable for use in forest restoration planning, as they can maintain long-term ecosystem balance (Chazdon and Guariguata 2016). Importantly, these species have demonstrated resilience to tropical humid forest environments, providing reassurance and confidence in their potential for planting in degraded lower montane forest restoration efforts (Blackham et al. 2014; Lázaro et al. 2020).

Similarity index between lower montane forest and forest restoration

Our similarity index analysis of the natural forest and restoration plantations revealed that the multi-species plantation had a higher similarity index (70.89%) than the mono-species plantation in comparison with the natural forest. This finding is consistent with previous findings that diverse planting schemes promote ecological resilience, enhance habitat complexity, and improve species interactions, all essential for long-term forest recovery (Chazdon 2014; Freitas et al. 2019). Although the mono-species plantation was less effective in mimicking a natural forest structure, it exhibited a substantial similarity index of 57.21% compared to the natural forest. This suggests that a mono-species plantation approach, under certain constraints, may also be a viable option for forest restoration, particularly in areas where resources or ecological constraints limit the feasibility of multi-species planting (de Morais et al. 2015; Ellis and Sloey 2024). This flexibility and potential for success under constraints should reassure the audience about the adaptability of the approach.

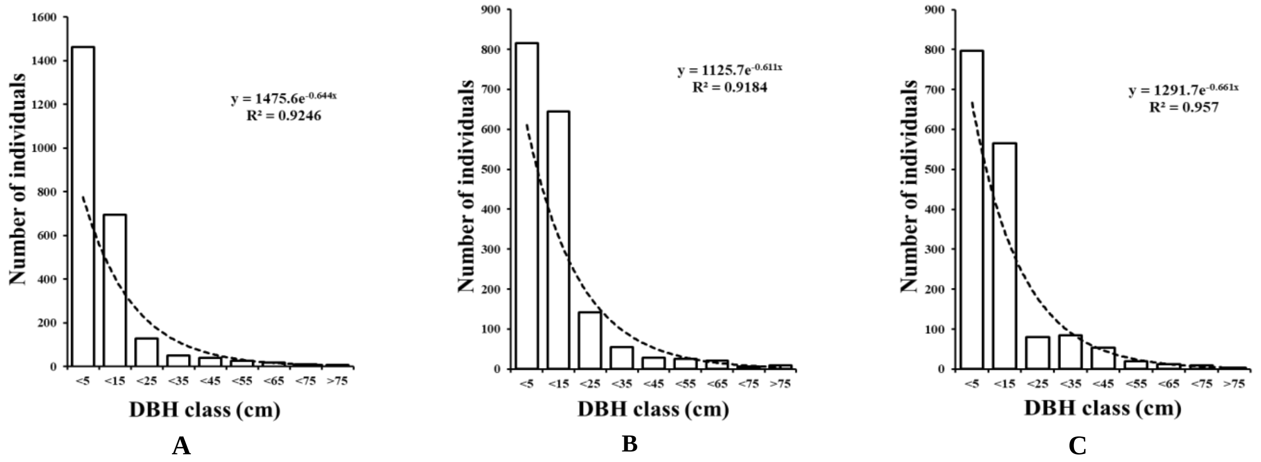


Figure 2. DBH class distributions plotted, showing an inverted J-shaped distribution. A. LMF; B. MOPR; C. MUPR

