Research Topics

Land degradation and forest management

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Land degradation and forest management

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Table of contents

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Editorial: Land degradation and forest management

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Editorial on the Research Topic
Land degradation and forest management

Forest land degradation poses a significant threat to forest ecosystems, and biodiversity including human livelihoods worldwide. This encompasses the deterioration of its land quality due to various factors, including deforestation, unsustainable land use practices, urbanization, and climate change. The implications are profound, leading to loss of soil fertility, reduced water quality, and increased vulnerability to natural disasters. As the global population continues to grow, the pressure on land resources intensifies, making effective management of our forests more crucial than ever. Forest management plays a pivotal role in addressing land degradation. Sustainable forest management (SFM) practices are designed to maintain and enhance forest ecosystems while, ensuring their capacity to provide essential services, such as biomass production, carbon sequestration, habitat for wildlife, and recreational spaces for communities. By adopting integrated approaches that combine conservation with responsible forest resource use, degraded forested lands can be restored while, improving resiliency against climate change. This editorial calls for a multifaceted approach to forested land management that recognizes the interconnectedness of forests and non-forest land use systems with community involvement, innovative technologies, and suitable policy frameworks.

Hashim et al. explores species diversity and vegetation pattern in temperate conifer forests along altitudinal gradients in the Western Himalayas. Using ordination (DECORANA) and classification techniques, no clear disjunct vegetation patterns were observed, though altitude and soil types influenced vegetation distribution. Factors such as soil chemistry, litter cover, and rockiness were linked to vegetation changes along the altitudinal gradient. The study highlights the importance of habitat heterogeneity in forest management and biodiversity conservation, emphasizing the need for broad-scale information to guide conservation efforts.

The research article by Kumar et al. highlights the factors influencing tree biomass and carbon stock in the Western Himalayas, India which are significant reservoirs of tree biomass and carbon stock. The carbon stock is primarily influenced by structural attributes like tree diameter (DBH), total basal area (TBA), and tree height while, species diversity, elevation, and climatic factors play a minor role. A few dominant large-diameter species contribute most of the carbon stock, stressing the need for regulated harvesting to ensure long-term ecosystem sustainability. The study concluded with emphasizing the importance of protected areas for achieving carbon neutrality of Western Himalayan Temperate forests. In addition to legal protection of these forests, stricter monitoring and periodic evaluation are recommended for regulating the human activities like tourism. Lastly, the study emphasized the need of further research to explore the factors influencing the forest biomass and carbon storage.

The research article by Bueno et al. highlights the potential of *Opuntia ficus-indica* (prickly pear cactus) toward ecosystem resilience, forest restoration, and restricting desertification for restoring Mediterranean forests. The species is highly drought-resistant, adaptable to arid conditions, and capable of improving soil fertility and structure. Planting the species in degraded areas can help combat soil erosion, increase organic matter, and enhance water retention, making the environment more suitable for the re-establishment of native forest species.

The research article by Li et al. emphasizes the need to understand the spatial patterns of ecological risk for effective management and conservation planning in Qilian Mountain National Park, particularly in addressing biodiversity protection and regulating anthropogenic activities. There is significant spatial variation in ecological risks, with higher risks in areas with more anthropogenic activities (like grazing and infrastructure development) as compared to remote, and undisturbed areas.

The research article by Negi et al. reports significant changes in species composition, plant diversity, and biomass distribution with altitude at mountain ecosystems of the eastern Himalayas, India i.e., vegetation shifts from diverse, dense forests to sparser, less diverse communities from lower to higher altitudes. These elevation-driven changes in vegetation are found closely linked to temperature, soil properties, and moisture availability.

The review article by Wani et al. presents a bibliometric analysis of studies on threat assessment and species prioritization for conservation discussing key methodologies used in threat assessments (including the IUCN Red List criteria), regions and species groups. The review emphasized conservation prioritization and better resource allocation using GIS and machine learning particularly for the underrepresented regions and species.

Zhumasheva et al. reports community-based traditional management practices are crucial in balancing forest conservation with resource use ensuring sustainable harvesting of nuts and fuelwood for improved livelihoods and forest health in the Jalalabad region of Kyrgyzstan. However, overharvesting, lack of infrastructure, and climate change were identified as major challenges to forest resiliency and sustainable resource management.

Aabeyir et al. through remote sensing identifies agricultural expansion, deforestation, and human encroachment as significant drivers of habitat degradation in terms of biodiversity and ecosystem services at Gbele Resource Reserve of Ghana's Upper West Region. The study recommends sustainable land management practices, regular monitoring and empowering forest-dependent communities to protect the Reserve's ecological integrity.

Similarly, in Heilongjiang Province, China Ren et al. also recommends empowering and participation of local community for sustainable forest management practices using technology for successful biodiversity conservation and ecological restoration. The study emphasized the support of adequate and pro-people policy frameworks with optimum and timely financial support for success of these efforts.

The study by Verma et al. reveals that parasitism affects the growth rates and physiological traits of sandalwood, leading to reduced biomass and altered nutrient uptake. These complex host-parasite interactions need a better understanding of host responses to improve sandalwood cultivation and management practices to improve plant health and productivity of the sandalwood.

The study by Sur et al. integrates multi-sensor data (e.g., satellite imagery from various sources) with advanced machine learning algorithms for achieving accuracy and reliability of vegetation degradation assessments. In addition to distinct patterns of degradation related to factors such as land use changes, urbanization, and climate variability, short-term fluctuations and long-term trends in vegetation health were also detected.

Understanding the plant distribution, ecological traits, and diversity patterns in subtropical managed forests are important for developing effective forest management strategies (Waheed et al.). Species diversity and composition are the function of management practices, soil characteristics, and climate. Adaptive management approaches considering these ecological insights can enhance biodiversity conservation, improve forest resilience, and optimize resource use.

Spatial patterns and quantum of deforestation in the Eastern Carpathians of Romania was assessed using fractal algorithms by Diaconu et al.. Superiority of fractal analysis in estimating forest loss over traditional methods that overlook complex patterns of fragmentation and degradation was highlighted. The study identifies land use changes and topographical variations as the primary drivers of irregular deforestation patterns and thus recommends fractal algorithms-based forest management and conservation strategies for effective forest restoration.

The review by Gunawardena et al. highlights the importance of integrating management options with collaborative frameworks to achieve both land degradation and carbon neutrality for improving ecosystem services and conserving biodiversity. Conflicting interests in land use, data gaps, and lesser stakeholder engagement were identified as the main challenges to achieve the neutrality goals.

Importance of seed traits, germination rates, and seedling growth in understanding the mechanism of adaptation by tree species in harsh treeline environment of western Himalayas was reported by Singh et al.. Seed size and dispersal mechanisms significantly influence the distribution and establishment of seedlings in this environment. Temperature, moisture, and soil conditions are critical for seedling success and survival. The study emphasizes the need for conservation strategies that consider these ecological dynamics to support the resilience of treeline ecosystems in the face of climate change and habitat disturbance.

Kryszk et al. examine the declining interest in afforestation within the framework of the Common Agricultural Policy (CAP) in Poland and Lithuania. Despite the potential benefits of afforestation for biodiversity and climate change mitigation, there is a notable reduction in landowners' engagement in afforestation initiatives in these two countries. Factors contributing to this decline include insufficient financial incentives, bureaucratic barriers, and a lack of awareness about the ecological and economic advantages of afforestation. The study recommends adequate pro-people policy support and targeted awareness campaigns to reinvigorate interest in afforestation practices among landowners in these countries.

Author contributions

GS: Conceptualization, Project administration, Writing – original draft. AN: Project administration, Writing – review & editing. SC: Project administration, Supervision, Writing – review & editing. PP: Supervision, Writing – review & editing. AS: Writing – review & editing.

Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Exploration of species diversity and vegetation pattern in temperate conifer forests along altitudinal gradients in the Western Himalayas

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The moist temperate forests extend along the whole length of outer ranges of Himalaya between the subtropical pine forests and sub-alpine timberline formation with a rainfall from 400 mm to 800 mm. The altitudinal range is from 1500 m to 3000 m. The floristic variation in these Himalayan forests is poorly understood especially in the study area. Species composition may be either unpredictable or it may correspond to environmental heterogeneity. Vegetation from 144 stands in between 2000 – 2700 m altitude was sampled. Soil samples were collected to document edaphic conditions. Soils were physically and chemically analyzed. Ordination (DECORANA) and classificatory techniques were used to analyze the vegetation data. No clear disjunct vegetation patterns emerged from these analyses. The major axes brought out by the ordination were related to altitude and although it is possible to relate the units of classification to broad soil types. Soil chemical properties, litter cover and rockiness were significantly associated with the vegetation variation along an altitudinal gradient. The application of classification to the ordination allowed the interpretations of the vegetation variation in terms of topography and predictable climatic factors such as rainfall, wind speed and extent of snow accumulation. The vegetation patterns revealed have been discussed in relation to the general problem of plant community definition in continuous forest type. Our results demonstrate the overlapping rather than clearly discrete boundaries between the vegetation types and species distributions. This overlapping nature of the vegetation types are discussed in terms of overlapping environmental preferences of the species. Our results are consistent with notion that species separate edaphically and landscape scale within the uniform looking forest. This view maintains that differences in soil within the forest are distinct enough to favor different species, and thus create numerous floristically differentiated forest patches along the altitudinal gradients. The findings that non obvious but distinct floristic and edaphic variation exist within the Himalayan moist temperate forests at the landscape scale have important practical implications for forest management.

In biodiversity conservation a high degree of habitat heterogeneity implies an increased need for wide-scale information on species distribution and endemism patterns to better assess where the different habitats are, which species they harbor, and where conservation efforts should be concentrated. The overlapping nature of vegetation types and hazy boundaries of the plant communities implies that plant ecologists must continue to attempt the difficult definition of hazy boundaries.

KEYWORDS

mountain ecosystem, species distribution, multivariate analysis, Western Himalayas, distribution pattern, composition

1. Introduction

In recent decades, vegetation ecologists have become increasingly interested in recording and understanding mountain vegetation species composition and spatial structure. Owing to the interference of several influences in the high mountain vegetation pattern, the driving factors of the vegetation pattern always remain a topic of debate (Wazir et al., 2008; Saima et al., 2009; Khan et al., 2011, 2012).

In mountain landscapes, altitude is the most important factor that influences habitats, climate, and flora contrasts. Altitude represents a complex combination of related variables such as topography, aspect, soil types, and other environmental features that further affect vegetation composition (Dasti et al., 2007; Wazir et al., 2008; Saima et al., 2009). Ellu and Obua (2005) described how different altitude and slope aspects encounter species diversity. Kharkwal et al. (2005) pointed out that the elevation gradient and its linked geo-climatic variables like precipitation and temperature determine species diversity and community composition. Topographic heterogeneity is also considered an important factor in temperate mountain landscapes that shape phytodiversity, as they strongly influence the length of the growing season related to temperature (Dasti et al., 2007; Wazir et al., 2008; Saima et al., 2009). Downslope movement and distribution of soil water contents depend on the ratio of a rocky surface area to soil volume and the spatial variation in runoff pattern. Redistribution of rainfall water by runoff from some areas depends on rainfall intensity and the surface's infiltration rate.

In most Himalayan forests, runoff accumulation occurs at various scales (Yair and Shahak, 1982; Wazir et al., 2008). Thus, niches and habitats of various kinds and sizes are formed, which determine the structure and composition of vegetation (Orshan, 1986). The distribution patterns of runoff determine the differences in vegetation type and influence plant assemblage (Dasti et al., 2007; Wazir et al., 2008). Besides the runoff pattern, strong winds and snowpack distribution at high altitudes significantly shape plant communities (Tanner et al., 1990; Ahmad et al., 2006; Wazir et al., 2008). The results of previous studies (Tanner et al., 1998; Wazir et al., 2008; Saima et al., 2009; Khan et al., 2012) showed continuous variations in composition across a broad range from lower blue pine canopied forest to upper, relatively open *Abies pindrow* formation.

We used modern data analysis techniques to assess the compositional patterns of plant communities in such a complex landscape. These approaches help summarize the gradients in data sets, assisting in formulating hypotheses and testing their validity (Odgaard and Rasmussen, 2000). Many studies have explored variations in species diversity and soil nutrient concentrations along altitudinal gradients using numerical techniques (Khan et al., 2011; Ping et al., 2013; Saima et al., 2018). Sharma et al. (2009) described the species distribution pattern in India's Garhwal forest of the Himalayas. Bhattarai et al. (2014) investigated species diversity in Nepal's Karnali River Valley in the Himalayas. Shaheen et al. (2011) studied species diversity, community structure, and distribution patterns in Kashmir, Pakistan's Western Himalayan Alpine pasture. Ummara et al. (2013) quantitatively analyzed understory vegetation from only 22 different Shogran Valley reserve forest sites. Raja et al. (2014) described the floristic composition of Ayubia National Park from 27 sampling stands (sites). Jadoon et al. (2017) described ecological gradient analyses of plant associations in the Thandiani reserve forests of the Western Himalayas, Pakistan, using 50 sampling stands. Saima et al. (2018) conducted research to quantify the diversity of species and boundaries of plant communities along the altitudinal gradient in the Shogran Valley reserve forest, while Rahman et al. (2019) explored the floristic diversity of Manoor Valley (Naran). Qadir et al. (2020) researched the anthropogenic effects on natural plant biodiversity distribution and diversity patterns, particularly along the sides of the road in pine forests. Noor et al. (2021) explored the structure and vegetation composition of the forest located in Murree Hills, Punjab, Pakistan. Ahmad et al. (2021) studied the structure and vegetation composition of the Himalayans' moist temperate forest, Kaghan Valley. In other parts of the world, Wani et al. (2023) conducted research to quantify the diversity and compositional patterns in the Kashmir Himalaya.

Although all these studies add significant literature to the topic, studies related to species diversity and richness using a multivariate analysis of the study area are missing and confined to the single forest type due to its remote location and inaccessibility of tracks. Moreover, no information is available regarding a long transect along the elevation gradient from low-level blue pine forests to upper timberline fir forests in these temperate mountain areas. Besides these, we have attempted to elucidate how vegetation changes along the pronounced environmental gradient across the three reserved forest types covering an area of 3,500 Km^2. We aimed to

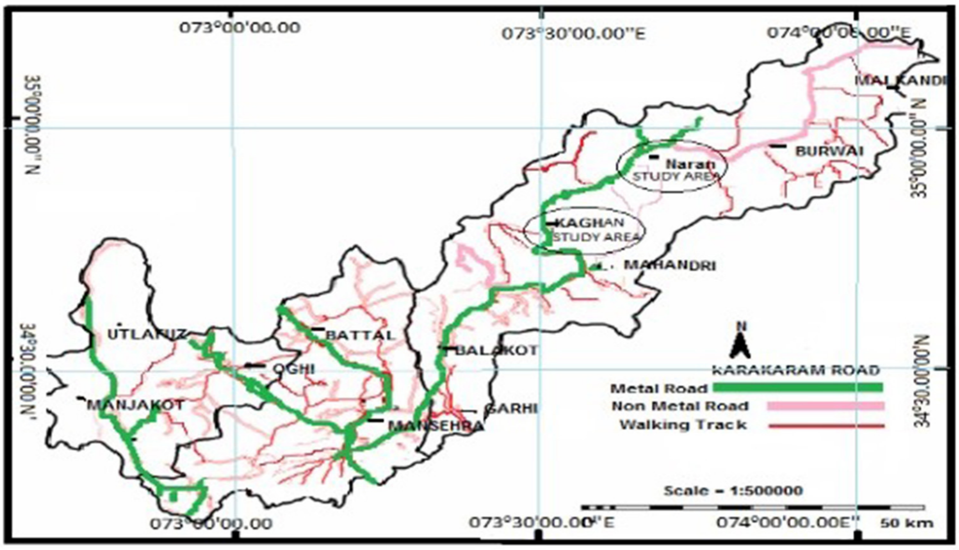

FIGURE 1
A map of the study area depicting sampling localities in circles. Sampling sites for lower Western Himalaya temperate forests (low-level blue pine forests), mid-*Cedrus deodara* forests, Kaghan, and upper-west Himalayan fir forests (upper *Abies pindrow* forests, Naran) are shown in circles.

relate vegetation distribution to topography and soils along an altitudinal gradient and examine some environmental factors for their correlation with the vegetation pattern in our study area.

2. Materials and methods

2.1. Study area

The valley of Kaghan (34.5417° N, 73.3500° E) is located in the mountains of the Western Himalayas, Pakistan with an elevation range of 650–4,170 m a.s.l (Figure 1). The valley is enveloped by the lower Himalayan mountain range, resulting in an alpine climate and the prevalence of pine forests and alpine meadows. The natural range of temperate forests covers an extensive area of the northwestern Himalayas in Pakistan, India, China, Nepal, and Bhutan. The area is located on gneisses and schists, extending over granites, limestones, and quartzite of the Precambrian age (Champion et al., 1965). The region's topography is a mosaic of rocky hills, rock outcrops, wetlands, and glaciated lakes of varying sizes (Hussain and Illahi, 1991).

The environment varies widely along the altitudinal gradient (Table 1). There is a considerable year-to-year climatic variation, especially in rainfall and winter snow accumulation. The overall climate of the study area is classified as temperate continental, with an average frost-free growing period of 204 days from April to October. The average annual precipitation for the area is 1,200 mm, with 65% falling as rain from May to August. However, an appreciable amount is brought in during the winter and spring. On average, precipitation is greater at low altitudes than at high altitudes, while temperatures show the opposite trend (Figures 2, 3).

TABLE 1 Plant families with the highest number of genera and species (dominant).

Family	Number of genera	%	Number of species	%
Asteraceae	14	22.22	16	10.39
Lamiaceae	12	19.05	14	9.09
Rosaceae	09	14.29	12	7.79
Fabaceae	08	12.70	09	5.84
Ranunculaceae	06	9.52	08	5.19
Polygonaceae	05	7.94	09	5.84

In the study area, there are certain dry inner valleys that remain unaffected by the monsoon, leading to the transition to dry temperate forests. In these areas, the snowmelt provides sufficient moisture to sustain the soil during the summer months. Most of the trees and shrubs found in the outer regions of the moist temperate forests do not extend into the inner dry valleys. Overall, the vegetation in these areas undergoes significant changes, warranting a distinct classification of dry and wet temperate forests separately. However, it is important to note that there is no definitive boundary between the moist and dry temperate forests as they gradually transition into one another.

The moist temperate conifer forests in Pakistan (N 34° 38.38′ latitudes and E 73° 33.11′ longitudes) lie within an elevation range of 1,500 to 3,000 m above sea level and receive 400–800 mm of annual rainfall. The topmost characteristic of these forests is the extensive development of coniferous species, which follows a typical structural sequence along the altitudinal gradients. Continuous canopy prevails at lower elevations (close forest zone).

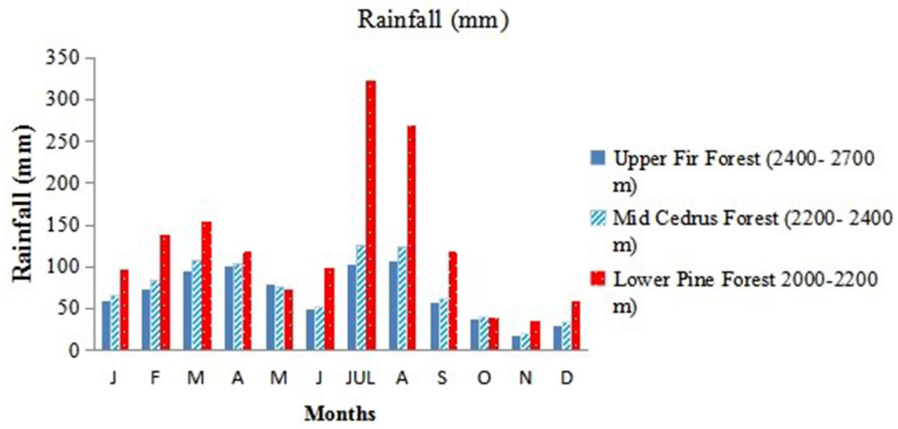

FIGURE 2
Mean monthly rainfall distribution of three forest types: low-level blue pine forests (lower pine forests), mid-*Cedrus deodara* forests (mid-*Cedrus* forests), and upper *Abies pindrow* forest (upper fir forests) along an altitudinal gradient.

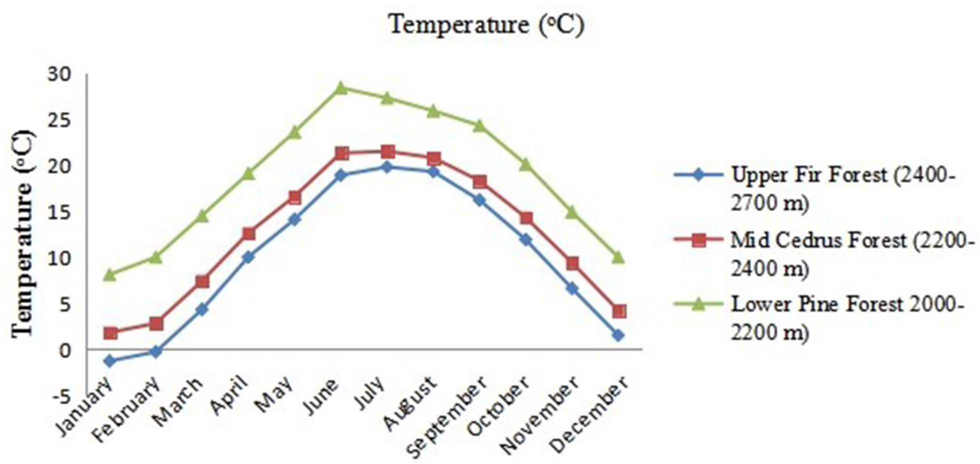

FIGURE 3
The mean monthly temperature of three forest types: low-level blue pine forests (lower pine forests), mid-*Cedrus deodara* forests (mid-*Cedrus* forests), and upper *Abies pindrow* forests (upper fir forests) along an altitudinal gradient.

At an altitude of 3,000 m, the forest canopy becomes fragmented, and the crowns of individual trees generally do not overlap forming woodland zone. Noticeably, this woodland zone was followed by herbaceous pastures (Alpine zone). Commonly, these forests consist of different plant species that exhibits considerable amount of overlap, because these species have specific site preferences like temperature regime and edaphic attributes.

The altitudinal range is from ∼1,371 to 2,743 m (a.s.l), with tree limits varying markedly with aspect and configuration. The transition from moist to dry is gradual, and many forests are intermediate, between the two (transitional/ecotone). It is considered that the abundance of tall shrub *Parrotiopsis jacquemontiana* marks the transition from the moist to the dry type, where *Artemisia brevifolea* tends to replace grass with distinctive grayish foliage (Parker, 1924). The moist temperate forests lie between 2,000 and 2,700 m and are located in the same geographical region (N 34° 38.38′ latitudes and E 73° 33.11′ longitudes). At ∼1,800 m, the tropical pine (*Pinus roxburgii*) forest ends, giving way to a belt of moist temperate forests followed by the subalpine *Krummholz* formation, above which alpine vegetation takes over until the beginning of the permanent snowline (Arnfield, 2023).

2.2. Selection of sites

The study area was subdivided into two major forest zones based on physiognomy and environmental conditions along an elevation gradient. These forest types are designated as lower

Western Himalayan temperate forests, which include (a) low-level blue pine (*Pinus wallichiana*) forests (2,000 m a.s.l.), (b) mid-level moist deodar (*Cedrus deodara*) forests (2,200 m a.s.l.), and (c) upper West Himalayan fir (*Abies pindrow*) forests (2,400 m a.s.l.). After repeated surveys, study sites were marked within each forest located in the Kaghan Valley. Low-level blue pine forest at Paprung (N 34° 37.511′, E 73° 18.311′), mid-level moist deodar forest at Kamal Ban (N 34° 42. 034′, E 073° 31. 232′), and upper Himalayan mixed fir forests at Naran (N 34° 55. 003′) were chosen for investigation. The chosen forests encompass a broad range of altitudes, from the lower close forest (2,000–2,200 m a.s.l) to the upper timberline forest (2,400–2,700 m a.s.l) with a relatively open canopy. The three forests selected during the survey exhibited contrasting environmental conditions. In addition, natural disturbances such as fire, insect infestations, and avalanches are relatively infrequent, as they have been protected from human interference. Nevertheless, illegal collection of litter and grazing by livestock had taken place at most of the sites. The exact magnitude of this influence is difficult to estimate.

2.3. Vegetation sampling

Based on elevation, physiognomy, and principal plant species, the coniferous forest was stratified into the blue pine zone (200 m), the cedar zone (2,200 m), and the *Abies* zone (2,400 m). Fieldwork was carried out from mid-July to mid-August, a time of highest cover for most flowering species. Vegetation sampling was conducted during the peak growing season, July–August 2021–2022. Sampling sites were visited regularly throughout the growing season to sample flowering individuals for identification. Vegetation data were obtained from three forest types, with two sites in each forest [the difference in altitude between the upper and lower sites was ∼100 m (Figure 4)].

Two sites (200 m) and three stands with an interval distance of 100 m were positioned in each forest type. In each stand, four quadrates were sampled. Three topographic positions and eight sampling stands (10 × 10 m) positioned perpendicular to the base of the slope minimized altitude and soil variations within each stand (Figure 4). A total of 144 stands were monitored. The stands were geo-referenced using GPS technology and listed. A total of 288 quadrates were sampled from 144 stands. The size of quadrate was 1 × 1 m^2 for herbs, 5 × 5 m^2 for shrubs and 10 × 10 m^2 for trees. Quantitative phytosociological techniques were used to record each species' density, frequency, and cover. The soil and the presence/absence of herbs and shrubs rooted in the 4 × 4 m quadrat were recorded and identified in the field. Voucher specimens of critical species were collected and identified later by matching the specimens in the Bokhari Memorial Herbarium, BZ University, Multan, Pakistan. Identifications and nomenclature were mainly based on the Flora of Pakistan (Nasir and Ali, 1972).

2.4. Soil sampling

Soil samples (0–15 cm) were randomly collected from each stand at three points and mixed into a composite sample. Pieces of roots, stones, and plant debris, if any, were carefully removed by sieving through a 2-mm mesh. Three subsamples were drawn from this composite sample. The collected 144 samples were oven dried at 105°C for 48 h before analysis. Soil pH was determined in 1 M $CaCl_2$ (7 gm of soil in 0.35 ml) using an HM Digital 10 K pH meter (Allen, 1989; Cramer, 2012). Soil electrical conductivity (EC) was measured on a water extract (7 gm of soil in 0.35 ml) using a CM-30 ET digital EC meter (Cramer, 2012). Soil organic matter (OM) was determined in 1 g of milled soil following the method of Walkley and Black (1934). Available phosphorus (AP) was determined following the Bray II method. Total nitrogen (TN) content was determined using the Kjeldahl method [BUCHI Kjeldahl Line B-324 (Cramer, 2012)], and the concentrations of exchangeable cations were analyzed after extraction with 1 N ammonium acetate at pH 7. Total sodium (TNa) and total potassium (TK) were measured using a flame photometer (Jenway - 500 701- 230 V). Mg^{++} and Ca^{++} were determined using an atomic absorption spectrophotometer, Model- GD-320 N. CO^{-3} and HCO^{-3} were measured using the titration method (Richards, 1954). Ordinal scales assessed the percentage rock cover, litter cover, and canopy openness. The point scale for rock cover ranged from 1 to 4, with 1 = (<75%), 2 = (25–75%), 3 = (>25%), and 4 = absence of rocks or stones. Point scales for litter cover per stand were 1–4, with 1 = (25%), 2 = (25–50%), 3 = (51–75%), and 4 = (>75%). The point scales for forest canopy openness structure also ranged from 1 to 3, with 1 = separated, 2 = side by side, and 3 = overlapped (Suarez et al., 2004).

2.5. Environmental data

We used several environmental variables as explanatory variables that are important for shaping the plant communities in a complex mountain landscape based on previous studies (Hall and Swaine, 1976; Tanner et al., 1992, 1998; Dasti and Angew, 1994; Herbert and Fownes, 1995; Reich et al., 1996; Vitousek and Farrington, 1997; Dasti et al., 2007; Wazir et al., 2008; Saima et al., 2009). Stands were geo-referenced (Supplementary Table S1) using the GPS Navigator (Garmin *e* Trex Legend H, Lenexa, Kansas). Climatic parameters such as temperature (°C), relative humidity (%), heat index (°C), and wind speed (m/s) were recorded at the time of sampling by using Kestral, Japan (4500NV Applied Ballistics Meter). The annual data for all the climatic variables were obtained from the Pakistan Metrological Department.

2.6. Statistical analysis

We used cluster analyses to classify the sampling units based on their similarity. Presence-absence data were used for this analysis using the default option "Farthest neighbor," incorporating the Pearson coefficient. Ordination analyses were conducted to visualize the floristic pattern of vegetation composition in relation to environmental factors. The main emphasis was on classification ordination, which was used in part to check whether the classification results adequately reflect the main floristically gradient in the data set and to detect the relation between

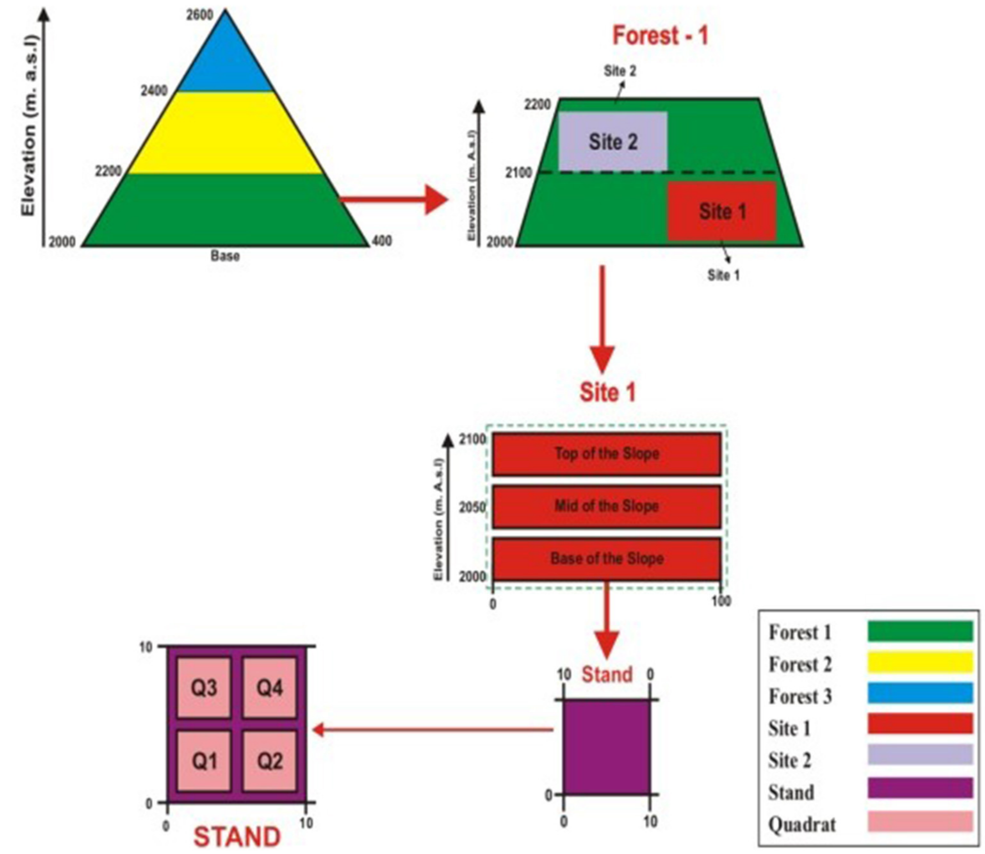

FIGURE 4
Depiction of sampling techniques. Forest 1 = low-level blue pine forest; Forest 2 = mid-*Cedrus deodara* forest; Forest 3 = upper *Abies Pindrow* forest, stand (10 × 10 m) and quadrate (4 × 4 m).

some environmental factors and the composition and structure of vegetation. Species and stand data were ordinated using detrended correspondence analysis (DCA) using the program DECORANA (Dasti and Angew, 1994; Wazir et al., 2008). Ordination axes 1 and 2 were used for data interpretation; scatters of the classification groups from both procedures were plotted on overlays of the ordination to assess the compatibility of the two methods of data simplification (Dasti and Angew, 1994; Wazir et al., 2008). The relationships between altitude, soil characteristics, and DCA axes were determined using the Pearson correlation (Saima et al., 2009). The calculations were made with the help of statistical computer software (MVSP Version 3.1 and MINITAB, Version 15.0). The variation between soil heterogeneity and plant communities was estimated using variance analyses (ANOVA). Duncan multiple range tests were used to detect and compare any significant difference between the means of soil parameters in different communities at the 5% significance level.

3. Results

3.1. Floristic structure and species diversity

A total of 154 plant species belonging to 50 families and 131 genera were recorded. Asteraceae (14), Lamiaceae (12), Rosaceae (09 species), Fabaceae (08 species), Ranunculaceae (06 species), and Polygonaceae (05 species) were found to be the dominant families in these forests (Table 1). The remaining families were represented by less than three genera and three species and thus contributed little to the floristic richness.

3.2. Vegetation structure and composition

Based on the numerical classification, six vegetation types were identified in Himalayan moist temperate forests: *Senecio chrysanthemoides* – *Clinopodium piperitum*; *Artemisia brevifolea* – *Fragaria nubicola*; *Parrotiopsis jacquemontiana* – *Indigofera gerardiana*; *Micromeria* – *Viburnum grandiflorum*; *Siline viscosa* – *Arisaema propinquum*; and *Adiantum venustum* – *Ajuga parviflora* (Table 2). The indicator values for each association are shown in Table 3.

3.2.1. Association A. Senecio chrysanthemoides – Clinopodium piperitum (Group I)

This association is generally found between 2,500 and 2,600 m (a.s.l.) on the mountain slopes, which receive ∼800 mm of

TABLE 2 Normal cluster analysis (NCA) of six recognized vegetation associations.

Associations	Name of association	Plot no.	Altitude (m a.s.l.)	Forest type/habitat
A	*Senecio chrysanthemoides – Clinopodium piperitum*	121–144	2,500–2,600	Upper *Abies pindrow* forest, upper slopes
B	*Artemisia brevifolea – Fragaria nubicola*	97–120	2,400–2,500	Upper *Abies pindrow* forest, lower slopes
C	*Parrotiopsis jacquemontiana – Indigofera gerardiana*	73–96	2,300–2,400	Mid-*Cedrus deodara* forest, upper slopes
D	*Micromeria – Viburnum grandiflorum*	49–72	2,200–2,300	Mid-*Cedrus deodara* forest, mid slopes
E	*Silene viscosa – Arisaema propinquum.*	25–48	2,100–2,200	Low-level *Pinus wallichiana* forests, lower slopes
F	*Adiantum venustum – Ajuga parviflora*	01–24	2,000–2,100	Low-level *Pinus wallichiana* forests, foot slopes

Each plant association has 24 stands (n =24). m a.s.l., meter above sea level.

TABLE 3 Indicator values of six plant associations.

Association	A	B	C	D	E	F
Species name	2,500–2,700	2,400–2,500	2,300–2,400	2,200–2,300	2,100–2,200	2,000–2,100
Senecio -Chrysanthemoides	**24**	•	•	•	•	•
Dryopteris remosa	**19**	13	5	16	15	17
Clinopodium piperitum	**17**	•	•	•	•	•
Viola biflora	**15**	17	4	16	13	17
Plectranthus rugosus	**12**	•	•	•	•	•
Ligularia amplexicaulis	**8**	•	•	•	•	•
Artemisia brevifolea	•	**24**	•	•	•	•
Fragaria nubicola	•	**20**	8	20	17	18
Myosotis alpestris	10	**16**	4	14	10	7
Parrotiopsis jacquemontiana	•	•	**24**	•	•	•
Indigofera gerardiana	•	•	**18**	16	2	4
Sambucus wightiana		1	**15**	12	6	13
Astragalus grahamianus	•	•	**9**	•	•	•
Pilea scripta	•	•	**9**	•	•	•
Micromeria biflora	1		1	**20**	16	16
Vibernum grandiflorum	•	•	5	**18**	17	17
Trifolium repens	6	2	11	**16**	4	7
Polygonum hydropiper	4	12	10	**16**	5	11
Adiantum venustum	8	14		**14**	13	18
Delphinium roylei	•	•		**11**	•	•
Geranium wallichiana	11	13	9	14	**19**	19
Hedera nepalensis	5	10	•	•	**11**	16
Silene viscosa	•	•	•	•	**9**	•
Arisaema propinquum	•	•	•	•	**8**	•
Pimpinella diversifolia	•	•	•	•	•	**3**

Bold values indicating dominant and distinct species.

average annual rainfall. The dominant *Senecio chrysanthemoides* and *Clinopodium piperitum* are perennial herbs. *Spiraea canescens*, *Rosa microphylla*, *Rubus biflorus*, *Lonicera quinquelocularis*, and *Jasminum humile* are important shrubs. Various ferns such as *Dryopteris remosa*, *Athyrium rupicola*, and *Adiantum venustum* are among the first to develop in late spring after the snow melts. The important climbers include *Clematis montana* and *Hedera nepalensis* (Table 3).

3.2.2. Association B. Artemisia brevifolea – Fragaria nubicola (Group II)

This association colony occupies the altitudinal range between 2,400 and 2,500 m (a.s.l.) and is distinctive in the combination of a high coverage of strongly aromatic, shrubby *Artemisia brevifolea* and procumbent, white-flowered *Fragaria nubicola* along with erect, shiny blue-flowered *Myosotus alpestris*. The former species dominate the open, dry stony, while the latter is widely distributed in forests and shrubberies. The dominance of the species can be ranked in the order *Artemisia brevifolea* > *Fragaria nubicola* > *Myosotus alpestris* (Table 3).

3.2.3. Association C. Parrotiopsis jacquemontiana – Indigofera gerardiana (Group III)

This association occurs at an altitude range of 2,300–2,400 m (a.s.l), and the diagnostic species of this association is *Parrotiopsis jacquemontiana*. The abundance of this species marks the transition from moist to dry forests, where *Artemisia* tends to dominate. The dominance of *Indigofera gerardiana* and *Sambucus wightiana* and the presence of *Sorbaria tomentosa* and *Spiraea vaccinifolia* indicate an ample moisture regime in this vegetation type (Table 3).

3.2.4. Association D. Micromeria – Viburnum grandiflorum (Group IV)

The component species of this association is distributed over the altitudinal ranges of 2,100–2,200 m (a.s.l.), corresponding to the lower part of the altitudinal range of *Cedrus deodara*. The dominant species of this association are *Vibernum grandiflorum*, *Micromeria biflora*, and blue-flowered *Delphinium roylei*. Among the component shrubs, *Vibernum grandiflorum* and *Indigofera gerardiana* are most common on stony pavements with shallow soils. *Aster falconeri*, *Geranium lucidium*, and *Trifolium repens* are common herbs found on the level of gently sloping wet meadows and pastures (Table 3).

3.2.5. Association E. Siline viscosa – Arisaema propinquum (Group V)

This association occupies the wetland sites of the valley highlands ranging from 2,200 to 2,300 m (a.s.l.) and is characterized by small white-flowered *Silene viscosa* and *Arisaema propinquum*. *Geranium wallichianum* assumes the leading role in the flora dominating the association. Among the shrubs, *Barbaris lyceum*, *Indigofera gerardiana*, and *Vibernum grandiflorum* contributes the most to this association. Among the ferns, *Dryopteris remosa* and *Adiantum venustum* occupy the micro-relives distributed along the wet, stony slopes. Among the component species, *Fragaria indica*, *Geranium wallichiana*, and *Micromeria biflora* are dominant on lower slopes, and *Silene viscosa and Viola biflora* are frequent on mid-slopes. At the same time, *Vibernum grandiflorum* is recorded at its maximum at the top of the slopes (Table 3).

TABLE 4 Pearson correlation coefficient between DCA axis scores of soil parameters and altitudes of three forest types (located along the altitudinal range 2,000–2,700-m a.s.l.).

Parameters	Coefficient axis 1	Coefficient axis 2
Altitude	0.673***	−0.486***
pH	−0.257**	0.051 NS
Electrical conductivity dS/m	−0.580***	0.324***
Organic matter %	−0.009 NS	0.043 NS
Nitrogen ppm	−0.731***	0.268**
Potassium ppm	−0.412***	0.312***
Calcium ppm	−0.431 ***	0.349***
Magnesium ppm	−0.358***	0.310***
Sodium ppm	−0.686***	0.380***
Chloride ppm	−0.159*	0.311***
Carbonates ppm	−0.385***	0.294***
Bicarbonates ppm	−0.432***	0.18**
Relative humidity %	0.077 NS	−0.048 NS
Heat index °C	−0.177 N	−0.020 NS
Average temperature °C	−0.195 NS	0.313 NS
Rainfall mm	−0.400***	0.587***
Winter snowfall cm	−0.748***	0.077***
Rock cover %	0.619***	−0.406***
Wind speed m/s	0.461***	−0.703***
Litter cover %	0.619***	−0.406***
Canopy openness	0.619***	−0.406***

*NS, non-significant, *,**, significant, ***, highly significant.
All the observed parameters are significant at a p-value of <0.001.

3.2.6. Association F. Adiantum venustum – Ajuga parviflora (Group VI)

This association is distributed in the low-level blue pine forest zone from 2,000 to 2,100 m (a.s.l.) and is characterized by *Pimpinella diversifolia* and *Pteracanthus urticifolia*, which were altogether absent from the rest of the study area. *Adiantum venustum*, *Ajuga parviflora*, and *Geranium tuberaria* are the conspicuous species, while *Primula rosea* and *Sambucus wightiana* are significantly frequent. Among shrubs, *Indigofera gerardiana*, *Lavatera kashimiriana*, and *Lonicera quinquelocularis* contribute sufficiently to community composition (Table 3).

3.3. Environmental determinants

In addition to the overriding importance of altitude in determining species distribution and shaping plant communities in mountain landscapes, most of the edaphic factors included have significant correlations with the first axis (Table 4). The importance of soil EC, K, Ca, Mg, Na, and N was notable. All these cations were negatively correlated with DCA axis 1 but were mostly much lower in values except EC, Na, and P,

TABLE 5 The mean values and standard deviation for all the soil variables for identified six associations.

Variables	Associations (m a.s.l.)						*D*
	A	B	C	D	E	F	
	(2,500–2,700)	2,400–2,500	2,300–2,400	2,200–2,300	2,100–2,200	2,000–2,100	
Soil pH	6.10^{a}	6.59^{c}	6.34^{b}	6.42^{b}	6.13^{a}	6.69^{d}	0.14***
	0.09	0.57	0.27	0.20	0.10	0.42	
Soil EC (dS/m)	1.33^{a}	2.15^{c}	1.80^{b}	2.69^{d}	2.08^{c}	2.69^{d}	0.13***
	0.13	0.33	0.58	0.12	0.25	0.06	
Organic matter %	2.31^{b}	2.59^{c}	2.45^{b}	2.40^{b}	2.51^{c}	2.05^{a}	0.14***
	0.37	0.33	0.38	0.46	0.28	0.05	
Nitrogen ppm	0.13^{a}	0.14^{a}	0.12^{a}	0.17^{a}	0.12^{a}	0.23^{c}	0.05***
	0.00	0.02	0.00	0.15	0.00	0.28	
Phosphorus ppm	19.35^{a}	24.13^{c}	23.34^{b}	23.65^{b}	23.42^{b}	25.91^{d}	0.43***
	1.23	0.82	0.78	0.68	1.03	1.16	
Potassium ppm	1.17^{a}	1.60^{c}	1.31^{b}	1.61^{c}	1.32^{b}	1.73^{c}	0.24***
	0.04	0.26	0.14	0.24	0.16	0.09	
Calcium ppm	0.25^{a}	0.49^{d}	0.35^{b}	0.40^{c}	0.39^{b}	0.52^{d}	0.04***
	0.04	0.09	0.12	0.16	0.02	0.06	
Magnesium ppm	0.16^{a}	0.22^{c}	0.19^{b}	0.22^{c}	0.18^{b}	0.26^{d}	0.01***
	0.02	0.03	0.02	0.02	0.03	0.03	
Sodium ppm	0.31^{a}	5.30^{b}	5.64^{b}	8.89^{c}	9.08^{c}	11.57^{d}	0.98***
	0.07	3.24	3.61	1.51	1.62	0.84	
Chloride ppm	0.38^{a}	0.40^{a}	0.37^{a}	0.41^{b}	0.36^{a}	0.55^{c}	0.02***
	0.03	0.05	0.05	0.06	0.07	0.05	
Carbonates ppm	2.01^{a}	2.40^{a}	2.78^{a}	3.26^{b}	3.49^{c}	4.63^{d}	0.42***
	0.84	1.13	1.12	1.15	0.86	0.33	
Bicarbonates ppm	11.38^{a}	11.38^{a}	34.65^{b}	36.78^{b}	30.00^{b}	62.21^{c}	7.43***
	1.35	1.35	21.94	31.51	7.91	11.25	
Relative humidity %	45.51^{b}	65.96^{c}	46.92^{b}	48.09^{b}	31.08^{a}	44.30^{b}	5.29***
	10.72	4.65	16.98	19.57	3.41	5.34	
Heat index °C	20.77^{b}	15.00^{a}	24.29^{c}	21.18^{b}	30.37^{d}	21.97^{b}	2.51***
	2.64	2.85	8.76	8.15	5.03	2.57	

m a.s.l., meter above sea level; ***, highly significant; *d*, Duncan value. Superscript alphabets indicating LSD = Least Significant Difference between the means at the required level of probability.

whose *r*- values exceeded 0.4. Regarding axis 2, the *r*-values of these factors were considerably low ($r < 0.3$). Most of the soil cations and soil EC exhibited a positive correlation, with axis 2 suggesting cation-rich soil conditions from the top to the bottom of the gradient. All chemical variables measured in the soil samples showed broad variations at the landscape scale (Tables 4, 5). There was a general increase in the concentration of all the solutes in the sequence from the upper *Abies pindrow* forest to the mid-*Cedrus deodara* forest to the low-level *Pinus wallichiana* forest.

3.4. Ordination

To identify a minimal set of geo-climatic variables and to know the relative importance of these factors, stands and species ordination (DCA) was performed, using geo-climatic parameters as response variables. The DECORANA (DCA) ordination results were used to plot the scatters of classificatory results, as shown in Figures 5, 6. DECORANA (axes 1 and 2) showed a plot of 144 stands from three forest types. Stands are plotted individually, and zones are shown where

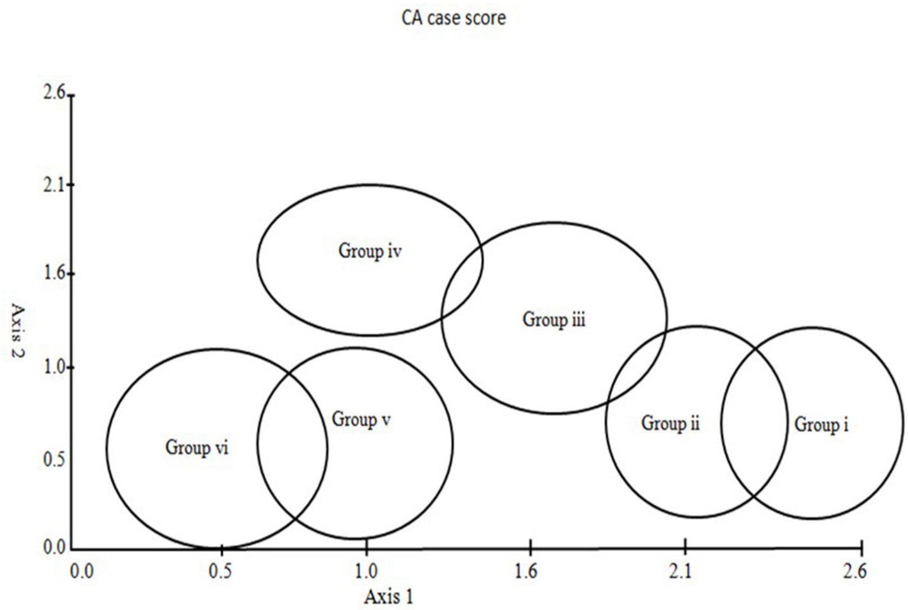

FIGURE 5
Sampling stand distribution in different communities exhibiting association delineated by ordination.

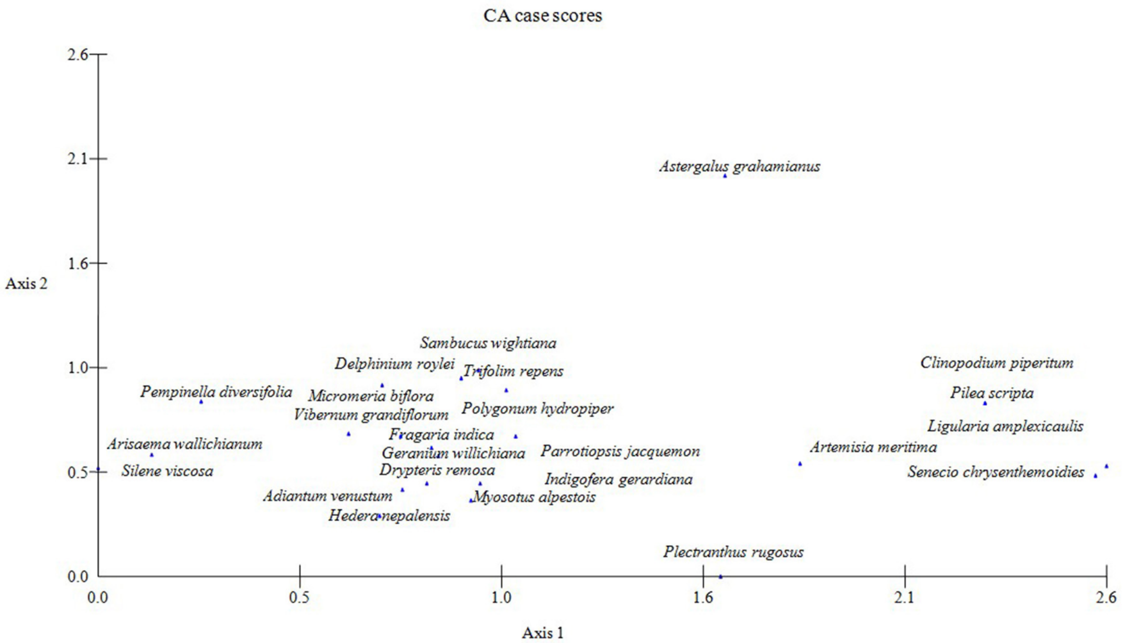

FIGURE 6
Detrended correspondence analysis (DCA) species ordination of the qualitative data of three temperate forests regarding ordination indicates species distribution along the elevation gradient (*Abies Pindrow* forest, *Cedrus Deodara* forest, and low-level *Pinus Wallichiana* forest).

the NCA segregates each of the six associations, as indicated in Figure 5. The eigenvalues of axes 1 and 2 were very large compared to the lower-order axis, and the latter were ignored (Figure 6).

3.5. Correlation of the ordination axes

The stands belonging to associations (A and B) delineated from the upper *Abies pindrow* forest were grouped at one end (high scores) and those of low-level *Pinus wallichiana* forest (associations E and F) on the other end (low scores) of the DCA axis 1. Stands belonging to the associations C and D identified in the mid-*Cedrus deodara* forest occupy the intermediate position along axis 1, reflecting the mid-elevations of the study area (Figure 5). The results depicted in Figure 6 confirm the overlapping nature of the associations in space defined by the ordination axes. The first ordination axis represents an elevation gradient, with the highest elevation on the right (high scores) and the lowest on the left (low scores). Within the first axes, species such as *Senecio chrysanthemoides*, *Ligularia amplexicaulis*, *Clinopodium piperitum*, and *Artemisia brevifolea* lie at the most elevated end of axis 1. At the same time, *Pimpinella diversifolia*, *Arisaema propinquum*, and *Silene viscosa* are confined to the lowest elevation end of the gradient. The diagram's midpoint is occupied by *Parrotiopsis jacquemontiana*, *Sambucus wightiana*, *Micromeria biflora*, *Trifolium repens*, and *Delphinium roylei*. Similarly, on axis 2, *Plectranthus rugosus* and *Hedera nepalensis* are at the lower end of the axis, while *Sambucus wightiana* and *Trifolium repens* are at the opposite end. The second axis represents increasing soil cations from the top to the bottom.

4. Discussion

Low-level *Pinus wallichiana* forest is located between 2,000 and 2,200 m (a.s.l); in this forest type, there is no definite evidence of basic changes through human activities. The forest under consideration is exclusive of blue pine with a small admixture of spruce (*Picea smithiana*), deodar (*Cedrus Deodara*), and fir (*Abies Pindrow*). The top canopy is more or less continuous. The mid-level moist deodar (*Cedrus deodara)* forest is nearly pure, but some spruce is also present in patches. Deodar extends into the inner drier ranges of the Himalayas, ultimately becoming sufficiently different to be described as dry deodar. Upper-west Himalayan fir and mixed broad-leaved forest represent the subalpine forests where regeneration of *Abies pindrow* is usually fairly abundant. The general climate of this forest is dry and temperate; however, soil moisture is adequate, which makes it difficult to distinguish it from most temperate forests.

The extensive growth of *Artemisia brevifolea* distinguishes the upper fir forests from the lower coniferous belt. Champion et al. (1965) and Hussain and Illahi (1991) described these forest types completely. The temperature, relative humidity, and wind speed were significantly related to DCA axes 1 and 2. Both the ordination axes are significantly influenced by altitude and the environmental queues generated by it, which indicated some important implications for these factors in determining vegetation attributes. For example, low species richness in associations belonging to upper forests may be attributed to increasingly strong winds, reduced precipitation, reduced temperature, increased soil acidity, and a reduced supply of nutrients with increasing altitude. These results confirm those of Pendry (1994).

Soil pH usually falls with altitude, and the most acidic soils are generally found in upper fir forests. These findings are consistent with those of Pendry and Proctor (1996) but contradict the results of Veneklaas (1991), who reported a higher pH of soils in upper forests than soils in nearby lower forests. Soil organic matter increased with altitude, partly because of reduced decomposition at lower temperatures but often due to temporary waterlogged conditions created by melting snow in the upper mountains. Extractable phosphorus is also low at higher altitudes. The elevational reduction in phosphorus might be partially due to a high concentration of organic matter, which reduced the potential of phosphorus fixation (Tanner et al., 1998). Differences in the concentration of cations among the three forests are noted. These differences may be largely attributed to the parent material (Pendry and Proctor, 1996), but the importance of the downslope movement of water cannot be ignored (Dasti et al., 2007, 2010; Wazir et al., 2008). The study demonstrates significant associations between the landscape-level patterns of environmental heterogeneity and the distribution patterns of plant communities and species across mountain landscapes. Two factors seem to be significant: altitude and substrate. The overriding importance of altitude as an environmental factor affecting plant species association is not surprising, considering its close correlation with temperature and rainfall (Danin et al., 1975; Dasti and Malik, 1998; Dasti et al., 2007; Wazir et al., 2008; Saima et al., 2009). The analysis and assessment of patterns and the zonation along the first axis suggest that the most important environmental gradients and boundaries across the landscape are associated with the downslope movement of water, soil chemistry, and organic matter. It is difficult to assess the relative importance of these factors in comparison with altitude. Still, the consistent negative interaction terms between these factors suggest that some combined effect is significant. The reasons for these correlations might be due to variations in runoff patterns.

The spatial distribution of soil moisture might correlate with rainfall in lower blue pine valleys, which receive an appreciable amount of runoff. However, the situation is quite different in the upper rocky mountains with patches of soil. Here, soil moisture distribution depends on the ratio of rock surface to soil volume (Olsvig-Whittaker et al., 1983; Dasti and Malik, 1998; Wazir et al., 2008). These patterns seem best explained by interactions between soil depth and the downslope redistribution of runoff. The lower forests gain water from run-on, while the upper forests lose water from run-off. This run-off will be distributed along a moisture gradient, along which vegetation change will be gradual. Besides this factor, snow accumulation and its gradual melting are of considerable importance in determining the vegetation type at high elevations. Apart from the differences in moisture regime, the topographically induced variation in soil EC plays a significant role in community composition and plant assemblage.

The species belonging to associations located at lower elevations occur on soil with relatively high EC, which decreases with an increase in altitude. Thus, the altitudinal floristic differences may partially be attributed to soil EC. The decrease in EC with increased altitude is largely due to the downward movement of nutrients, which affects spatial distribution along the altitudinal transect (Pendry and Proctor, 1996; Dasti and Malik, 1998; Raja et al., 2014). The correlation of soil pH with the distribution and association of plant species is not surprising but has already been reported in several studies (Pendry and Proctor, 1996). We have demonstrated that the topographically induced edaphic patchiness governs the distribution of the species across the landscape. A clear relationship between the characteristics of the substrate and associations defined by numerical analysis shows that these groupings of plant species are not arbitrary assemblages.

Differences in soil within the forest are distinct enough to favor different species at different locations and thus create numerous floristically differentiated forest patches. These findings confirm the edaphic patchiness model rather than the random walk or regional homogeneity model (Tuomisto et al., 2003). In this respect, the spatial pattern of environmental conditions along the elevation gradient appears particularly important to plant communities' composition and distribution. Altitude and topography govern the properties of the soil, which are of overriding importance in determining the variations in floristic composition. Sharp species turnover occurs when the environmental conditions change, and different (dominant) species are found at sites with different environmental conditions. These findings are consistent with the findings of Lieberman et al. (1985) and Clark et al. (1998, 1999). The dominance of only one plant species at most sites was obvious in the field, and this is a notable feature of the vegetation of Himalayan moist temperate forests. However, a few associations, defined by cluster analysis and confirmed by DCA, showed that some species were restricted to the upper forests and others to the low or mid forests. Still, several have distribution ranges that extend into all associations.

The visual representation of the results resemble the continuous nature of the vegetation more than the grouping produced by classification. However, describing the salient features of associations between stands and species with ordination is easy and worthwhile. The scatters of the DCA demonstrate the overlapping rather than discrete boundaries between the vegetation types and species distributions. This overlapping nature of the vegetation types might be due to the species overlapping environmental preferences. The substrate of all these communities is found through the interaction of topography and related basic geomorphological processes of surface movement through wind and downslope movement of water, which are expected to bring about arbitrary changes in dominance within the association. Finally, the altitude limits the vegetation observed along an altitudinal gradient. For example, the dominance of *Senecio chrysanthemoides* and *Clinopodium piperitum* is restricted to high altitudes (2,400–2,700 m a.s.l). These species were absent from altitudes below 2,400 m a.s.l. Similarly, the species dominating the low forest (*Silene viscosa* and *Arisaema propinquum*) were not found in the upper fir forest zone.

5. Conclusion

The present study described the vegetation patterns and community composition in three different forest zones along the altitudinal gradient. Our findings revealed that these forest types significantly differ physiognomically and structurally. The special nature of the mountain landscape, with its variable altitude and topographically induced environmental variations, is the main factor that determines the prominent vegetation zones in these mountain forests. The soil's physical and chemical properties appear to be the most highly influential factors in the distribution and diversity of species along the altitude. The present investigation showed that the distribution of community types, species composition, and diversity are better understood by studying geo-climatic factors along the elevational gradient on the mountain landscape. Based on the comprehensive botanical survey and early descriptive accounts of the study area, the vegetation in the temperate forest zone was stratified into three physiognomic units: (a) low-level *Pinus wallichiana* forest, (b) mid-*Cedrus deodara* forest, and (c) upper *Abies pindrow* forest. As Himalayan moist temperate forests exhibit a rather stable floristic composition and slow species dynamics, the different years in which the vegetation was sampled were not considered a serious methodological problem.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary material, further inquiries can be directed to the corresponding authors.

Author contributions

MH, EB, AD, MK, TF, and AAb contributed to the study's concept, design, and statistical analysis. MK, SM, MH, AAs, and AD contributed to the analysis and interpretation of the data. EB, MK, and HA contributed to the investigation and resources. MH, AAb, HA, and ZA contributed to the drafting of the manuscript. SM, EB, TF, AAs, MK, and AD contributed to this manuscript's review, editing, and proofreading. HA contributed to funding acquisition and study supervision. All authors contributed to the article and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2023.1195491/full#supplementary-material

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Factors influencing tree biomass and carbon stock in the Western Himalayas, India

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The assessment of tree biomass and its carbon (C) stock at the local and regional level is considered a crucial criterion for understanding the impact of changing environments on the global carbon cycle. In this context, we selected three sites in the western Himalayas, covering parts of Himachal Pradesh and north-eastern Haryana. Each study site experiences distinct climatic conditions, vegetation types, and elevations. We seek to elucidate the determinants of tree biomass and carbon stock across different forest types in the Western Himalayas. We found that temperate forests contributed the most biomass and carbon stock, with *Cedrus deodara* attaining the highest values of 782.6 ± 107.9 Mg/ha and 360 ± 49.7 Mg C/ha. In contrast, *Quercus leucotrichophora* mixed temperate had the lowest 286.6 ± 57.2 and 128.9 ± 25.7 Mg/C ha, respectively. Only a few species, such as *Abies pindrow*, *Cedrus deodara*, *Quercus floribunda*, and *Quercus semecarpifolia*, accounted for significant biomass and carbon stock. The lower elevation subtropical forests had the highest species richness (8–12 species) and stem density (558.3 ± 62.9 to 866.6 ± 57.7 trees/ha). Furthermore, tree diameter, total basal cover, and height emerged as the strongest predictors of biomass and C stock. The remaining variables showed no significant associations, including species diversity, climatic attributes and elevation. Thus, our study extended the assertion that vegetation composition and structural attributes, apart from climatic and topographic factors, are equally important in determining biomass and C stock in forest ecosystems. Our study indicated that the temperate forests in the western Himalayas possess significant carbon storage and climate change mitigation potential.

KEYWORDS

carbon stock, climate change, Western Himalayas, forest types, structural attributes

1 Introduction

Anthropogenic activities have raised the earth's temperature by 1°C beyond pre-industrial levels, and this is expected to climb to 1.5°C by 2,052 if current emission rates continue (IPCC, 2018). The 2°C of global warming is anticipated to negatively influence livelihood, food security, health, and biodiversity (Smith et al., 2018). Therefore, countries have set targets to keep global temperatures below 2°C within the framework of the Paris Agreement. Furthermore, the pact requires countries to reach carbon (C) neutrality (zero C emissions) by the second half of this century (UNFCCC, 2015). As a result, mitigation techniques focus primarily on CO_2 removal from the atmosphere and its secure storage. In this context, forests provide a viable solution because they cover around 31% of the earth's surface area (FAO, 2022) and sequester 15–20% of annual human C emissions (Le Quéré et al., 2018; Case et al., 2021). Furthermore, around 80% of Earth's total plant biomass is confined to forests (Kindermann et al., 2008) and they hold a more significant amount of carbon in their biomass and soil than stored in the atmosphere (Pan et al., 2013). Contrastingly, tropical forests have the highest C storage capacity (471 Pg C), while boreal and temperate forests have 272 Pg C and 119 Pg C, respectively (Pan et al., 2011). Most carbon is stored in aboveground biomass (AGB) components in tropical forests, whereas C is limited to belowground, primarily soils, in boreal and temperate forests (Malhi et al., 1999). Given their high C sequestration potential, the assessment of biomass and C stock inventories at local or regional scale is crucial for understanding the contribution of forests to the global carbon cycle, particularly in the context of changing climatic conditions (Gibbs et al., 2007; Huynh et al., 2023). Furthermore, this information is valuable in attaining the objectives of global obligations such as the "Reduction of Emissions from Deforestation and Forest Degradation (REDD)" initiative, which aims to offset forest loss and earn carbon credits (Lung and Espira, 2015; Sahoo et al., 2021).

The carbon pool in forests varies widely on a regional and global scale. Various factors, including vegetation composition, structural attributes, topography, climatic conditions, disturbances, and stand age, are attributed to this variation (Pregitzer and Euskirchen, 2004; Islam et al., 2017; Kothandaraman et al., 2020; Gogoi et al., 2022). Tree C storage, especially aboveground, is a function of various structural attributes, such as stem density, mean tree diameter (Poorter et al., 2015; Islam et al., 2017), height (Moles et al., 2009), and basal area (Mensah et al., 2016, 2020). The role of DBH in predicting C stock is more pronounced in temperate forests, as a significant fraction of the AGB (40%) is comprised of large diameter (>60 cm) trees (Lutz et al., 2018). Apart from this, species diversity influences C stock in many ways as positive (Mensah et al., 2016; Lie et al., 2018; Kaushal and Baishya, 2021), negative (Jerzy and Anna, 2007), and no relationships (Khanalizadeh et al., 2023; Pinto et al., 2023) have been observed globally.

In addition to these, environmental variables also play a crucial role in forest C stock. For example, topographic elements (elevation, slope and aspect) create microclimatic conditions and regulate soil moisture, light availability and vegetation patterns, ultimately determining biomass. Among topographic elements, elevation is the most studied factor (Singh, 2018; Cheng et al., 2023) due to its role in vegetation patterns and productivity through temperature and precipitation effects (Xu et al., 2017; Sanaei et al., 2018). Studies have shown that climatic variables (temperature and precipitation) influence forest biomass through direct and indirect effects on species diversity (Stegen et al., 2011; Mensah et al., 2023).

The Indian Himalayan region (IHR), a biodiversity hotspot, is home to varied species of flora and fauna and provides numerous ecosystem services (Negi et al., 2019; Ahirwal et al., 2021) such as carbon sequestration, water regulation and livelihood to a million of people. Considering its vast natural wealth and unique environmental conditions, IHR is reported to sequester 65 million tonnes of carbon annually (Tolangay and Moktan, 2020) and possess more significant climate change mitigation potential. In this context, the Central and Western Himalaya has been investigated extensively for biomass and carbon stock estimation (Sharma et al., 2010, 2016, 2018; Gairola et al., 2011; Dar and Sundarapandian, 2015; Dar et al., 2017; Kaushal and Baishya, 2021; Dar and Parthasarathy, 2022; Haq et al., 2022; Tiwari et al., 2023). Meanwhile, in terms of factors influencing C storage, Himachal Pradesh and the bordering Siwalik ranges in the Western Himalayas are less explored. Despite the limited number of studies (Nagar, 2012; Banday et al., 2017; Chisanga et al., 2018; Singh and Verma, 2018; Bhardwaj et al., 2021; Kumari et al., 2022), the region still lacks a thorough understanding of the variables impacting biomass and C stock development over a wide spatial scale. Therefore, the current research aims to understand the carbon stock dynamics in different forest ecosystems, each with contrasting climatic conditions and elevations. Furthermore, due to the inherent vulnerability of the Himalayan region and its significant degree of disturbance, we have chosen three designated protected areas (Wildlife Sanctuaries) as the focal sites for our investigation. Because protected sites bear high species richness and offer multiple ecosystem services including C sequestration (Collins and Mitchard, 2017). A recent study showed that approximately 26% of terrestrial woody C is present in AGC stock of protected areas (Duncanson et al., 2023). The current investigation addresses the following questions: (1) What is the tree biomass and carbon stock status across the studied region? (2) What are the determinants of tree biomass and carbon stock? (3) How do vegetation types and forest attributes influence biomass and C stock?

2 Materials and methods

2.1 Study sites

The study sites are located in the lesser Himalayan region of Himachal Pradesh (H.P.) (32.1024° N, 77.5619° E) and the Siwalik region of Haryana (29.0588° N, 76.0856° E) states, which are situated in the north-western part of India (**Table 1**; **Figure 1**). The chosen study sites are part of three Wildlife Sanctuaries (WLS), specifically Khol Hi-Raitan, commonly referred to as Morni Hills (hereafter KHR) (300–800 m), Chail WLS (900–2,100 m), and Churdhar WLS (1,900–3,600 m) (**Figure 1**). The former falls within the jurisdiction of the Haryana State Forest Department, while the latter two are under the control of the H.P. Wildlife Department. These study sites displayed considerable heterogeneity in climatic conditions, topographic features, and elevation levels (**Figure 2**). The KHR WLS experiences a subtropical monsoonal climate characterized by seasonal patterns of hot summers, wet monsoons, and cold winters. The annual temperature ranges from 3°C during the winter season to 44°C during the summer season. The annual precipitation exhibited an average of 1,200 mm, with most rainfall from July to September. The Chail WLS represents a transition zone between subtropical and temperate environments. It has a subtropical climate at lower elevations and a temperate one at higher reaches. The annual precipitation averaged 1,700 mm during the monsoon season (July to September). The maximum temperature can reach 35°C in summer, whereas the minimum temperature can drop below 0°C in winter. The Churdhar WLS spans across a wide elevation range of 1,900–3,600 m with temperate humid climates. It experiences cool, pleasant summer

TABLE 1 An overview of selected study sites in the Western Himalayas.

Forest type/ abbreviation	Forest class*	Latitude	Longitude	Mean elevation (m)	Locality	Dominant/associated vegetation
Anogeissus latifolia dominated stand (ALD)	–	30.68816	76.92409	500	KHR WLS (Panchkula, Haryana)	*Anogeissus latifolia, Acacia leucophloea, Acacia catechu, Ziziphus mauritiana, Lannea coromandelica*
Lower Siwalik dry deciduous (LSDD)	Northern Dry Mixed Deciduous Forests (5B/C2)	30.69857	76.93636	650	KHR WLS (Panchkula, Haryana)	*Acacia modesta, Mallotus philippensis, Grewia asiatica, Tectona grandis, Randia tetrasperma*
Quercus leucotrichophora -mixed temperate (QLMT)	Ban Oak (*Quercus leucotrichophora*) Forests (12/C1a)	30.73103	77.00824	2,014	Chail WLS (Solan, H.P.)	*Quercus leucotrichophora, Rhododendron arboreum, Pinus roxburghii, Pyrus pashia*
Cedrus deodara pure stand (CDP)	Moist Deodar (*Cedrus deodara*) Forests (12/C1c)	30.95179	77.20097	2,080	Chail WLS (Solan, H.P.)	*Cedrus deodara, R. arboreum, Q. leucotrichophora, Daphne papyracea*
Quercus floribunda – mixed (QFM)	Moru Oak (*Quercus floribunda*) Forests (12/C1b)	30.96104	77.19838	2,753	Churdhar WLS (Sirmaur, H.P.)	*Q. floribunda, Abies pindrow, Pinus wallichiana*
Abies pindrow dominated stand (APD)	Upper West Himalayan fir (*Abies pindrow*) forests (12/C2b)	30.97661	77.19857	2,965	Churdhar WLS (Sirmaur, H.P.)	*Abies pindrow, Q. semecarpifolia, Prunus cornuta*
Quercus semecarpifolia-Abies mixed (QSAM)	Kharsu Oak (*Quercus semecarpifolia*) Forests (12/C2a)	30.89442	77.48392	3,163	Churdhar WLS (Sirmaur, H.P.)	*Q. semecarpifolia, Abies pindrow, Picea smithiana, Sorbaria tomentosa*
Abies spectabilis pure stand (ASP)	West Himalayan Fir Forest (14/CIb)	30.83619	77.4369	3,235	Churdhar WLS (Sirmaur, H.P.)	*Abies spectabilis*

*Forest classification as per Champion and Seth (1968).

weather, where the temperature hardly exceeds 23°C, whereas it drops below freezing during winter. The study area receives annual precipitation averaging 1,600 mm, typically in rainfall and snow. The higher reaches of Churdhar WLS remain snow-covered from early December to March.

2.2 Sampling design and data collection

A preliminary survey was undertaken in 2020-21 across the proposed sampling sites to evaluate various factors, including vegetation types, elevation, and topography. The vegetation at the three study sites was categorized into eight forest types (FT) based on visual observation, following the classification of Indian forests by Champion and Seth (1968) (**Table 1**; **Supplementary Figure 3**). Tree inventory was conducted using square plots of 31.6 m × 31.6 m (equivalent to 0.1 hectares) (FSI, 2002). A total of 3 plots (0.3 ha) were established at each forest type. However, in a few forests at KHR, where the topography exhibited undulating or steep slopes, a plot measuring 10 m × 10 m was chosen as the optimal size. Plot selection was based on considering the threat of disturbance, as forests near human settlements or the periphery of WLS were avoided. To investigate the impact of elevation on stand structural metrics, a minimum distance of 100 meters was maintained between consecutive plots, if feasible. Geographical coordinates and elevation data were collected via a portable Global Positioning System (GPS) device (Garmin eTrex 10 model). The identification of trees that fell within each sampling plot was done with established sources such as the Flora of Himachal Pradesh (Chowdhery and Wadhwa, 1984), the Floristic reconnaissance of Churdhar Wildlife Sanctuary (Choudhary and Lee, 2012; Subramani et al., 2014), and the Flora of Haryana (Kumar, 2001). In each sampling plot, all trees with a girth of more than 10 cm were measured for circumference over bark at a height of 1.37 m. The measured values were then converted to diameter by dividing the circumference by 3.14. The tree height was measured using a BLUME-LEISS altimeter. The mean diameter of trees at breast height (DBH) was used to calculate tree basal area. The tree basal cover within each plot was summed up to get the total basal cover (TBC), expressed as basal cover/hectare.

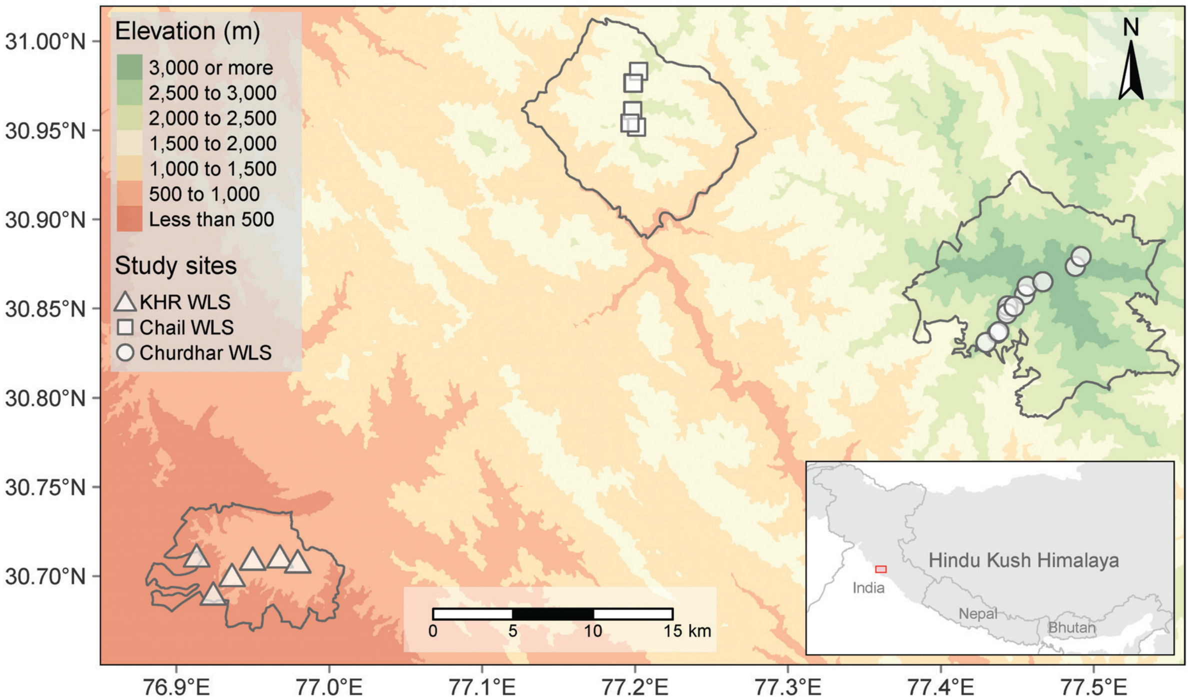

FIGURE 1
A location map of the study area depicting sampling points within each study site.

The stem density or tree density (SD) represents the number of tree individual per area is expressed as individuals/hectare. Species richness was determined using Margalef's richness index (SR) = S-1/ln N (Margalef, 1958), where S = total number of species, ln = natural log and N = total number of individuals. For the determination of species diversity, the Shannon-Weiner (H') index was used: $H' = -\sum n_i/N \ln n_i/N$ (Shannon and Weaver, 1949), where n_i represents the number of individuals of the ith species, and N is the number of individuals of all species in the population. The climate data including mean annual temperature (MAT) and total annual precipitation (MAP) for the study sites was accessed at 30-arc-sec (~1 km) resolution from the CHELSA[1] database version 2.1 (Karger et al., 2017). The values of these variables were extracted for a particular sampling plot using the *R* package *terra* version 1.7.39 (Hijmans, 2023).

2.3 Estimation of tree biomass and carbon stock

A non-destructive approach based on allometric equations was used to estimate aboveground tree biomass. Firstly, the volume of individual tree species was computed using species-specific volumetric equations developed by the Forest Survey of India (FSI, 1996; **Supplementary Table 1**), and the volume of all the tree species in a plot was summed up to get the growing stock volume density (GSVD m^3 ha). To get aboveground biomass (AGB), the GSVD was multiplied with the appropriate biomass expansion factor (BEF) available for hardwood and coniferous species (Pine and Spruce-fir), as given in Sharma et al. (2010, 2016) (**Supplementary Table 1**). For the estimation of belowground biomass (BGB), the following equation given by Cairns et al. (1997) was used: BGB = exp [−1.059 + 0.884 × ln (AGB) + 0.284]. Both AGB and BGB were summed to get total biomass (TB). For the calculation of carbon stock, the following formula was used: Carbon stock (Mg C/ha) = Biomass (Mg/ha) × C (%). When evergreen coniferous species comprise more than 50% of forest types, a carbon factor of 46% was employed. Conversely, a carbon factor of 45% was used where broadleaved species were predominant (Negi et al., 2003).

2.4 Data analysis

Before subjecting the statistical analysis, the data were checked for normality assumptions using the Shapiro-Wilk test. Differences between forest attributes (stem density, diameter at breast height, tree height, total basal cover and biomass) among forest types were tested through one-way analysis of variance (One-way ANOVA) and Tukey's HSD *post-hoc* test. Pearson's correlation test was performed to check if there is a significant association between carbon density and its candidate variables. The "corrplot" package in R was used for correlogram preparation (Wei and Simko, 2021). Linear regression analyses were used to study the effect of explanatory variables on C stock. In addition, we also performed the principal component analysis (PCA), as it can reduce a large number of variables to a few main variables without compromising the data originality (Jolliffe and Cadima, 2016) and effectively remove multicollinearity among variables. The PCA was performed

1 https://chelsa-climate.org/

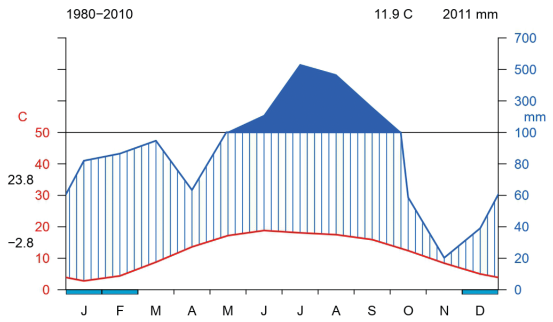

FIGURE 2
Walter and Lieth climate diagram for the sampling plots illustrating the variation in average monthly temperature (red) and precipitation (blue) from 1980 to 2010.

using the package "factoextra" in R (Kassambara and Mundt, 2020). All the statistical tests were performed using packages "stats," and "multcompView" in R programming language 4.3.0 (Graves et al., 2023; R Core Team, 2023).

3 Results

3.1 Stand composition and structural attributes

A total of 29 tree species were recorded across the studied forest types. The species richness (no. of species) varied from 8 to 12 species in subtropical forests and 3–6 species in temperate forests. The ALD and LSDD forests attained the highest stem density (SD) values (558.3 ± 62.9 and 866.6 ± 57.7), respectively (Figure 3A; Supplementary Table 2). Across the temperate forests, the SD value ranged from (303 ± 20.8 to 573.3 ± 55), with the lowest in CDP and the highest in QSAM. The SD values differed considerably between forest types ($F = 18.261$, $p < 0.001$). In terms of DBH, the ALD-dominated stand had the lowest value (23.3 ± 1.6), while the CDP stand had the greatest (46.8 ± 11.6) (Figure 3B). A similar trend was observed for height, with values lowest in ALD (12.1 ± 1.7) and highest in CDP (29.3 ± 1.1) (Figure 3C; Supplementary Table 2). The variation in mean DBH and mean height data across different forest types was statistically significant ($F = 4.741$, $p = 0.005$; $F = 11.538$, $p < 0.001$). Similarly, the TBC differed significantly among forest types ($F = 3.541$, $p = 0.01$), with the highest value (74.4 ± 25.9) in QSAM, while the lowest (24.9 ± 4.01) in ALD (Figure 3D; Supplementary Table 2).

3.2 Mapping of biomass and carbon stock

The biomass (AGB, BGB, TB) and carbon stocks were highest in *Cedrus deodara* forest (CDP) with values ranging from (641.7 ± 53.3, 140.9 ± 15.5, 782.6 ± 107.9 Mg/ha and 360 ± 49.7 Mg C/ha), respectively (Supplementary Table 2; Figures 4A–D). Apart from this, temperate forests such as APD (322.1 ± 154.1), QFM (309.2 ± 84.5), and QSAM (284.9 ± 84 Mg C/ha) contributed substantially to C stock formation (Supplementary Table 2; Figure 4D). Among the studied forest types, the QLMT contributed the least to C stock formation (128.9 ± 25.7 Mg C/ha) (Figure 4D). The C stock in subtropical forests, ALD and LSDD, varied between (207.03 ± 19.5 and 250.9 ± 41.4 Mg C/ha), respectively (Supplementary Table 2; Figure 4D).

3.3 Species contribution in C stock formation

In terms of species-wise contribution across all forests, *Abies pindrow* contributed the maximum to C stock formation (375.1 Mg C/ha), followed by *Cedrus deodara* (353.5 Mg C/ha), *Quercus semecarpifolia* (282 Mg C/ha) and *Q. floribunda* (250 Mg C/ha) (Supplementary Figure 1). In the case of subtropical species, *Anogeissus latifolia* (143.8 Mg C/ha) contributed the highest in C stock followed by *Mallotus philippensis* (110.9 Mg C/ha), and *Lannea coromandelica* (57.8 Mg C/ha) (Supplementary Figure 1).