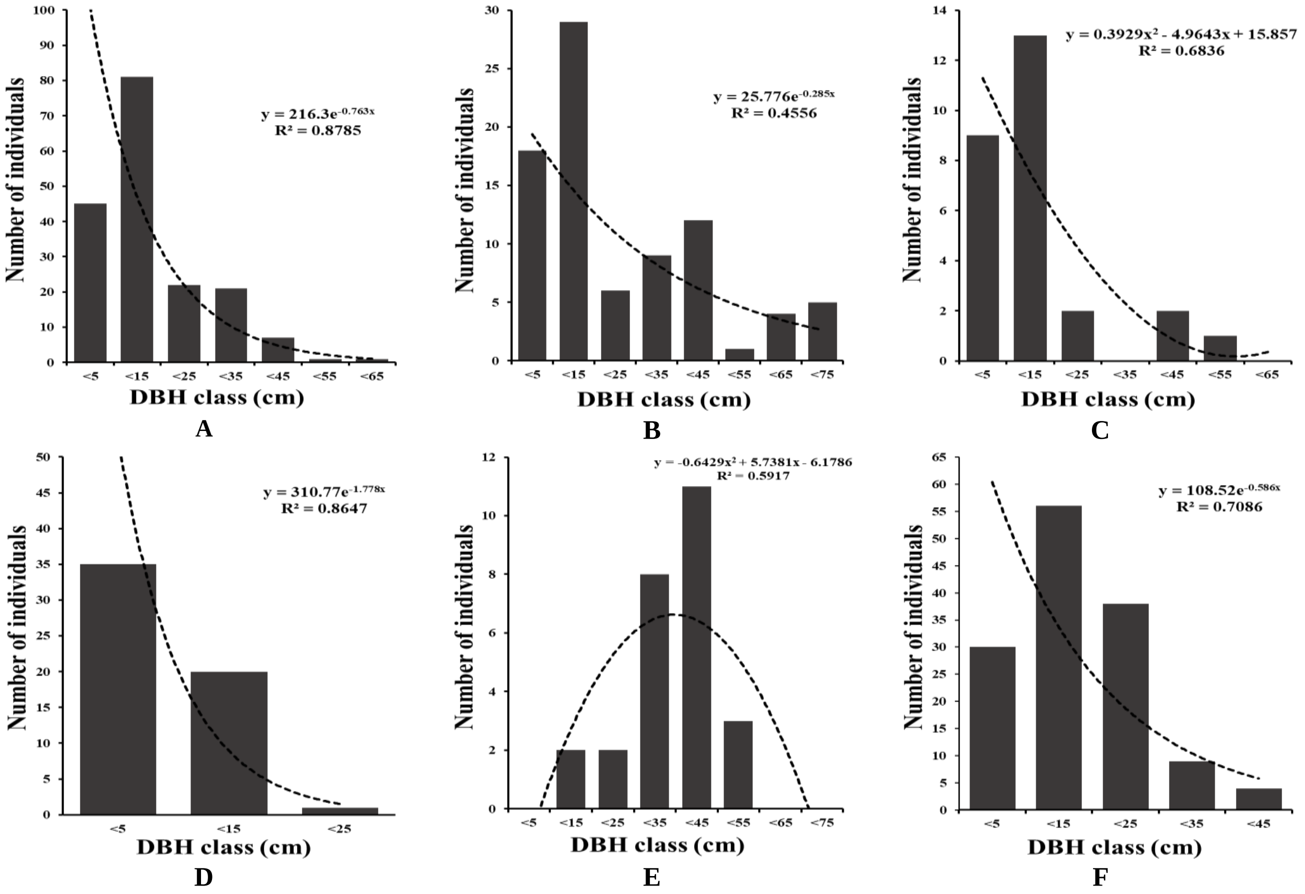
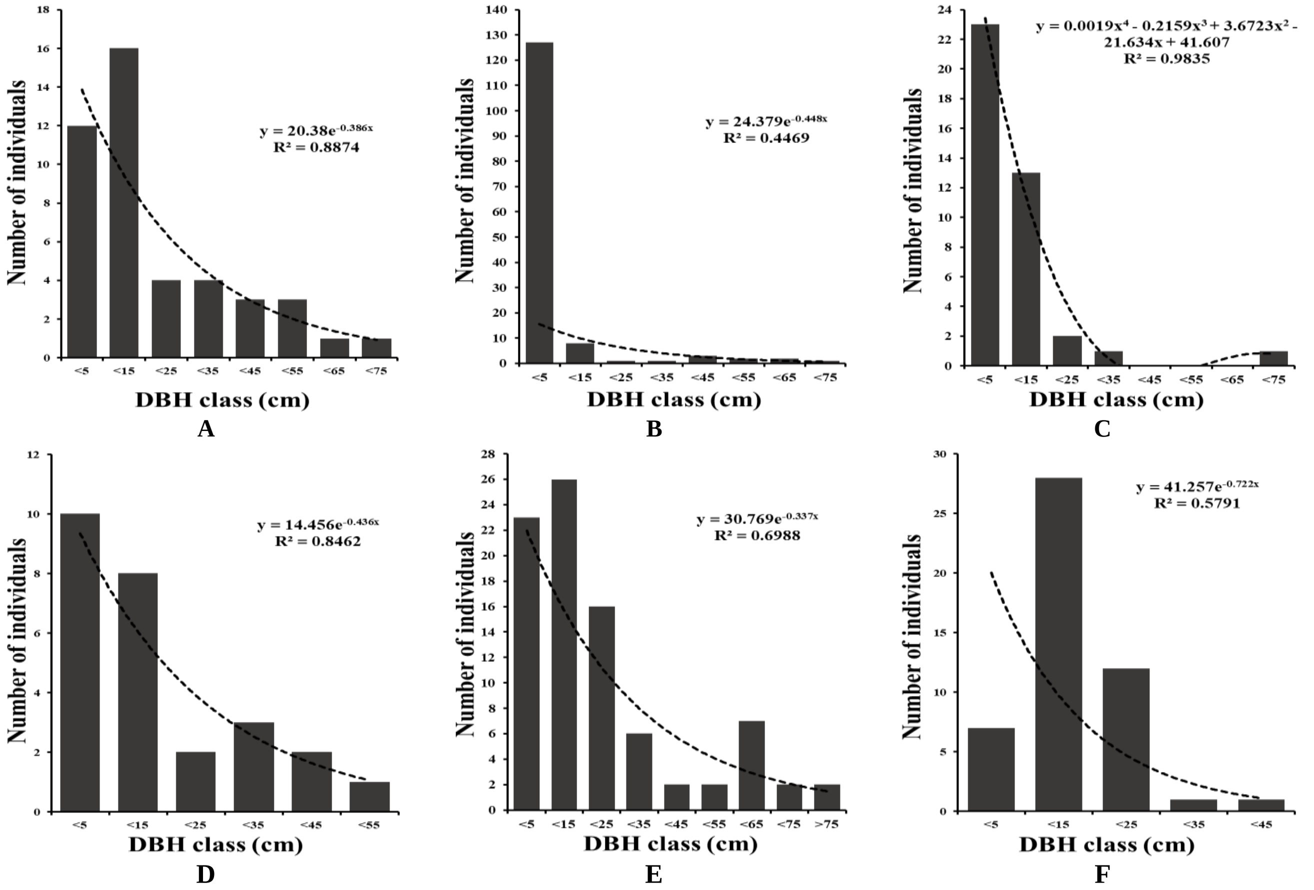


Figure 3. Diameter class distribution of tree species planted in MUPR: A. *C. acuminatissima*; B. *C. armata*; C. *D. glandulosa*; D. *C. iners*; E. *B. alnoides* and in MOPR: F. *P. cerasoides*



A B C

D E F

Figure 4. Diameter class distribution of some native species in MUPR: A. *S. wallichii*; B. *B. gammieana*; C. *P. declinata* and in MOPR; D. *C. acuminatissima*; E. *C. armata*; F. *D. glandulosa*

However, the long-term success of mono-species plantations can be hindered by lower ecological resilience, limited species interactions, and greater vulnerability to environmental changes (Bauhus et al. 2017). Our findings emphasize the importance of adopting a holistic and biodiversity-inclusive approach to forest restoration, particularly in biodiversity-rich ecosystems such as lower montane forests. Future restoration projects should prioritize multi-species planting to enhance ecosystem function and resilience, particularly under climate change and other environmental pressures.

Climate change

During the period 2010-2022, the annual rainfall (mean ± Standard Deviation [SD]) was 1,047.53 ± 266.38 mm, reaching a maximum of 1,702.3 mm in 2022 (Figure 5.A). The dry season was from November to April, and the wet season from May to October; 53.3% of the rainfall was during the wet season on average (Figure 5.B). There were 123 ± 12.89 rainy days per year from 2010 to 2022 (Figure 5.A). There was no record of extreme events such as severe storms or extreme droughts; the maximum temperature was 33.29 ± 0.45°C, and the minimum temperature was 22.30 ± 0.39°C. The highest April temperature was 36.96 ± 1.95°C, and the lowest January temperature was 17.15 ± 1.24°C (Figure 5.B).

The climate data indicated a generally moderate to high level of precipitation, aligning with the monsoonal climate of Northern Thailand. Recent increases in moisture may have influenced forest growth and restoration dynamics (Hamilton et al. 2020). The temperature data reflect the tropical climate of the study area, which is important for assessing stress and adaptation in plant species (Chapman et al. 2012). The observed climatic changes, including seasonal rainfall patterns and temperature ranges, have direct implications for forest restoration strategies. Ample rainfall during the wet season supports the establishment and growth of tree seedlings, and the dry season necessitates considerations for water availability and drought tolerance in restoration planning (Chirino et al. 2009). An absence of extreme weather events promotes long-term restoration success, reducing the risk of large-scale disturbances.

The 12-month SPEI values demonstrated significant variability between -2.0 and 2.0, fluctuating between drought and wet conditions across years (F = 28.062, $P < 0.0001$), largely due to ENSO events. During the 10 years from 2010 to 2021, climate conditions in the study area were classified as normal for 68.2% of the time, moderate for 18.2%, and severe for 13.6% (Figure 6.A), according to the classification scheme of Li et al. (2015). A severe drought was recorded from 2015 to 2016, followed by moderate conditions. Global ENSO data derived from the MEIv2 product indicated that the moderate phase was most frequent (70.83%), followed by the cold phase (La Niña, 20.00%) and warm phase (El Niño, 9.17%) (Figure 6.B). A strong El Niño event occurred during 2015-2016, followed by more moderate conditions in 2018-2019. Dry conditions in the study area are often linked to El Niño events in this region (Goddard and Gershunov 2020). However, variations were observed in certain years, such as 2019, when wetter conditions prevailed despite the occurrence of El Niño, implying that other climatic factors, such as the Indian Ocean Dipole or local monsoonal variation, may have played a mitigating role (Zhang et al. 2022). The variability in SPEI values in the study area also reflected the significant influence of ENSO events on local climate conditions, which is consistent with the findings of other studies that highlighted the impact of ENSO on precipitation and temperature patterns across Southeast Asia (Timmermann et al. 2018). Due to the complexity of climate systems, global indices such as ENSO should be interpreted alongside regional climatic influences when considering their effects on forest ecosystems. The implications of these findings are crucial for forest management and restoration in the study area. Understanding the relationship between ENSO phases and local climate variability can be useful in predicting drought risks, planning for tree species adaptation, and guiding restoration efforts under future climate change scenarios. Further research on interactions between ENSO and other regional climate drivers could enhance predictive models and improve resilience strategies for forest ecosystems in Southeast Asia.

Forest dynamics

Based on the Shannon-Weiner Diversity Index, among trees with DBH ≥ 2 cm, the number of species in each plantation tended to increase with each census, with a higher rate of increase (99.33 ± 9.50) in the multi-species plantation than in the mono-species plantation (86.17 ± 3.66) (Figure 7.A). The basal area showed the same trend, with higher values in the multi-species plantation (31.30 ± 2.02 m2 ha^{-1}) than in the mono-species plantation. In contrast, tree density decreased over time, with higher values in the multi-species plantation (Figure 7.B, Table 2 and Table 3).

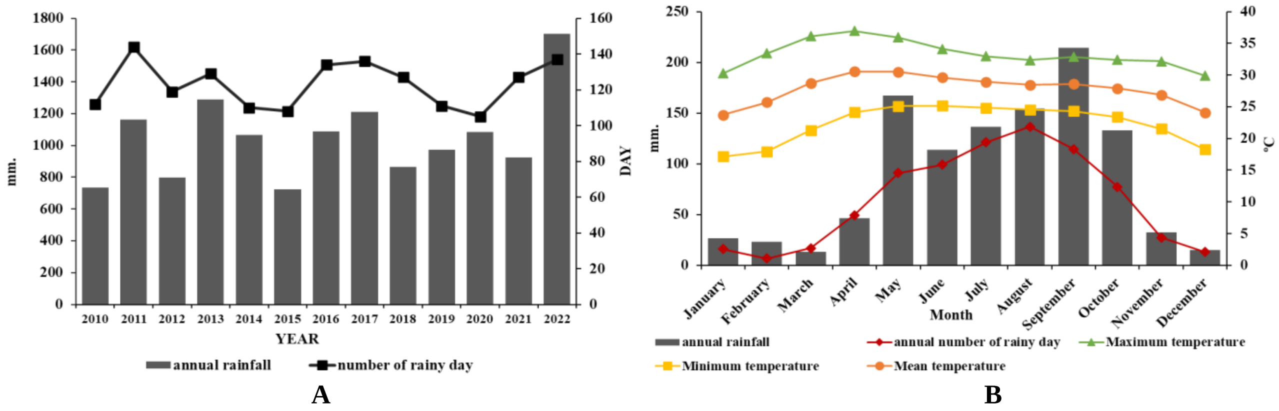


Figure 5. Climatic data from the DS-PNP, Chiang Mai Province, Northern Thailand, from 2010 to 2022: A. Annual rainfall and number of rainy days and B. Average monthly rainfall, the average number of rainy days, maximum temperature, minimum temperature, and mean temperature