3.4 Relationships between forest attributes and C stock

Pearson's correlation matrix showed that C stock is linked with different forest attributes. Among the structural variables, only the TBC ($r = 0.80$, $p < 0.001$), DBH ($r = 0.65$, $p < 0.001$), and height ($r = 0.62$, $p < 0.01$) showed significant association with C stock (Figure 5). In contrast, the other variables directly or indirectly influenced C stock but were not significantly associated. For instance, stem density ($r = 0.10$, $p = 0.64$) showed a positive association, whereas diversity attributes (species richness index and Shannon index) were negatively correlated ($r = -0.36$, $p = 0.09$; $r = -0.35$, $p = 0.09$) (Figure 5). Among climatic variables, we found that MAP was positively correlated ($r = 0.27$, $p = 0.19$), whereas MAT ($r = -0.29$, $p = 0.17$) had a negative association. Similarly, elevation ($r = 0.31$, $p = 0.14$) showed a positive association with C stock, but its effect was not significant.

The candidate variables were found to exhibit a correlation with each other. For example, stem density showed a negative correlation with DBH ($r = -0.54$, $p < 0.01$), whereas it was positively associated with species richness ($r = 0.68$, $p < 0.001$) and Shannon diversity ($r = 0.66$, $p < 0.001$). Elevation displayed a significant influence on climatic variables and diversity attributes, such as MAP ($r = 0.85$, $p < 0.001$) and MAT ($r = -1$, $p < 0.001$), SR ($r = -0.72$, $p < 0.001$) and H' ($r = -0.43$, $p < 0.05$) (Figure 5).

3.5 Factors influencing biomass and C stock

The bivariate analyses revealed that structural attributes except for stem density (Supplementary Figure 2A) explained the most significant variation among all the variables tested against C stock. The total basal cover (TBC) emerged as the strongest predictor of C stock (64%), followed by DBH (42%) and height (39%)

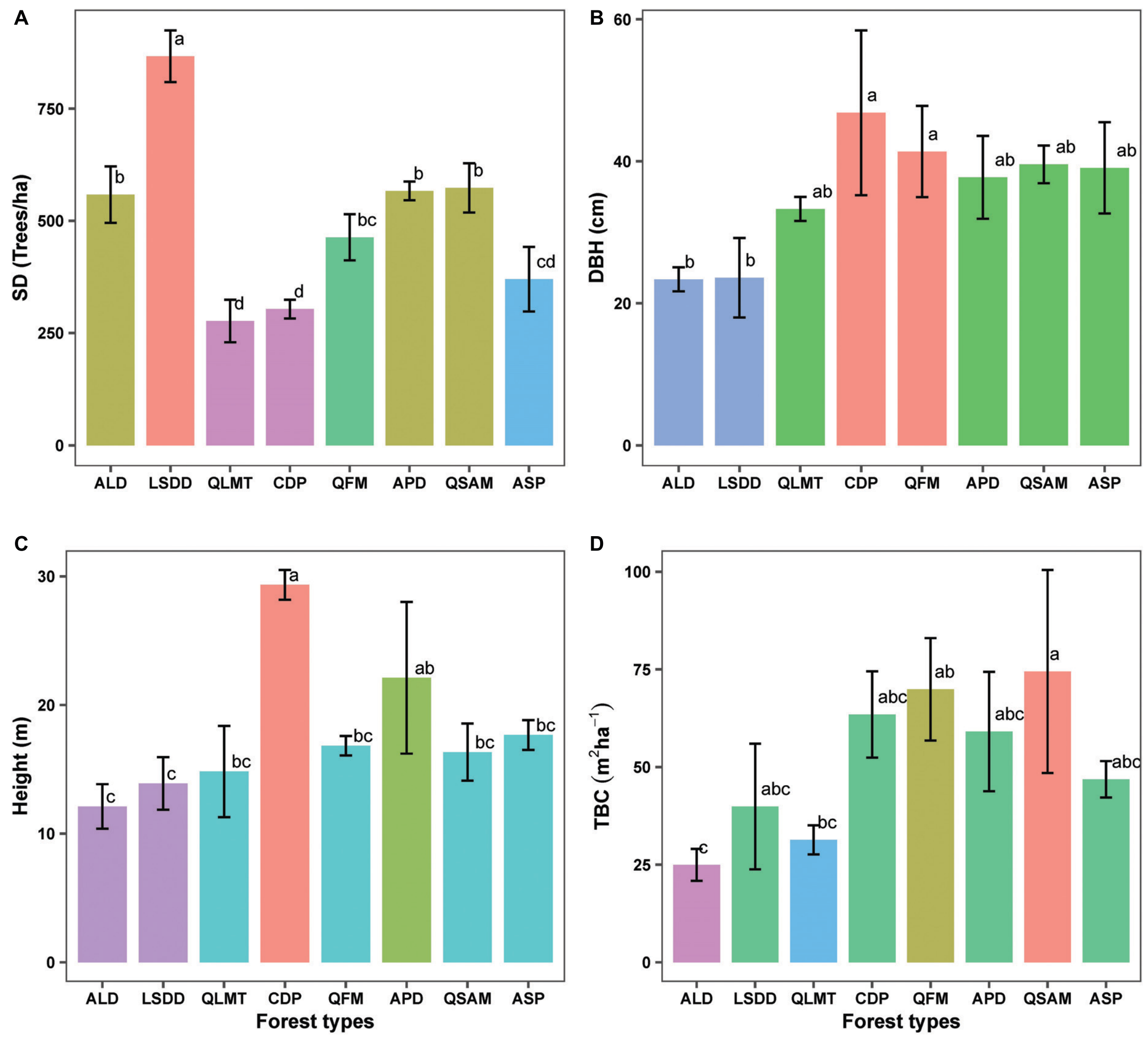

FIGURE 3
Variations in stand structural attributes **(A)** stem density, **(B)** diameter at breast height, **(C)** tree height, and **(D)** total basal cover across different forest types. Error bars represent the standard deviation of the data. Forest types are listed in ascending order of elevation. Different letters on bar tips indicate significant differences between mean values by Tukey's HSD *post-hoc* test, ($p < 0.05$).

(Figure 6). The rest of the tested variables showed no significant variation in C stock, with a contribution of SR and H′ varied from 13% and 12%, respectively (Supplementary Figures 2B, C). Climatic variables, including MAP and MAT, displayed only 7.5% and 8.5% of the variation in C stock (Supplementary Figures 2D, E). Among the topographic factors, elevation explains 9% of the variance (Supplementary Figure 2F). Furthermore, the principal component analysis (PCA) biplot explains the factors affecting biomass and C stock. It demonstrates that factors such as TBC, DBH, height, and MAP are grouped with biomass and C stock, whereas MAT, SD, species richness (SR), and Shannon diversity (H′) are oppositely placed. The principal component axes (PC1 and PC2) explained 77.5% of the variance (Figure 7).

4 Discussion

The current study unravels the factors affecting biomass and C stock across the different forest types in the Western Himalayas. The biomass and C stock values ranged from 286.6 Mg/ha to 782.6 Mg/ha and 128.9 Mg C/ha to 360 Mg/C ha, respectively (Supplementary Table 2). The current observations are comparatively higher but within the range of previous studies in similar environments. Carbon stocks in the Indian Himalayas have been reported to vary between 59.20–245.31 Mg C/ha (Sharma et al., 2010), 107.8–234.1 Mg C/ha (Gairola et al., 2011), 85.22–234.32 Mg C/ha (Sharma et al., 2018), 22.7–236.8 Mg C/ha (Haq et al., 2022), 133.04–273.28 Mg C/ha (Dar and Parthasarathy, 2022) and 207.32–270.98 Mg C/ha (Tiwari et al., 2023). However, our results are comparatively on the lower side, contrary to the study of Kaushal and Baishya (2021), wherein they reported that the total biomass density and C stock varied (566.17–1280.79 and 258.22–577.77 Mg C/ha) in different forests. The reported range of biomass C stock at a global scale varied from 506–627 Mg C/ha in the USA (Smithwick et al., 2002), 58.9–386.5 Mg C/ha in NE China (Wei et al., 2013), and 12.96–856.50 Mg C/ha in Panama (Ruiz-Jaen and Potvin, 2011) across temperate and tropical forests, respectively. A recent study by Di Matteo et al. (2023) showed that

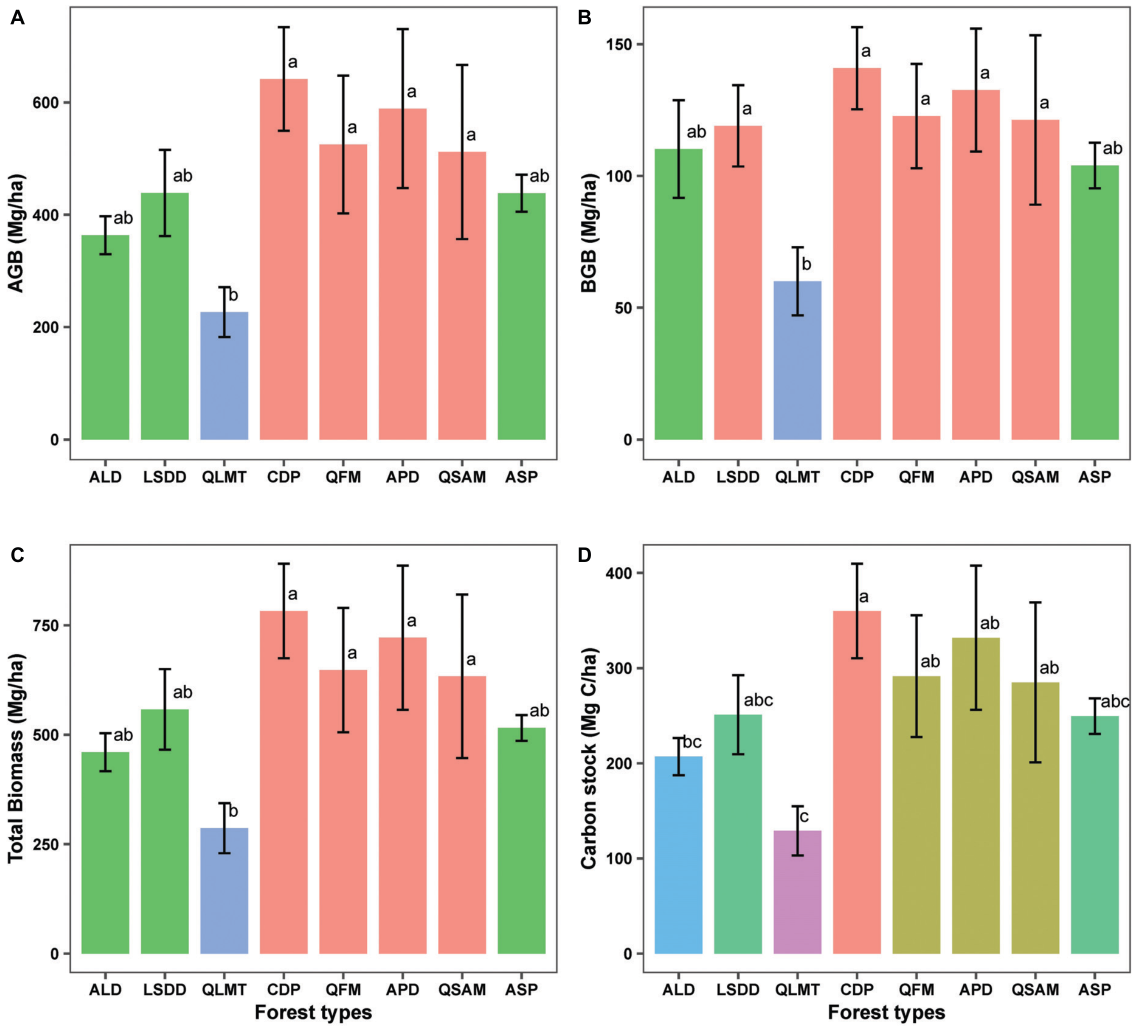

FIGURE 4
Variations in biomass and C stock **(A)** aboveground biomass, **(B)** belowground biomass, **(C)** total biomass, and **(D)** carbon stock among different forest types. Error bars represent the standard deviation of data. Forest types are listed in ascending order of elevation. Different letters on bar tips indicate significant differences between mean values by Tukey's HSD *post-hoc* test, ($p < 0.05$).

the tree biomass (living + root) ranged from 546.7 to 695.1 Mg/ha in temperate old-growth forests of Italy. The variation in results is attributed to stand age, edaphic conditions, vegetation type, disturbance, and topography.

Trees provide a win -win strategy to mitigate global climate change as they regulate light availability, litter quantity and quality and ultimately govern C dynamics (Shirima et al., 2015). Carbon storage in forest ecosystems, especially in temperate forests where only one or few species are dominant, is mainly contained in large diameter, and old-age species. Furthermore, our results revealed that a more significant fraction of biomass and C stock is contributed by temperate species including *Cedrus deodara*, *Abies pindrow*, *Quercus floribunda* and *Quercus semecarpifolia* (**Supplementary Figure 1**), highlighting the fact that large trees contribute disproportionally to stand biomass compared to small trees (Poorter et al., 2015). In our study, the strong positive influence of DBH and TBC on C stock is in conformity with previous studies in tropical (Gebeyehu et al., 2019; Saimun et al., 2021) and temperate environments (Yuan et al., 2018; Bisht et al., 2022).

Another crucial structural attribute, stem density, showed no relationship with C stock. This probably due to the fact that a greater stem density may induce a crowding effect, due to which plant species compete for resource acquisition, ultimately causing reduced tree growth (Sullivan et al., 2017; Bhandari et al., 2021; Ulak et al., 2022). In terms of species diversity (species richness and Shannon diversity), we found no significant effect on C stock. The negligible role of species diversity on biomass C stock reflected the influential role of a few dominant species (Larsary et al., 2021). Generally, two hypotheses, niche complementarity and selection effect hypotheses (Tilman et al., 2001; Cardinale et al., 2009) explain how species diversity promotes biomass production. In diverse communities (tropical regions), plant species prefer niche partitioning for maximum utilization of resources and to facilitate each other, unlike communities where only a few dominant species are present. In our case, the lack of significant

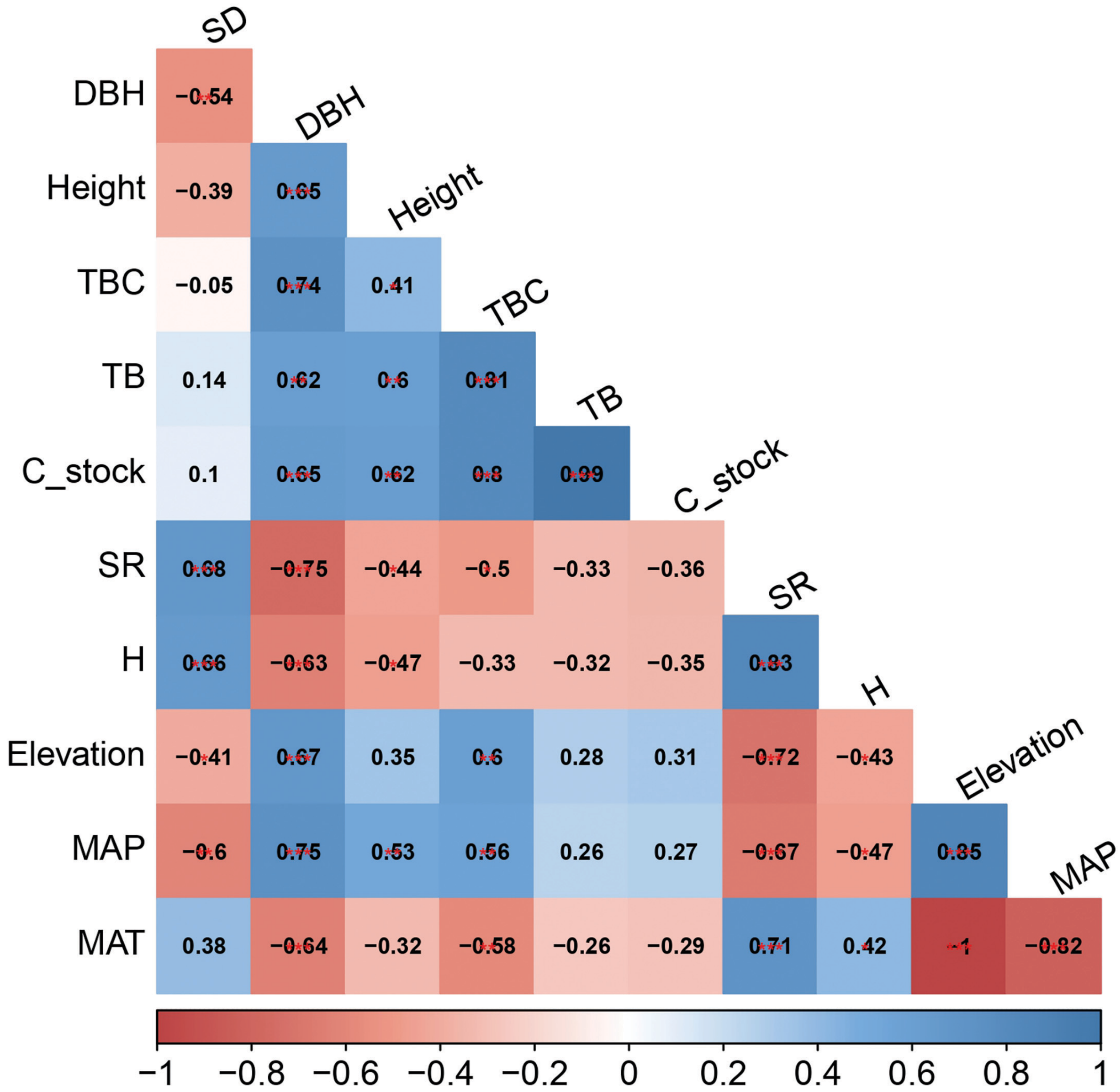

FIGURE 5
Pearson correlogram matrix between variables and carbon stock. SD, stem density; DBH, diameter at breast height; TBC, total basal cover; TB, total biomass; SR, species richness index; H, Shannon diversity index; MAP, total annual precipitation; MAT, mean annual temperature.

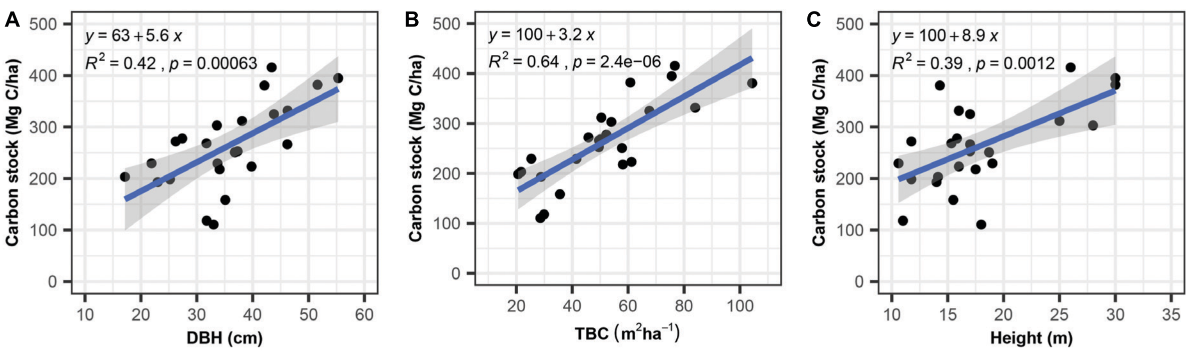

FIGURE 6
Bivariate relationships of carbon stock with diameter at breast height **(A)**, total basal cover **(B)** and tree height **(C)**.

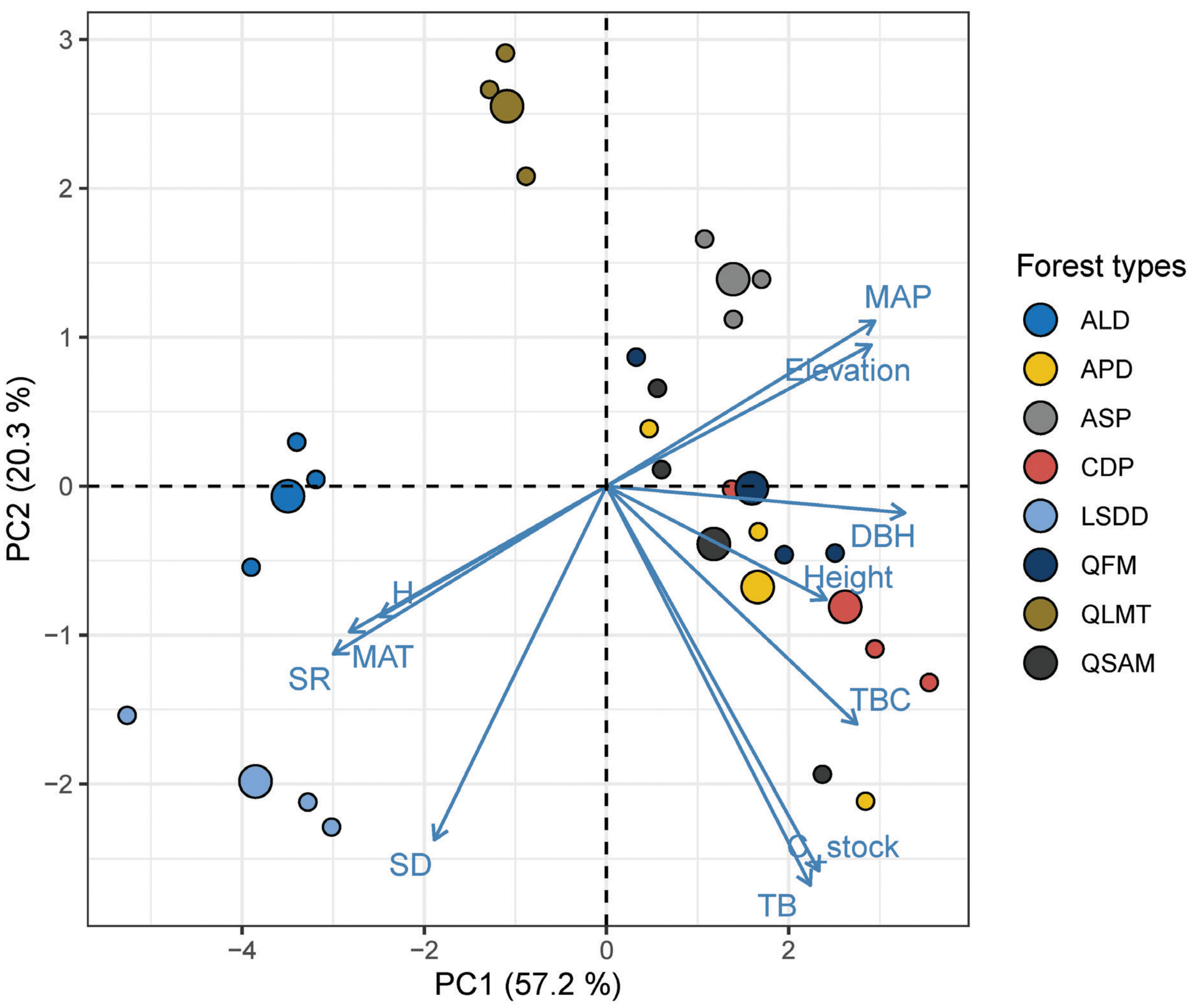

FIGURE 7
A principal component analysis between structural variables and forest types depicting a correlation biplot matrix diagram. The points represent forest types and the arrow indicates variables. SD, stem density; SR, species richness; H, Shannon index; MAT, mean annual temperature; DBH, diameter at breast height; MAP, total annual precipitation; TB, total biomass; TBC, total basal cover.

association of species diversity with biomass is probably due to the selection effect where dominant large tree species outline the other species. Previous studies (Paquette and Messier, 2011; Arasa-Gisbert et al., 2018) have observed that competitive exclusion, rather than species diversity, is the relevant explanation for high biomass and C stock in temperate forests growing under favorable conditions. Furthermore, the diversity-productivity hypothesis is reported to be scale-dependent. Generally, the positive influence of diversity on biomass/carbon is restricted to smaller sampling plots (0.1 ha), whereas negative or neutral relationships at a larger scale (0.25–1 ha) (Chisholm et al., 2013; Poorter et al., 2015; Fotis et al., 2018).

In our study, we expected elevation to be one of the critical elements in biomass and C stock formation. However, its effect was insignificant (r = 0.31; p = 0.13). Several reasons could be attributed to this: (1) our sampling plots are distantly related and span over different environmental conditions, so maybe that role of elevation is overcome by other factors, like forest types and species attributes; (2) we have analyzed our data across sites not within a single site, this might have caused sizeable environmental heterogeneity where the influence of elevation is negated. Previous studies in mountainous areas (Sharma et al., 2018; Kaushal and Baishya, 2021; Maza et al., 2022) have also observed similar trends. Although elevation didn't play a role in biomass and C stock development, its significant effect can be followed on other forest attributes. For example, tree species diversity decreases with a rise in elevation, a trend previously observed in other studies in the Western and central Himalayas as well (Sharma et al., 2018; Kaushal and Baishya, 2021; Wani et al., 2022; Tiwari et al., 2023). This could be due to the harsh climatic conditions at higher elevations, which retard tree growth and development (Wieser et al., 2014; Wani et al., 2023).

In bivariate analysis, none of the climatic variables (precipitation and temperature) showed significant association with C stock. Precipitation is a crucial environmental factor that governs moisture availability for plant growth and development and indirectly drives biomass production (McCarthy and Enquist, 2007; Lie et al., 2018). The positive but insignificant role of precipitation on C stock could be explained by the fact that the selected sites receive abundant rainfall as a whole. Therefore, moisture may not be the limiting factor. On the other hand, temperature has a predominant role in biomass and carbon stock formation, whereas in montane and temperate forests, studies have shown that it doesn't influence much (Selmants et al., 2014; Yue et al., 2018). Given that most of our sampling plots fall under a temperate environment, experiencing more or less the same temperature probably leads to homogeneity in temperature and ultimately negates its effect.

Overall, our finding revealed that structural variables (DBH, TBC, and tree height) override the role of abiotic (MAT, MAP, elevation) variables in biomass and C stock formation. Our results are in line with previous studies in the Himalayas (Sharma et al., 2010; Kaushal and Baishya, 2021; Dar and Parthasarathy, 2022) and (Poorter et al., 2017; Balima et al., 2021; Maza et al., 2022) elsewhere in the world. The PCA biplot also showed that biomass and C stock were positively associated with structural variables such as DBH, TBC and height. In contrast, stem density, diversity attributes and MAT were negatively correlated (**Figure 7**). Furthermore, the PCA biplot revealed that certain variables like elevation and MAP were positively associated with biomass and C stock despite insignificant effects in bivariate analysis.

Management activities at the community or regional levels are attributed to enhanced carbon storage (Adekunle et al., 2014; Solomon et al., 2017). In our study, we can say that, besides forest attributes, the management regime may be one of the contributing factors in biomass and C stock. Because each forest type is legally protected, there is less likelihood of external disturbance. As a result, it can be asserted that protected areas provide a conducive environment for the growth of plants and biodiversity conservation. Numerous studies have supported the significance of protected areas in shaping and maintaining the structure and functioning of ecosystems. For instance, a study by Keith et al. (2014) in the montane ash forests of southern Australia observed that the biomass carbon stock of logged forests was 55% lower than that of old-growth forests. The study of Dimobe et al. (2019) in W National Park in Burkina Faso, Western Africa, reported higher species richness (89 sp.) and carbon density (94.73 Mg/ha) compared to non-protected sites.

In another study, Måren and Sharma (2021) in the temperate forests of Nepal revealed that protected forests exhibited a more significant carbon stock (163.71 Mg C/ha) in comparison to unprotected forests (114 Mg C/ha). A separate investigation was carried out by Poudel et al. (2020), who conducted a study in the reserved oak forests of Nepal, whereby they determined that the carbon stock within this ecosystem exhibited a range of 52.8–194 Mg C/ha. Recently, a study conducted by Chaudhury et al. (2022) in North-east India found higher stand composition, biomass and C stock under protected forests compared to reserve forests and village forests, highlighting the fact that enhanced management practices in disturbed forests can lead to more significant CO_2 sequestration and climate change mitigation.

Nevertheless, it is worth noting that anthropogenic disruption cannot be disregarded, even considering the legal status of the study sites. For example, illegal deforestation, fire, and agriculture expansion within and near protected sites may lead to forest loss (Collins and Mitchard, 2017). Apart from this, unregulated tourism is also a significant concern for protected sites. For example, in the present study, we observed that Chail and Churdhar WLS, are major tourist hotspots in north-western India. Hence, they incur a heavy tourist influx during the summer, posing a substantial burden on fragile ecosystems. Indeed, the investigated ecosystems have an appreciable amount of biomass C stock, but its long-term persistence requires integrated efforts of authorities and the local population.

5 Conclusion

The present study's findings demonstrate that the selected sites in the Western Himalayas serve as a substantial repository of tree biomass and carbon stock. Temperate forests account for greater biomass and C stock than subtropical forests. We found that structural attributes govern the C stock in selected forest types mainly mean tree DBH, total basal cover (TBC), and tree height. However, the role of species diversity, elevation and climatic attributes in determining C stock was insignificant. Furthermore, in agreement with previous studies in the Himalayas, we found that only a few dominant species with large diameters account for the majority of the C stock of these forests. Hence, the cutting and felling of these species must be regulated for long-term ecosystem sustainability. Our findings highlight the role of protected sites in achieving carbon neutrality and the effective implementation of sustainable development goals (SDGs) strategies. At the same time, despite the legal status of these study sites, there is an urgent need to regulate permissible human activities such as tourism. Furthermore, the current findings may be useful to policymakers and stakeholders in developing management plans and climate change mitigation strategies for the Western Himalayas. Further studies are recommended to understand the detailed mechanism of factors involved in biomass and C storage in the Western Himalayas.

Data availability statement

The original contributions presented in the study are included in the article/**Supplementary material**, further inquiries can be directed to the corresponding author.

Author contributions

PK: Conceptualization, Data curation, Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Resources, Software, Validation, Visualization, Writing—original draft, Writing—review and editing. AK: Formal analysis, Funding acquisition, Investigation, Methodology, Validation, Visualization, Writing—original draft, Writing—review and editing. MP: Data curation, Formal analysis, Funding acquisition, Methodology, Validation, Visualization, Writing—review and editing. SH: Formal analysis, Funding acquisition, Investigation, Methodology, Validation, Visualization, Writing—review and editing. ANS: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Project administration, Resources, Supervision, Validation, Visualization, Writing—review and editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2023.1328694/full#supplementary-material

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A crop for a forest: *Opuntia ficus-indica* as a tool for the restoration of Mediterranean forests in areas at desertification risk

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Introduction: The Mediterranean is the European region with the lowest woody cover and the highest level of habitat degradation, being highly susceptible to climate change effects and desertification risk. In such worrying conditions, increasing woody cover and restoring forests is a major goal established in several international commitments. However, recruitment limitation of woody species is rather frequent both within natural regeneration processes and active restoration programs, particularly due to drought, overgrazing, and a lack of post-planting tending operations. Therefore, finding suitable tools to improve the recruitment success of native woody species is of crucial importance.

Methods: We assessed woody natural regeneration under abandoned prickly pear orchards, olive trees, and nearby open areas in three sites under high desertification risk in central Sicily (Italy). Then, we tested for differences in density, richness, diversity, height, and basal diameter of the woody recruiting species between these three habitats.

Results and discussion: Natural regeneration was widespread under prickly pear, with 94.6% of the sampled plots showing at least one recruit, in comparison to 61.6% of plots under olive and 22.3% in open areas. Natural regeneration density under prickly pears (114 ± 99 individuals m^{-2}) was significantly higher ($p < 0.001$) than under olive trees (60.4 ± 76.4) and open areas (4.6 ± 9.3). Recruits' diversity, basal diameter, and height were also significantly higher under prickly pear, concentrating 94.4% of the individuals higher than 100 cm and all late successional species. Our results indicate a great potential for prickly pears to accelerate the natural regeneration of Mediterranean woody species in areas under desertification. However, a site-specific evaluation must be made taking into account prickly pear's historical presence, temporary income as a crop, management capacity and, especially, its invasive potential.

KEYWORDS

desertification, ecological restoration, Mediterranean forests, oak, plant–plant facilitation, recruitment limitation, seed dispersal, *Quercus*

Introduction

Large areas in the Mediterranean have been historically deprived of the original woody cover, reducing biodiversity and exposing bare lands to increasing soil erosion and desertification risk (Prăvălie et al., 2017; Pausas and Millán, 2019). On the one hand, such degradation processes are bound to increase in the next decades as a consequence of human population growth as well as land use and climate change (Reynolds et al., 2007; Mulligan et al., 2016). On the other hand, the progressive abandonment of agricultural lands and the reduced pressure on woodlands in the Mediterranean could progressively provide the opportunity to restore native vegetation and correlated ecosystem services in increasingly larger areas (Plieninger et al., 2014; Novara et al., 2017; Bueno et al., 2020b). However, recruitment limitation is often a huge barrier to forest restoration, particularly in areas under desertification threat, and is caused by several factors (Acácio et al., 2007; Granda et al., 2014). Seed limitation is a first bottleneck that can depend on the lack of mother plants (source limitation), seed dispersers, and/or their interactions (Valiente-Banuet et al., 2015; La Mantia et al., 2019). Even when seed limitation is overcome, the high mortality of planted or naturally regenerating seedlings and saplings seems to be more the rule than the exception in the Mediterranean (Duponnois et al., 2009; Mendoza et al., 2009; Andivia et al., 2017). Drought and herbivory have been found to be the main factors causing such limitations; consequently, plant–plant facilitation (e.g., nurse plants, biogroups) may become essential for ecological restoration of degraded and/or harsh environments (Castro et al., 2002; Gómez-Aparicio et al., 2004; Padilla and Pugnaire, 2006; Brooker et al., 2008). This occurs, for instance, in xeric mountain areas, where shrub and tree species were found growing clustered together and around a main and larger species in biogroups (Pedrotti, 2019). Hence, these plant ensembles may allow the progressive spread of woody vegetation into open areas where single woody species are unable to establish and persist. Many studies have investigated plant–plant facilitation in the Mediterranean, although the balance between positive and negative effects is still not straightforward to predict due to largely species-specific and context-dependent outcomes, particularly in drought-prone ecosystems (Gómez-Aparicio, 2009; Filazzola and Lortie, 2014; Gonzalez and Ghermandi, 2019). For example, the facilitative effect of native shrubs can be reduced as aridity increases due to competition (Andivia et al., 2017), or nurse plant functional traits or growth form (e.g., shrub or tree) may generate contrasting effects, either positive or negative, along plant ontogeny (Gómez-Aparicio, 2009; Rolo et al., 2013). However, a critical issue is that if several degraded areas are totally deprived of any woody cover, then the *ad-hoc* implementation of nurse plants becomes the only chance. Such strategy, in turn, implies careful planning and crucial post-planting tending operations, which are not always carried out due to economic reasons, often undermining the efficacy of restoration interventions (Le Houerou, 2000; Gómez-Aparicio, 2009; Meli et al., 2017). In several cases, non-native plant species are faster-growing or more stress-tolerant than natives, making them more appealing to be used for ecological restoration (Krumm and Vítková, 2016; Badalamenti et al., 2020c; Suzuki et al., 2021). On the one hand, such exotic species may become invasive and seriously threaten native species and habitats, thus leading to expensive control and eradication programs (Simberloff et al., 2013). On the other hand, even invasive species may be useful for restoration or conservation purposes under some circumstances, therefore generating a conservation trade-off and calling for a better understanding of their functional role in ecosystem dynamics and plant community assembly (Schlaepfer et al., 2011; Vimercati et al., 2020; Badalamenti et al., 2020a). Although the risk of a generalized and uncritical approach to non-native species still exists, there is increasing consensus in the scientific community on the need for evidence-based assessment of alien species' invasiveness and related harmful impacts (e.g., Kumschick et al., 2023). In this research, we assessed the possible role of prickly pear [*Opuntia ficus-indica* (L.) Mill.] as a tool for forest restoration in Mediterranean areas. The prickly pear is a cactus species native to Mexico but cultivated over 1 million hectares in the Mediterranean basin, mainly due to its edible fruits but also for livestock fodder and fencing (Le Houérou, 1996). In Europe, Italy plays a leading role in prickly pear cultivation, hosting 8,614 hectares of plantations, mostly of spineless varieties located in Sicily, producing up to 87,000 tons a year, more than 12% of the world production (Erre et al., 2009; ISTAT, 2022). Due to their extreme capacity to thrive in harsh and dry conditions and ease of reproduction, opuntias, particularly the spinier species *Opuntia maxima* Mill. and *O. stricta* (Haw.) Haw., have also become strongly invasive in some Mediterranean regions, particularly in small islands, rocky habitats, and cliffs (Vilà et al., 2003; Padrón et al., 2011; Guarino et al., 2021; Tesfay and Kreyling, 2021). In turn, positive effects of prickly pear on different ecosystem services such as soil protection, nutrient cycling, carbon sequestration, and refuge to native grasses, forbs, and argan trees have also been reported (Génin et al., 2017; Oduor et al., 2018; Novoa et al., 2021; Stavi, 2022; Jorge et al., 2023). Additionally, prickly pear is being used as a restoration tool in large-scale projects in Africa (Neffar et al., 2018), and its cultivation contributes to sustaining the agricultural socio-economical tissues in many arid areas, therefore representing a typical conservation trade-off (Shackleton et al., 2011; Stavi, 2022). Despite this growing wealth of knowledge, the potential facilitative effects of prickly pear on the natural regeneration of Mediterranean woody species are still unknown. To fill this gap, we aimed to quantify the density, richness, and size of the woody species recruiting underneath prickly pear individuals in comparison with olive trees and nearby abandoned open areas. We hypothesized that, due to its functional and structural characteristics (CAM metabolism, low water requirements, moderate shading effect, and protection against herbivores), prickly pear will significantly facilitate the recruitment of a wide range of native woody species and allow their full establishment beyond the sapling stage.

Materials and methods

Study sites

Field surveys were carried out in three study sites in Sicily (Italy), all classified at a critical risk of desertification (Calvi et al., 2016; Figure 1; Supplementary Table S1). The first prickly pear orchard (PPR) is localized in Roccapalumba (Palermo province), whose landscape is dominated by clay soils and cultivated lands, especially annual crops, olive, and prickly pear orchards. The study site falls within the upper meso-Mediterranean upper dry bioclimatic belt (Bazan et al., 2015), at an altitude of 500 m a.s.l., on slopes less than 20%. Mean annual precipitation is 561 mm, mean annual temperature is 15.7°C, and soils are classified as Typic pelloxererts (Fierotti, 1988). The second prickly

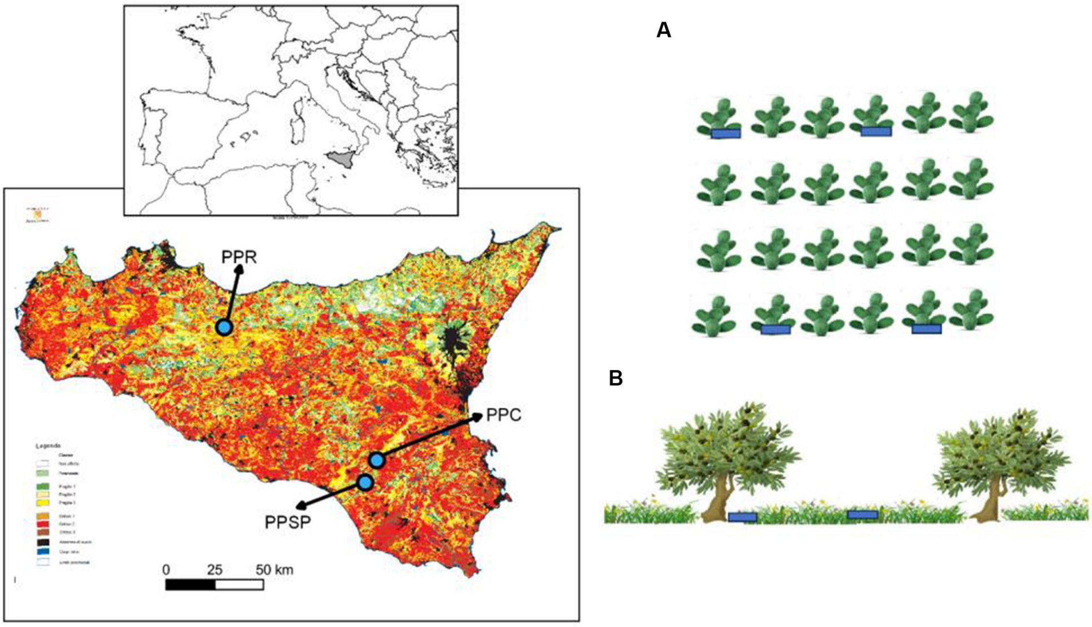

FIGURE 1
Location of the three prickly pear orchards overlaid onto the desertification risk map of Sicily (Calvi et al., 2016), with study sites located in areas classified as critical 1 and 2 (red/brown color). From left to right: PPR, Roccapalumba; PPSP, Santo Pietro; PPC, Caltagirone. **(A)** Example of the random distribution of the sampling plots (blue) under the prickly pear individuals, avoiding consecutive individuals and rows and **(B)** sampling plots under olives and open areas, avoiding nearby individuals. Images: Freepik.com.

pear orchard (PPSP) is localized in Santo Pietro (Catania province), in hilly areas of south-eastern Sicily, dominated by sandy soils and covered by cultivated areas, including annual crops, vineyards, pasturelands, and hardwood forests and shrublands. The study site falls within the upper thermo-Mediterranean with a lower sub-humid bioclimate (Bazan et al., 2015), at an altitude of 262 m a.s.l., on slopes less than 10%. Mean annual precipitation is 690 mm, mean annual temperature is 17.0°C, and the soils are Typic xerochrepts (Fierotti, 1988). The third prickly pear orchard (PPC) is localized in Caltagirone (Catania province), in inner hilly areas dominated by clay soils and cultivated lands, subject to an upper thermo-Mediterranean lower sub-humid bioclimate (Bazan et al., 2015), at an altitude of 427 m a.s.l., on slopes less than 10%. Mean annual precipitation in this area is 690 mm, mean annual temperature is 17.0°C and the soils are Typic xerochrepts (Fierotti, 1988). All the surveyed prickly pear orchards were planted approximately 30 years ago. Most of the surface of the PPR and PPC orchards is currently managed, with the natural regeneration controlled through mowing; however, from 2005 onwards, local farmers have been leaving some rows unmanaged that were selected for sampling. PPSP orchard was totally abandoned after 2005, although it is currently accessed by horses, sheep, and cattle. The PPSP falls within the Site of Community Importance "Bosco di Santo Pietro" (ITA 070005), which hosts some of the most significant cork oak stands in Sicily.