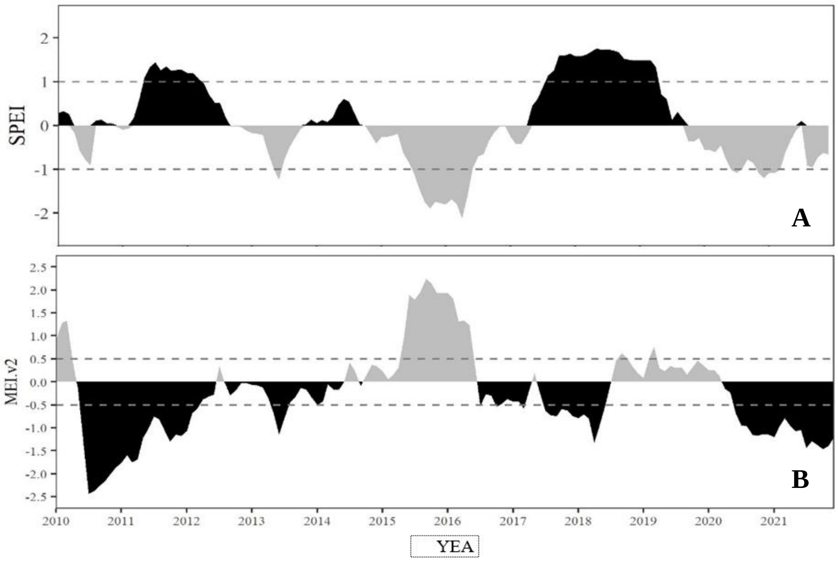


Figure 6. The wet and drought conditions are based on A. SPEI at DS-PNP, Chiang Mai Province, Northern Thailand, and B. The global ENSO oscillation events from 2010 to 2021. Black and grey represented drought/El Niño and wet/La Niña, respectively

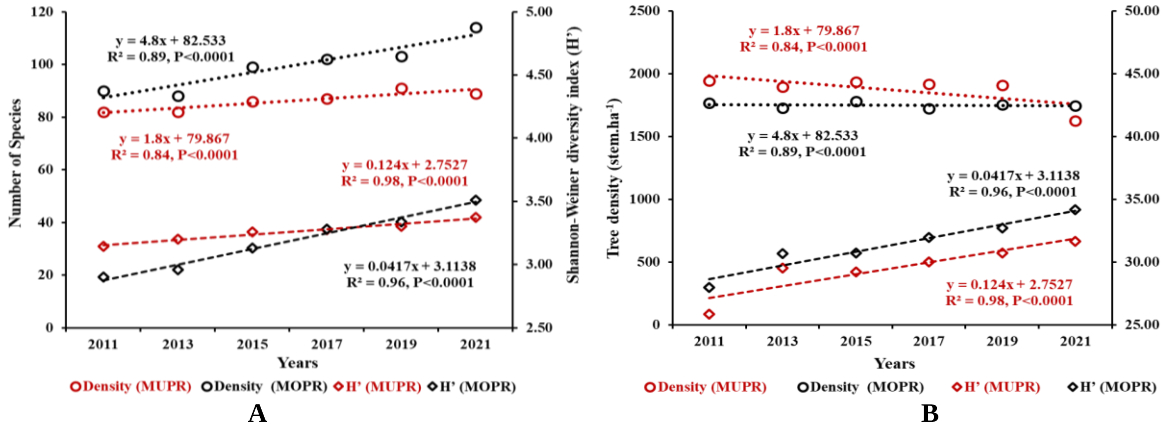


Figure 7. Changes in the forest structure from 2011 to 2021 in FDPs of MOPR and MUPR: A. The number of species and the Shannon-Weiner Diversity Index)H'(and B. The basal area and tree density. The regression lines indicate a significant level at $p < 0.05$

These species richness and diversity findings imply positive species recovery and ecosystem restoration. The higher species number and basal area of the MUPR indicate that its recovery was more robust than that of the MOPR. This trend could be attributed to several factors, such as differences in restoration practices, local environmental conditions, or historical land use impacts (Smith et al. 2016; Lin et al. 2018). However, the decreasing trend in tree density in both plantations, despite increases in species richness and basal area, may imply a shift towards fewer but larger trees as the forest matures. This pattern aligns with the findings of other studies that reported that forest succession often involves an initial increase in tree density, followed by stabilization or reduction as competition for resources intensifies (Chazdon 2014; Arroyo-Rodríguez et al. 2017). Overall, our results underscore the importance of ongoing monitoring and adaptive management in forest restoration projects to ensure that species diversity and forest structure are effectively supported as the ecosystem evolves.

In the MUPR, average mortality (5.89 ± 5.02% year^{-1}) among trees with DBH ≥ 2 cm was higher than average recruitment (4.10 ± 1.78% year^{-1}) during the 10-year study period (2011-2021); the highest mortality rate was observed in the third period (Y4-Y6) (Table 4), coinciding with a long drought from November 2015 to May 2016. In the fifth period (Y8-Y10), mortality (14.74% year^{-1}) was again higher than recruitment (6.70% year^{-1}), which was possibly related to a forest fire that burned approximately 30% of the total plot areas. The forest fires at Doi Suthep-Pui National Park in 2020 have severely impacted the environment and public health. More than 320 hectares of land in the Huai Teng Tao area were destroyed, and the fires spread to important areas, including the lower mountain forest (Earth.Org 2020; Mongabay 2020). Among trees with DBH ≥ 5 cm, average recruitment (3.90 ± 4.16% year^{-1}) differed only slightly from average mortality (3.77 ± 3.03% year^{-1}). However, this relationship varied among periods; in the first period (Y0-Y2), recruitment was higher than mortality, and in the third and fifth periods (Y4-Y6, Y8-Y10), mortality was higher than recruitment (Table 4). Among saplings with DBH < 5 cm, the same trend was observed, with mortality and recruitment of 8.31 ± 7.50 and 6.98 ± 3.48% year^{-1}, respectively, with an unusually high mortality rate in the fifth period (21.54% year^{-1}).

Thus, in the MUPR, average mortality was higher than average recruitment throughout the study period. Previous studies have reported that higher mortality rates than recruitment rates in forest ecosystems may be the result of climate change and/or disturbance (Chazdon 2014). The elevated mortality rates observed during the third period (Y4-Y6) may be attributable to a prolonged drought, which had a significant impact on tree mortality, particularly affecting pioneer species such as *M. ramentacea*, *V. volkameriiolia*, and *S. roxburghii* and some evergreen tree species such as *D. glandulosa*, *C. armata*, and *H. nilagirica*, as drought significantly affects tree mortality (Ansley et al. 2018; Dow 2023). Forest fires, which impacted approximately 30% of the study area, were also a major factor contributing to higher mortality than recruitment during the fifth period (Y8-Y10). The effects of fires on forest structure and tree growth rates can inhibit tree regeneration (Ziegler et al. 2017). Recruitment rates were higher than mortality rates in the early period of tree growth (Y0-Y2), whereas mortality rates were higher in the later period (Y8-Y10), perhaps due to species adaptation or environmental changes over time (Shimamoto et al. 2018; Bieng et al. 2021).

We considered all trees DBH ≥ 2 cm in MOPR, and the mortality and recruitment rate during the 10-year study period (2011-2021) showed that the average mortality rate was higher than the average recruitment rate, 4.55 ± 2.00 and 4.44 ± 1.65 %/yr^{-1}, respectively. In particular, during the third period (Y4-Y6) highest mortality rate was found (Table 5), according to a long period of drought around 7 months, November 2015-May 2016. Considering trees DBH ≥ 5 cm, we found the average recruitment rate was higher than with average mortality rate (3.94 ± 4.37 and 3.16 ± 1.35 %/yr^{-1}, respectively). However, it varied among periods; in the first period (Y0-Y2) recruitment rate was higher than the mortality rate, and in the third and fifth periods (Y4-Y6, Y8-Y10), it was found mortality rate was higher than the recruitment rate (Table 5). Considering for sapling DBH < 5 cm, the same trend was also found (6.86 ± 2.74 and 6.86 ± 2.74%/yr^{-1}, respectively); in particular, the highest recruitment rate was found in the fifth period (10.56 %/yr^{-1}).

The forest dynamics based on mortality and recruitment rates in the MOPR revealed a generally positive trend in forest regeneration, as the net average recruitment rate slightly exceeded the net average mortality rate. However, fluctuations during the third period (Y4-Y6), attributed to a prolonged drought, which had a significant impact on tree mortality, particularly affecting pioneer species (*E. subumbrans*, *V. volkameriiolia,* and *S. roxburghii*) and some evergreen tree species (*D. glandulosa,* and *C. armata*) influenced by the severe drought from November 2015 to May 2016, significantly impacted mortality rates. This period of low precipitation (<50 mm/month) highlighted the sensitivity of both pioneer and evergreen species to climatic extremes (Ansley et al. 2018; Dow 2023). For trees with a DBH ≥ 5 cm, the higher recruitment rates compared to mortality rates, especially in the first (Y0-Y2) and fifth periods (Y8-Y10), suggest stable or expanding canopy cover. Nevertheless, the increased mortality rates in the fifth period indicate potential disturbances or competitive pressures (Shimamoto et al. 2018). Saplings (DBH < 5 cm) exhibited higher recruitment rates than mortality rates, particularly during the fifth period. This suggests successful regeneration despite some periods of higher mortality, reflecting favorable conditions for young trees (Bieng et al. 2021).

Relationship between recruitment and mortality rates

Among trees with DBH ≥ 2 cm in the MUPR, we selected at least 15 individuals from 24 species for further forest dynamics analysis. Variability in the relationship between mortality and recruitment rates was observed among species and periods. These patterns were categorized into three groups based on average changes within 10-year periods (Figure 8.A). Group 1, in which recruitment was higher than mortality, consisted of seven species: *L. auriculatus*, *Syzygium tetragonum* Wall., *Bridelia glauca* Blume, *B. gammieana, C. iners, T. pomifera,* and *T. gymnanthera*, most of which are native species. This group showed relatively low variability in the relationship between mortality and recruitment rates over time; however, from 2019 to 2021 (Figure 8, Y8-Y10), three species (*S. tetragonum*, *T. pomifera,* and *T. gymnanthera*) exhibited higher mortality than recruitment due to the impact of a wildfire in 2020. Group 2, in which mortality was higher than recruitment, included 11 species, *M. ramentacea, V. volkameriiolia, D. glandulosa, S. roxburghii, E. acuminata, C. armata, E. spicata, W. paniculata, P. cerasoides, H. nilagirica,* and *B. alnoides,* many of which are pioneer species, such as *M. ramentacea, V. volkameriiolia, S. roxburghii, E. acuminata,* and *W. paniculata*. However, the relationship between mortality and recruitment rates varied among species and periods within this group. For example, *P. cerasoides* had higher recruitment than mortality (Figures 8.D, E), but during 2019-2021 (Figure 8.F), all 11 species exhibited higher mortality than recruitment due to the 2020 wildfire. Group 3, which exhibited a balance between mortality and recruitment, comprised six species, *P. gamblei, Pittosporum napaulense* (DC.) Rehder & Wilson, *L. grandis, P. declinata, C. acuminatissima,* and *S. wallichii*, most of which are native species. However, the pioneer species *P. napaulense* showed higher variability in the relationship between mortality and recruitment. In addition, from 2019 to 2021, five species (*P. gamblei*, *P. napaulense, L. grandis, P. declinata,* and *S. wallichii*) had higher mortality than recruitment (Figure 8.F) due to the wildfire in 2020.