Sampling design

Regeneration assessment

In spring 2021, within each of the three prickly pear orchards, we established 50 rectangular plots (3 × 2 m) around the trunk of prickly pear plants, for a total of 150 plots (Figure 1). Prickly pear trunks represented the center of the plots, with the longer axis established along the row and the smaller axis directed to the inter-row. To reduce spatial autocorrelation, a plot was separated by two individuals (i.e., considering only the individuals n. 1, n. 4, n. 7, etc.), and 10 individuals in each row were surveyed, with a minimum distance of three rows (i.e., considering only the rows n. 1, n. 4, n. 7, etc.; Figure 1).

Natural regeneration under olives and in open areas

To allow for comparisons with prickly pears, we established the same rectangular plots (3 × 2 m) around the trunk of 120 randomly selected olive individuals (*Olea europaea*) distributed in one abandoned orchard nearby PPC (*N* = 40) and two abandoned orchards nearby PPSP (*N* = 40 each), with all olives located from 130 to 550 m from the respective prickly pear orchards. The higher number of plots in PPSP was to compensate for the sampling effort once we did not find abandoned olive orchards in the surroundings of PPR. As a control, we also randomly established the same rectangular plots at open areas nearby (±3 m) for each olive tree in each abandoned olive orchard (*N* = 50 in PPC and *N* = 100 in PPSP) and in an abandoned field on the side of the PPR orchard (*N* = 50), with a minimum distance of 5 m among plots (Figure 1). Once PPSP was totally abandoned (i.e., no inter-row management as in PPC and PPR), we also established 50 plots in the prickly pear inter-rows, complementing the sampling in open areas. We checked that these open areas were neither managed (i.e., mowing and tilling) nor burned at least in the last 15 years, based on interviews with the landowners, field observations, and the analysis of satellite images (Google Earth Pro®). At each plot, we counted and identified all woody individuals higher than 10 cm, shrub and tree individuals while height and basal diameter were measured only for shrub and tree individuals exceeding 1 m in height. The second edition of Flora d'Italia (Pignatti et al., 2017–2019) was used as a reference for plant identification and nomenclature, while the classification of the main seed dispersal vector of the species was from Jordano (2014) and Bueno et al. (2021).

Characteristics of the nearest forests

Since we were mostly interested in evaluating the facilitation effect of prickly pear to promote forest restoration, we recorded the presence of the nearest patches of native forests as a reliable proxy of the potential seed source for colonization through seed dispersal. Natural forests were searched within a 2-km radius area around each study site, considering a minimum forest area of 5,000 m^2. We obtained the forest cover and type from the most recent regional forestry inventory (Camerano et al., 2011). Furthermore, we also assessed the recent wildfire occurrence (from 2007 to 2020) in the same natural forests because a high fire frequency may seriously compromise seed production and subsequent seed dispersal chances (data from the headquarters of the Forest Service of Sicily).[1]

Statistical analysis

In order to check whether our sampling was representative of the overall natural regeneration richness, we first performed rarefaction tests for each habitat in each study site with the *specaccum* function and 100 permutations within the *vegan* package (Supplementary Figure S1). To test for differences in richness, density (number of plants per 100 m^2), height, and basal diameter of natural regeneration underneath prickly pear, olive trees, and in open areas, we used generalized linear models (GLM). First, we ran two global models pooling the data from the three sites to account for the potential natural variability across sites. In the first model, we included all species found in the survey (lianas, small shrubs, shrubs, and trees), whereas in the second, we included only the shrub and tree species. Then, to check for local scale differences, we ran separate models for each site, again one model with all species and a second with shrubs and trees. For richness analysis, we used a Poisson distribution with log link, and for density, height, and basal diameter, we used a negative binomial distribution once these data showed a non-normal distribution (Shapiro–Wilk $p < 0.05$), resulting in high overdispersion in the Poisson and quasi-Poisson models. All analyses were performed with R v4.2.1 (R Core Team, 2021) with the *MASS* package for the GLM.

Results

Density and species richness of woody natural regeneration

Overall, 1,516 individuals from 25 different woody species were recorded in the sampling plots, with an average density of 53.7 individuals per 100 m^{-2} (Table 1). The rarefaction analysis indicated that our sampling effort can be considered satisfactory to represent the plant community of recruits at our sites (Supplementary Figure S1). Colonization under prickly pear was widespread, with 94.6% of the plots containing at least one recruiting individual, while this percentage dropped to 61.6% under olives and to 22.3% in open areas. Average natural regeneration density under prickly pear was almost double that under olive trees ($114 \pm 99 \times 100\,m^{-2}$ vs. 60.4 ± 76.4, respectively) and almost 30 times higher than in open areas (4.6 ± 9.3), with GLM indicating significant differences both considering all plant species and only shrubs and trees (Tables 2, 3; Figure 2). In turn, density and richness were higher under olives than in open areas. Prickly pear's highest woody species density and richness were confirmed by the separated GLM models, indicating significantly higher values in all intra-site comparisons (Tables 2, 3; Figure 2).

Recruitment of established woody species

Out of the 1,516 woody individuals, 202 (12.9%) were shrubs and trees higher than 100 cm, belonging to 15 species, and 94.4% of them occurred under the prickly pears. Indeed, only six plots in open sites and five plots under olives hosted such recruiting individuals, which were, conversely, widely spread over 117 prickly pear plots, accounting for 1.4, 10.9, and 19.7% of all woody individuals in the three habitats, respectively. Recruits were also significantly higher and larger under prickly pear than under olives and open areas (Table 4), including 53 individuals higher than 300 cm and with basal diameters up to 39 cm (Figure 3). Such differences were also confirmed by the separated GLM models (Table 4; Figure 3).

Natural regeneration species composition

Wild asparagus (*Asparagus acutifolius* L.) was the most abundant species (28.9% of all the individuals) and, together with olive (*Olea europaea* L. s.l., 18.2%), mastic (*Pistacia lentiscus* L., 15.5%), and brambles (*Rubus* spp., 2.6%), occurred in all three habitats (Table 1). Seventy-six individuals of three oak species, i.e., downy oak (*Quercus pubescens Willd. s.l.*), cork oak (*Q. suber* L.), and holm oak (*Q. ilex* L.), were observed, but only under prickly pears. Downy oak was the most abundant oak species (80.3%), occurring at relatively high densities ($19 \times 100\,m^{-2}$) and even reaching heights up to 7.5 m and diameters of 21 cm in PPC. Olives, almonds (*Prunus dulcis* (Mill.) D.A. Webb), mastic and downy oaks, (*Quercus pubescens*) accounted for 91% of the established individuals, although only the first two occurred in all prickly pear orchards. The large majority of the recruits were from fleshy-fruited species (92.6%), indicating an active role of animal seed dispersal networks.

Characteristics of the nearest forests

Since we could not perform a statistical analysis of the influence of the nearest forest stands due to low sampling sites, we report here, and in Supplementary Table S2, a descriptive assessment. In PPC, the area with almost all oaks recruited, there were four downy oak stands with average size of 3.7 hectares, occurring 500 m away, and no wildfires were registered in these areas in the last 14 years. In PPR, there were three downy further than oak stands with average size of 8

1 https://sifweb.regione.sicilia.it/arcgis/rest/services/Censimento_Incendi/MapServer

TABLE 1 Relative abundance and density of the woody plant species found in the study areas.

Family	Species	Plant habit	Main seed dispersal vector*	RA	Density ($N \times 100^{-2}$)			Relative abundance (%)		
					Open	Olive	Prickly pear	Open	Olive	Prickly pear
Ulmaceae	*Celtis australis* L.*	Tree	Birds	0.07			16.7			0.1
Ericaceae	*Erica arborea* L.	Shrub	Auto	0.07		16.7			0.2	
Lauraceae	*Laurus nobilis* L.*	Tree	Birds	0.07			16.7			0.1
Leguminosae	*Robinia pseudoacacia* L.*	Tree	Wind	0.07			16.7			0.1
Oleaceae	*Fraxinus angustifolia* Vahl*	Tree	Wind	0.13	16.7		16.7	1.8		0.1
Pinaceae	*Pinus halepensis* Mill.	Tree	Auto	0.13		16.7			0.5	
Fagaceae	*Quercus ilex* L.	Tree	Birds	0.13			16.7			0.2
Leguminosae	*Cytisus infestus* (C.Presl) Guss.	Shrub	Auto	0.20	16.7			5.5		
Fabaceae	*Ceratonia siliqua* L.	Tree	Mammals	0.20		16.7	16.7		0.2	0.2
Cistaceae	*Cistus creticus* L.	Small shrub	Auto	0.20		50.0			0.7	
Thymelaeaceae	*Daphne gnidium* L.	Small shrub	Birds	0.40		25.0			1.4	
Ephedraceae	*Ephedra fragilis* Desf.*	Shrub	Birds	0.40			16.7			0.6
Leguminosae	*Ononis natrix* L. subsp. *ramosissima* (Desf.) Batt.	Small shrub	Auto	0.40		16.7			1.4	
Thymelaeaceae	*Thymelaea hirsuta* (L) Endl.	Small shrub	Auto	0.40		33.3			1.4	
Oleaceae	*Phyllirea latifolia* L.*	Shrub	Birds	0.79	16.7	33.3	16.7	1.8	1.8	0.3
Lamiaceae	*Stachys major* (L.) Bartolucci & Peruzzi	Shrub	Birds	0.99		50.0	25.0		2.1	0.6
Fagaceae	*Quercus suber* L.*	Tree	Birds	0.99			16.7			1.5
Rosaceae	*Rubus* spp.*	Shrub	Mixed	2.64	16.7	16.7	18.8	7.3	0.2	3.4
Rosaceae	*Prunus dulcis* (Mill.) D.A. Webb*	Tree	Mixed	3.50	16.7		16.7	5.5		4.9
Fagaceae	*Quercus pubescens* Willd. s.l.*	Tree	Birds	3.89			16.7			5.8
Lamiaceae	*Thymbra capitata* (L.) Cav.	Small shrub	Auto	5.74		131.8			20.0	
Anacardiaceae	*Pistacia lentiscus* L.*	Shrub	Birds	15.50	23.8	46.7	34.0	18.2	29.7	9.4
Rubiaceae	*Rubia peregrina* L.	Liana	Birds	16.03		50.0	44.9		7.6	20.5
Oleaceae	*Olea europaea* L. s.l.*	Tree	Birds	18.21	20.0	33.3	41.7	43.6	2.8	23.4
Liliaceae	*Asparagus acutifolius* L.	Liana	Birds	28.89	21.4	50.8	42.8	16.4	30.1	29.0

RA, overall relative abundance in ascending order. *Information from Jordano (2014) and Bueno et al. (2021).

hectares, but all of them were further than 1.5 km and had burned at least twice in the last 14 years. In PPSP, we found two *Q. suber* forests (average size of 10 hectares) and one large *Q. ilex* forest (>10 hectares), although 1.8 km distant.

Discussion

In the last century, massive reforestation projects have significantly increased woody cover in the Mediterranean, while millions of hectares of agricultural fields have been abandoned, although many of them did not evolve toward late successional stages composed of native tree species. In our study, we found that prickly pear strongly facilitated the recruitment and fostered the growth of a wide variety of Mediterranean woody species, including late successional tree species, in comparison with abandoned olive trees and open areas.

Woody species density and richness patterns

Recruits were widely distributed under prickly pear in all three study sites, with 94.6% of the 150 sampled individuals hosting at least one woody species. Olive trees also showed a high frequency of woody recruits (62%), in comparison to open areas (22%). However, virtually all fully established (≥1.0 m high) shrubs and trees were found under prickly pear. In a study comparing the natural regeneration under a native shrub [*Retama sphaerocarpa* (L.) Boiss.] and paired open sites along an environmental gradient in Spain, just about 10% of the 1,263 *Retama* shrubs had one recruit, for a total of 211 woody individuals belonging to four species (Andivia et al., 2017). The authors also observed that Retama's facilitative effect was negatively correlated with aridity and herbivory pressure. Due to the CAM metabolism, the high water-use efficiency, the shallow root system, and the high level of herbivory protection, prickly pear is expected to cause a positive

TABLE 2 Results of the generalized linear models comparing density ($n \times 100\ m^{-2}$) and richness considering all plant species with all sites pooled and within each study site.

	Habitat	Density				Richness			
		Estimate	SE	*z*	*p*	Estimate	SE	*z*	*P*
All sites	Intercept (olive)	4.10	0.18	22.62	<0.001	0.17	0.08	2.01	0.044
	Open	−2.61	0.23	−11.26	<0.001	−1.64	0.17	−9.66	<0.001
	Prickly pear	0.63	0.24	2.60	0.009	0.92	0.10	9.43	<0.001
Caltagirone	Intercept (olive)	−0.69	0.22	−3.10	0.001	−0.69	0.22	−3.10	0.002
	Open	−1.14	0.42	−2.72	0.006	−1.14	0.42	−2.72	0.006
	Prickly pear	1.91	0.24	8.08	<0.001	1.91	0.24	8.08	<0.001
Roccapalumba	Intercept (open)	0.94	0.27	3.45	<0.001	−1.83	0.35	−5.18	<0.001
	Prickly pear	2.39	0.38	6.33	<0.001	2.04	0.38	5.41	<0.001
Santo Pietro	Intercept (olive)	4.37	0.20	21.55	<0.001	0.42	0.09	4.66	<0.001
	Open	−2.63	0.27	−9.58	<0.001	−1.63	0.20	−7.98	<0.001
	Prickly pear	1.05	0.32	3.24	0.001	1.02	0.11	9.02	<0.001

TABLE 3 Results of the generalized linear models comparing density ($n \times 100\ m^{-2}$) and richness of shrub and tree species with all sites pooled and for each study site.

	Habitat	Density				Richness			
		Estimate	SE	*z*	*p*	Estimate	SE	*z*	*p*
All sites	Intercept (olive)	3.05	0.21	14.80	<0.001	−0.71	0.13	−5.45	<0.001
	Open	−1.73	0.26	−6.60	<0.001	−0.92	0.21	−4.48	<0.001
	Prickly pear	0.97	0.28	3.53	<0.001	1.25	0.14	8.63	<0.001
Caltagirone	Intercept (olive)	2.55	0.36	7.08	<0.001	−1.29	0.30	−4.28	<0.001
	Open	−1.72	0.49	−3.52	<0.001	−1.01	0.54	−1.88	0.061
	Prickly pear	1.36	0.48	2.82	0.004	1.98	0.32	6.25	<0.001
Roccapalumba	Intercept (open)	0.81	0.32	2.56	0.0105	−1.97	0.38	−5.20	<0.001
	Prickly pear	2.02	0.44	4.61	<0.001	1.74	0.41	4.23	<0.001
Santo Pietro	Intercept (olive)	3.23	0.24	13.60	<0.001	−0.51	0.14	−3.54	<0.001
	Open	−1.58	0.32	−4.93	<0.001	−0.80	0.24	−3.32	<0.001
	Prickly pear	1.37	0.38	3.61	<0.001	1.35	0.17	7.88	<0.001

correlation between increasing aridity and herbivory and its facilitative effect. Olive trees showed significantly fewer woody recruits than prickly pears, particularly those lacking established shrub and tree species (Figure 3; Table 4), possibly resulting from a higher competition for light and space because abandoned olive trees often have a high density of lower branches and root shoots. Conversely, prickly pear has a more open architecture, giving support to lianas and allowing the full growth of trees and high shrubs. A facilitative effect was also observed in the dynamics of biogroups in Italian mountain ecosystems, where junipers (such as *Juniperus deltoides* R.P. Adams and *Juniperus hemisphaerica* Presl) have been found to act as the development center for several woody species (Lapenna and Fascetti, 2010; Pedrotti, 2019). Indeed, a global meta-analysis indicated that shrubs tend to provide higher facilitative effects than trees, especially when considering later developmental stages (Gómez-Aparicio, 2009), although such ontogenetic trade-off was also observed within the same shrub species (Rolo et al., 2013). Indeed, recruitment was significantly higher under olives than in open areas, and the establishment of lianas such as *Asparagus acutifolius* and wild madder (*Rubia peregrina* L.) was widespread there. Interestingly, Andivia et al. (2017) also found that *Asparagus acutifolius* was the most abundant species occurring under *Retama* shrubs. The second most abundant recruiting species was olive, occurring in all prickly pear orchards, with some individuals higher than 6 m. A high olive seedling density has also been verified under Mediterranean pine plantations, although saplings and adults were very rare (Badalamenti et al., 2018). In turn, olive recruits were absent or rare under other olives and open areas, indicating both high intra-specific competition and susceptibility to drought and herbivory.

Prickly pear contribution against recruitment limitation

The positive role of prickly pear in the ecological restoration of semi-arid ecosystems has been previously acknowledged (Le Houérou, 1996; Neffar et al., 2013; Génin et al., 2017; Neffar et al., 2018) and the utility of prickly pear to improve soil conditions in order to favor the

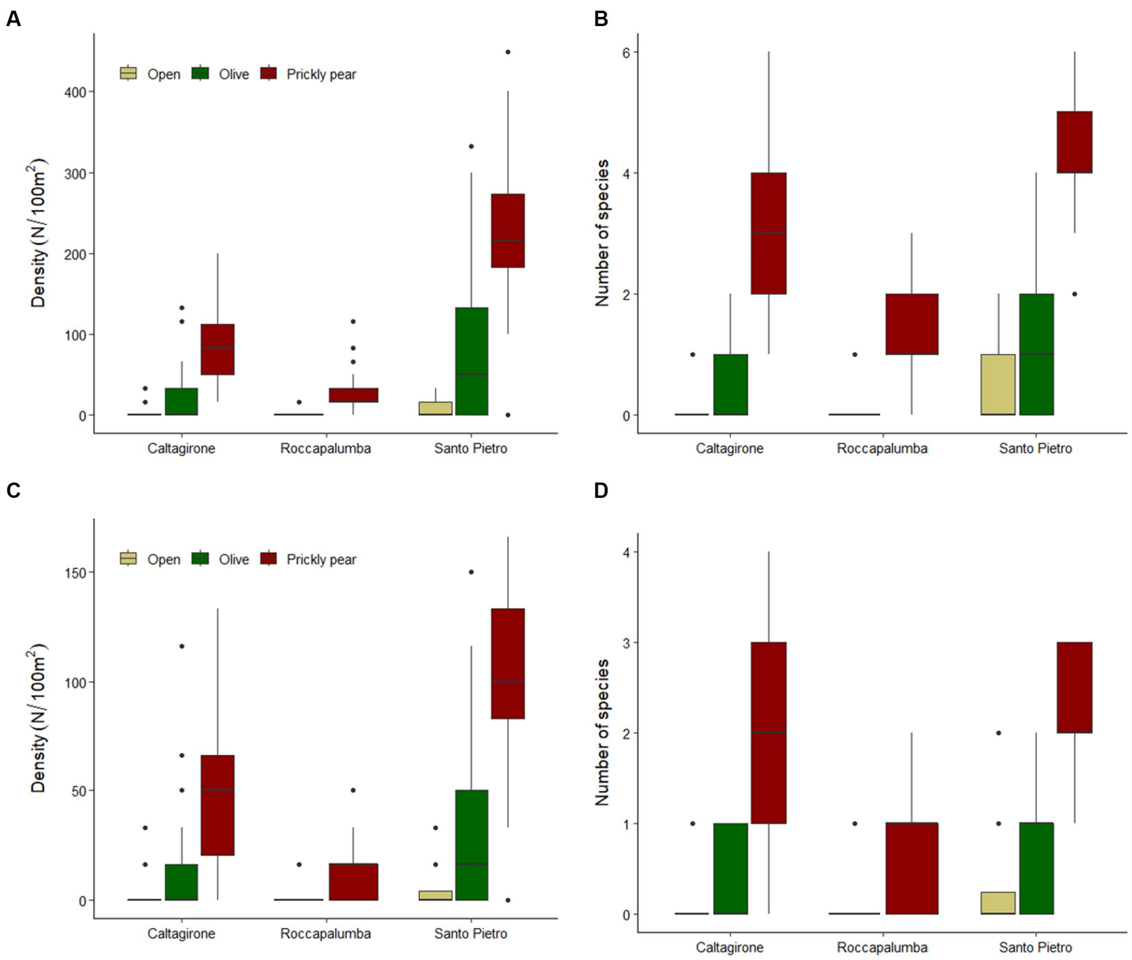

FIGURE 2
Boxplots showing the natural regeneration density and richness of all woody species **(A,B)** and of shrubs and trees **(C,D)** across open areas, under abandoned olive trees, and in prickly pear orchards in the three study sites.

TABLE 4 Results of the generalized linear models comparing the height and basal diameter of shrub and tree species across habitats.

Habitat	Height				Basal diameter			
	Estimate	SE	*z*	*p*	Estimate	SE	*z*	*p*
Intercept (olive)	3.70	0.09	40.51	<0.001	0.79	0.41	1.90	0.057
Open	0.23	0.15	1.57	0.117	0.31	0.95	0.33	0.745
Prickly pear	1.52	0.11	13.94	<0.001	1.06	0.42	2.52	0.012

cultivation of other species has been highlighted in the past, but with exclusive reference to agricultural contexts (Nocito, 1844). Due to its peculiar plant structure, the prickly pear could also be a useful plant for combating erosion in riparian contexts, where, moreover, its invasive potential is effectively zero (Stavi, 2022). However, its possible facilitative role in the natural regeneration of Mediterranean woody species has been barely documented. In one of the few study cases, prickly pear has been reported to aid the development of the argan tree (*Sideroxylon spinosum* L.) in pre-Saharan Morocco, with an increasing effect of old prickly pear individuals resulting in soil organic matter accumulation (Génin et al., 2017; Hassan et al., 2019). Prickly pear has also been facilitating the natural regeneration of common walnut (*Juglans regia* L.) and almond (*Prunus dulcis*) in Mediterranean agroecosystems (Bueno et al., 2020a; Badalamenti et al., 2022). Recently, a study in the savanna ecosystem showed that the invasive *Opuntia stricta* can create fertility islands that facilitate the establishment of native plants, improving soil abiotic and biotic conditions (Novoa et al., 2021). In another study, Oduor et al. (2018) found that the abundance and diversity of native species were not affected by prickly pear invasion, regardless of the cover percentage, showing that the negative effects exerted by alien plants are strongly context-dependent.

The rarefaction analysis indicated that our results are representative of the plant community of recruits present at our

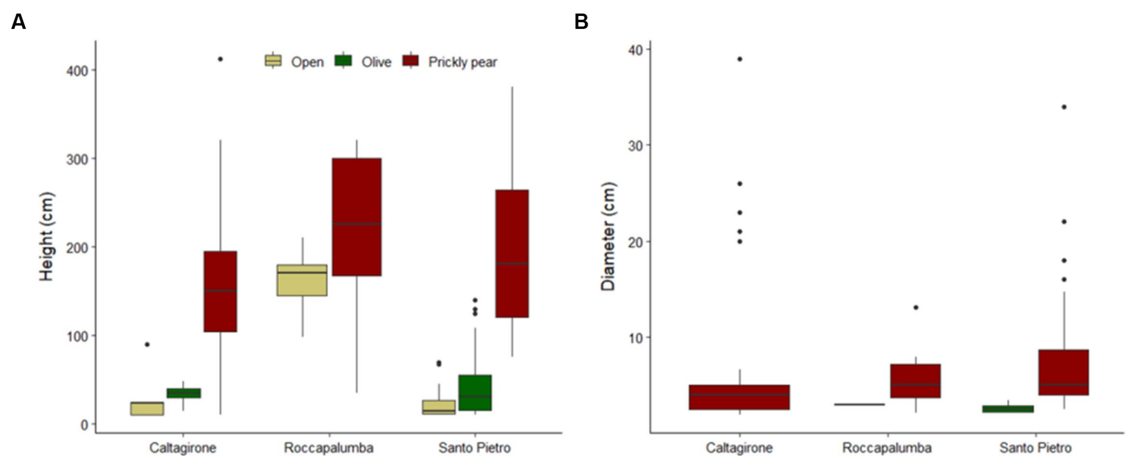

FIGURE 3
Boxplots showing the height of all shrub and tree individuals **(A)** and the basal diameter of individuals with a height of ≥100 cm **(B)** in the open areas, under abandoned olive trees, and in prickly pear orchards in the three study sites.

study sites (Supplementary Figure S1). The vast majority of the woody species found in our surveys (Table 3) have fleshy fruits and rely on vertebrates, particularly mammals and especially birds, for seed dispersal (Bueno et al., 2021). This evidence implies a high incidence and effectiveness of animal-mediated seed dispersal networks. This is particularly the case for birds, as they tend to avoid open areas or to stay on the ground, relying both on natural and artificial perches (Pons and Pausas, 2006; La Mantia et al., 2019). Although we did not assess the seed rain, our personal observations clearly indicate that birds frequently used both olive trees and prickly pear individuals. Indeed, during our surveys, we observed many seed dispersers, such as warblers (*Sylvia* spp.), thrushes (*Turdus* spp.), robins (*Erithacus rubecula*), corvids (*Corvus cornix*, *Pica pica*, and *Garrulus glandarius*), pigeons (*Columba palumbus*), and starlings (*Sturnus* spp.) perching or foraging inside the prickly pear orchards. Due to its quick development, prickly pear can soon offer suitable perches for birds, consequently fostering secondary succession underneath. Since herbivory and trampling are key biotic factors affecting the recruitment of woody species, the physical protection offered by a nurse plant is of paramount importance (Padilla and Pugnaire, 2006; Andivia et al., 2017). The PPC and PPR orchards are protected to prevent the access of large animals, whereas the PPSP and respective olives and open areas are frequently accessed by livestock (cattle, horses, and sheep), while wild herbivores such as deer are not present. Since spineless prickly pears are the most used varieties in our study sites, the main protective mechanism is the physical barrier created by the cladodes, particularly when they become hard with aging. Despite such protection was also provided by olives and can be offered by other native shrubs (Smit et al., 2008), the multi-stemmed architecture of prickly pear, similar to a candle holder, seemed to represent a great trade-off between protection and space and light competition (Rolo et al., 2013). Effectively, we observed several recruits showing a typical tree habit with a streamlined trunk, in comparison with the shrubby aspect of recruits in open areas that is typical of sites under high herbivory pressure.

Oak regeneration and the role of the remnant forest patches

Oak forests (dominated by *Quercus* spp.) represent the late successional stages in some areas of the Mediterranean basin, including the surroundings of our study sites. Since oaks are highly susceptible to recruitment limitations and are often lacking in reforestation interventions, they are frequently used as indicators in facilitation studies (Acácio et al., 2007; Gómez-Aparicio, 2009; Bobiec et al., 2018). The main abiotic constraints hampering the natural regeneration of Mediterranean oaks are poor soil conditions and summer drought. These two conditions may not only kill the seedlings but also affect the viability of acorns when moisture content drops below 26% (Acácio et al., 2007; Ganatsas and Tsakaldimi, 2013; Matías et al., 2019). Light limitation is another relevant factor, with reduced development in high-tree-density plantations or dense shrublands (Pausas et al., 2006). To overcome these limiting factors, prickly pear was found to be of crucial importance. Indeed, not only did we find a high oak seedling density under prickly pears, but they also hosted all the established oak individuals that reached up to 7 m high, indicating a good balance between soil organic matter content, water availability, space, and light competition (Bautista-Cruz et al., 2018; Hassan et al., 2019). However, our results also suggest a central role of oak seed limitation conditioned by the characteristics of the nearest forest fragments (especially size and distance) because the vast majority of high oak individuals were found in PPC. Jays and the wood mouse, both observed in our study sites (Cairone et al., 2020; Bueno et al., unpublished data), are the main seed dispersers of Mediterranean oaks. They act at very different spatial scales, with wood mice operating at short distances (10–50 m) and jays at medium distances (approximately 300 m), with rare long-distance dispersal events (Gómez, 2003; La Mantia and Bueno, 2016). Despite patches of oak forests being present in the surroundings of all the study areas and relatively large oak forest patches (up to 18 ha) being present inside the 12.5 km^2 surveyed area, only at PPC were they within the mean seed dispersal range of the Eurasian jay (*Garrulus glandarius*) (as far as 500 m), which seemed to be one major factor driving the highest oak density there. Interestingly, some oak individuals in PPC

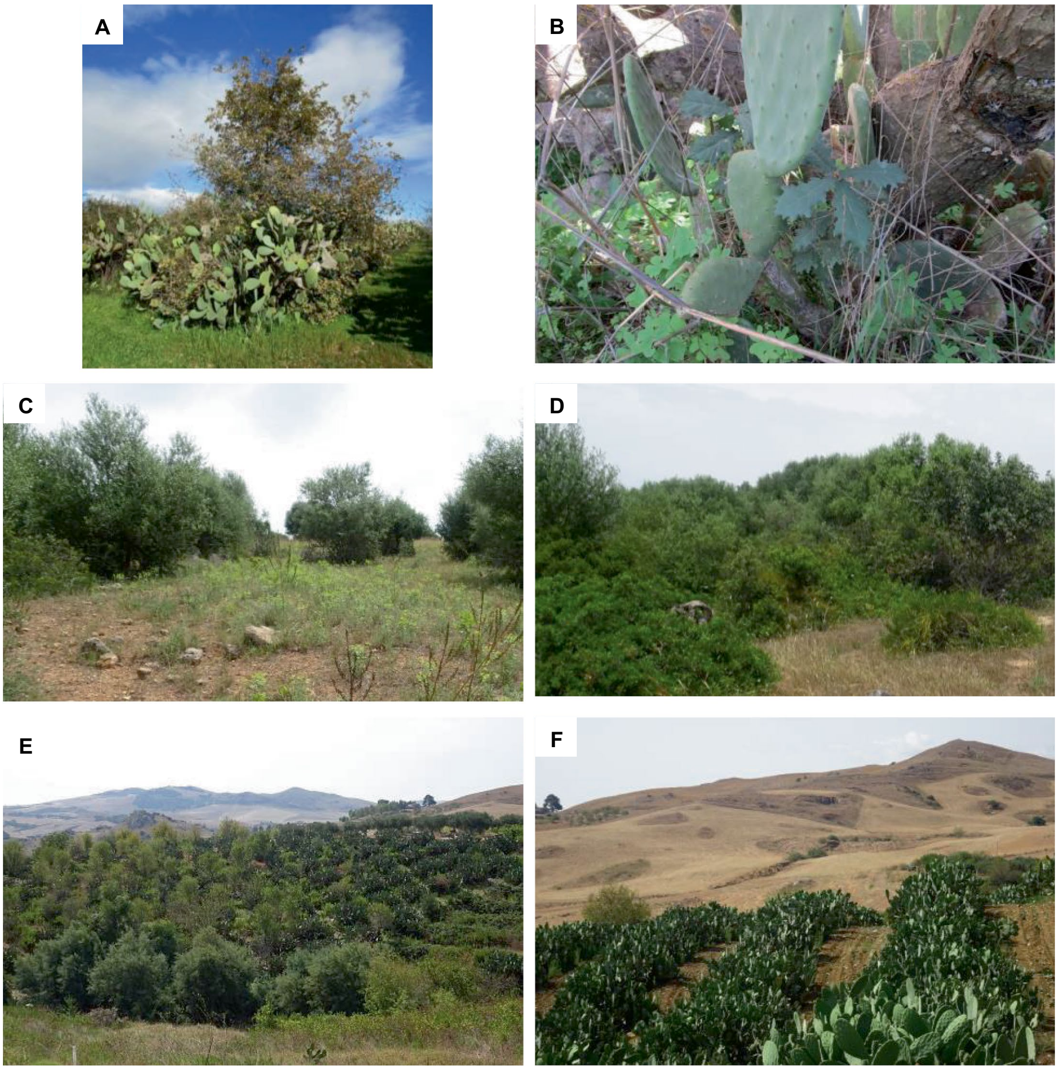

FIGURE 4
One mature *Quercus pubescens* individual **(A)** and one *Quercus pubescens* sapling growing at the PPC site **(B)**. Abandoned olive trees and respective open areas at PPSP **(C)**, current appearance of the abandoned prickly pear orchard at PPSP, dominated by native woody species **(D)**, aspect of the PPR site with woody species covering the prickly pears in the unmanaged part on the left **(E)**, one managed prickly pear orchard and nearby arable land intermingled with degraded patches that could be suitable to use prickly pear to foster native woody species colonization **(F)**.

were already reproductive (Figure 4), indicating a further successional step where seed dispersal from outside may become increasingly less important. Conversely, the very high fire frequency affecting oak forests in the surroundings of PPR is a major detrimental factor, as recurrent wildfires generally curb seed production for several years (Trabaud and Galtié, 1996; Badalamenti et al., 2020b), therefore reducing the chances of long-term seed dispersal. The suitability of prickly pear for oak seedling establishment, as widely observed in PPC, would suggest, hence, that the direct seeding of acorns could be successfully implemented in PPR and PPSP sites in order to overcome seed limitation.

Conservation trade-offs for the use of prickly pear

The limited number of studies evaluating prickly pear as a restoration tool largely depends on its prevalent use as a productive crop, so unwanted plant species in prickly pear orchards are usually removed. For this reason, we did not find other suitable areas for our research, which limited the number of sampling sites. However, we clearly recognize that the major constraint that may limit the use of prickly pear as a restoration tool is its invasive potential in the Mediterranean, where it represents a serious ecological threat, particularly for chasmophytic vegetation in rocky habitats (Vilà et al., 2003; Guarino et al., 2021). Even if in our study areas and in nearby abandoned fields and forest patches (e.g., pine and eucalyptus reforestation, oak forests, and maquis), we found no evidence of invasive behavior, we do not recommend its use in areas where it is not already present or if careful management is not feasible. From this side, it should be kept in mind that prickly pear is an agricultural crop already covering millions of hectares and sustaining the economy of entire regions, so that in many areas, including Sicily, there are no restrictions on its cultivation. Most importantly, its management is really feasible, as knocking down all the first fruits when they are unripe is a common agricultural practice made by local farmers, preventing their diffusion through seed dispersal. Prickly pear is also highly sensitive to shading, so that the cover by adult tree individuals, once living underneath its canopy, naturally prevents its future development. In turn, when native woody plants are fully established, prickly pear can be cut, and the residuals can be used to feed livestock (another common practice), mulch, increase soil organic matter, or be used as a natural biogel for reforestation purposes (Le Houérou, 1996; Vimercati et al., 2020; Stavi, 2022). Considering the several biotic and abiotic constraints hampering the restoration success of Mediterranean degraded areas, particularly in arid and semi-arid climates, our results indicate that prickly pear may represent a valid tool to facilitate the restoration of Mediterranean woodlands, provided that its invasive potential is carefully considered.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary material, further inquiries can be directed to the corresponding author.

Author contributions

RB: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Validation, Visualization, Writing – original draft, Writing – review & editing. EB: Data curation, Investigation, Methodology, Writing – original draft, Writing – review & editing. GS: Investigation, Methodology, Writing – review & editing. TM: Conceptualization, Funding acquisition, Investigation, Methodology, Project administration, Supervision, Writing – review & editing.

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In memoriam

One of the authors (Tommaso La Mantia) dedicates this article to his friend Enza Chessa, who studied prickly pear throughout her life.

Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

The author(s) declared that they were an editorial board member of Frontiers, at the time of submission. This had no impact on the peer review process and the final decision.

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Supplementary material

The Supplementary material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2024.1343069/full#supplementary-material

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An analysis on the spatial heterogeneity characteristics of landscape ecological risk in Qilian Mountain National Park

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As a key ecological function area and a priority area for biodiversity conservation in China, Qilian Mountain National Park is facing a severe test of its ecological environment, and the study of its landscape ecological risk is of great significance to the construction and high-quality development of the Qilian Mountain National Park. In this research, based on land use data from six periods (i.e., year in 1995, 2000, 2005, 2010, 2015, and 2020) in the Qilian Mountain National Park, we divided the ecological risk plots, calculated the landscape pattern, and constructed the landscape ecological risk index to deeply explore the temporal and spatial heterogeneity of landscape ecological risk in Qilian Mountain National Park by using ArcGIS, Fragstats and GeoDa. The results showed that: Grassland is the predominant land use type, the area covered by woodland and grassland have exhibited a significant increase since 1995. Landscape fragmentation and disturbance indices exhibit fluctuations across different years, but showed an overall decreasing trend, and landscape stability was improved in the study area. There were obvious differences in the disturbance indices of different landscape types, with grassland and bare land having the highest values. Ecological risk in the study area is heterogeneous, with an overall low ecological risk and a shift to a lower risk level, and a decreasing trend in ecological risk, which is positively correlated spatially and mainly manifested as a "low-low" aggregation. Global warming and unreasonable human activities have exacerbated the ecological degradation of Qilian Mountain National Park, but a series of ecological restoration strategies after the establishment of the national park have gradually improved the regional ecological environment.

KEYWORDS

land use type, landscape pattern, ecological risk index, spatial autocorrelation, Qilian Mountain National Park

1 Introduction

Ecological risk refers to the potential negative impacts on ecosystem structure， function and stability resulting from external natural and anthropogenic activities, either in the present or future (Depietri, 2019; Zhang et al., 2020, 2023). Over the past few decades, ecosystem structure and function have experienced strong changes in response to global environmental change and intensified human activities, and ecological risks to various ecosystems have continued to increase (Wang et al., 2020; Xie et al., 2020). Landscape ecological risk, as an important part of ecological risk, is based on landscape ecology to construct the link between landscape patterns and ecological processes. Different from the traditional ecological assessment, landscape ecological risk assessment focuses more on the spatial and temporal heterogeneity of risk and scale effects (Han et al., 2022; Zhang et al., 2023). And it also provides a comprehensive characterization, dynamic patterns and spatial visualization of multisource risks (Suter et al., 2003; Peng et al., 2015; Wang et al., 2020).

In terms of previous research on landscape ecological risk assessment, there are both regions affected by climate change and human activities, such as administrative regions, coastal cities, and mining areas, as well as more environmentally fragile areas such as highlands, wetlands, and nature reserves (Mikhailov et al., 2015; Ersayin and Tagil, 2017; Lin et al., 2019; Zhang et al., 2020). Currently, landscape pattern index and source-sink risk are the methods for landscape ecological risk assessment, among which the method based on land-use and landscape indices is the most widely used (Ayre and Landis, 2012; Peng et al., 2015; Li et al., 2017; Wang et al., 2020). For example, Zhang et al. (2020) analyzed the temporal and spatial characteristics of landscape ecological risk during the urbanization process and discussed the regularities of landscape ecological risk changes in 48 coastal cities in China using ecological risk assessment based on land use data (Zhang et al., 2020). Wang et al. (2020) quantitatively assessed the landscape ecological risk of the Koshi River Basin using landscape type data and the landscape ecological risk model, and analyzed the temporal–spatial evolution characteristics of landscape ecological risk for the study period and for over different elevation and slope intervals, then explored the spatial clustering characteristics of landscape ecological risk using spatial autocorrelation analysis (Wang et al., 2020). Qi (2019) developed an evaluation index system to quantitatively assess the ecological risk by employing landscape pattern indices and investigated the evolutionary characteristics and patterns of ecological risk in various periods. However, this research failed to examine the variations of ecological risks among different landscape patterns and regions (Qi, 2019).