The regeneration of selected tree species can provide valuable insights into forest ecosystem responses to environmental disturbance and the wildfire (Ainsworth and Drake 2023). By categorizing species based on mortality and recruitment rates, we can identify distinct ecological strategies that reflect either resilience or vulnerability to disturbances such as wildfires (Banks et al. 2011). For example, *L. auriculatus* and *S. tetragonum* had higher recruitment than mortality, indicating effective regeneration and favorable growth conditions. Their stable mortality–recruitment relationship implies successful adaptation to environmental factors (McGranahan and Wonkka 2020). However, the wildfire in 2020 disrupted these conditions, leading to increased mortality in *S. tetragonum, T. pomifera*, and *T. gymnanthera*, whereas *M. ramentacea* and *D. glandulosa* experienced higher mortality rates, highlighting their vulnerability to severe disturbances. Nonetheless, *P. cerasoides* showed a temporary increase in recruitment, implying that some species may still regenerate if environmental conditions improve. Species such as *P. gamblei* and *S. wallichii* maintained a balance between mortality and recruitment, resulting in stable populations. However, the increase in mortality during wildfires illustrates that even established species can be significantly impacted by ecological disturbances (Banks et al. 2011). Overall, these findings emphasize the importance of monitoring tree species responses to disturbances to develop effective forest management and restoration strategies. By leveraging the resilience of certain native species while recognizing the vulnerabilities of others, it becomes clear that proactive measures are crucial in enhancing forest recovery amid ongoing environmental challenges (Kalapodis and Sakkas 2024).

Among trees with DBH ≥ 2 cm in the MOPR, we selected at least 15 individuals from 22 species for further forest dynamics analysis. Again, variability in the relationship between recruitment and mortality was observed between species and periods. Pattern patterns were categorized into three groups based on average changes within the 10-year periods (Figure 9.A). Group 1, which had higher recruitment than mortality, consisted of 13 species: *C. polyanthum, S. sigun, L. grandis, P. gamblei, C. iners, L. martabanica, Q. oidocarpa, T. pomifera, Dehaasia kurzii* King ex Hook.fil.*, B. glauca, M. ramentacea, C. acuminatissima,* and *S. wallichii*, most of which are pioneer species. The relationship between mortality and recruitment rates showed relatively low variability over time.

Table 4. Variables of MUPR dynamics during 10 years at DS-PNP, Chiang Mai Province, Northern Thailand

Variables	Year											Net average 2011-2021
	Y0		Y2		Y4		Y6		Y8		Y10	
BA m^2/ha	25.86		29.56		29.23		30.04		30.75		31.66	29.52 ± 1.99
Loss)m^2.ha^{-1}(		0.52		1.76		1.65		0.91		2.00		1.37 ± 0.62
Gain)m^2.ha^{-1}(		4.22		1.44		2.45		1.61		2.91		2.53 ± 1.12
Overall Individual density)/ha(	1,946		1,897		1,937		1,921		1,911		1,627	1873.17 ±121.87
M)%.yr^{-1}(		3.54		3.27		5.05		2.87		14.74		5.89 ± 5.02
R)%.yr^{-1}(		2.26		4.31		4.63		2.60		6.70		4.10 ± 1.78
Individual density)/ha( )Tree(	873		1051		1,038		1012		1,008		884	977.67 ± 78.54
M)%.yr^{-1}(		2.05		2.34		3.23		2.12		9.12		3.77 ± 3.03
R)%.yr^{-1}(		11.32		1.72		1.96		1.92		2.55		3.90 ± 4.16
Individual density)/ha( )sapling(	1073		846		899		909		903		743	895.50 ±107.20
M)%.yr^{-1}(		4.79		4.45		7.00		3.76		21.54		8.31 ± 7.50
R)%.yr^{-1}(		4.13		7.49		7.78		3.38		12.13		6.98 ± 3.48

Table 5. Variables of MOPR dynamics during 10 years at DS-PNP, Chiang Mai Province, Northern Thailand

Variables	Year											Net average 2011-2021
	Y0		Y2		Y4		Y6		Y8		Y10	
BA m^2/ha	28.01		30.71		30.75		31.95		32.71		34.17	31.29 ± 2.10
Loss)m^2.ha^{-1}(		1.46		1.81		1.25		0.67		1.39		1.32 ± 0.42
Gain)m^2.ha^{-1}(		4.17		1.85		2.45		1.43		2.85		2.55 ± 1.05
Overall Individual density)/ha(	1,769		1,727		1,780		1,726		1,757		1,748	1,751 ± 21.96
M)%.yr^{-1}(		3.63		3.57		6.50		2.25		6.82		4.55 ± 2.00
R)%.yr^{-1}(		2.43		5.08		4.96		3.14		6.56		4.44± 1.65
Individual density)/ha( )Tree(	902		1,090		1,054		990		987		976	999.83 ± 65.43
M)%.yr^{-1}(		2.09		2.83		4.83		1.75		4.28		3.16 ± 1.35
R)%.yr^{-1}(		11.56		1.15		1.70		1.60		3.72		3.94 ± 4.37
Individual density)/ha( )sapling(	867		637		726		736		770		772	751.33 ± 74.97
M)%.yr^{-1}(		5.35		4.23		8.41		3.08		7.56		5.85 ± 2.43
R)%.yr^{-1}(		4.60		4.86		9.72		5.27		10.56		6.86 ± 2.74