The Qilian Mountain National Park, as one of the 10 pilot national parks in China, is an important water source and biodiversity conservation priority area in Northwest China, and an important ecological barrier in the Silk Road Economic Belt. This region plays an important role in maintaining the ecological balance, guaranteeing runoff replenishment, and sustaining the sustainable development of the region (Shan et al., 2023). However, the ecological environment in some areas of the Qilian Mountain has deteriorated and ecological risks have increased under the influence of a variety of factors such as global climate change, overloaded grazing, man-made destruction, insufficient means of protection, etc. (Liu et al., 2023; Ma et al., 2023). The solution of the eco-environmental problems of Qilian Mountain is of great significance to the sustainable socio-economic development of the Hexi Corridor in the Northwest China, and to the regional or national ecological security (Li et al., 2021). Therefore, many scholars have carried out in-depth research on the ecological and environmental problems of Qilian Mountain National Park. Liu et al. (2023) found that the risk of desertification in Qilian Mountain National Park experienced three distinct phases from 2000 to 2020, characterized by a decline, an increase, and another decline. The primary cause of the increased risk of desertification in the region is the pressure-driven effect brought about by climate change and natural disasters (Liu et al., 2023). Ma et al. (2023) found that the NDVI of Qilian Mountain National Park showed an increasing trend over time from 2000 to 2020. However, gradual development in the study area was highlighted by anthropogenic disturbances and local development of related economies at the cost of ecological damage from 2010 to 2020 (Ma et al., 2023). Shan et al. (2023) found that the ecological carrying capacity of Qilian Mountain National Park decreased from 2000 to 2014 due to the destruction of ecological environment (Shan et al., 2023). Despite many studies have been reported on the ecological environment of the Qilian Mountain, there has been limited research on the spatial and temporal evolution of landscape ecological risk features.

In light of the current national development policy emphasizing the importance of “Lucid waters and lush mountains are invaluable assets,” there is significant theoretical significance in assessing the spatial and temporal heterogeneity as well as the evolutionary trends of ecological risks within the Qilian Mountain National Park. Therefore, this research aims to analyze the landscape ecological risk within the Qilian Mountain National Park by constructing an Ecological Risk Index (ERI) based on the landscape pattern indices (landscape fragmentation and disturbance indices) from 1995 to 2020 and reveal the temporal and spatial patterns of landscape ecological risk within the Qilian Mountain National Park. This study will provide scientific reference for the ecological protection and restoration of Qilian Mountain National Park.

2 Materials and methods

2.1 Study area

The Qilian Mountain National Park is located at the borders of Gansu and Qinghai provinces in China, on the northeastern edge of the Tibetan Plateau (93.51–103.90°E, 35.83–39.98°N), presenting one of the initial pilot regions established within the national park system of the country. With an average elevation surpassing 4,000 m, the area exemplifies the distinctive geological structure of the Kalidong graben within the Kunlun-Qinling geosyncline, belonging to the typical semi-arid alpine continental climate (Liu et al., 2023). The multi-year average annual precipitation in the study area ranges from 96.15 to 728.46 mm, with a spatial distribution trend gradually decreasing from east to west. The annual average temperature changes significantly with altitude, ranging from −14.97°C to 6.48°C, with the low-altitude areas being much warmer than the high-altitude areas. Additionally, the total solar radiation in the study area exceeds 5,000 MJ/m^2. Due to strong vertical zonation and topographic relief, this region showcases a rich variety of ecosystems, including forests, grasslands, glaciers, and other natural landscapes (Figure 1).

2.2 Data sources

The land use data utilized in this research is derived from the annual China Land Cover Dataset (CLCD), which presents the first Landsat-based dataset developed by scholars at Wuhan University using Google Earth Engine (GEE) platform (Yang and Huang, 2021). The dataset possesses aspatial resolution 30 m. One advantage of this dataset is its continuous land use classification results spanning 30 years, updated until 2020, providing a high temporal resolution. In this research, the land use data at a spatial resolution of 30 m for six periods (i.e., 1995, 2000, 2005, 2010, 2015, and 2020) were selected. The land use types within the study area were extracted based on the boundary mask of the Qilian Mountain National Park. The land use types include arable land, forest, shrubland, grassland, water bodies, glaciers and permanent snow cover, bare ground, impervious surfaces,

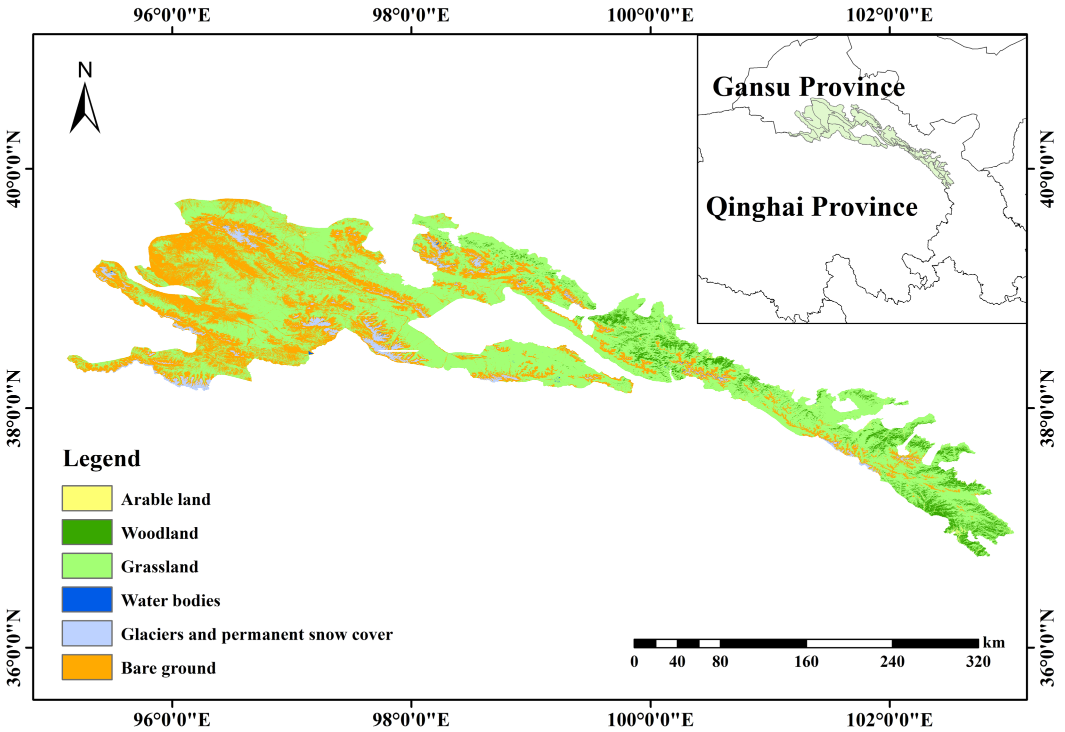

FIGURE 1
Location map of the study area.

and wet-lands. Based on actual conditions of the Qilian Mountain National Park and the classification methods employed, the land use types within the study area were reclassified into six landscape elements: woodland, grassland, bare ground, arable land, water bodies, glaciers and permanent snow cover (Figure 1).

2.3 Methods

2.3.1 Segmentation of ecological risk regions

To investigate the regional heterogeneity of landscape ecological risk within the Qilian Mountain National Park, the study area needs to be divided into several ecological risk plots, each plots were calculated by landscape pattern index and ecological risk index. The size of ecological risk plots is usually determined according to the actual situation of the study area or by using the standard of 2–5 times of the average patch area of the landscape (Bartolo et al., 2012; Cui et al., 2018; Zhang et al., 2018). In this study, we divided the study area into 1,106 ecological risk plots, each of which is 10 km × 10 km, as shown in Figure 2.

2.3.2 Landscape pattern indices

Landscape pattern indices serve as quantitative measures that integrate information on landscape structure and spatial characteristics, providing valuable insights into the features of landscape patterns (Gong et al., 2015; Fan et al., 2016; Corsi et al., 2020). These indices play a crucial role in understanding the stability, fragmentation, and heterogeneity of landscapes. At the patch level, metrics such as patch density and patch number are adopted to assess the landscape heterogeneity, providing an indication of the variety of patches within the landscape. The landscape segmentation index measures the degree of patch separation, indicating the level of isolation among patches. Fragmentation, which encompasses the spatial complexity and fragmentation of the landscape structure, is often characterized by combining patch number and segmentation index (Xie et al., 2013). The landscape fractal dimension index is applied to quantify the patch complexity of patch shape based on their perimeter and area dimensions, offering insights into the naturalness or irregularity of patch shapes within the landscape. The landscape disturbance is a fundamental concept in landscape ecology and captures the various risks posed by diverse sources of disturbance to ecological landscapes. In this research, the landscape disturbance index was calculated by incorporating the landscape fragmentation index, landscape segmentation index, and landscape fractal dimension index (Table 1).

2.3.3 Construction of landscape ERI

The landscape *ERI* was formulated to assess the ecological risk based on by leveraging the landscape pattern indices, which capture the variations in landscape structure within the selected study area. The landscape *ERI* is calculated using the following equation:

$$ERI = \sum_{i=1}^{n} \frac{A_i}{A} R_i \qquad (1)$$

where, *ERI* is the ecological risk index for the sampling area, i is landscape type, A_i is the area of landscape type i in the n sample area, A is the total area of sample n, R_i is the landscape disturbance index of type i.

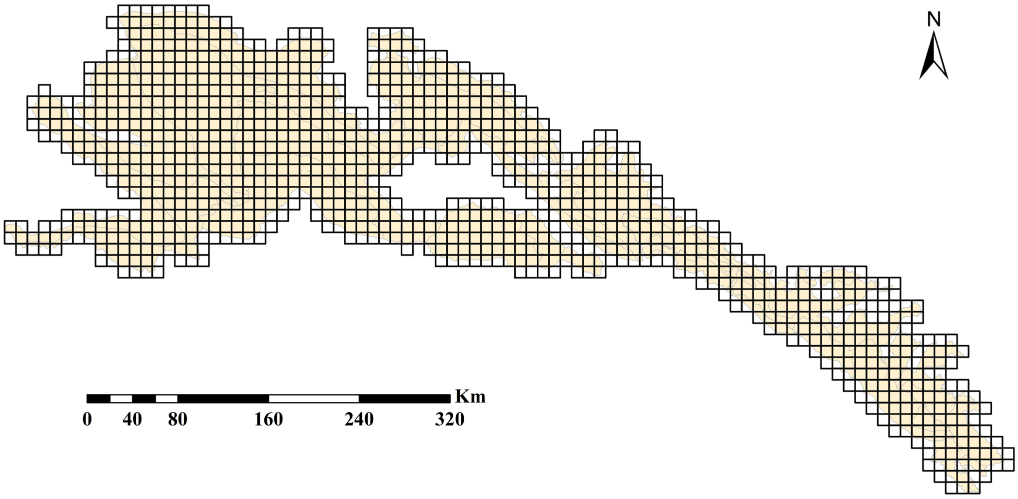

FIGURE 2
Segmentation of ecological risk regions.

TABLE 1 Landscape pattern indices and their definitions.

Index name	Calculation equation	Parameter description
Landscape fragmentation index	$C_i = N_i/A_i$	N_i: the number of patches of landscape type i; A_i: the area of the landscape type i.
Landscape segmentation index	$S_i = E_i/P_i$	E_i: the distance index of the landscape type i; P_i: the area index of the landscape type i.
Landscape fractal dimension index	$D_i = 2\ln(M_i/4)\ln(A_i)$	M_i: the perimeter of landscape type i.
Landscape disturbance index	$R_i = aC_i + bS_i + cD_i$	a, b, and c: the weight factors of C_i, S_i, and D_i, which were designated as 0.5, 0.3, and 0.2, respectively

2.3.4 Spatial autocorrelation analysis

Spatial autocorrelation analysis is undertaken to examine the significant relationships between attribute values of a specific feature and those of neighboring features in space to the objective of such analysis is to uncover the spatial correlation patterns of attribute characteristics between spatial reference units and their adjacent units. It encompasses two key indicators: global spatial autocorrelation index and local spatial autocorrelation index (Bosso et al., 2017; Wang et al., 2020; Zhang et al., 2020). The former is commonly quantified using the Moran's I, which enables the assessment of the spatial correlation of ecological risk across the entire study area. The value of Moran's I is between −1 and 1, with a positive value indicating a positive spatial correlation. As the value of Moran's I increases, the spatial correlation becomes more significant, providing valuable insights into the overall spatial pattern of ecological risk (Zhang et al., 2020). The latter, on the other hand, is visualized through the Local Indicators of Spatial Association (LISA) maps, thus depicting whether attribute values exhibit distinct patterns of high-high clustering and low-low clustering within the local areas.

3 Results

3.1 Changes in land use

Through the analysis of land use types and areas for the six study periods, several observations can be made. Firstly, woodland is primarily concentrated in the central and eastern parts of the study area, bare land is predominantly located in the western region. The glaciers and permanent snow cover is mainly distributed in the high mountain areas of the western and southeastern regions. Water bodies occupy a relatively small area and are predominantly situated beneath mountains covered with ice and snow. Cultivated land is scattered and occupies a smaller area, primarily distributed in the marginal areas in the eastern and northern regions (Figure 3). Upon examining the land use transition matrix spanning from 1995 to 2020 (Table 2), notable observations could be made regarding the substantial increase in areas of grassland and woodland over the years, grassland has been converted mainly from bare ground, and woodland has been converted mainly from grassland. There has been a significant reduction in the area of bare ground, which has been converted mainly to grassland. Glacier and permanent snow cover has also decreased, mainly converting to bare ground. The areas of land use types transferred out in descending order was as follows: bare land, grassland, glaciers and permanent snow cover, forests, water bodies, and cultivated land. Conversely, that of land use transferred in was as follows: grassland, bare land, forests, glaciers and permanent snow cover, water bodies, and cultivated land. In this transformation process, there is a significant exchange of land use between bare ground and grassland, with a clear mutual conversion trend. The area of bare ground resulting from the conversion of ice and snow landscapes and grasslands in 2020 accounted for the highest proportion.

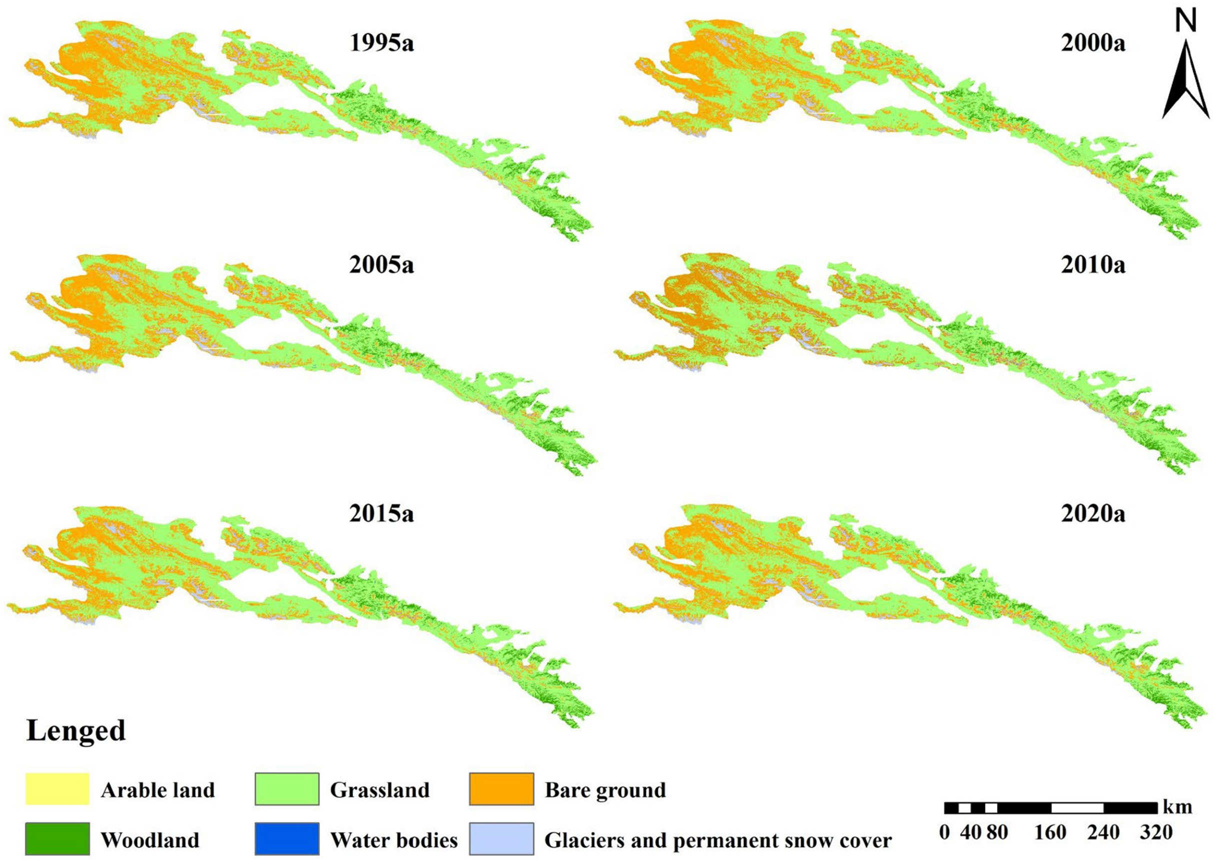

FIGURE 3
Land use types of Qilian Mountain National Park from 1995 to 2020.

TABLE 2 Land use type transfer matrix in the study area during 1995–2020 (unit: km^2).

Year		Area of each land use type in 2020						
		Arable land	Woodland	Grassland	Water bodies	Glaciers and permanent snow cover	Bare ground	Total
Area of each land use type in 1995	Arable land	51.63	2.92	27.92	0.21	0.00	0.09	82.77
	Woodland	0.55	2216.16	122.31	0.00	0.00	0.00	2339.03
	Grassland	46.62	414.83	26875.11	22.11	19.29	1867.12	29245.08
	Water bodies	0.00	0.10	5.97	43.23	7.39	34.26	90.95
	Glaciers and permanent snow cover	0.00	0.00	4.07	22.68	1484.31	284.66	1795.73
	Bare ground	1.59	0.20	3665.34	19.19	262.48	13038.85	16987.66
	Total	100.39	2634.21	30700.74	107.42	1773.47	15224.98	50541.21

3.2 Analysis of landscape pattern index changes

The landscape pattern index for different land cover types in the Qilian Mountain National Park in 1995, 2000, 2005, 2010, 2015, and 2020 were calculated using Fragstats 4.2 software. Table 3 reveals noteworthy patters regarding the patch number and fragmentation of different land types. Arable land initially experienced a decrease in patch number and fragmentation, followed by an increase, although the subsequent increase in fragmentation was not significant, indicating an overall reduction in fragmentation. Conversely, forests, grasslands, water bodies, glaciers and permanent snow cover, and bare land exhibited fluctuations in patch number and fragmentation with periods of decrease, increase, and subsequent decrease. However, the overall trend indicates a decrease in the level of fragmentation.

TABLE 3 Calculation results of landscape pattern indices for different land use types.

Landscape type	Year	Number of patches	Degree of fragmentation	Degree of segmentation	Degree of fractal dimension	Degree of disturbance
Arable land	1995	7,355	0.0886	1.0000	1.3729	0.6189
	2000	5,961	0.0718	1.0000	1.3646	0.6088
	2005	5,195	0.0626	1.0000	1.3604	0.6034
	2010	4,982	0.0600	1.0000	1.3654	0.6031
	2015	5,382	0.0649	1.0000	1.3684	0.6061
	2020	5,707	0.0688	1.0000	1.3505	0.6045
Woodland	1995	44,419	0.5353	1.0000	1.3785	0.8434
	2000	40,723	0.4907	1.0000	1.3646	0.8183
	2005	41,800	0.5037	1.0000	1.3618	0.8242
	2010	42,103	0.5074	1.0000	1.3567	0.8250
	2015	40,876	0.4925	1.0000	1.3596	0.8182
	2020	39,372	0.4739	1.0000	1.3498	0.8069
Grassland	1995	187,766	2.2627	0.9176	1.4435	1.6953
	2000	158,492	1.9099	0.9165	1.4313	1.5162
	2005	140,560	1.6938	0.9105	1.4205	1.4042
	2010	115,126	1.3874	0.9004	1.4227	1.2484
	2015	120,373	1.4506	0.8995	1.4194	1.2790
	2020	100,037	1.2055	0.9078	1.3624	1.1476
Water bodies	1995	20,355	0.2453	1.0000	1.4354	0.7097
	2000	15,807	0.1905	1.0000	1.4138	0.6780
	2005	19,963	0.2406	1.0000	1.4150	0.7033
	2010	19,676	0.2371	1.0000	1.4132	0.7012
	2015	13,919	0.1677	1.0000	1.4141	0.6667
	2020	11,175	0.1347	1.0000	1.3800	0.6434
Glaciers and permanent snow cover	1995	22,416	0.2701	1.0000	1.3776	0.7106
	2000	20,505	0.2471	1.0000	1.3426	0.6921
	2005	27,763	0.3346	1.0000	1.3987	0.7470
	2010	34,257	0.4128	1.0000	1.4040	0.7872
	2015	28,235	0.3403	0.9999	1.4005	0.7502
	2020	12,438	0.1499	0.9999	1.3136	0.6376
Bare ground	1995	152,031	1.8321	0.9486	1.4432	1.4893
	2000	133,568	1.6096	0.9575	1.4420	1.3805
	2005	129,000	1.5545	0.9631	1.4414	1.3545
	2010	149,512	1.8018	0.9786	1.4423	1.4829
	2015	130,328	1.5705	0.9777	1.4347	1.3655
	2020	96,555	1.1636	0.9802	1.3739	1.1506

It is evident that the disturbance index displays an overall decreasing trend. Significant differences in disturbance levels are observed among different landscape types within the study area, with grassland and bare land exhibiting the highest disturbance indices. When comparing the disturbance indices with those of the previous year, slight increases in disturbance are noticeable for cultivated land and grassland in 2015, for forests and glaciers and permanent snow cover in 2005 and 2010, for water bodies in 2005, and for bare land in 2010. These findings suggest a slight increase in external disturbances to various ecological landscapes during those specific years. Landscape fractal dimension index fluctuated across different years, but the overall trend showed a decrease, indicating a reduction in the complexity of patch shapes.

In summary, from 1995 to 2020, the patch number, fragmentation, and disturbance indices for different landscape patterns in the study area exhibited an overall decreasing trend. This indicates a reduction

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in landscape fragmentation and complexity of patch shapes, resulting in increased resistance to disturbances and enhanced landscape stability.

3.3 Analysis of changes in landscape ecological risk

The study area was classified into low risk (ERI < 0.01), relatively low risk (ERI = 0.01–0.02), moderate risk (ERI = 0.02–0.04), relatively high risk (ERI = 0.04–0.08), and high risk (ERI > 0.08), using the natural break method and the landscape ERI (calculated from Equation 1). This classification allowed for the generation of an ecological risk level map and facilitated the assessment of changes in the area for each risk level.

With Figure 4 and Table 4, it becomes apparent that the study area is predominantly characterized by low risk, which are extensively distributed and exhibit a substantial annual increase in area. Notably, an expansion of low risk can be observed in the central and western regions of the study area in 2000 and 2010. Furthermore, from 2015 to 2020, a significant transfer from relatively low risk to low risk occurred on a large scale in the western region.

Relatively low risk zones also have a broad distribution, primarily concentrated in the western region. However, a gradual decrease was observed in the area of relatively low risk zones over the years, as revealed in Table 4. The transfer from relatively low risk to low risk zones has occurred gradually, with the most significant transfer taking place between 1995 and 2000 and between 2015 and 2020. The southern glaciers and permanent snow cover regions, bare land areas, and the northern border areas characterized by bare land, arable land, and forests are predominantly occupied by moderate risk zones. An increase in the area of moderate risk zones was observed in 2000, with notable growth in the southwestern region (Figure 4). The southeastern region experienced a transfer from moderate to relatively high risk. From 2005 to 2010, the area of moderate risk decreased in the southwestern region but increased in the northern region, transferring to relatively high risk. Areas characterized by high and relatively high ecological risks occupy a smaller area and are primarily located in the southwestern edge regions with glaciers and permanent snow cover and the forested regions in the eastern regions. The relatively high risk area experienced a great increase in area in 2010 (Figure 4). Nevertheless, the expansion of relatively high risk zones was effectively controlled in 2015 and 2020, resulting in a reduction in area. The area of high-risk zones fluctuated in different years but remained relatively stable overall.

Combining the aforementioned analysis with Table 5, it can be observed that from 1995 to 2020, the area of low-risk zones significantly increased, while the areas of other risk levels decreased. This overall trend indicates a transfer toward lower risk levels, with the most notable transition observed from relatively low-risk to low-risk zones. In conclusion, the ecological risk within the study area exhibited a consistent and significant decrease over time.

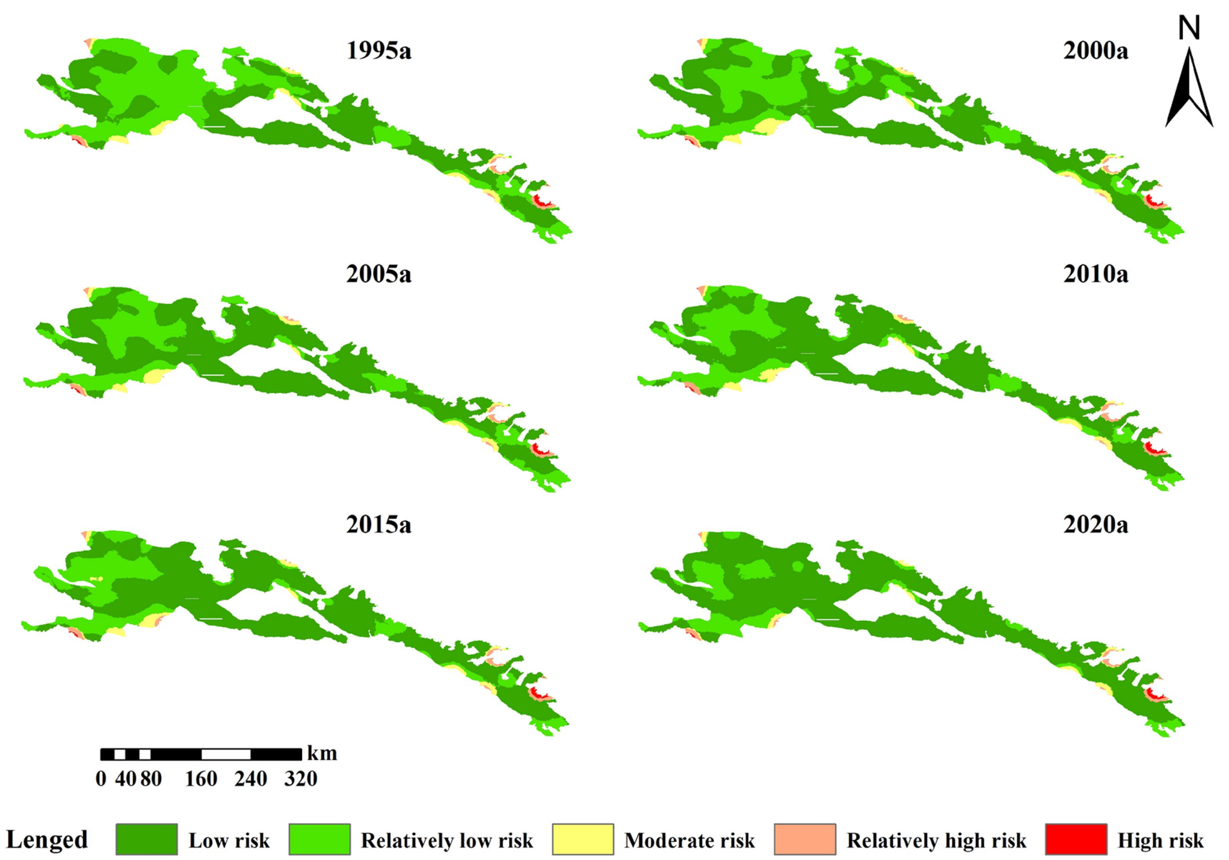

FIGURE 4
Ecological risk level of Qilian Mountain National Park from 1995 to 2020.

TABLE 4 Area of the region with different ecological risk levels in the study area (unit: km²).

Risk level	1995	2000	2005	2010	2015	2020
Low risk	25296.21	31150.77	33544.99	35116.630	35354.71	40949.67
Relatively low risk	22908.00	16821.80	14464.70	12880.20	12728.10	7799.92
Moderate risk	1534.40	1844.40	1696.24	1540.52	1574.24	1003.56
Relatively high risk	619.72	558.68	659.52	802.84	715.70	619.20
High risk	182.92	165.64	175.84	201.04	172.67	168.96

TABLE 5 Ecological risk transfer matrix in the study area from 1995 to 2020 (unit: km²).

		2020					
		Low risk	Relatively low risk	Moderate risk	Relatively high risk	High risk	Total
1995	Low risk	24165.55	1130.56	0.00	0.00	0.00	25296.11
	Relatively low risk	16771.50	6123.52	13.00	0.00	0.00	22908.02
	Moderate risk	12.52	545.84	910.80	65.24	0.00	1534.40
	Relatively high risk	0.00	0.00	79.76	539.72	0.24	619.72
	High risk	0.00	0.00	0.00	14.24	168.68	182.92
	Total	40949.57	7799.92	1003.56	619.20	168.96	50541.21

3.4 Spatial autocorrelation analysis

3.4.1 Global spatial autocorrelation analysis

The values of Moran's I values were calculated using the GeoDa software to analyze the spatial distribution of landscape ecological risk from 1995 to 2020, as depicted in Figure 5. The Moran's I provides insights into the average spatial clustering of similar ecological risk attributes within the study area. The figure reveals that the values of Moran's I for all six periods were positive but relatively small, indicating the presence of spatial positive correlation, although not statistically significant. This suggests that areas with similar ecological risk levels tend to exhibit spatial clustering. Overall, the values of Moran's I displayed a decreasing trend from 1995 to 2020, with a slight increase in 2015 but with minimal variation. This indicates a decrease in the spatial clustering of ecological risk within the study area. Notable, the most substantial decrease in the values of Moran's I was found between 2015 and 2020.

3.4.2 Local spatial autocorrelation analysis

Local spatial autocorrelation analysis was performed using GeoDa software to generate LISA maps with a significance level of 0.05 for each period, as depicted in Figure 6. These LISA maps illustrate the spatial clustering patterns of ecological risk within the study area. Meanwhile, the Figure 6 reveals that the "high-high" clusters primarily appeared in the peripheral areas and regions characterized by elevated ecological risk. The number of clustering areas generally decreased over time, with exceptions observed in 2000 and 2015. Moreover, in 2020, the "high-high" clusters disappeared from the upper portion of the study area. Throughout the period spanning from 1995 to 2020, the study area mainly featured "low-low" clusters, indicating that areas with low ecological risk were surrounded by neighboring areas exhibiting similarly low ecological risk levels. Furthermore, in 2000, there was a notable increase in "low-low" clusters in the western and central regions. However, in 2005 and 2010, the number of "low-low" clusters decreased in the western region but increased in the central region. Nevertheless, the overall number of "low-low" clusters exhibited a decreasing trend but increased slightingly in 2015 and 2020, which was mainly concentrated in the central region. Considering the land use types and corresponding risk levels, it becomes apparent that the "low-low" cluster structure was primarily observed in grassland areas characterized by low ecological risk. Such observation indicates a higher level of internal landscape stability within these regions. Conversely, the "high-high" cluster structure was predominantly found in forested areas, water bodies, and glaciers and permanent snow cover, which were associated with medium to high ecological risk. This suggests lower internal landscape stability within these areas.

4 Discussion

The growing attention to the ecological environment by the government has led to increased research interest in landscape ecological risk within the academic community. Investigating landscape ecological risk allows for the evaluation and analysis of ecological risk heterogeneity from both temporal and spatial perspectives, providing valuable data for ecological conservation endeavors. The variations of Land use type reflect the regional environmental changes and directly affect ecosystem health (Cantarello et al., 2011). The study area is dominated by grassland and bare land, which are also the two land use types that have changed most significantly in the last 20 years. The expansion of grassland and woodland could be attributed to various ecological projects implemented by the government, such as the "Natural

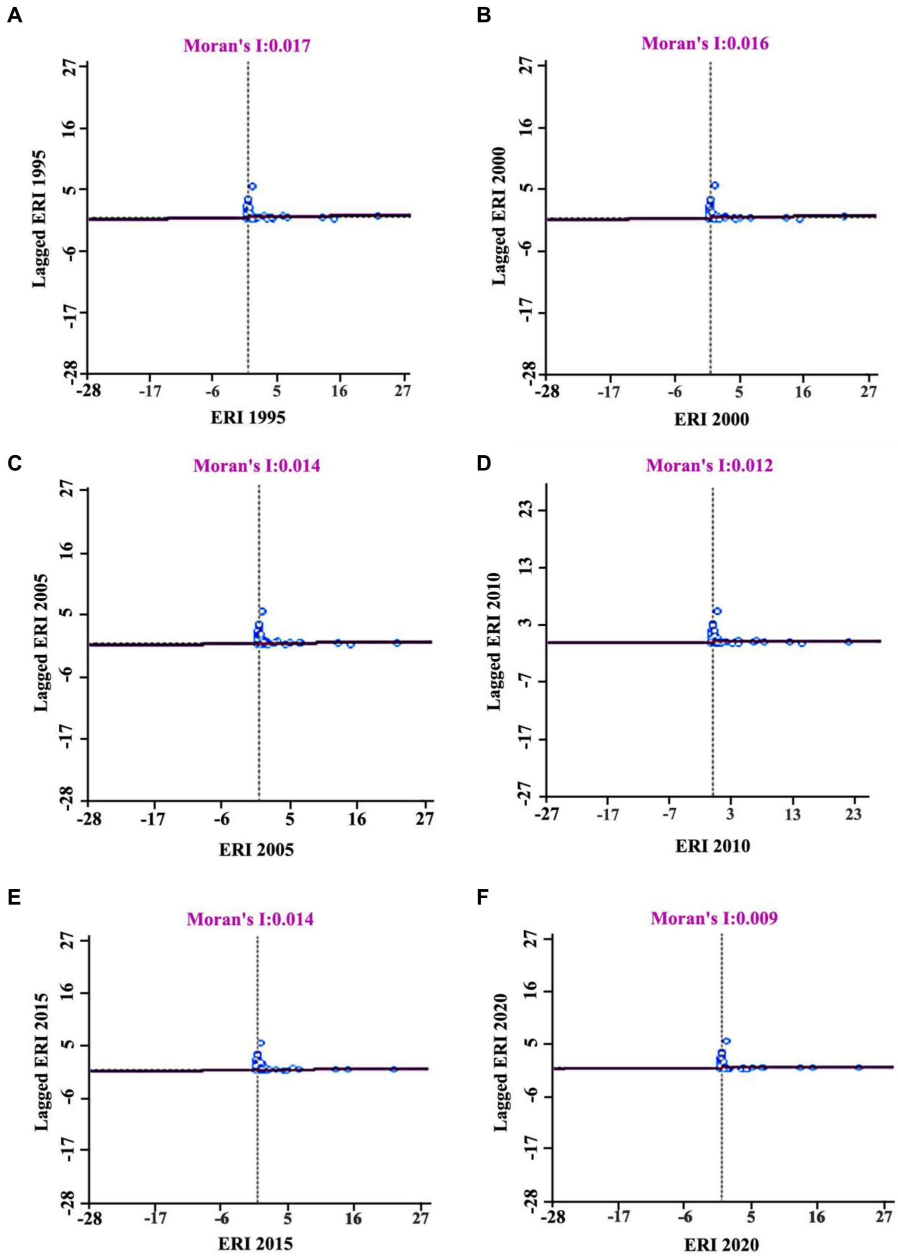

FIGURE 5
(A–F) Scatterplots of the Moran's I of the Ecological Risk Index(ERI) in Qilian Mountain National Park from 1995 to 2020.

Forest Protection Project," the "Return of Cultivated Land to Forests," and the "Three North Protection Forest Project" (Mu et al., 2022; Liu et al., 2023; Xue et al., 2023). In addition, a series of ecological restoration policies after the establishment of the National Park Reserve in 2017, including land restoration in mining areas and the conversion of cultivated land back to forests and grasslands, have resulted in a significant transfer from bare land to grassland. Global warming has caused the melting of glaciers and permanent snow at altitudes above 4,000 m, and has also directly transformed glacial snow-covered areas into bare ground (Xiao et al., 2021; Liu et al., 2023). Meanwhile, 1867.12 km^2 of grassland has been transformed into bare ground due to human unreasonable development and utilization, over-grazing and other factors (Peng et al., 2022).

Landscape fragmentation index, landscape fractal dimension index and landscape disturbance index in the study area showed a decreasing trend, increased disturbance resistance, significant improvement in landscape stability, gradual decrease in ecological risk, and high spatial similarity. From 1995 to 2015, the landscape pattern and ecological risk of the Qilian Mountain fluctuated mainly due to the climate warming, frequent occurrence of extreme weather events, and severe impacts of human activities such as mining and indiscriminate logging. Since the establishment of the Qilian Mountain National Park, significant grassland degradation due to the

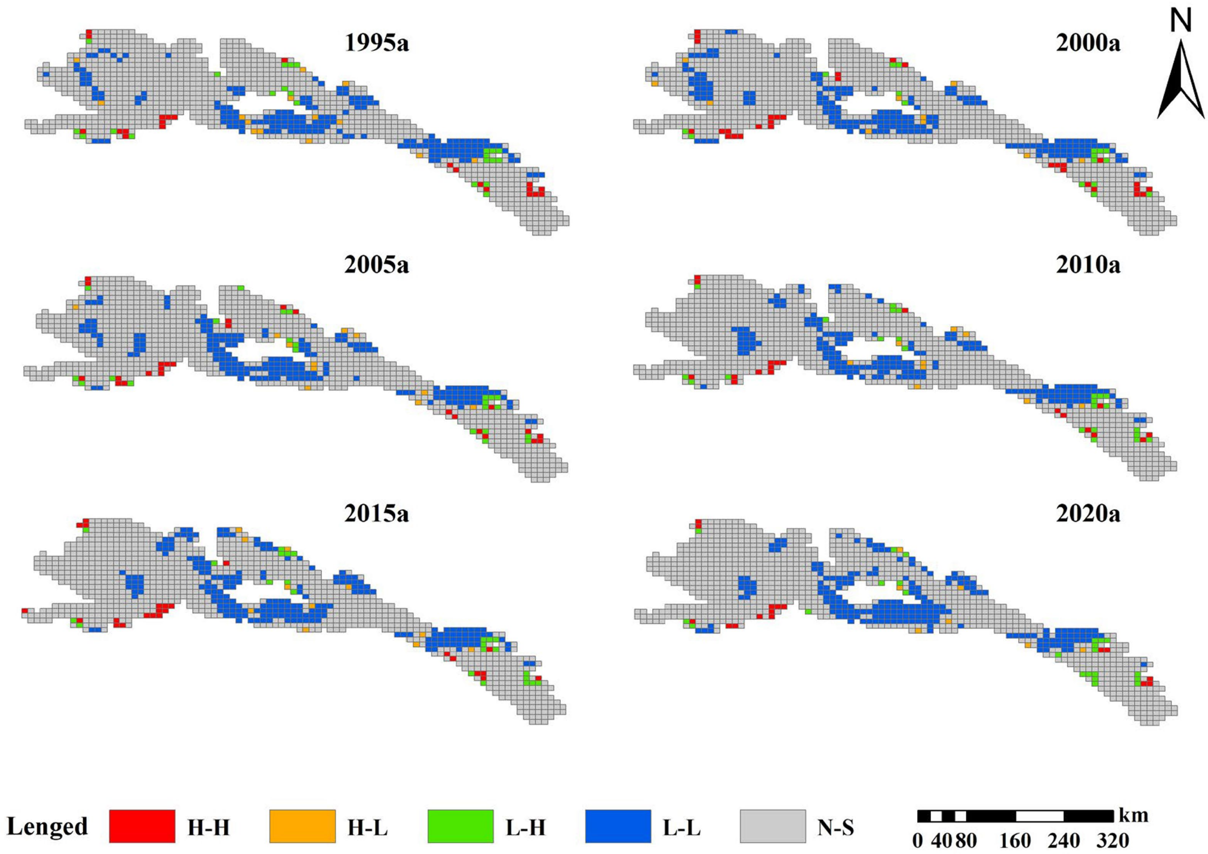

FIGURE 6
LISA maps of Qilian Mountain National Park from 1995 to 2020.

proliferation of wild animals in the region, and the population explosion of herbivorous animals, such as wild deer and rock sheep (Liu et al., 2023). However, a series of ecological protection and restoration measures have had a positive impact on mitigating and controlling ecological damage. As a result, the ecological risk level of the study area has been significantly reduced and the ecological environment has gradually gotten better during the period of 2015–2020. Yu et al. (2022) found that the landscape fragmentation index of woodland in Qilian Mountain increased significantly, and ecological risk increased due to indiscriminate logging, unorganized mining, etc. (Yu et al., 2022). The reason for the difference with the results of this study may be the inconsistency between the land use type classification standard and the ecological risk level classification standard. Meanwhile, the study period of Yu et al. (2022) is 2000–2018, Qilian Mountain National Park has just been established, and the effect produced by ecological measures has not yet appeared.

Although the ecological risk in the study area is dominated by low ecological risk zones, the area of the moderate risk zones has been significantly reduced in 2020. Some areas in the southern and northern regions have already transitioned to relatively low risk. The ecology of the study area is moving in a good direction, but which is still fragile, as found by Liu et al. (2023), the study area experienced the highest risk of desertification during the period of 2015–2020 (Liu et al., 2023), due to the rapid conversion of the snow/ice area to barren due to increased temperatures, and the risk of soil erosion in grassland areas with high topographic relief was exacerbated due to the increased erosive power of rainfall in the context of warming and humidification (Du et al., 2022; Zhang et al., 2022). In the future, Qilian Mountain National Park should be in line with the principle of appropriate to herd animals and forests, respect the self-repair of nature and so on, to strengthen ecological education, rodent and pest management, and rotational grazing. Comprehensive planning and integrated management of the Qilian Mountain National Park, and do our best to harmonize development with the natural environment.

5 Conclusion

Based on the analysis of land use data and the calculation of ERI using landscape pattern indices, the ecological risk in the Qilian Mountain National Park from 1995 to 2020 was analyzed and visualized. The main findings are as follows:

(1) From 1995 to 2020, the study area was primarily characterized by grassland and bare land as the dominant land use types. Ecological engineering and a series of ecological restoration policies after the establishment of the National Park Reserve have resulted in a massive shift from bare land to grassland. However, as a result of global warming, glaciers and permanent snowpack continue to melt into bare land.

(2) The landscape fragmentation and disturbance indices exhibited different patterns in different years, but the overall trend was

decreasing from 1995 to 2020. This indicates an improvement in landscape stability in the study area. Among the different landscape types, grassland and bare land exhibited the highest disturbance indices.

(3) The landscape ecological risk of Qilian Mountain National Park shows a decreasing trend, and the study area is primarily characterized by low ecological risk, while the high-risk areas are relatively small in proportion and mainly distributed in the eastern edge regions. Although the areas of moderate risk, relatively high risk, and high risk fluctuated from 1995 to 2000, effective ecological restoration and protection measures have led to a reduction in their areas and a continued expansion in the area occupied by low ecological risk zones by 2020. Meanwhile, there is a positive spatial correlation of ecological risk in the study area, and the landscape ecological risk mainly shows a "low-low" clustering structure.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

YL: Writing – original draft. QQ: Conceptualization, Writing – review & editing. DW: Data curation, Writing – review & editing. WA: Formal analysis, Writing – review & editing. XH: Conceptualization, Writing – review & editing. TY: Data curation, Writing – review & editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

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Elevation gradients alter vegetation attributes in mountain ecosystems of eastern Himalaya, India

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The present study describes how vegetation (the tree layer) is shaped along the elevation gradients in the eastern part of the Indian Himalayan Region. Various vegetation attributes, distribution, population structure, and regeneration patterns of 75 tree species belonging to 31 families were studied. Tree species richness shows a low plateau (peaked between 1,300–1,500m) with a linearly decreasing pattern above 1,500m asl. Ericaceae was found as the dominant family, followed by Lauraceae and Rosaceae. The distributional pattern of species-to-genera ratio (S/G) did not follow any particular trends, while β-diversity increased along the elevation gradient. The Margalef index of species richness, the Menheink index of species richness, and the Fisher alpha were found to be highest at lower altitudes (1,000–1,500m), while the Simpson index was highest at middle altitudes (2,600–3,000m). Random distribution was shown by maximum tree species (47.3%), followed by a contagious distribution (42.9%), and regular distribution (10.8%). The regeneration of tree species was found to be good with a healthier number of seedlings (10.2%), fair (43.5%), poor (30.3%), while 16% were observed not regenerating. *Acer laevigatum* (1,500m), *Prunus nepalensis* (3,300m), *Viburnum sympodiale* (3,400m) were among the new regenerating species at the respective altitudes. The population structure of tree species in terms of proportion of individuals in seedlings, saplings, and the adult class varied in all the elevation transects. Species with better regeneration on upper distribution limits have been recognized as probable for upward movement.

KEYWORDS

elevation gradient, species richness, population structures, regeneration, conservation

1 Introduction

Mountain ecosystems cover about 27% of the world's land surface and support approximately one-quarter of terrestrial biological diversity (Korner and Spehn, 2019). Mountain ecosystems are characterized by steep environmental gradients, including temperature, pressure, and moisture (Spehn, 2011; Antonelli et al., 2018). Abiotic and biotic factors influence the patterns of diversity and distribution of taxa along altitudinal gradients (McCain and Grytnes, 2010). Along an elevation gradient, environmental variables directly affect species composition, growth patterns, and ecosystem functioning, which leads to a change in the vegetation composition (Guo et al., 2013; Krömer et al., 2013; Malizia et al., 2020). The elevation regulates several abiotic factors (i.e., soil parameters, atmospheric pressure, humidity, cloudiness, solar radiation, light availability, pH, etc.) that control the composition of vegetation and the ecology of mountain forests (Cirimwami et al., 2019). Species from different taxa, families, and life forms respond specifically to these factors according to their eco-physiological properties and sensitivity (Vetaas and Grytnes, 2002; Korner, 2007; McCain and Grytnes, 2010).

Among mountain ecosystems across the globe, Asia has the highest and most populated mountain ecosystem—the Himalaya. Besides, it is endowed with an overwhelming richness, representativeness, and uniqueness of biodiversity (Rawal et al., 2018). Elevation and abiotic factors are the governing drivers for differences in species richness and composition in the Himalaya (Lee and Chun, 2016; Pandey et al., 2018a). The species composition depends directly on temperature and air pressure, which decrease along the elevation gradient (Whittaker et al., 2001; Bhattarai and Vetaas, 2006; Wani et al., 2023). It is well established that diversity declines linearly along the elevation gradient (MacArthur, 1972; Korner, 2000). However, recent studies highlight that plant diversity often peaks at mid-elevations (Rahbek, 2005; Kessler et al., 2011). This may vary among taxa and mountain ranges (Cardelus et al., 2006; Körner et al., 2017).

The elevation gradient in the Himalayan region support diverse vegetation types, from tropical forest to alpine meadow, comprised of an unusually extensive elevation and vegetation gradient (Singh et al., 1994; Tang et al., 2014; Wani et al., 2022). Among different life forms, tree species are an essential component of forest composition. Tree diversity is reported to be responsible for structural complexity and environmental heterogeneity in the mountain ecosystem (Gaston, 2000; Oommen and Shanker, 2005). Tree dynamics, particularly distribution along the elevation gradient, are important to understand in biodiversity conservation studies and ecological processes (Gaire et al., 2014). Among various ecological processes, regeneration of tree species is a crucial process for their existence in a community under varied environmental conditions (Negi et al., 2018b). The regeneration pattern depicts the current status of forest health and indicates the future composition of a particular community (Tesfaye et al., 2002; Gairola et al., 2014; Negi et al., 2018a). The size-class distribution of forest trees is a prime indicator of forest structure and dynamics and is widely used to examine the forest's health, including regeneration and recruitment of species (Forrester et al., 2017; Bhutia et al., 2019). The stand density and basal area are excellent indices for estimating forest biomass and carbon sequestration potential.

The eastern part of the Himalayan biodiversity hotspots (Mittermeier et al., 1999; Mayer et al., 2000) is among the most diverse regions with high endemism (Behera et al., 2002). However, basic knowledge on the structure and composition of the Himalayan forests is still limited in many regions, particularly in the remote eastern parts (Chakraborty et al., 2018). Arunachal Pradesh and Sikkim are known for having the highest species diversity in the north-eastern Himalayan region (Wani et al., 2024). Few studies attempted to assess the floristic diversity pattern in the eastern Himalayan region along the elevation gradient (Behera and Kushwaha, 2006; Acharya et al., 2011; Sinha et al., 2018). However, the range of altitudinal amplitude among these studies was very limited because of the remoteness and difficult terrain. To fill this lacuna, the present study explores the distribution of woody taxa along the elevation gradients in the region. A wide range of elevation gradients (1,000–4,000 m) from subtropical forests to treeline ecotones (sub-alpine forests) has been covered in the study. The objectives of the present study include: (i) assessment of species richness, distribution pattern and relative dominance of tree species, and (ii) study the population structure and regeneration pattern of tree species, along the elevation gradients.

2 Methodology

2.1 Study area

Indian Himalayan Region (IHR) extends across nine states fully, hilly districts of two states (Assam, and West Bengal) and 02 Union Territories (Ahirwal et al., 2021). It is broadly divided into two zones; Western Himalaya and Eastern Himalaya, and represents about 16.2% of the total geographical area, 44% of the total biodiversity, and 100% of the alpine and glacial systems of the country. In present study, the upper Teesta valley of Sikkim state from Singtam (1,000 m asl) in East to Yumesamdong (4,000 m asl), Lachung North Sikkim considered as a representative site for intensive field studies along a wide elevation gradient (1,000–4,000 m asl). The dissected topography and significant variations in climatic conditions along the elevation gradient resulted in diverse vegetation types. The major forest types along the studied transect includes tropical moist deciduous forest, tropical semi-evergreen forest, and tropical wet evergreen forest, subtropical forests, temperate forests, subalpine forests and alpine meadow (Prabha and Jain, 2016).

2.2 Methods

Standard quadrat method was used for vegetation sampling along the elevation gradient. We established 31 sites between 1,000–4,000 m in the upper Teesta valley keeping a minimum of 100 m elevation difference between two sites (Figure 1). Random sampling method was used to investigate the vegetation along the elevation gradient and 50 m × 50 m plots were marked randomly at each site. Within each 50 m × 50 m plot, ten random quadrats (10 m × 10 m) were laid for enumerating trees (adult), twenty (5 m × 5 m) for saplings and forty (1 m × 1 m) for seedlings, respectively (Figure 1), following previous studies from the Himalayan region (Reshi et al., 2017; Pandey et al., 2018b). Besides, for calculating different diversity indices and data analysis, the whole altitudinal gradient (1,000–4,000 m asl) was divided into six major Forest Types (FT). These include 1,000–1,500 (FT 1-deciduous-broadleaf forest), 1,600–2,000 (FT 2-deciduous-broadleaf forest), 2,100–2,500 (FT 3-deciduous-broadleaf forest), 2,600–3,000 (FT 4-broadleaf-evergreen forest), 3,000–3,500 (FT 5-broadleaf-evergreen and coniferous-evergreen forest) and 3,600–4,000 (FT 6-coniferous-evergreen forest—Supplementary Table S1).

The forest structure and composition were determined following Misra (1968) and Mueller Dombois and Ellenberg (1974). Tree species diversity was determined using the Shannon-Wiener index [H′ = −Σpi

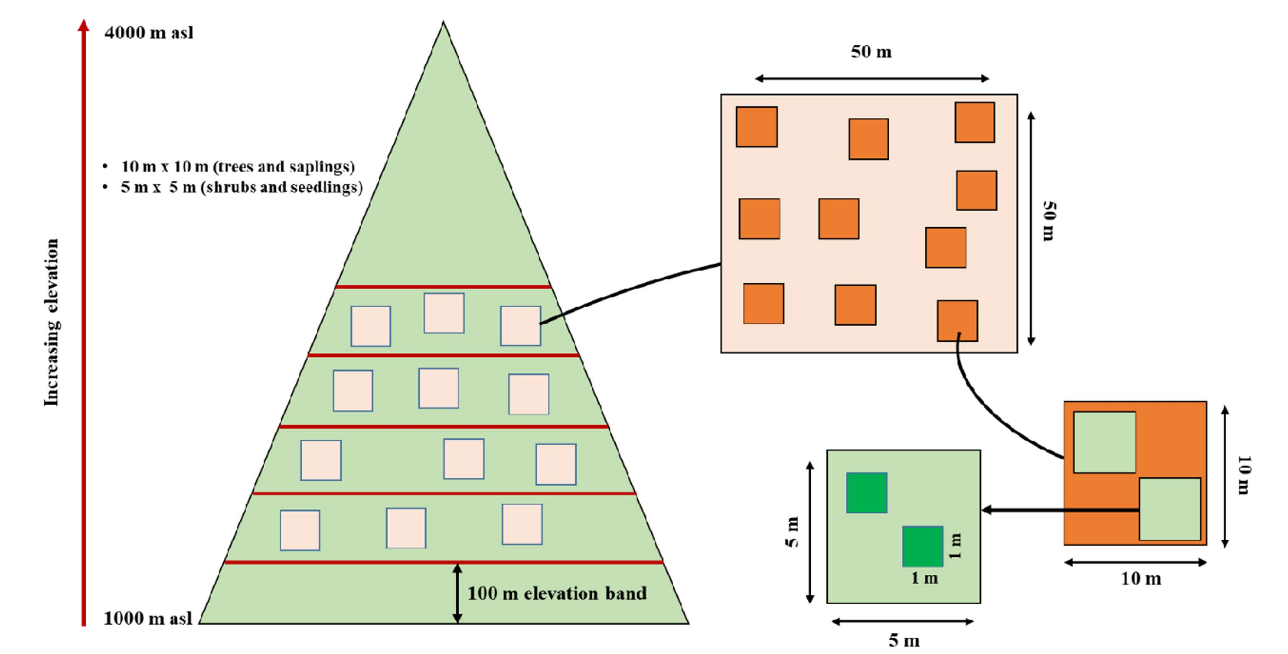

FIGURE 1
Pictorial diagram of altitude transect sampling method and representation of sample quadrat and plots within the altitude transect.

* ln (pi)] where ln: natural log; pi: proportion of the entire community made up of ith species (Shannon and Weaver, 1963). Shannon diversity provides information on both species' richness and relative abundance among plots and is thus sensitive to the sampling effort or the number of individuals sampled. The ratio of abundance to frequency (A/F) for different species was determined to elicit distribution patterns following Whitford (1949). This ratio has indicated regular (<0.025), random (0.025–0.05) and contagious (>0.05) distribution patterns. Circumference at breast height (cbh = 1.37 m) was taken to determine tree basal area (Odum, 1971). The basal area of all trees within the quadrats was calculated at each elevation band and the values were summed up to obtain the basal area (m^2/ ha) of the elevation site following Greig-Smith (1983). Basal area was used to determine the relative dominance of a tree species (Curtis and McIntosh, 1950). Beta diversity (β) was measured using the Whittaker (1975) formula as given in Mena and Vazquez-Dominguez (Mena and Vázquez-Domínguez, 2005): $\beta = (s/\alpha) - 1$; where "α" is the mean number of species per altitude belt and "s" is the total number of species recorded across the study system (i.e., altitude). We also estimated species richness, genus richness, and family richness for the overall forest community along the altitudinal gradient. We also estimated essential stand structural parameters: the median DBH, basal area (m^2/ha), and size class distribution of forest trees.

The regeneration status of dominant trees was assessed based on the proportional distribution of individuals in each seedling, sapling, and adult stages (Saxena and Singh, 1984; Khan et al., 1987): Good regeneration is considered if number of seedlings > saplings >trees; fair regeneration is considered if number of seedlings > or ≤ saplings≤ trees, or if the species survives only in sapling stage, but no seedlings (saplings may be <, > or = trees) following Bhandari (2020) and Wani and Pant (2023). If a species is present only in tree form (absent seedling and sapling stages), it is considered not regenerating, while species having no trees but only seedling stages are considered as "new" species. Individuals of tree species measuring >30 cm diameter (diameter at breast height—130 cm above ground level) were considered trees, individuals between >11–30 cm diameter as saplings, and individuals <10 cm diameter as seedlings. The basic Circumference at Breast Height (CBH) information of individual trees generated from each quadrat was used for the development of population structures. The individuals of each tree species were grouped into seven arbitrary CBH classes (A: <10; B: 10–30; C: 31–60; D: 61–90; E: 91–120; F: >121–150; G: >150 cm) for generating demographic profiles following Saxena and Singh (1984). Class A and B represent seedlings and saplings, respectively, and other classes (C-G) represent trees with different girth size classes.

Regression analysis was used to study the relationship between elevation and species richness using SPSS version 10.0. For Alpha diversity, the total number of species at each transect was calculated and analyzed. Correlations were developed using SPSS_16 software. The Sorenson similarity index was calculated using the Estimate S_9 software. Quadratic models were fitted between altitude and species distributions. The selection of the quadratic model was made based on its performance.

3 Results

The result reveals quantitative detail on the availability, distribution, population density, and abundance of 75 tree species belonging to 31 families (Figure 2). A total of 3,439 tree individuals were measured in 93 plots along the elevation gradient. Ericaceae was found to be the dominant family (14 spp.), followed by Lauraceae (7 spp.) and Rosaceae (5 spp.).

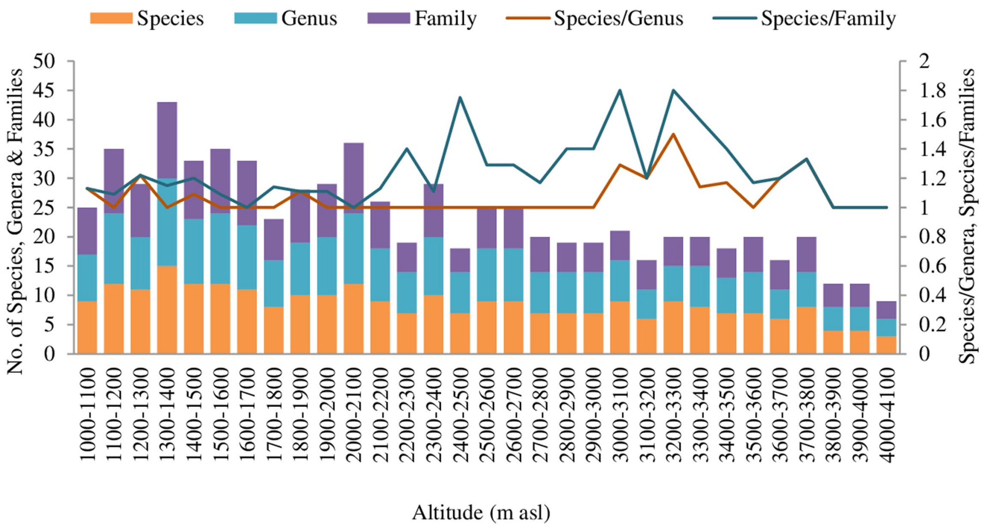

FIGURE 2
Floristic diversity pool across altitudinal gradient in Teesta valley, Sikkim.

3.1 Species richness and diversity measures

The distribution of tree species at each elevation level is provided in Table 1. The Margalef Index of Species Richness, Menheink Index of Species Richness and Fisher Alpha were found to be maximum at lower altitudes (1,000–1,500 m), while the Simpson Index was found to be maximum at middle (2,600–3,000 m) altitudes. Species richness indices in different forest types along the elevation in Teesta Valley are given in Table 2. A significant decreasing trend is observed for most diversity indexes (MeI, MI, FA; Table 3). Species richness was found to be maximum (Curtis and McIntosh, 1950) at 1,300 m followed by 1,500 m (Cirimwami et al., 2019) and minimum (Antonelli et al., 2018) between 1,200–1,500 m (Figure 3A). Species richness with respect to forest type was recorded maximum (39 species) in FT1 and minimum (9 species) in FT 6 (Supplementary Table S1). Tree species richness shows a low plateau (peaked between 1,300–1,500 m) with a linear decreasing pattern above 1,500 m (Figure 3A), while species diversity (H′) was observed maximum (2.3) at 1,300 m and minimum (0.7) at 4,000 m (Figure 3B). Considering the importance of β-diversity in conservation planning, it is evaluated separately for each study site along the elevation gradient (Figure 3C). β-diversity has a positive relationship with increasing elevation. It was recorded maximum for high altitude sites; > 3,400 m (Figure 3C). Pearson correlation coefficients between various phytosociological parameters are presented in Table 3. Berger Paker Index and β-diversity showed a positive correlation, and SR, D, H, MeI, MI, while FA showed a negative correlation with the elevation.

3.2 Species distribution pattern, stand basal area and density

Rhododendron arboreum, Alnus nepalensis, Syzygium balsameum, Cryptomeria japonica, Viburnum cordifolium, Salix babylonica, Daphnephyllum himalayense, and *Populus ciliata* showed regular distribution along the Teesta valley. Random distribution was shown by the maximum species (47.31%), followed by a contagious distribution (42.92%), and minimum (10.77%) by regular distribution. *Abies densa, Alnus nepalensis, Betula alnoides, Rhododendron arboretum, Mallotus nepalensis* and *Schima wallichii* have a wider distribution, with appearances at eleven, ten, nine, six, and five sites, respectively. Total basal area (TBA) was found to be maximum (47.48 m^2/ha) at 1,101–1,200 m followed by 41.30 m^2/ha (3,100–3,201 m) and 40.52 m^2/ha (3,000–3,101 m), and minimum (10.88 m^2/ha) at 2,300–2,401 m (Table 1). The average range of TBA was found to be between 10.88 and 47.48 m^2/ha along the elevation gradient in the valley (Figure 3D); it was found to be maximum for *Abies densa* at higher altitudes. There was no specific pattern observed for tree density; it was found to be maximum (1,253 Ind/ha) at 3,100 m and minimum (473 Ind/ha) at 2,100 m (Figure 4A); maximum density was observed for *Abies densa* (693 Ind/ha) at 3,500 m.

3.3 Regeneration pattern

The overall seedling density with relation to altitude ranged between 473–3,060 Ind/ha, while the sapling density ranged from 260–1,760 Ind/ha (Figures 4A,B). Maximum seedlings (3,060 Ind/ha) were observed at 1,100 m, and minimum (473 Ind/ha) at 2,900 m (Figure 4C). The regeneration of tree species was found to be fair (43.5%) at most of the attitudes (i.e., 1,100–1,200, 1,400–1,500), followed by poor (30.3%) at 1,000–1,100, 1,200–1,300, and not regenerating (14%) at 1,100–1,200, 1,300–1,400, while it was found to be good (10.2%) at higher altitudes (3,200 m, 3,700 m and 3,800 m; Supplementary Table S2). The results further reveal that 14.2% of tree species are not regenerating at their respective elevations. Regeneration of *Abies densa* was found to be good at higher altitudes (>3,400 m), and of *Salix babylonica* at middle altitude (2,300–2,800 m). *Acer laevigatum* (1,500 m), *Terminalia myriocarpa* (1,500 m) *Prunus nepalensis* (3,300 m), *Viburnum sympodiale* (3,400 m) are among the new regenerating species in the respective elevation, as they were found only in the seedling stage.

TABLE 1 Distribution of tree species along elevation gradient in Teesta valley (Sikkim).

Elevational transects (m asl)	Tree species in a particular 100 m transect	Total number of individuals (Ind/hectare)	TBA
1,000–1,100	*Castanopsis hystrix, Actinodaphne sikkimensis, Holmskioldia sanguinea, Ostodes paniculatus, Schima wallichii, Castanopsis tribuloides, Calophyllum polyanthum, Lyonia ovalifolia, Sapondias axillaris*	866.7	36.20
1,101–1,200	*Schima wallichii, Syzygium balsameum, Castanopsis hystrix, Sarcosperma arboreum, Barchemia floribunda, Erythrina stricta, Ostodes paniculatus, Macaranga indica, Elaeocarpus lanceaefolius, Brassaiopsis mitis, Morus laevigata, Engelhardtia spicata*	993.3	47.48
1,201–1,300	*Schima wallichii, Castanopsis hystrix, Mallotus roxburghianus, Alnus nepalensis, Castanopsis tribuloides, Engelhardtia acerifolia, Ficus bengalensis, Saurauia napaulensis, Prunus cerasoides, Phoebe attenuate, Eurya japonica, Castanopsis tribuloides*	900.0	24.91
1,301–1,400	*Schima wallichii, Mallotus nepalensis, Rhus insignis, Engelhardtia acerifolia (spicata), Syzygium balsameum, Daphnephyllum himalayense, Erythrina arborescens, Alnus nepalensis, Acer oblongum, Terminalia myriocarpa, Machilus edulis, Prunus cerasoides, Castanopsis tribuloides, Juglans regia, Albizia chinensis*	793.3	26.98
1,401–1,500	*Mallotus philippensis, Schima wallichii, Castanopsis hystrix, Syzygium balsameum, Engelhardtia acerifolia, Prunus cerasoides, Castanopsis tribuloides, Machilus edulis, Rhus insignis, Daphnephyllum himalayense, Litsea cubeba, Eurya japonica*	806.7	25.49
1,501–1,600	*Alnus nepalensis, Mallotus nepalensis, Engelhardtia acerifolia, Terminalia myriocarpa, Betula alnoides, Andromeda elliptica, Machilus duthiei, Albizia chinensis, Magnolia pterocarpa, Ficus nemoralis, Terminalia myriocarpa, Glochidion triandrum, Acer laevigatum*	653.3	36.89
1,601–1700	*Mallotus nepalensis, Alnus nepalensis, Rhus insignis, Cedrela toona, Ficus nemoralis, Pentapanax leschenaultia, Acer laevigatum, Erythrina arborescens, Angiopteris evecta, Cryptomeria japonica, Prunus cerasoides*	773.3	33.04
1,701–1,800	*Alnus nepalensis, Mallotus nepalensis, Engelhardtia acerifolia, Acer laevigatum, Betula alnoides, Leucosceptrum canum, Rhus insignis, Prunus cerasoides*	620.0	13.18
1,801–1,900	*Alnus nepalensis, Engelhardtia acerifolia, Erythrina arborescens, Rhus insignis, Eurya japonica, Acer laevigatum, Mallotus nepalensis, Sarcosperma arboreum, Rhus chinensis, Ficus nemoralis*	653.3	11.41
1,901–2,000	*Cryptomeria japonica, Prunus cerasoides, Alnus nepalensis, Acer laevigatum, Engelhardtia acerifolia, Rhus insignis, Barchemia floribunda, Saurauia grimthii, Betula alnoides, Leucosceptrum canum*	620.0	25.24
2,001–2,100	*Alnus nepalensis, Rhus insignis, Mallotus nepalensis, Leucosceptrum canum, Prunus nepalensis, Engelhardtia acerifolia, Cryptomeria japonica, Erythrina arborescens, Andromeda elliptica, Cedrela toona, Glochidion triandrum, Ficus hookeri*	680.0	14.34
2,101–2,200	*Alnus nepalensis, Rhododendron arboreum, Cryptomeria japonica, Debrigeasia longifolia, Litsea cubeba/ Actinodaphne citrate, Erythrina arborescens, Acer laevigatum, Juglans regia, Betula utilis*	473.3	12.07
2,201–2,300	*Rhododendron arboreum, Alnus nepalesis, Salix daltoniana, Acer laevigatum, Picea smeathiana, Populus ciliate, Lyonia villosa*	573.3	14.86
2,301–2,400	*Rhododendron arboreum, Salix babylonica, Cryptomeria japonica, Betula alnoides, Alnus nepalensis, Erythrina arborescens, Acer cambelli, Evodia fraxinifolia, Litsae citrate, Viburnum cordifolium, Zanthoxylum acanthopodium*	493.3	10.88
2,401–2,500	*R. arboreum, Populus ciliate, Salix babylonica, Alnus nepalensis, Acer cambelli, Betula alnoides, Zanthoxylum acanthopodium, Acer cambelli, Lyonia ovalifolia*	720.0	14.65
2,501–2,600	*Tsuga dumosa, Acer cambelli, Salix babylonica, Alnus nepalensis, R. arboreum, Litsae citrate, Evodia fraxinifolia, Betula alnoides, Lyonia ovalifolia*	673.3	19.52
2,601–2,700	*Tsuga dumosa, Salix babylonica, Alnus nepalensis, Cupress* sp., *Prunus nepalensis, Evodia fraxinifolia, Lyonia ovalifolia, R. arboreum, Betula alnoides*	700.0	19.55
2,701–2,800	*Tsuga dumosa, Salix babylonica, Daphnephyllum himalayense, Viburnum cordifolium, Larix griffithii, R. arboreum, Abies densa, Juniperus recurva*	620.0	13.23
2,801–2,900	*Daphnephyllum himalayense, Salix babylonica, Larix griffithii, R. arboreum, Viburnum cordifolium, Juniperus recurva, Abies densa, Tsuga dumosa*	580.0	12.41
2,901–3,000	*Daphnephyllum himalayense, Salix babylonica, Larix griffithii, R. arboreum, Tsuga dumosa, Viburnum cordifolium, Abies densa, Juniperus recurva*	593.3	15.57
3,001–3,100	*Tsuga demosa, Larix griffithii, R.arboreum, Abies densa, Daphnephyllum himalayense, Rhododendron niveum, R. grefianum, Salix babylonica, Viburnum cordifolium, Juniperus recurva, Viburnum sympodiale*	606.7	40.52

(Continued)

TABLE 1 (Continued)

Elevational transects (m asl)	Tree species in a particular 100 m transect	Total number of individuals (Ind/hectare)	TBA
3,101–3,200	*Abies densa, Rhododendron niveum, Viburnum cordifolium, Betula utilis, Lyonia villosa, viburnum sympodiale, Prunus nepalensis*	1253.3	41.13
3,201–3,300	*Abies densa, Rhododendron hodsonii, Lyonia villosa, Betula utilis, Rhododendron nevium, Acer pectinatum, Acer campbellii, Prunus nepalensis, Rhododendron arborium, R. grandii*	820.0	20.17
3,301–3,400	*Abies densa, Lyonia ovalifolia, Larix griffithii, Rhododendron falconery, Salix babylonica, Rhododndron hodgsonii, Prunus nepalensis, Betula alnoides*	760.0	26.75
3,401–3,500	*Abies densa, Larix griffithii, Lyonia ovalifolia, Prunus rufa, Betula utilis, Viburnum sympodiale, Betula alnoides*	660.0	16.73
3,501–3,600	*Abies densa, Viburnum sympodiale, Sorbus microphylla, Prunus rufa, Betula alnoides, Acer campbellii, Lyonia ovalifolia*	993.3	30.24
3,601–3,700	*Abies densa, Rhododenderon hodgsonii, Betula alnoides, Acer campbellii, Salix babylonica, Betula utilis*	793.3	13.76
3,701–3,800	*Salix babylonica, Abies densa, Betula utilis, Rhododendron fulgens, Rhododendron lanatum, Acer cambelli, Betula alnoides, Sorbus macrophylla*	960.0	28.42
3,801–3,900	*Abies densa, Rhododendron lanatum, Betula utilis, Sorbus microphylla,*	940.0	21.31
3,901–4,000	*Abies densa, Salix babylonica, Rhododendron lanatum, Betula utilis*	693.3	12.19
4,001–4,100	*Abies densa, Sorbus microphylla, Salix babylonica*	660.0	18.63

TABLE 2 Species richness indices in different forest types in Teesta valley, Sikkim, eastern Himalaya.

Altitude	SR	SD	TBA	D	H′	MeI	MI	FA	BP
FT 1 (1,000–1,500)	39	802	31.13	0.86	2.62	1.74	6.11	9.89	0.32
FT 2 (1,600–2,000)	22	668	17.70	0.83	2.27	1.20	3.61	5.29	0.34
FT 3 (2,100–2,500)	21	582	13.20	0.86	2.38	1.23	3.53	5.19	0.30
FT 4 (2,600–3,000)	16	620	15.46	0.89	2.39	0.91	2.62	3.58	0.17
FT 5 (3,000–3,500)	18	896	24.63	0.67	1.82	0.85	2.79	3.76	0.56
FT 6 (3,600–4,000)	9	808	17.28	0.63	1.49	0.45	1.33	1.63	0.59

SR, Species Richness; SD, Stem Density; TBA, Total Basal Area; H, Shannon–Wiener diversity index; D, Simpson Index; MI, Margalef Index of Species Richness; MeI, Menheink Index of Species Richness; FA, Fisher Alpha; BP, Berger Paker Index.

TABLE 3 Correlation among various parameter in the Teesta valley, Sikkim, eastern Himalaya.

	Altitude	SR	SD	TBA	D	H′	MeI	MI	FA
Altitude	1								
SR	−0.890*	1							
TBA	−0.369	0.712	1	1					
D	−0.780	0.561	0.724	−0.089	1				
H	−0.886*	0.771	−0.740	0.183	0.956**	1			
MeI	−0.960**	0.965**	−0.721	0.510	0.721	0.882*	1		
MI	−0.917*	0.997**	−0.188	0.659	0.611	0.809	0.982**	1	
FA	−0.910*	0.998**	−0.877*	0.666	0.597	0.795	0.976**	0.998*	1
BP	0.615	−0.378	−0.004	0.230	−0.971**	−0.873*	−0.545	−0.429	−0.415

*Significant at the 0.05 level; **Significant at the 0.01 level. SR, Species Richness; SD, Stem Density; TBA, Total Basal Area; H, Shannon–Wiener diversity index; D, Simpson Index; MI, Margalef Index of Species Richness; MeI, Menheink Index of Species Richness; FA, Fisher Alpha; BP, Berger Paker Index.

3.4 Population structure

The population structure of the tree community and the distribution of seedlings, saplings, and adults in the forest varied along the elevation gradient (Figure 5). The high accumulation of saplings of dominant species (*Schima wallichii, Castanopsis hystrix,* and *Engelhardtia acerifolia*) between 1,000–1,500 m, the sharp decline in seedlings, and high tree-size class were characteristic of

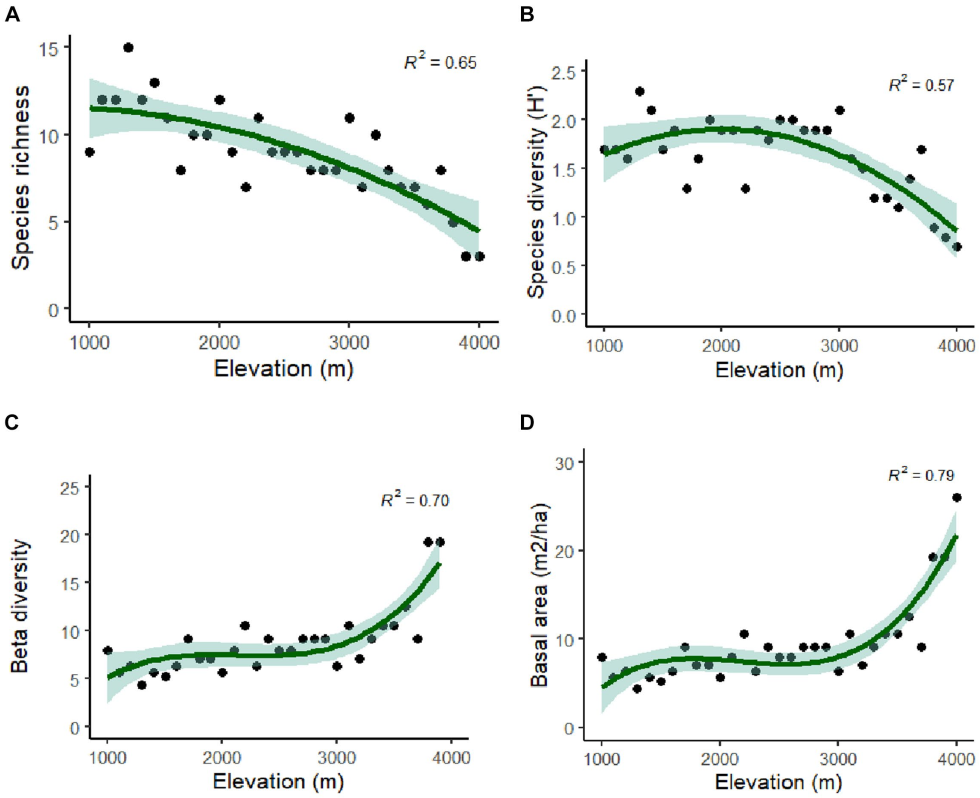

FIGURE 3
Polynomial regression analysis **(A)** between elevation and tree species richness; **(B)** tree species diversity and elevation; **(C)** between altitude and Beta diversity; and **(D)** between altitude and TBA.

this altitudinal transect (Figure 5A). However, the overall composition of tree species forms a reverse j-shaped structure, which is reported as a progressive population structure. *Alnus nepalensis* showed a reverse hill-shaped structure with high seedlings, a sudden decline in saplings, and a further increase in the higher girth size class between 1,600–2,000 m. *Mallotus nepalensis* showed a very low number of seedlings and a sudden increase in saplings and higher girth class, while *Engelhardtia acerifolia* showed a hill-shaped structure with an accumulation of individuals at saplings stage, and decline in seedling and higher girth classes (Figure 5B). Population structure showed a greater proportion of individuals (*Rhododendron arboreum, Alnus nepalensis*) in the seedling and sapling stages between 2,100–2,500 m, and a decline in tree-size classes formed a reverse j-shaped structure (Figure 5C). *Salix babylonica* showed a hill-shaped structure with an accumulation of saplings, and a decline in seedlings and higher girth classes. The higher proportion of individuals *(Salix babylonica, Larix griffithii)* in the sapling stage was evident between 2,600–3,000 m (Figure 5D). At the temperate zone (3,100–3,500 m), the dominant species (*Abies densa, Betula utilis*) showed a higher number of individuals in sapling stages and adult classes, and decreasing numbers in higher tree-size classes form a bell-shaped structure (Figure 5E). Population structure at treeline ecotone (3,500–4,000 m) shows a greater proportion of individuals in the sapling stage (*Betula utilis, Rhododendron nevium*) and a decline in seedlings and higher tree-size classes form a hill-shape structure (Figure 5F). *Abies densa* showed a progressive population structure with sufficient seedlings, saplings, and individuals in higher girth size classes.

4 Discussion

The patterns of species-genus ratio (S/G) at the local scale in light of the hypothesis were described as part of evolutionary dynamics, wherein these ratios are related to speciation or diversification rates (Krug et al., 2008). The altitudinal decrease in the S/G ratio of trees in the Teesta valley would imply their phylogenetic over-dispersion towards the highest altitudes. Species diversity is the most crucial indices, which not only captures information on species richness but also indicates the relative abundance of species in a forest (Shaheen et al., 2012). Higher species richness between 1,300–1,500 m may be because of the presence of a slightly warmer temperature, and hence these species are localized only to specific habitats (Jetz and Rahbek, 2002; Sinha et al., 2018). In a study, Singh et al. (1994) observed an increase in tree species richness with elevations up to 1,500 m in mixed *Pinus roxburghii* broadleaved forests in central Himalaya. In the mountains, this trend is well reported due to various environmental factors, including decreasing temperature and air pressure along an elevation gradient (McCain and Grytnes, 2010).

It is well established that elevation itself is not causing any changes in the distribution of species but is influenced by climate and environmental factors (Bhattarai and Vetaas, 2006; Krömer et al.,

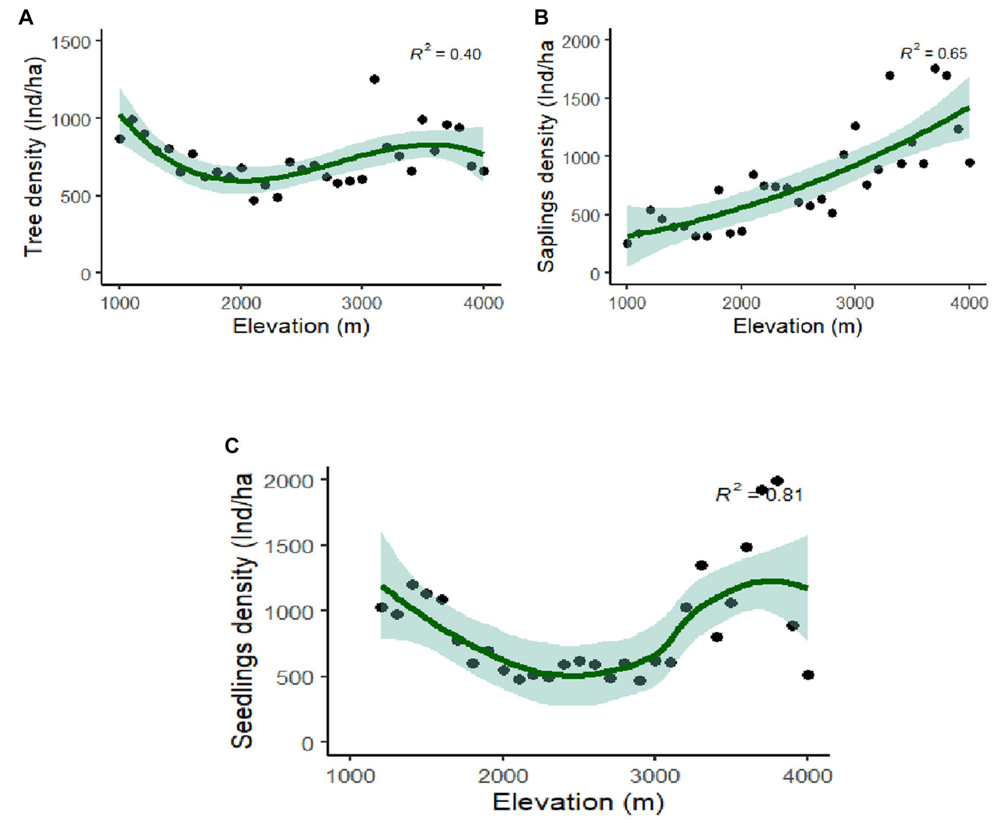

FIGURE 4
Polynomial regression analysis between altitude and density **(A)** tree; **(B)** saplings; and **(C)** seedlings.