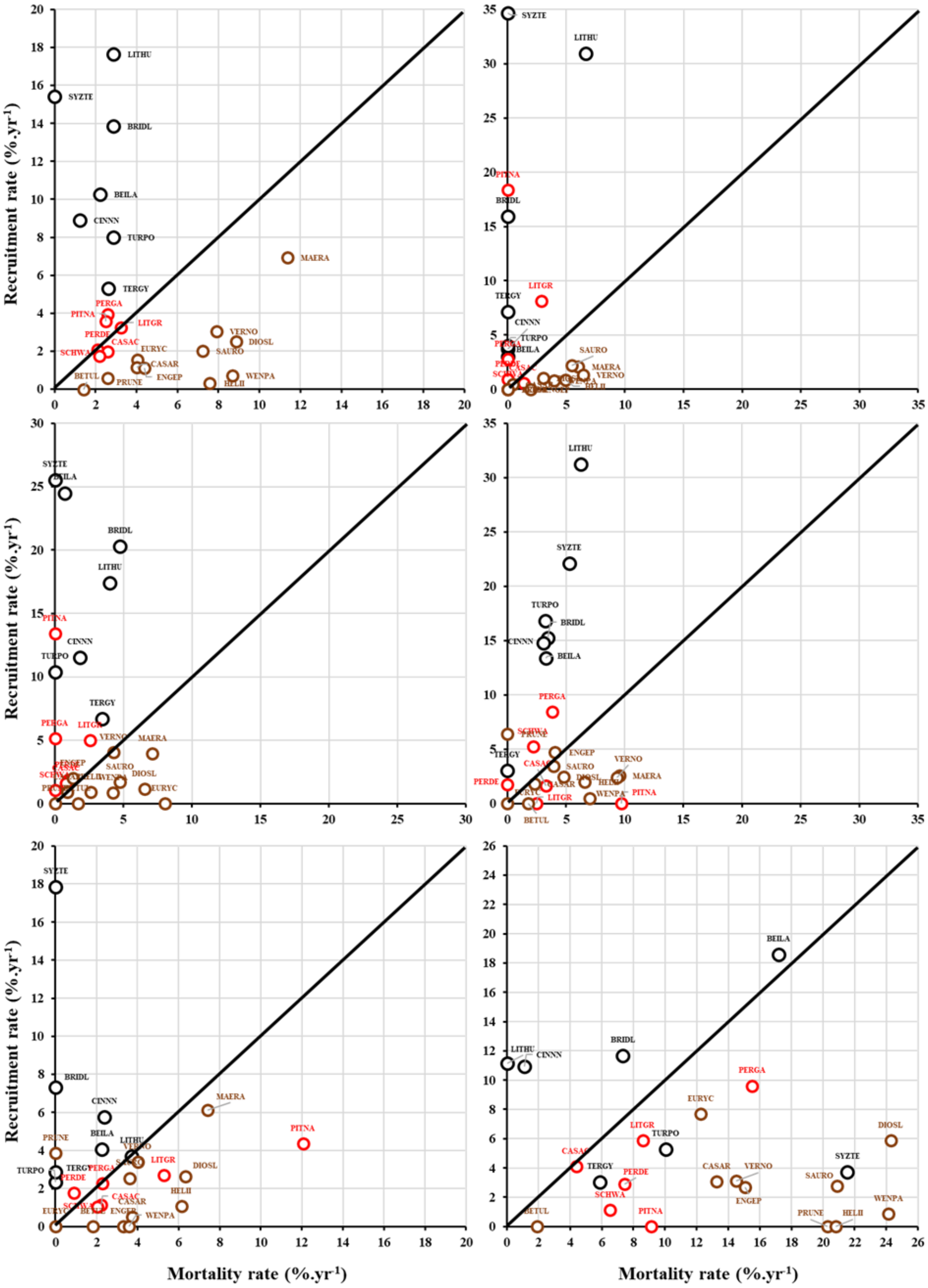


Figure 8. Relationships between recruitment and mortality rates during the 10 years)2011-2021(. The black line is an aspect ratio of 1:1; black dots are a group of species with higher recruitment than mortality, grey dots are a species with the balance between mortality and recruitment, and red dots are a group of species with higher mortality than recruitment

However, *C. acuminatissima* and *P. gamblei*, which are dominant canopy species in the lower montane forest, consistently showed higher recruitment than mortality in all periods, indicating strong regeneration and establishment of both species. Group 2 had higher mortality than recruitment in species such as *E. subumbrans*, *D. glandulosa*, *V. volkameriiolia*, *M. pinnata*, and *S. rugosus*, respectively. Many of the species with high mortality rates are pioneer species, such as *E. subumbrans*, *V. volkameriiolia*, *M. pinnata*, and *S. rugosus*. However, the relationship between mortality and recruitment varied depending on the species and period, with most species consistently having higher mortality than recruitment within the 10-year periods, but a few, such as *S. rugosus*, having higher recruitment (Figure 9.E). Group 3 exhibited a balance between mortality and recruitment, including *P. cerasoides*, *C. armata*, *S. roxburghii*, and *M. indica*. However, the pioneer species *M. indica* exhibited higher recruitment than mortality during the early period but higher mortality than recruitment during later periods (Figure 9). Selected tree species in the MOPR showed significant recovery patterns between 2011 and 2021, with variations in recruitment and mortality rates among different species. For example, *C. polyanthum*, *S. sigun*, and *P. gamblei* had higher recruitment than mortality, indicating strong recovery and establishment and effective adaptation to environmental factors (McGranahan and Wonkka 2020) whereas *E. subumbrans* and *D. glandulosa* exhibited higher mortality than recruitment, highlighting their vulnerability. High mortality can hinder recovery and impact the structural integrity and resilience of the forest ecosystem (Ainsworth and Drake 2023). Additionally, *P. cerasoides* and *C. armata* showed a balance between mortality and recruitment. Although some species, such as *M. indica*, initially exhibited higher recruitment, subsequent trends indicated increased mortality, possibly. Overall, this study emphasizes the importance of understanding species responses to environmental pressures and forest restoration practices within the lower montane forest ecosystem. The variability in mortality and recruitment rates highlights different ecological strategies employed by species, forming a forest landscape shaped by both resilience and vulnerability (Kalapodis and Sakkas 2024). This understanding is crucial for engaging in effective forest management practices, sparking interest in the development of sustainable approaches that will further improve forest recovery and promote a diverse, resilient ecosystem over the long term.