2013). In a review, Rahbek (2005) showed that a decrease in species richness along the elevation gradient is not the rule, as the result of the review indicates that approximately half of the studies had a mid-elevation peak in species richness. Tree species diversity decreases with increasing elevation due to the fact that ecosystems at higher elevations have been mainly colonized by species that are tolerant of extreme weather conditions (Gaston, 2000; Trigas et al., 2013). The range of diversity (0.7–2.3) is consistent with the value reported in previous studies from the region (Singh et al., 1994; Behera and Kushwaha, 2006; Acharya et al., 2011; Pandey et al., 2018a). The average value of species diversity was analysed between 1.5 and 3.5 in the present study, which is comparable with the previous studies from the region [Ghildiyal et al., 1998 (1.86–2.73); Uniyal et al., 2010 (0.70–3.08); Raturi, 2012 (0.78–3.45); Pant and Sammant, 2012 (0.74–2.66)].

The range of the Simpson index (1-D) was found between 0 and 1 according to Simpson (1949), where zero represents no diversity and 1 represents maximum diversity. Among other diversity indices, Whittaker (1977) and Clifford and Stephenson (1975) demonstrated Margalef's diversity index (D_{Mg}) and Menhinick's index (D_{Mn}) as two of the best known of species richness indices. The results of D_{Mg} and D_{Mn} are consistent with the earlier studies. The result of β diversity in our study is consistent with Magurran (2004), according to which β diversity increases in heterogeneous landscapes and declines in homogenous ones. The patterns of β diversity differ considerably across the sites, and the recorded maximum is for high-altitude ones. B-diversity patterns revealed that the species replacement rate was lower in the mid-altitude compared to lower and higher altitudes. Adjacent transects along the altitudinal zone showed low differences in the values of β diversity, which indicates species composition does not vary significantly across adjacent forest types. The range of Concentration of dominance (Cd) in the study region (i.e., 0.1–0.6) is more or less similar to the values (0.19–0.99) reported by Whittaker (1977), however higher as compared to previous studies from western Himalaya [Gairola et al., 2011 (0.12–0.25); Raturi, 2012 (0.09–0.63); Malik and Bhatt, 2015 (0.06–0.37)].

The distribution pattern of tree species in any forest ecosystem indicates its adaptability to various environmental drivers. In our study, random distribution was shown by the maximum number of species in the temperate zone, followed by contagious distribution, and the minimum by regular distribution. The species in temperate zones have been reported (Körner, 1998) to have larger environmental tolerances and may follow a random distribution, as also found in the present study. The contagious distribution pattern was more evident in the tropical zone, may be due to narrower tolerances to environmental variation and being likely to be affected by a steep temperature gradient (Oommen and Shanker, 2005). According to Odum (1971), contagious distribution has been accepted as a characteristic pattern of plant occurrence in nature. It may be due to an insufficient mode of seed dispersal or when the death of trees creates a large gap, encouraging the recruitment and growth of numerous saplings (Richards, 1996; Hubbell et al., 1999).

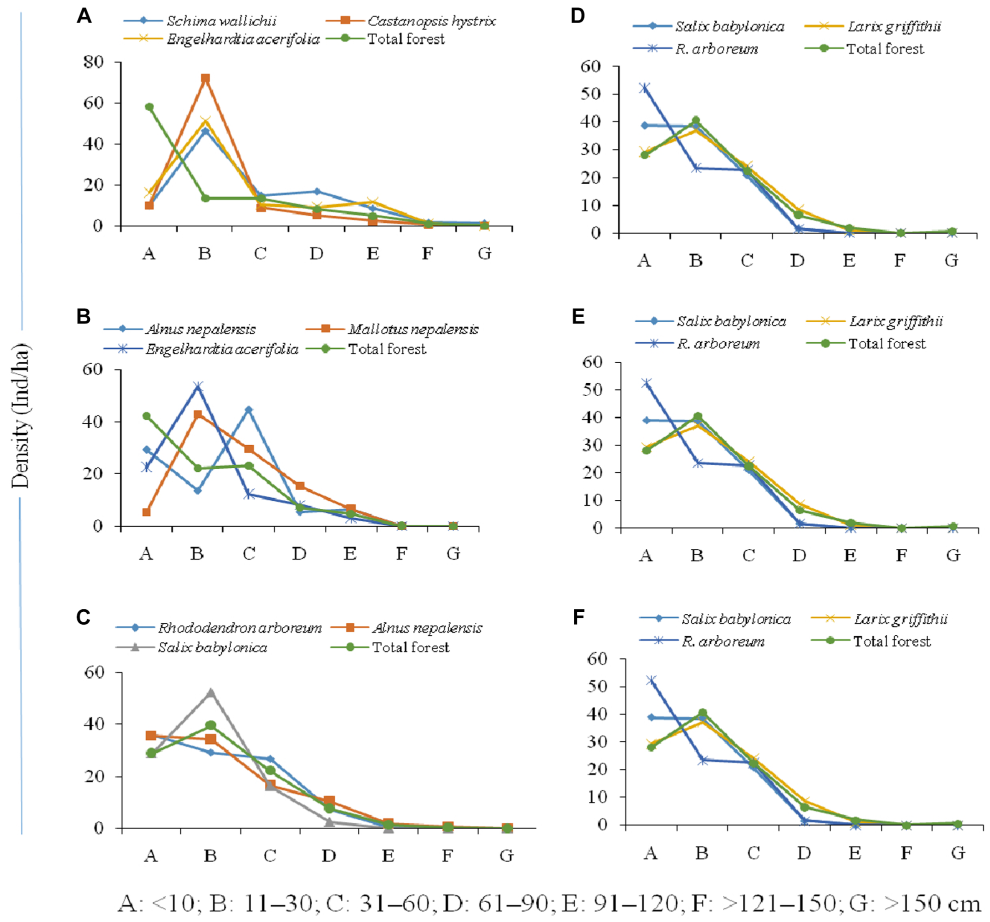

FIGURE 5
Population structure of forests [**(A)** 1,000–1,500, **(B)** 1,600–2,000, **(C)** 2,100–2,500, **(D)** 2,600–3,000, **(E)** 3,100–3,500, **(F)** 3,600–4,000 m] across altitudinal strata.

Among species having a wider distribution, *Abies densa* forms the dominant forest >3,000 m to 4,100 m, and *Schima wallichii* between 1,000–1,400 m. Many species such as *Actinodaphne sikkimensis*, *Barchemia floribunda*, and *Brassaiopsis mitis* are unable to extend their ranges beyond certain elevation bands probably due to their narrow tolerance to climatic variations (Shaheen et al., 2012).

The regeneration of *Rhododendron* species and *Abies densa* was seen to be good, with a relatively greater number of seedlings, saplings, and individuals in higher girth classes above 3,000 m. The overall increase in the density of dominant species (i.e., *Abies densa*) at higher elevation compensates for the reduction in the number of rare species, thereby increasing overall density, as also reported by Scott (1976). The variation in tree density along the elevation transect is attributed to forest community type, age, and site characteristics (Gaire et al., 2010; Pandey et al., 2018b). According to Acharya et al. (2011), high density at higher elevations is an adaptation of species to withstand cold climatic conditions and strong wind currents, thus opting for alpine refugia (Gentili et al., 2015). Tree density in temperate coniferous forests (>3,000 m) was significantly higher as compared to sub-tropical or broad-leaved forests, which may be due to the better regeneration potential of Pinaceae (Begon et al., 2006). *Abies densa* showed greater recruitment on the upper distribution limit (>3,900 m) in EH, and *Betula utilis* in WH indicated shifting of the species towards higher altitude.

The range of Total Basal Area (TBA) in our study is similar to those reported in previous studies: Gairola et al. (2011) (35.08–84.25 m^2/ha); Raturi (2012) (3.18–43.62 m^2/ha); Pandey et al. (2016) (10.43–248.41 m^2/ha); Malik and Bhatt (2015) (10.49–42.92 m^2/ha) from various parts of IHR. The maximum number of individuals of any species in a particular forest, if represented by higher diameter classes, is considered the population of that species on the verge of population decline (Benton and Werner, 1976). The dominance of tree individuals in medium to lower diameter classes suggests that the forest is still in an evolving stage. TBA of tree species was found to be comparatively low compared to previous studies, which may be due to (i) the sparsely distribution of tree species with low girth classes (Gaire et al., 2010; Rai et al., 2012), (ii) human disturbance (Benton and Werner, 1976; Rawal et al., 2018), and (iii) higher rates of mortality relative to tree growth (Acharya et al., 2011). The relatively lower basal cover of the trees >3,500 m may be the result of the higher density of young individuals and the effect of cold and harsh climatic conditions.

Size class distribution of trees provides the population structure of forests (Saxena and Singh, 1984), and is extensively used to understand regeneration status (Veblen, 1997). The health of any forest depends on the regeneration potential of tree species and their proportional distribution among different age classes (seedlings, saplings, juveniles, and trees) in space and time (Enright and Watson, 1991; Negi and Maikhuri, 2017; Rawal et al., 2023). The proportion of different life stages of tree species facilitates possible future forest composition and also indicates their stability (Gairola et al., 2011; Negi et al., 2018a). In our study, a fair regeneration pattern was exhibited at most sites, but was also found to be poor for many species, particularly in sub-tropical zone. The poor regeneration at lower altitudes near settlements can be attributed to various environmental factors, including anthropogenic activities. Heavy browsing by animals at seedling and sapling stages is also responsible for poor recruitment of seedlings (Negi et al., 2018b). The presence of a higher density of seedlings and saplings signifies a healthy, growing forest. Good regeneration of a few species, such as *Abies densa, Betula utilis, Rhododendron lanatum, Rhododendron arboreum,* and *Betula alnoides* at higher altitudes (> 3,200 m), is attributed to low competition among the species and the complete absenence of anthropogenic disturbances. Another reason is the fact that Pinaceae have better regeneration potential compared to other tree species and are less likely to be affected by herbivores, according to Begon et al. (2006). *Acer laevigatum*, *Prunus nepalensis*, and *Viburnum sympodiale* were among the new regenerating species at 2,100, 3,300, and 3,400 m, respectively, indicating a shift of these species towards higher altitudes. Among these, *Viburnum sympodiale* was present at 3,100 m, and the appearance of the seedlings of this species at 3,400 m indicates upward movement of the species. The climatic variables prevailing above 3,500 m and up to 4,000 m are favorable for the growth of *Abies densa*. The species shifting towards higher elevations, coping with the changing climate, can be a factor in the accumulation of a higher number of young individuals in these elevations.

Population structure studies of any forest are keys to understanding the mechanisms of species coexistence, development within the community, and long-term ecological processes (Enright and Watson, 1991; Rai et al., 2012). Further, the size class distribution of trees provides the population structure of forests and is extensively used to understand regeneration status (Bhutia et al., 2019). Tree species that are represented by all girth classes constitute continuous regeneration, while reverse trends constitute discontinuous regeneration (Benton and Werner, 1976). The reverse J-shaped population structure in the present study shows these species are in the most dominant form in the stand and indicate a good regeneration pattern, as indicated in earlier studies (Saxena and Singh, 1984; Tesfaye et al., 2002). The reverse J-shaped distribution of species indicates uneven-aged forests (Vetaas, 2000), with a sufficient number of young individuals to replace the old mature stands. A hill-shaped curve was observed between 2,100–3,500 m due to the accumulation of individuals in the sapling stage and their decline towards both higher tree classes and seedling; this is a scenario undesirable for a sustainable forest. This type of structure indicates replacement in tree size classes, and if the current state of seedling recruitment does not improve, the population may face problems in the long-term (Negi and Maikhuri, 2017; Negi et al., 2018a). One potential factor for the absence of large trees could be the practice of cardamom (*Amomum subulatum* Roxb.) cultivation mainly in the low-altitude forest of Sikkim (Bhutia et al., 2019), as forests are partially cleared for its cultivation (Kanade and John, 2018). Furthermore, the drying of cardamom demands a substantial supply of fuelwood, often generated from the nearby forests in lower elevations.

A large number of juveniles relative to adults indicates that a population is in a stable or growing stage, while a few juveniles suggest a decline trend in the population. The forest structure above >3,500 m is expanding in nature, with more individuals in the seedling and sapling stages, followed by a decline in the tree size classes, indicating the movement of dominant tree species towards higher altitudes. This could be due to the convergence point of two different forest types or ecotones. However, long-term ecological monitoring is required to documents the spatial changes in the forest dynamics and to suggest better management and conservation strategies (Negi et al., 2023). Benton and Werner (1976) reported that the higher number of individuals of any species in higher girth classes represents a declining population of the dominant species in a particular forest stand. Many tree species in the study area showed 'poor' regeneration; even some species were not regenerating at their respective transects, which is a matter of concern.

5 Conclusion

Tree species diversity, size class distribution, and basal area are the essential attributes that describe a forest's ecosystem, and measuring these attributes is fundamental to designing conservation strategies. Our study is able to provide an understanding of the forest composition, diversity, structure, and regeneration of forest trees along the altitudinal gradient. The study concludes the following: (i) patterns of vegetation vary considerably across sites and along elevations, thereby suggesting a stronger influence of micro-level factors and climatic conditions, (ii) tree species richness and diversity shows a low plateau with a linear decreasing pattern above 1,500 m along the elevation gradient, (iii) density did not follow a uniform pattern; however, it was found to be highest at higher altitudes due to the minimum level of anthropogenic pressure and uniform climatic conditions, (iv) tree species exhibits fair regeneration at many sites in the sub-tropical zone, and good regeneration in the temperate zone. Many species were not regenerating, which indicated need for special conservation measures, (iv) disparity in the size class distribution was observed among forests along the altitudinal gradient, (v) β-diversity patterns revealed that the species replacement rate was less in the mid-altitude compared to the lower and higher altitudes, and (vi) a reverse J-shape distribution of tree diameter, signifying the unevenness of age, particularly at lower altitudes. Our study highlights conservation concerns for the low-altitude forests that record high species diversity, although they lack large-diameter trees. Population structure above >3,500 m is expanding in nature, with more individuals in seedling and sapling stages, followed by a decline in tree size classes. This indicates an upward movement of tree species towards higher altitudes. However, long-term monitoring of seedling dynamics would help to predict potential changes in tree species distribution and the stability of the forest ecosystem under climate change regimes.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary material, further inquiries can be directed to the corresponding authors.

Author contributions

VN: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Project administration, Supervision, Writing – original draft, Writing – review & editing. AP: Writing – review & editing. AS: Writing – review & editing, Formal analysis, Methodology. AB: Writing – review & editing, Data curation. DP: Writing – review & editing, Data curation. KG: Software, Writing – review & editing. ZW: Writing – review & editing. JB: Writing – review & editing. SS: Funding acquisition, Writing – review & editing. HY: Writing – review & editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

Supplementary material

The Supplementary material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2024.1381488/full#supplementary-material

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Bibliometric analysis of studies on threat assessment and prioritization of species for conservation

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The present study investigated the evolution and current situation of research on threat assessment and prioritization of species for conservation at a global level by analyzing bibliometrically the most relevant and productive authors, sources, and countries, most cited papers, country collaborations and most frequent keywords as reflected in the scientific literature using the Web of Science database. From 1989–2022, a total of 315 relevant documents were retrieved from 129 sources. Results revealed that since 1989, there has been an increase in the number of publications on threat assessment and prioritization of species for conservation. A total of 1,300 authors have contributed to the field through their research contributions. Among the 129 sources, the journals 'Biodiversity and Conservation' and 'Biological Conservation' are the most relevant and productive. Among countries, the USA has produced the highest number of publications, whereas Benin has the highest Multiple Country Production with a rate of 71.4%. Among the authors, 'Keith DA' has received the most citations, and among the sources, the journal 'Biological Conservation' received the highest number of citations. Conservation, biodiversity, conservation priorities, species richness, and threatened species are the most frequently used keywords and follow power-law distribution. The present study will be useful to the researchers in determining which journals to target and how to identify potential research partners in the concerned field. It is recommended that institutions in developed countries be encouraged to lead research programs in developing and underdeveloped countries so that such studies will be carried out at local, regional, and global scale, as biodiversity loss is a global issue.

KEYWORDS

bibliometric, threat assessment, prioritization, conservation, biological conservation

Introduction

Detailed information on different parameters of biodiversity is vital for ecological stability and balance (Mehta et al., 2020; Wani and Pant, 2023). However, biodiversity at a global level has been undergoing a critical phase owing to various drivers threatening the survival of species (Tilman et al., 2017; Ripple et al., 2019; Nic Lughadha et al., 2020;

Pouteau et al., 2022). A driver is any natural or anthropogenic factor that directly or indirectly causes an alteration in an ecosystem and may threaten biodiversity by increasing extinction probabilities (Chase et al., 2020; Ngodhe, 2021). A direct driver has an unambiguous impact on ecosystem processes, while an indirect driver has a more diffused impact by fluctuating one or more direct drivers (Branquinho et al., 2019). The primary goal of the Convention on Biological Diversity (CBD) and the United Nations Sustainable Development Goals (UNSDGs) is to reduce or control global biodiversity loss and increasing species extinctions; however, success to date has been considerably inadequate (Dad and Rashid, 2022). During the 20*th* century, the earth lost 50% of its wetlands and 40% of its forests, and around 60% of global ecosystem services were halted (Mehta et al., 2020), with a loss of 137 species per day during the later decades of the century (Moram et al., 2011). Species extinction at such a rapid rate is considered 1,000–10,000 times than the natural extinctions in the past (Hilton-Taylor, 2000). Thus, understanding the distribution and composition of species assemblages and their prediction in space and time are extremely imperative errands to look into the providence of biodiversity from the perspective of current global change (Bhat et al., 2020; Rawat et al., 2021; Thakur et al., 2021). In order to develop efficient conservation and management plans for these biodiversity-rich areas, extensive information on species, communities, and habitats is required. Taking note of the biodiversity loss, there is an increasing array of regional, national, and international awareness and policy mechanisms aimed at the conservation of biodiversity globally (Kullberg and Moilanen, 2014).

A well-established method for identify areas with underrepresented biodiversity and taking cost-effectiveness into account when making conservation plans is systematic conservation prioritization (Karimi et al., 2023). Prioritization of species, habitats, and communities for conservation is pre-requisite for biodiversity conservation and management planning at the local, regional, and global scales (Singh and Samant, 2010). The conservation status of taxonomic units must be evaluated locally using the IUCN criteria in an appropriate manner (Rodrigues et al., 2006; Miller et al., 2007; Abeli et al., 2009). Broad, ecosystem-based conservation techniques are also necessary to stop the extinction of species; these strategies don't rely on taxonomic data or identifying specific species, but rather on the composition of local communities and the types of habitats they occupy. When determining where to focus their efforts, organizations working to protect biodiversity are forced to make tough trade-offs (Sinclair et al., 2018). Although experts can offer valuable advice to decision makers, their capacity to handle intricate spatial optimization issues is limited (Martin et al., 2012). Prioritization has therefore been developed to deal with this issue. According to Kukkala and Moilanen (2013), prioritization is the "biogeographic-economic activity of identifying important areas for biodiversity; where, when and how we might efficiently achieve conservation goals." Developing a Conservation Priority Index of unique species, communities, and habitats at local, regional, national, and global levels is a crucial step in planning conservation and management strategies (Wani et al., 2022). Over the past 20 years, spatial prioritizations have aided in the decision-making process regarding land use, forest planning, and conservation (Karimi et al., 2023). Global spatial conservation prioritizations have been carried out with great effort, offering comprehensive insights for global protection in the future. However, nations must decide which species to target for conservation and what national priorities should be set for expanding protected areas if they are to make significant progress toward accomplishing sustainable development goals. However, lack of sufficient funding and rapid biodiversity loss make such programs difficult for the researchers especially in the developing and underdeveloped countries. Thus, such nations should be the hotspots for these studies and sufficient funding should be allocated to these countries. Further, researchers in such countries should identify possible collaborator institutions and countries for their research projects on biodiversity conservation.

Bibliometric analysis studies have played a significant role in science and technology management and decision-making (Aleixandre-Benavent et al., 2018). Such studies are receiving considerable attention, as they provide valuable information on scientific research and its progression in a specific field of study (Aleixandre-Benavent et al., 2018). Several papers have been published employing bibliometric analysis techniques to evaluate a particular subject area or topic of research at qualitative and quantitative level: Tsunami research (Chiu and Ho, 2007; Jain et al., 2021; Suprapto et al., 2022); water research (Wang et al., 2010); biotechnology (Dalpe, 2002; Vain, 2007), deforestation (Aleixandre-Benavent et al., 2018); biodiversity loss (Tan et al., 2022); renewable energy (Rosokhata et al., 2021); ecotourism (Liu and Li, 2020; Khanra et al., 2021); climate change (Wang et al., 2014; Rana, 2020; Fu and Waltman, 2022); and COVID-19 research (Chahrour et al., 2020; Wang and Tian, 2021). In this paper, we perform a bibliometric analysis of published literature on threat assessment and prioritization of plants for conservation for the time period 1989–2022. Present study will add a new perspective to the current status and may help to identify hot spots in the field of global biodiversity conservation. More specifically, the article aims to identify the most relevant and productive sources, authors, institutions and countries, most cited authors, sources, institutions and countries, most influential articles, country collaborations and most frequently used keywords along with their growth trends from 1989–2022 in publications on threat assessment and prioritization for conservation. Present study will be useful to the researchers in determining which journals to target and how to identify potential research partners in the concerned field. Further, it will help researchers, managers, agencies and conservationists for planning better strategies to conserve and manage biodiversity.

Materials and methods

Bibliometrics is a quantitative method characterized by applying statistics and econometrics that draw on publication and citation data to determine the evolutionary structure of a research topic or field (Baker et al., 2020). It not only provides a more reliable analysis but has the potential to launch a systematic, apparent, and reproducible appraisal process based on the statistical measurement of science, scientists, or scientific activity (Aria and Cuccurullo, 2017). Many online bibliographic databases, like Web of Science, Scopus, Google Scholar, and

Science Direct, where metadata concerning scientific works is stocked can be sources of bibliographic information (Cobo et al., 2011). For retrieving the relevant literature in the present study, a search was made from the Web of Science (WoS) database by using the keywords, 'Plant threat assessment' OR 'Conservation prioritization' OR 'Conservation priority'. The database WoS was preferred as it is the most authentic and popular database among academicians (Gulhan and Kurutkan, 2021). Our initial search generated a total of 6,057 documents including research papers, reviews, editorial, letters, and proceedings. 12 review articles, 5 proceedings and 3 editorials were excluded and the remaining documents were screened through titles and abstracts to find out their suitability for our study. As, the main focus of our study was to identify such studies in which species/habitats or communities have been prioritized for conservation, a total of 315 articles formed our final dataset and information of only such papers was retrieved in bibtex format. For carrying out the bibliometric analysis of retrieved published literature on threat assessment and prioritization of plants for conservation, '*bibliometrix*' tool developed through the R programming language was used (Ingale and Paluri, 2020; Zhang et al., 2021; Majiwala and Kant, 2022). It is a state-of-the-art tool that follows the classical bibliometric workflow (Aria and Cuccurullo, 2017). Data was analyzed for the most relevant and productive sources based on number of articles published within a source, authors, affiliations, and countries; top cited articles, authors, sources, and countries; countries collaborations; keyword analysis; and source and keyword growth trends.

Results and discussions

Publication output

From 1989–2022, a total of 6,057 documents were generated in initial search for the keywords, out of which 315 relevant documents were selected after screening of titles and abstracts. The selected scholarly documents were retrieved from 129 sources in the web of science database. A total of 14,285 references have been used in these documents. Since 1989, there has been an increase in the number of publications, from a minimum (01) publication in 1989 to a maximum (24) publication in 2019 and 2021 (**Figure 1**). This may be due to the fact that researchers and policy-makers have understood that there is a need for more reasonable proactive tactics, seeking to categorize and protect at-risk species in a timely manner (Walls, 2018; Le Breton et al., 2019). Still, the enormous species yet to be assessed for extinction risk, coupled with the limited resource availability for such works, necessitates rapid appraisals as a primary step toward recognizing which species, habitats, communities, and areas should be prioritized for conservation. Thus, an approach that identifies at-risk species on the basis of threshold elements of IUCN Red List criteria has the potential to amplify the speed of species prioritization for conservation (Le Breton et al., 2019). However, prioritization is incomplete without consideration of the conservation actions required to conserve the assets at particular locations (Wilson et al., 2009).

Most productive authors, sources, countries, and institutions

Most productive authors, countries, and institutions are the chief indicators in the bibliometric studies that emphasize leading contributors within a particular research topic or field (Sharma et al., 2020). It helps scholars and practitioners looking for collaboration and higher studies in relevant fields (Singh et al., 2021). The present study reveals that a total of 1,300 authors have contributed to the targeted research field through their research contributions. Out of these, 28 have contributed to single-authored documents and 1,272 to multi-authored documents. 'Maxted N' from the University of Birmingham, UK has produced the maximum number of publications (12), followed by 'Bacchetta G' from the University of Cagliari, Italy (08 publications), and 'Fenu G' from the Italian Botanical Society Onlus, Italy (07 publications) (**Table 1**). On examining articles and authors within the framework of Lotka Law, it is revealed that 88.3% authors contributed with a single publication, 8% with 2 publications, 2.5% with 3 publications and only 1.1% authors with more than 3 publications (**Table 2**). Lotka's Law is a power-law distribution that describes the relationship between the number of authors and number of articles published by them (Lotka, 1926). It implies that few authors, (known as core authors) bear responsibility for most of the published articles, while the majority of authors only publish a small number of articles (Ridwan et al., 2023).

Of the 129 sources, the journal 'Biodiversity and Conservation' has been the most relevant and productive with 50 publications, followed by 'Biological Conservation' (39 publications), 'Oryx' (12 publications), 'Conservation Biology' and 'Plos One' (11 publications each), 'Journal for Nature Conservation' (9 publications), and 'Diversity and Distributions' (8 publications) (**Figure 2**). Based on Bradford's law, of the total 129 sources, only four were found to be the core sources (**Figure 3**). Bradford's law is a bibliometric principle that describes the relationship between journals and the articles published on a specific topic. It states that a small group of journals (known as core sources) contain a significant proportion of the articles related to a particular topic (Ridwan et al., 2023). Journals, Biodiversity and Conservation, Biological Conservation, Oryx, and Conservation Biology have been identified as the most important and basic sources for studies on threat assessments and conservation prioritization. These journals have published most number of articles on prioritization of conservation and thus are considered core journals for the topic. 'Biodiversity and Conservation' is an international journal that publishes articles on all aspects of biological diversity, its conservation, and sustainable use. On the other hand, 'Biological Conservation' is a leading international journal in the discipline of conservation science, publishing articles that contribute to the biological, sociological, ethical, and economic dimensions of conservation. Its primary aim is to publish high-quality papers that advance the science and practice of conservation or demonstrate the application of conservation principles and policy. Although the source dynamics analysis revealed that the journal 'Biodiversity and Conservation' remained the most relevant in terms of publications on threat assessments and prioritization for conservation up to the year 2013. But, since 2013, the number of relevant publications in 'Biodiversity and Conservation' has constantly decreased, and on

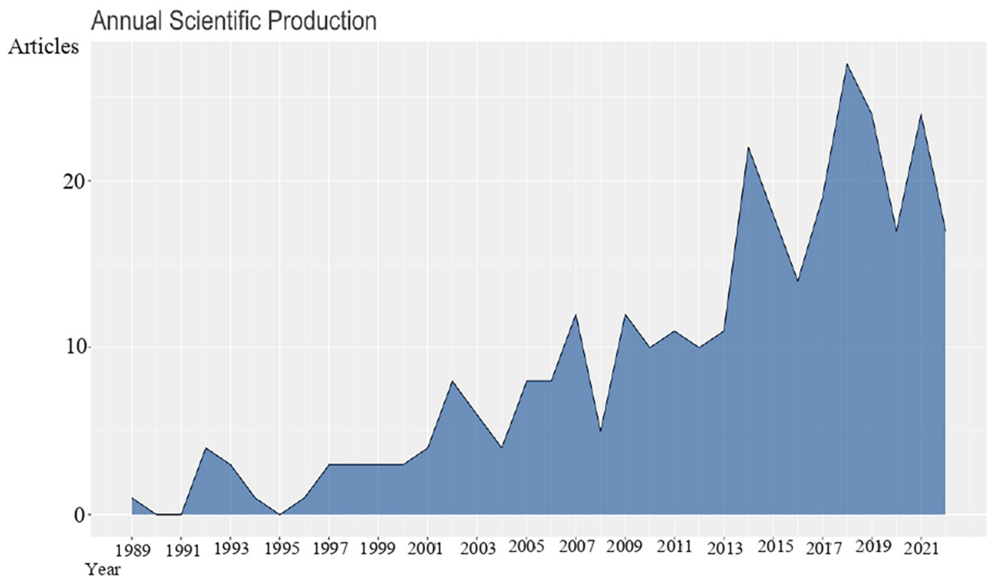

FIGURE 1
Annual scientific production of articles on prioritization for conservation from 1989–2022.

TABLE 1 Top most productive and cited authors from 1989–2022.

Author	h-index	Total Citations	Number of Publications
Maxted N	7	428	12
Bacchetta G	7	328	8
Fenu G	7	269	7
Albuquerque UP	5	100	5
Brehm JM	4	68	5
Domina G	2	148	5
Li J	4	77	5
Achicanoy HA	4	321	4
Araujo EL	3	89	4
Carta A	4	133	4
Huang J	3	64	3
Keith DA	3	439	4
Khoury CK	4	305	4
Yu S	1	7	4
Abeli T	3	154	3
Assogbadjo AE	3	38	3
Blasi C	3	155	3
Burgess ND	3	226	3
Burgman MA	3	376	3
Fensham RJ	3	39	3

the other hand, the number of relevant publications has increased in ‘Biological Conservation’ (Figure 4).

Most of the global biodiversity loss is concentrated in nine countries, viz., Australia, Brazil, China, Colombia, Ecuador, Indonesia, Malaysia, Mexico, and the USA (Mendoza-Ponce et al., 2020). This highlights the necessity for these nations to implement effective monitoring and policy enforcement for species conservation (Alroy, 2017). The present study reveals that most of these countries are working to implement effective monitoring and management policies for biodiversity conservation. Based on the country production analysis, it was revealed that authors from 76 countries have contributed to the field. The USA has the highest frequency of publications on threat assessment and prioritization for conservation (212 publications), followed by China (165 publications), the UK (106 publications), Italy (101 publications), India (79 publications), Australia (76 publications), Brazil (66 publications), Germany (46 publications), and Spain (44 publications) (Figure 5). In Himalayan Biodiversity Hotspot, China and India are the only countries to produce a significant number of publications in the relevant field. Other Himalayan countries like Pakistan and Nepal have produced 04 and 03 publications, respectively; whereas Bhutan has not produced any publication in the relevant field.

TABLE 2 Lotka law and the number of articles by the authors.

Documents written	No. of Authors	Proportion of Authors
1	1,148	0.883
2	106	0.082
3	32	0.025
4	7	0.005
5	4	0.003
7	1	0.001
8	1	0.001
12	1	0.001

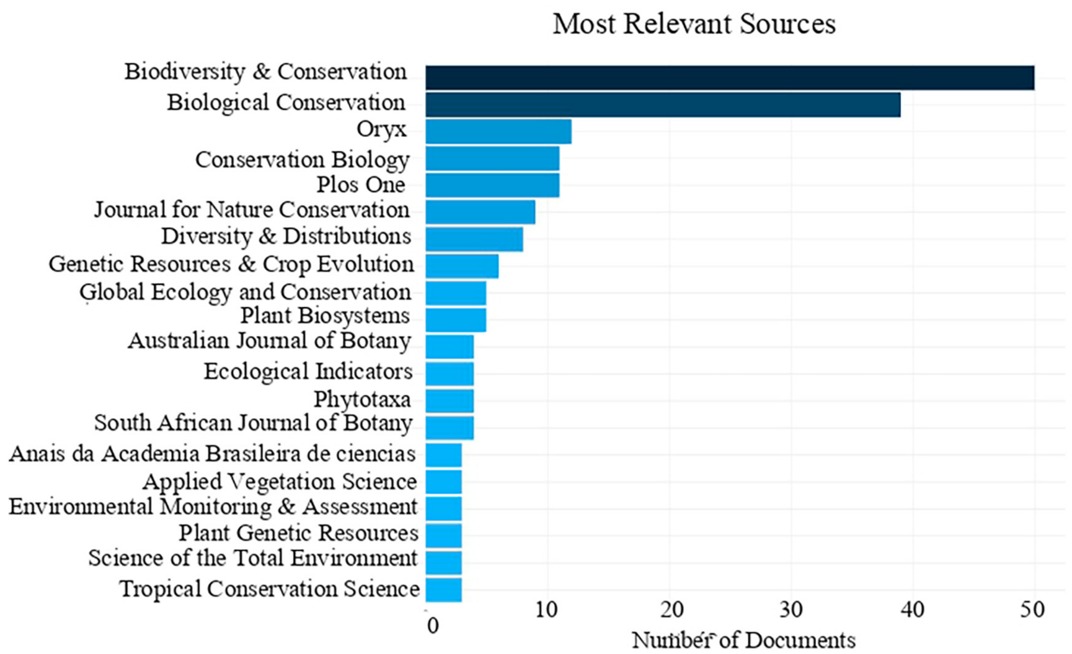

FIGURE 2
Top most relevant journals for the studies on prioritization for conservation.

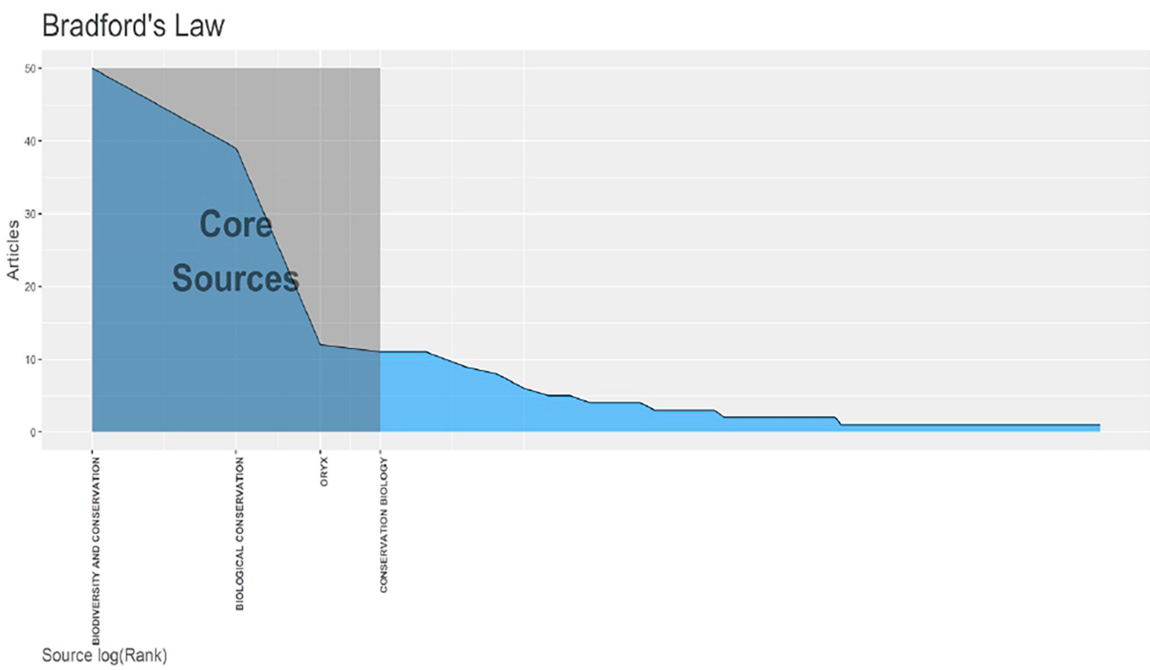

FIGURE 3
Bradford's law showing the most basic and core sources.

Citation analysis

Citation analysis is a fundamental method for science mapping that works on the conjecture that citations reflect intellectual linkages between publications that are formed when one document cites the other (Appio et al., 2014). The analysis enables the most influential publications in a research field to be determined (Donthu et al., 2021). In the present study, on average, each document has received 84.68 citations, with 5.50 citations per year per document. The most global cited document on threat assessment and prioritization for conservation include Schnittler and Gunther, 1999; Dhar et al., 2000; Myers et al., 2000;

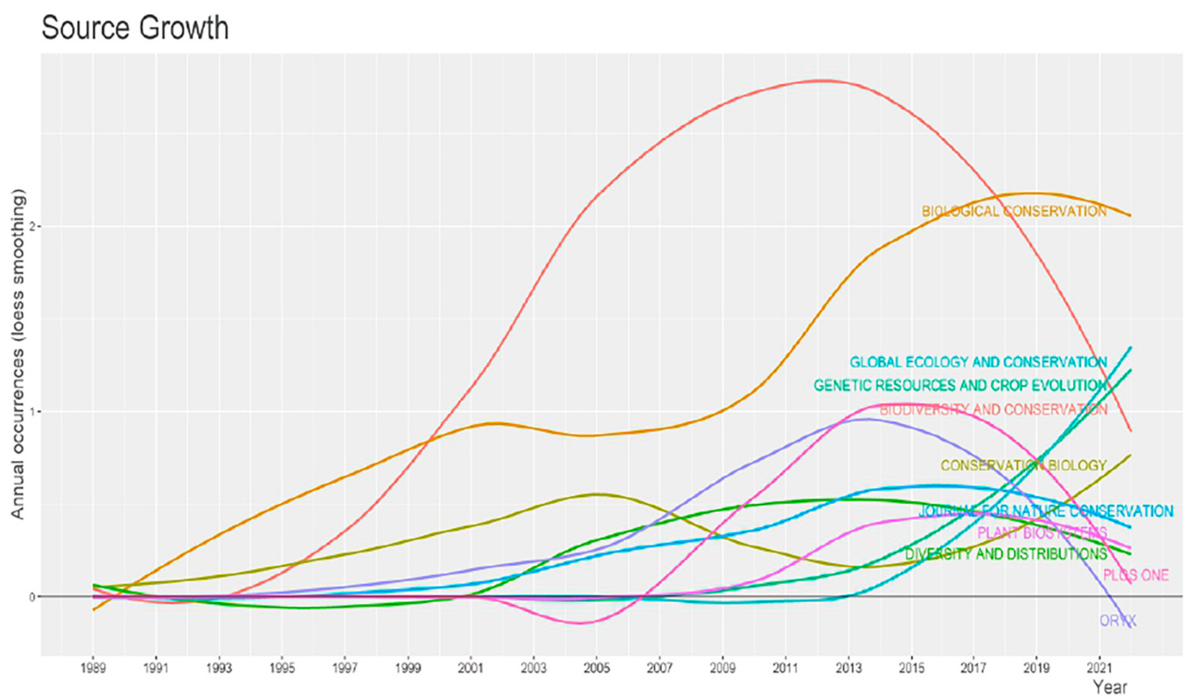

FIGURE 4
Growth of journals with respect to publications on prioritization for conservation from 1989–2022.

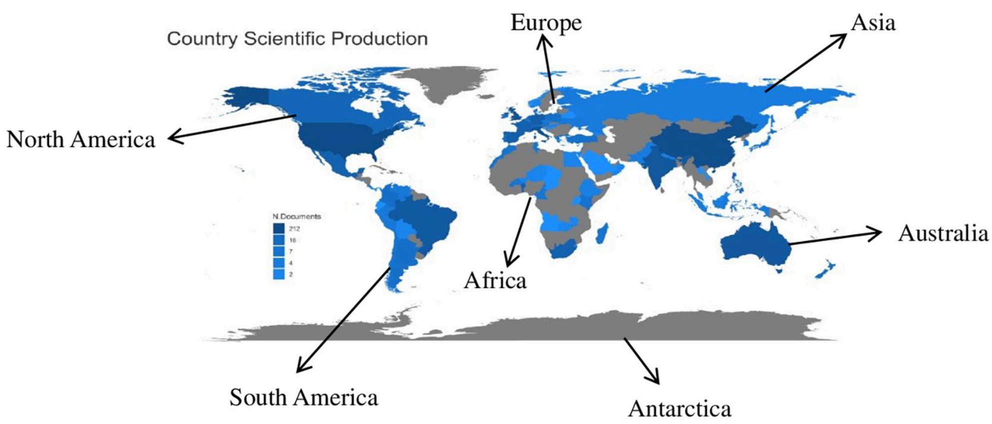

FIGURE 5
Global scientific production of scholarly documents on prioritization for conservation.