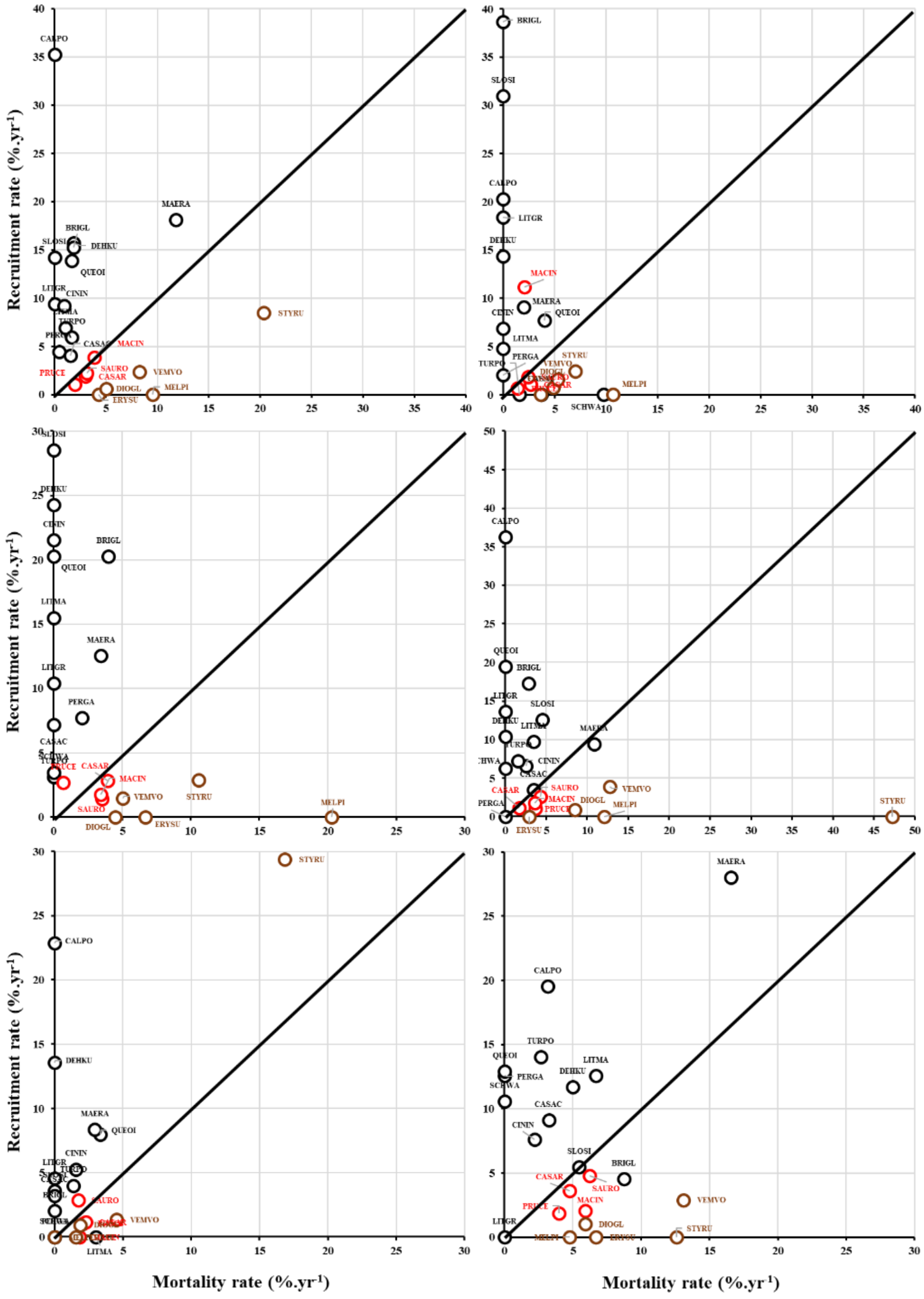


Figure 9. Relationships between recruitment and mortality rates during the 10 years (2011-2021(. The black line is an aspect ratio of 1:1; black dots are a group of species with higher recruitment than mortality, grey dots are a group of species with the balance between mortality and recruitment, and red dots are a group of species with higher mortality than recruitment

In conclusion, the use of multi-species plantations has emerged as a highly effective forest restoration strategy, with the potential to mirror the natural climax state of the forest closely. This approach offers renewed hope for revitalizing ecological processes and enhancing biodiversity within degraded lower montane forests. The forest dynamics of the study area revealed significant variations across different species and periods that were closely linked to environmental changes, particularly climate events such as drought, El Niño, and forest fires. These factors exerted a substantial influence on forest structure and species composition. Certain species within the natural forest, including *D. glandulosa*, *S. benzoides*, *C. armata*, and *L. mekongensis*, exhibited heightened vulnerability to drought. The interplay between forest dynamics and environmental changes underscores the susceptibility of certain species to drought and the challenges posed by forest fires. These findings deepen our understanding of the complex relationships governing forest ecosystems and offer valuable insights for guiding conservation and management efforts in lower montane forests.

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