Coates and Atkins, 2001; Hartley and Kunin, 2003; Kala et al., 2004; Keller and Bollmann, 2004; Partel et al., 2005; de Oliveira et al., 2007; Zhang and Ma, 2008; Brehm et al., 2010; Gauthier et al., 2010; Bacchetta et al., 2012; Brummitt et al., 2015. Among the authors, 'Keith DA' from the New South Wales National Parks and Wildlife Service, Australia has received the maximum impact with 439 citations and h-index of 3 (Number of publications = 4), followed by 'Maxted N' from the University of Birmingham, UK with 428 citations and 7 h-index (Number of publications = 12), 'Burgman MA' from the University of Melbourne, Australia with 376 citations and 3 h-index (Number of publications = 3), and 'Bacchetta G' from the University of Cagliari, Italy with 328 citations and 7 h-index (Number of publications = 8) (**Table 1**). Among the sources, the journal Biological Conservation tops the list with highest number of citations (1,406 citations; 22 h-index), followed by Biodiversity and Conservation (1,280 citations; 21 h-index), Plos One (701 citations; 9 h-index), Conservation Biology (577 citations; 8 h-index), and Diversity and Distribution (478 citations; 8 h-index) (**Table 3**). The h-index calculates an author's number of publications and citations for those articles (Kumar et al., 2023). It gives a breakthrough in the research community for assessing the scientific impact of an individual or source (Bihari et al., 2023). Among the most cited countries, the UK ranks at the top with 17,753 citations, followed by Australia, the USA, Germany, Brazil, China, Italy and India with 1,738, 1,242, 882, 683, 561, 540, and 444 citations, respectively (**Figure 6**).

TABLE 3 Top most productive and cited sources from 1989–2022.

Sources	h-index	Total Citations	Number of Publications
Biodiversity and Conservation	21	1,280	50
Biological Conservation	22	1,406	39
Oryx	7	301	12
Conservation Biology	8	577	11
Plos One	9	701	11
Journal for Nature Conservation	8	216	9
Diversity and Distributions	8	478	8
Genetic Resources and Crop Evolution	4	52	6
Global Ecology and Conservation	3	22	5
Plant Biosystems	3	95	5
Australian Journal of Botany	4	46	4
Ecological Indicators	2	22	4
Phytotaxa	2	14	4
South African Journal of Botany	3	66	4
Anais Da Academia Brasileira De Ciencias	2	23	3
Applied Vegetation Science	3	64	3
Environmental Monitoring and Assessment	3	85	3
Plant Genetic Resources-Characterization and Utilization	3	18	3
Science of the Total Environment	2	24	3
Tropical Conservation Science	2	16	3

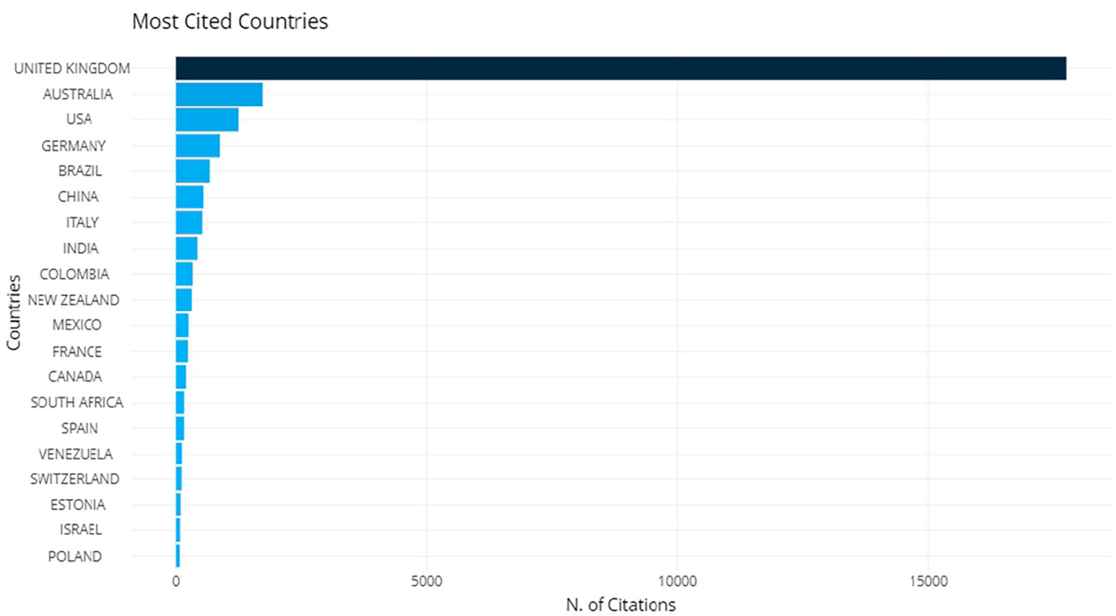

FIGURE 6
Top 20 most cited countries.

Collaboration analysis

Collaboration analysis is another important science mapping procedure to reveal how contributors are linked to each other in a particular research field. It determines the pertinent contributors and their relationships (Aria and Cuccurullo, 2020). Of the total 315 documents, 28 are single-authored documents, and the average number of documents per author is 0.242. The average number of authors and co-authors per document is 4.13 and 4.86, respectively, with a collaboration index of 4.43. Countries

TABLE 4 Most relevant countries by collaborating authors.

Country	Articles	Frequency	SCP	MCP	MCP Ratio
USA	42	0.13462	28	14	0.3333
China	37	0.11859	24	13	0.3514
United Kingdom	24	0.07692	9	15	0.625
India	21	0.06731	20	1	0.0476
Italy	21	0.06731	16	5	0.2381
Australia	19	0.0609	14	5	0.2632
Brazil	18	0.05769	17	1	0.0556
Spain	14	0.04487	11	3	0.2143
Mexico	10	0.03205	8	2	0.2
Canada	8	0.02564	6	2	0.25
France	8	0.02564	5	3	0.375
South Africa	8	0.02564	7	1	0.125
Benin	7	0.02244	2	5	0.7143
Germany	6	0.01923	4	2	0.3333
New Zealand	4	0.01282	3	1	0.25
Turkey	4	0.01282	4	0	0
Argentina	3	0.00962	3	0	0
Ireland	3	0.00962	3	0	0
Israel	3	0.00962	3	0	0

SCP, Single Country Production; MCP, Multiple Country Production.

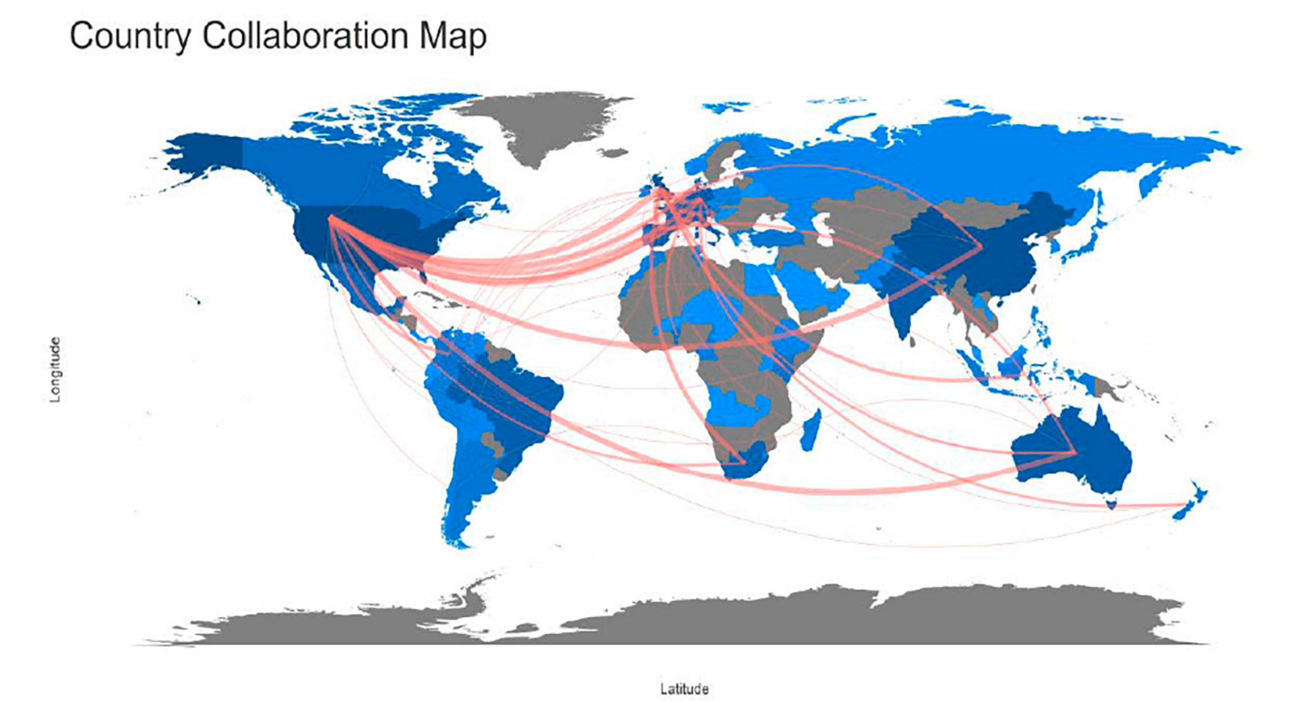

FIGURE 7
Country collaboration map (Intensity of blue color shows number of publications and the pink line depicts the connections).

with a Multiple Country Publications (MCP) rate ≥ 50% are the countries with high international cooperation in the field (Gulhan and Kurutkan, 2021). In the present study, Benin has the highest MCP with a rate of 71.4%, followed by the UK (62.5%) and France (37.5%). India has an MCP rate of only 4.76% (Table 4). Figure 7 depicts the country collaboration in

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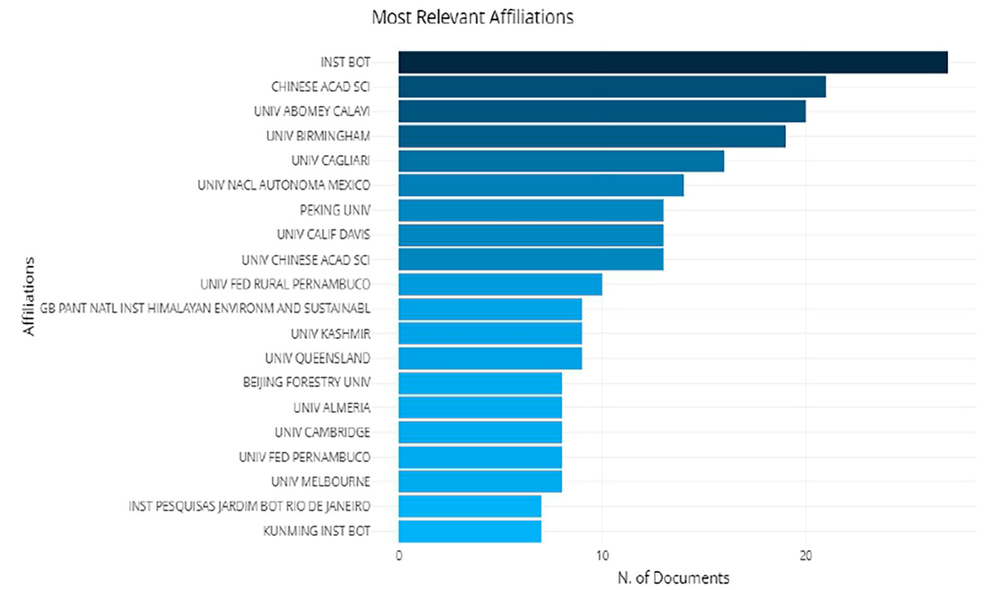

FIGURE 8
Top 20 most relevant affiliations.

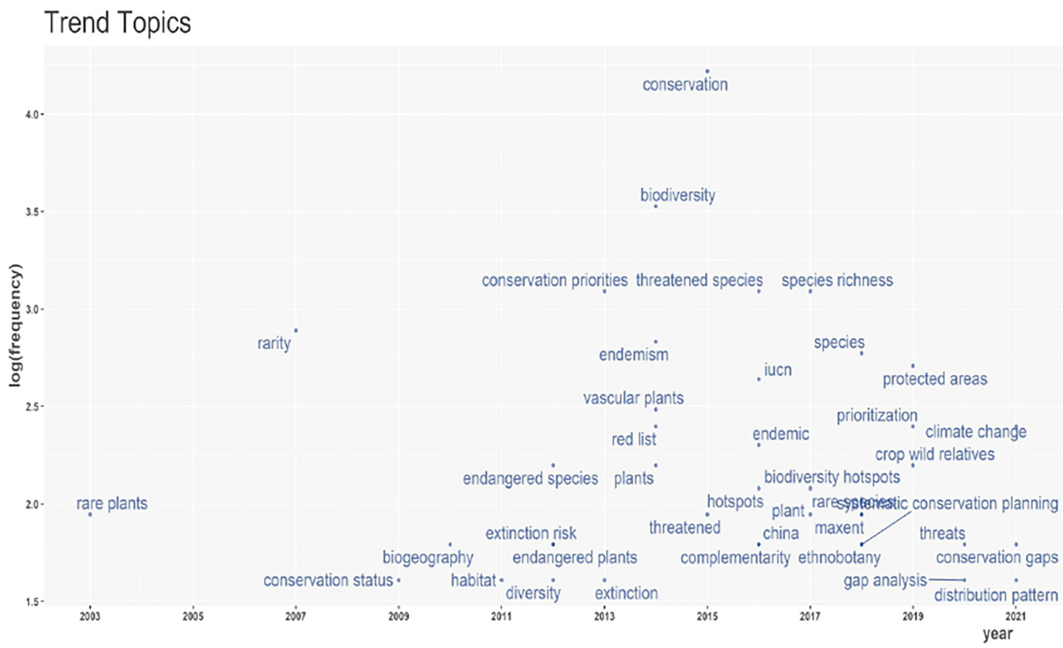

FIGURE 9
Trends in the topics (keywords) from 1989–2022.

threat assessment and prioritization for conservation studies, in which the blue color indicates the countries that have published the articles, and its intensity is proportional to the number of publications. The pink color line represents the connection between the countries, and its thickness depicts the level of collaboration. Collaboration is essential for conservation (Lloyd et al., 2023), as it will allow successful implementation of biodiversity conservation programs throughout the globe. Based on the most contributing institutes, the Institute of Botany has produced the maximum number of documents (27 publications), followed by the Chinese Academy of Sciences (21 publications), the University of Abomey-Calavi (20 publications), the University of Birmingham (19 publications), and the University of Cagliari (16 publications). Figure 8 shows the top twenty relevant affiliations of the corresponding authors with Institute of Botany, China, Chinese Academy of Sciences, China, University of Abomey-Calavi, Benin, and University of Birmingham, England as the most relevant affiliations.

Keywords analysis

Author keywords identify the content and theme of the published document (Sharma et al., 2020). For each article dealing with the threat assessment and prioritization for conservation, the original author keywords, i.e., used by the authors in the articles, were examined. A total of 1,008 keywords have been used by the authors to classify their studies from 1989–2022. **Figure 9** depicts a scatter plot of the most trending topics in threat assessment and the prioritization for conservation studies from 1989–2022. The height of keywords shows their frequency of occurrence in a particular year. The most frequently occurring keywords typically express the most trending topics of the year. The most frequently used keywords include conservation (68 in 2015), biodiversity (34 in 2014), conservation priorities, species richness and threatened species (22 each in 2013, 2017, and 2016, respectively), and rarity (18 in 2007). The frequency of keywords and their ranks follow a power-law distribution, with a few keywords used frequently whereas most of the keywords are not used so frequently, which is consistent with earlier studies (Li et al., 2008; Liu et al., 2011).

Limitations of the study

In the present study, data has been exclusively drawn from WoS data source, thus not representing the comprehensive literature in the field. Although, WoS is the most authentic and popular database among academicians, it is highly recommended that alternative data sources like Scopus and Google Scholar should be included in future studies for more thorough analysis of the available research documents on this topic. Besides using bibliometrix tool, analysis of data through BibExcel, CiteSpace, Hist Cite, and Pajek would be a better option for providing further detailed information on literature. Further, as in the present study, the search for data was conducted only in the English and there is a possibility that relevant publications in other languages may have been missed. By limiting the search to only English only, there is a risk of missing out some valuable findings published in other languages. Therefore, researchers should consider the possibility of including publications in other languages in future studies for more vivid and all-inclusive analysis of data. Despite of these limitations, our study provides imperative insights on research trends and directions in the conservation of biodiversity elements. By addressing these limitations in future investigations, researchers can further enhance and expand the knowledge base in this field.

Conclusion

The present study has investigated the evolution and current situation of research on threat assessment and prioritization of plants for conservation at a global level by analyzing bibliometrically the most relevant and productive authors, sources, and countries, most cited papers, country collaborations, and most frequent keyword aspects as reflected in the scientific literature. The present study will be helpful to the researchers to find out which journal should be targeted and to find out their research collaborators in the relevant field. Further, the study may be useful to identify the hotspots for further such studies. Our bibliometric analysis indicated that output in the relevant field has significantly increased since 1989. A total of 1,300 authors have contributed to 315 articles published in 129 journals. The journal Biodiversity and Conservation and Biological Conservation have produced the highest number of articles, whereas the journal Biological Conservation has received the maximum number of citations. The countries where most biodiversity loss is concentrated, like the USA, China, Australia, Brazil, and Mexico, has produced the maximum number of publications on threat assessment and prioritization for conservation. Benin, a developing country of West Africa has the highest rate of Multiple Country production indicating that both developed and developing countries are working together to tackle the global biodiversity loss. However, as GDP growth is the prime goal of most local governments in developing and underdeveloped countries institutions in developed countries should be encouraged to lead research programs in such countries. Global conservation needs far exceed any one organization's capacity and resources. Conservationists prioritize species, resources, and actions every day, but only through a structured decision-making process can strategic decisions be made in an explicit, transparent, and effective manner. This will also facilitate potential partnerships among conservation organizations, philanthropists, and other stakeholders.

Data availability statement

The original contributions presented in this study are included in this article/supplementary material, further inquiries can be directed to the corresponding author.

Author contributions

ZW: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Writing—original draft, Writing—review and editing. SP: Supervision, Writing—review and editing. JB: Writing—review and editing. MT: Writing—review and editing. SS: Funding acquisition, Writing—review and editing. MA: Writing—review and editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Sustainable forest management for nut and fuelwood production in the Jalalabad region, Kyrgyzstan: insights from local communities

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Jalalabad region in the Kyrgyz Republic is home to the world's largest natural walnut (*Juglans regia* L.) forests and pistachio (*Pistacia vera* L.) forests. These nut-fruit forests serve as the primary source of income for local people; however, deforestation has led to a decline in the availability of these resources. Wood from the forest is also used as a crucial energy source for cooking and heating in this region, despite state protection of the walnut forest due to a lack of alternative energy sources. This study aimed to explore solutions to restore nut-fruit forests while providing a fuelwood source and improving the income of local people. Qualitative research methods were employed, including semi-structured interviews and survey questionnaires with farmers and central and local government officials. The study investigated the main environmental problem of forest degradation and the challenges of forest management, such as livestock and complicated lease arrangements, and determined possible government support and incentives for local communities to participate in forest rehabilitation, such as planting instead of paying lease fees. The findings indicated that a tree-based farming approach can be a promising alternative land-use solution. Seventy-four percent of local farmers expressed their readiness to plant fruit trees and fast-growing tree species to meet their demand for fuelwood and improve their livelihoods if the government can provide temporary fencing support. The study also highlighted the need for capacity building for farmers to learn proper planting, managing, and harvesting for more sustainable practices. Policymakers need to modify legislation through simplified and incentive-based forest lease arrangements.

KEYWORDS
wood, fuel, forest rehabilitation, walnut forest, Kyrgyzstan, sustainable forest management, agroforestry

1 Introduction

Wood and coal are the primary sources of cooking and heating fuel for the majority of households in Kyrgyzstan, and their importance has significantly increased over time (Gottschling and Lazkov, 2005). The use of wood fuel not only serves as a vital energy source but also impacts public interests such as the environment, public health, rural development, employment, and foreign exchange (Githiomi, 2011). Since only approximately 17% of Kyrgyz households have access to modern district heating services, mainly in urban centers such as Bishkek, the majority of households depend on individual heating solutions, with coal being the primary fuel for 60% of them, followed by electricity (15%), wood and dung (5%), and gas (1%) (World Bank, 2020). Although alternative energy sources could help reduce the consumption of firewood in the region, they are limited, particularly for low-income households, due to affordability and infrastructure limitations (Balabanyan et al., 2015). The lower the income, the higher the dependence on solid fuels such as coal, dung, and wood, with 97% of households in the lowest income tertile relying on such fuels for their heating needs.

This overuse of natural forests has led to the depletion of resources and a decrease in the genetic diversity of species in Kyrgyzstan (Rehnus et al., 2013). The harvesting of timber for wood products and grazing remains a major driver of forest degradation, particularly in forests consisting of spruce, juniper, walnut (*Juglans regia* L.), fruit trees, and pistachio (*Pistacia vera* L.; Lindquist and Annunzio, 2016). Moreover, the rapid increase in the number of livestock in Kyrgyzstan has led to increased pressure on natural ecosystems, resulting in the degradation of grassland areas near villages (UNECE, 2019).

Previous studies have shown that the Jalalabad region in Kyrgyzstan is a biodiversity hotspot of international significance due to its natural walnut and other fruit forests (Fisher and Christopher, 2007; Venglovskii, 2015). Unsustainable harvesting of non-timber forest products (NTFPs) such as walnut and wild-crafted medicinal plants for export, and the overgrazing of livestock have led to the loss of 90% of fruit and nut forest habitats in the region over the last 50 years (Eastwood et al., 2009; Martínez de Arano et al., 2021). This, coupled with the increasing scarcity of firewood around settlements, has resulted in villagers having to venture further into the forests to meet their daily fuel needs.

In addition to these challenges, climate change is expected to significantly impact the living conditions and health of the population, with water resources being the most vulnerable (International Monetary Fund, Middle East and Central Asia Department, 2023). This will result in a reduction in the development of hydropower, which is the main source of energy in the country (Global Forest Resources Assessment, 2020). Given that people in the region depend on tree resources for quality water, timber, wood fuel, and food, it is crucial to find alternative energy sources and reduce dependency on natural forests as the primary source of fuelwood to prevent desertification and land degradation and mitigate the adverse impacts of climate change.

In order to propose viable solutions to the complex environmental challenges and energy source issues in this region, this study aimed to identify the local community's willingness to participate in forest rehabilitation by planting fast-growing tree species for fuelwood production to meet the growing demand for fuel. Furthermore, the study examined the perceptions of local forest users regarding the environmental problems related to forest degradation and the challenges in forest management, including complicated lease arrangements and livestock. This study mainly focused on questions about whether forest users were willing to cultivate fast-growing trees and what the locals' perceptions were regarding environmental problems. Additionally, the study determined the potential government support and incentives available to local communities by asking questions about what incentives the government could potentially provide for forest users.

2 Materials and methods

2.1 Study area

The research was conducted in the Nooken district of the Jalalabad region, as shown in Figure 1. In Kyrgyzstan, all forests and non-forest lands designated for forestry purposes, excluding forests under municipal and private ownership, collectively form the State Forest Fund. The State Forest Fund in this district covers approximately 247,342 hectares, yet only 67,011.9 hectares (27%) are covered with trees, indicating an uneven distribution. They are situated in a ribbon-like formation up to 20 km wide in the northeast, below the belt of alpine and subalpine meadows. The elevation of the district gradually increases from the lower regions, which lie below 1,000 m, to high peaks exceeding 4,000 m.

In order to better understand which factors promote farmers' environmental behavior, we selected two case study regions to identify similarities and differences in two geographical locations. The first study area selected as a forest user, Toskool-Ata, is located at 41°12′10″N, 72°41′30″E (Figure 2A). These GPS coordinates were received from the local government in charge of forest management in the study area. As illustrated in Figure 2A, areas closer to settlements exhibit degradation as a result of illegal logging and overgrazing, as noted by Beer et al. (2008). In Toskool-Ata, there is a population of 2,180 people (National Statistic Committee, 2021), living on the lower mountain slopes at an altitude of approximately 1,300 m to 1,800 m a.s.l. Forests in the Toksool-Ata region comprise both naturally occurring and human-modified walnut, pistachio, apple (*Malus Sieversii* M. Roem), and other fruit-bearing tree species distributed in different types of forests (Table 1). These forests are considered a biodiversity hotspot of international importance and have a special regime of use. The area has favorable agro-climatic conditions, with up to 850–1,200 mm of precipitation per year and an average temperature of 20.5°C in July and 3.1°C in January. The region's natural resources, including agriculture and walnut collection, are important sources of income for local households, along with pasture-based livestock. The usability of pastures depends on altitude, exposure, and water sources. Wood and coal are the main sources of energy for cooking and heating.

Toskool-Ata *leshoz*, a state forest enterprise that is responsible for the protection and management of forests and state-owned non-forested land located on *leshoz* territory in the Kyrgyz Republic, was established on 12 March 1997, under Decree No. 142 of the Government of the Kyrgyz Republic. It is located on a single massif, stretching 37 km from north to south and 21 km from west to east. The total area of Toskool-Ata *leshoz* is 71,723.3 hectares, of which 37,501 hectares (53%) are forests (Table 2). As a government body, the *leshoz* is responsible for implementing forest policies in line with the Forest Code, with the aims of protecting the forest resources, carrying out reforestation and afforestation, and enhancing the quality of life for the local population (Forest Service of Kyrgyz Republic, 2021).

For comparison with Toskool-Ata, Arimdzhan (41°02′07.0"N, 72°34′55.5″E) was chosen as a sample area representing non-forest users (Figure 2B). Arimdzhan has a population of 2,141, with most individuals engaged in agricultural activities.

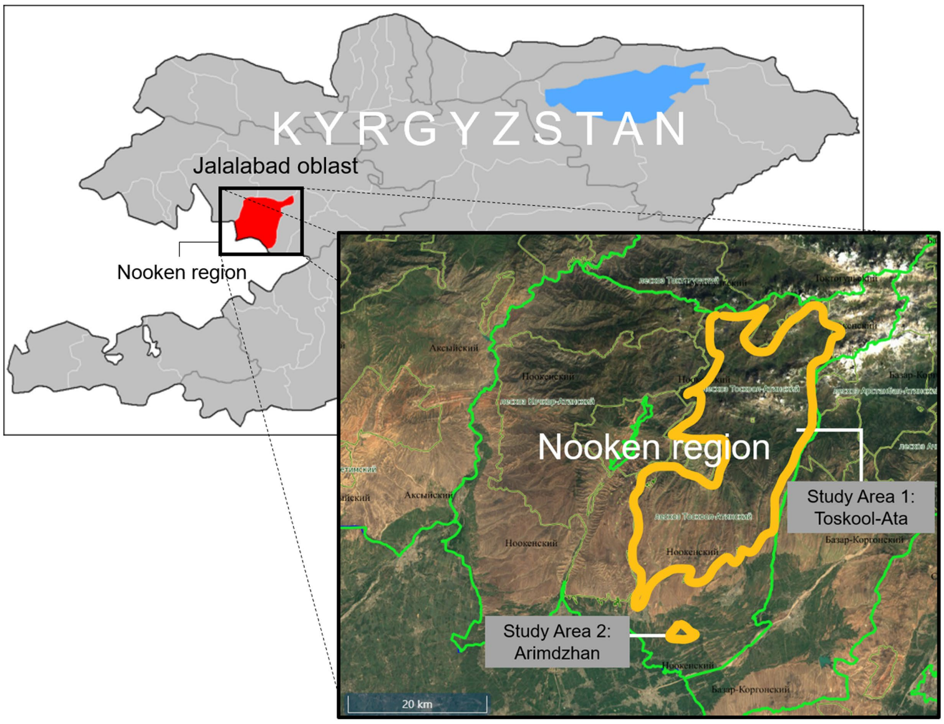

FIGURE 1
Location of the study area (Jalalabad, Kyrgyzstan).

FIGURE 2
Google Earth images of Toskool-Ata **(A)** and the Arimdzhan region **(B)**.

2.2 Data collection

Survey questionnaires were used to gather data on forest loss due to illegal harvesting of fuelwood and also to investigate government support and incentives for local communities to participate in forest rehabilitation. Respondents included forest-user households, non-forest-user households, and central and local government officials working in the forestry sector (Table 3). Separate questionnaires were developed for each group and translated into local Kyrgyz and Russian languages for better participation. The survey questionnaires were pre-tested in the field for verification and modifications.

Semi-structured interviews and focus group discussions with experts and stakeholders were also conducted during this period. The

TABLE 1 Distribution of types of forests.

S/N	Types of forests	Area (ha)
1	Walnut (*Juglans regia* L.) forest	2861.1
2	Pistachio (*Pistacia vera* L.) forest	38659.1
3	Almond (*Amygdalus communis* L.) forests	346.6
4	Protected fruit forests	346.6
7	Juniper (Juníperus) forest	6388.8
8	Erosion control forests	23467.7
9	Other trees	23467.7
	Total	71,723,3

Source: Toskool-Ata leshoz (state forest enterprise).

TABLE 2 Forest land category in the study area.

Land category	Hectares	Tenured land	% Of used land	Number of renters (tenure)
Forest land covered with tree	37,501	2,066.38	5.5	550
Irrigated arable land	21.1	6.45	30.6	21
Rainfed arable land	460.2	101	21.9	85
Hayfield	485	122.5	25.3	23
Pasture land	24,043.9	12,019.5	50	159
Gardens, vineyards	84.5	11.9	14.1	10
Homestead	465.2			
Roads	38.1			
Water	28.7			
Other types of land (rocks etc.)	8,595.6			
Total	71,723.3			

Source: Toskool-Ata leshoz (state forest enterprise).

TABLE 3 Number of respondents surveyed in this study.

Division	N	% Among respondents
Central government	10	12.3
Local government	6	7.4
Forest-user household	31	38.2
Non-forest-user household	34	41.9

aim of the discussions was to understand issues related to access to forest resources, resource use, and recommendations for improving resource governance. Considering approximately 400 households in the study area, 65 heads of households (13% of total households) were interviewed using a simple random sampling based on annual *leshoz* logs of lease agreements from the past year. This produced a selection of users who have official permits for the use of forest resources and those who do not but still consume them. Sixteen respondents, comprising 10 from the central government and 6 from the local government, were engaged as a sample to gather policy-related information and identify possible incentives for farmers from the government.

2.3 Data analysis

This study employs the theory of planned behavior (TPB) to examine the factors influencing farmers' perceptions and behaviors regarding tree planting. Developed by Ajzen (1991). TPB is a widely utilized behavioral framework that is considered instrumental in understanding and predicting farmers' behaviors related to environmental conservation and livestock welfare (Beedell and Rehman, 2000). As an extension of the "theory of reasoned action model," TPB offers valuable insights into predicting farmers' behavior, surpassing other socioeconomic variables (Lam, 2006).

Drawing upon questionnaire results, a hypothetical model of farmers' behavior toward tree planting was constructed. The questionnaire comprised 12 questions assessing perceptions and 12 assessing attitudes, predominantly utilizing a "Yes" or "No" scale. Additionally, warm-up questions, representing various social psychology constructs, were included in interviews to gauge farmers' perceptions, attitudes, and satisfaction.

Discourse analysis was conducted through interviews with the 16 government officials to explore the types of incentives that the government could offer to encourage local forest users to participate in forest restoration efforts. Both quantitative and qualitative data were edited, numbered, and entered into Statistical Package for Social Sciences (SPSS) 23 software. Multiple response questions were analyzed for frequencies and percentages. Tables and bar charts/graphs were used to present different variables and facilitate the results of the analysis and interpretation of data in Microsoft Excel.

3 Results

3.1 Demographic characteristics of the respondents

3.1.1 Respondents among forest users (Toskool-Ata)

The number of respondents from the forest users was 31, which consisted of 54.9% men and 45.1% women (Table 4). Age groups included 25.8% aged 40–49, 19.3% for both 30–39 and over 60, and 16.1% aged 50–59. Respondent incomes ranged from 1,000 to over 20,000 KGS (equivalent to USD 11–229). Most households near forests had low incomes (15,000 KGS or less, equivalent to USD 172), compared to Kyrgyzstan's *per capita* income of USD 1,323 (World Bank, 2020). Household heads were, on average, under 45.7 years old and had a high school education. Approximately 71% of forest users had only a high school education, while 29% held university degrees. As shown in Table 4, 45.1% of respondents had 5–6 family members, and 22.5% had 7–8 family members living together. All respondents reported full involvement in forest-related activities to support their livelihoods. The overall average landholding was approximately 2.8 hectares per household.

3.1.2 Respondents among non-forest users (Arimdzhan)

The number of respondents from non-forest users was 34. Among non-forest users, 61.7% were women and 38.3% were men (Table 5). Age groups included 20.5% aged 40–49, 23.5% aged 30–39, 17.6% over 60, and 26.4% aged 50–59. While 47% were government employees, all had

TABLE 4 Socioeconomic characteristics of the respondents for forest users.

Variables (n = 31)	Frequency	Percent (%)	Mean	SD	Min.	Max.
Gender						
Men	17	54.9				
Women	14	45.1				
Age group			45.7	10.9	28	63
20–29	2	6.5				
30–39	6	19.4				
40–49	10	32.3				
50–59	7	22.6				
≥60	6	19.4				
Level of education						
High school	22	71				
Greater than or equal to university	9	29				
Household size			5.36	1.6	1	8
1–2	1	3.2				
3–4	3	9.7				
5–6	14	45.2				
7–8	7	22.6				
N/A	6	19.4				
Income level (Kyrgyzstan Som, KGS)			14,417.2	12,981.5	3,800	75,000
1,000–5,000	4	12.9				
5,001–10,000	7	22.6				
10,001–15,000	10	32.3				
15,001–20,000	5	16.1				
20,001–25,000	2	6.5				
75,000 and over	1	3.2				
N/A	2	6.5				
Farm size category			2.8	1.4	1	5
≤1 ha	11	35.5				
2 ha	5	16.1				
3 ha	5	16.1				
≥4 ha	10	32.3				
Lease period			21.7	18.8	5	50
≤5 years	15	48				
≤25 years	8	26				
≤50 years	8	26				

agricultural lands and derived income from them. Respondents' income ranged from 10,000 to over 100,000 KGS (equivalent to USD 115–1,145). In terms of education, 38.2% graduated from high school, and 58.8% held university degrees. As for family size, 55.8% had 5–6 members, 22.5% had 3–4 members, and 17.6% had 7–8 members living together.

3.1.3 Respondents among central and local government officials

A total of 16 government officials participated in the sampling, including officials from both central and local government institutions. Among the central government officials, 30% were women and 70% were men (Table 6). Their working periods were distributed as 40% with 5–10 years of experience, 30% with over 10 years, and 30% with 1 year or less. In contrast, local government officials were all men (Table 7), and their working periods were divided as 50% with over 10 years, 33.3% with 5–10 years, and 16.6% with 1 year or less.

3.2 Forest use of local communities for economy and fuelwood

The primary economic activity among forest users predominantly involved the collection of nut fruits from the forests, whereas non-forest users overwhelmingly relied on agriculture, constituting

TABLE 5 Socioeconomic characteristics of the respondents for non-forest users.

Variables (*n* = 34)	Frequency	Percent (%)	Mean	SD	Min.	Max.
Gender						
Men	13	38.3				
Women	21	61.7				
Age group			47.3	13.3	27	68
20–29	4	11.8				
30–39	8	23.5				
40–49	7	20.6				
50–59	9	26.5				
≥60	6	17.6				
Level of education						
High school	14	41.1				
Greater than or equal to university	20	58.9				
Household size			5.05	1.4	2	8
1–2	2	5.9				
3–4	7	20.6				
5–6	19	55.9				
7–8	6	17.6				
≥9	0	0.0				
Income level (Kyrgyzstan Som, KGS)			48,789.2	3,9437.8	6,000	140,000
N/A	6	17.6				
5,000–10,000	4	11.8				
10,001–15,000	6	17.6				
15,001–20,000	3	8.8				
50,000–100,000	11	2.9				
120,000 and over	2	35.3				

TABLE 6 Gender, periods of working, and residence of interviewees in the central government.

Variables (*n* = 10)	Frequency	Percent (%)	Mean	SD	Min.	Max.
Gender						
Men	7	70				
Women	3	30				
Working period			16.1	17.3	1	29
≤1 year	3	30				
5–10 years	4	40				
≥10 years	3	30				

100% of their main economic activity. Within the forest user group, a significant majority, accounting for 74.2% of respondents, highlighted walnut or pistachio collection as their primary income source (Figure 3), indicating a substantial dependency on forest products within the study area.

Among forest users, a significant proportion, comprising 90.3% of respondents, reported utilizing wood for heating and cooking purposes (Figure 4). Of these, 48.4% stated that they source wood from the nearest state forests, while 38.7% indicated collecting wood from their leased forest lands, and 3.2% acquired it from the market. In contrast, respondents among non-forest users mentioned a practice of blending wood with coal and cotton stem, a proportion notably higher than households in non-forest areas, which stands at approximately 20.5%.

Both forest users and non-forest users have expressed concerns about the diminishing availability of fuelwood in the area over time, attributing this trend to deforestation. According to farmers, deforestation primarily stems from overgrazing and inadequate livestock management practices. This indicates a worrying cycle where

TABLE 7 Gender, periods of working, and residence of interviewees in the local government.

Variables (n = 6)	Frequency	Percent (%)	Mean	SD	Min.	Max.
Gender						
Men	6	100				
Women	0	0				
Working period			31.5	15.5	6	44
≤1 year	1	16.7				
5–10 years	2	33.3				
≥10 years	3	50				
Years lived in the area			58.1	11.2	36	68
36 years	1	16.7				
≥60 years	5	83.3				

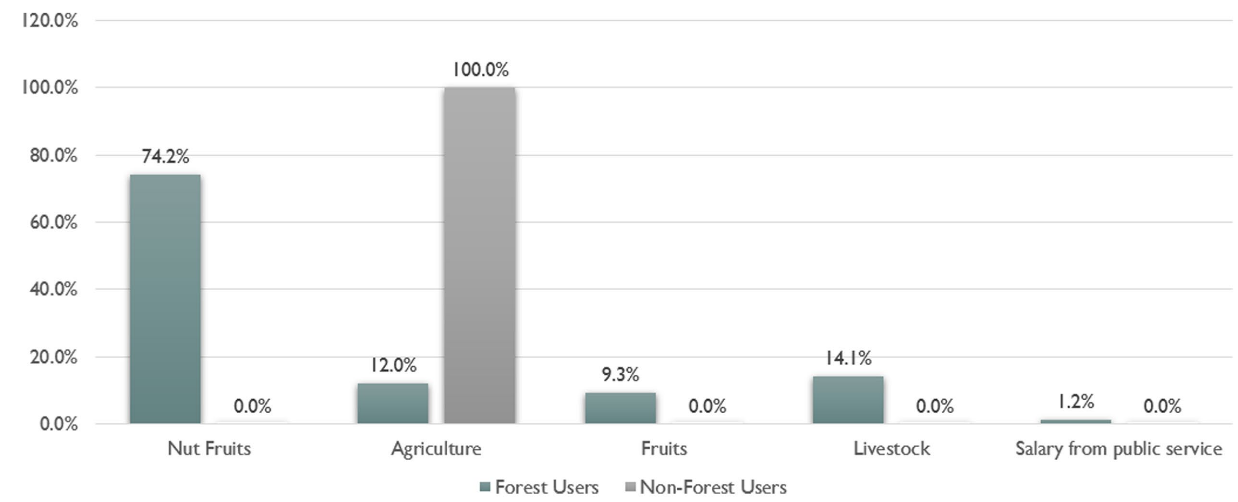

FIGURE 3
Main source of income for both forest users and non-forest users.

fuelwood collection and overgrazing contribute to the degradation of forest resources in state forests.

The survey findings highlight that a majority of fuelwood consumption occurs on people's leased forest lands or nearby forests, where they typically gather wood from dead shrubs or nut-fruit trees. However, it is noted that the availability of dead wood is declining, and people cannot rely on it consistently every year (Gottschling and Lazkov, 2005). Comparatively, among households in forest-user areas, only 9.7% resort to using coal for heating and cooking, sourced from the local market (refer to Figure 4). Respondents attribute their preference for fuelwood over alternative energy sources to the lack of viable alternatives (World Bank, 2019).

3.3 Perception of local people on challenges and capacity building in forest use

3.3.1 Perception of environmental problems

Respondents also displayed a clear understanding of the ecological challenges facing the area, with deforestation emerging as the primary concern. Among forest users, a notable proportion (48.8%) identified forest degradation as the main problem (see Figure 5), while among non-forest user farmers, 52.9% cited water shortage as the primary ecological issue. This discrepancy can be attributed to their main activity, which is agriculture. During interviews, an overwhelming majority of respondents (87.1%) identified overgrazing as the primary driver of deforestation. In contrast, only a minority (12.9%) considered fuelwood collection as the secondary cause of deforestation in the study area.

3.3.2 Growing livestock in the study area

Interviewees emphasized the need for fences on their leased forest land because cattle and other livestock from other households enter the forest land and eat all the young vegetation (Levy Bridges, The World Organization, 2022). This is one reason why households do not want to plant trees on their lands.

3.3.3 Willingness of local people to increase forest land and capacity building

Juldashev and Messerli (2000) identified a major challenge impeding agroforestry adoption in Kyrgyzstan as the perception by

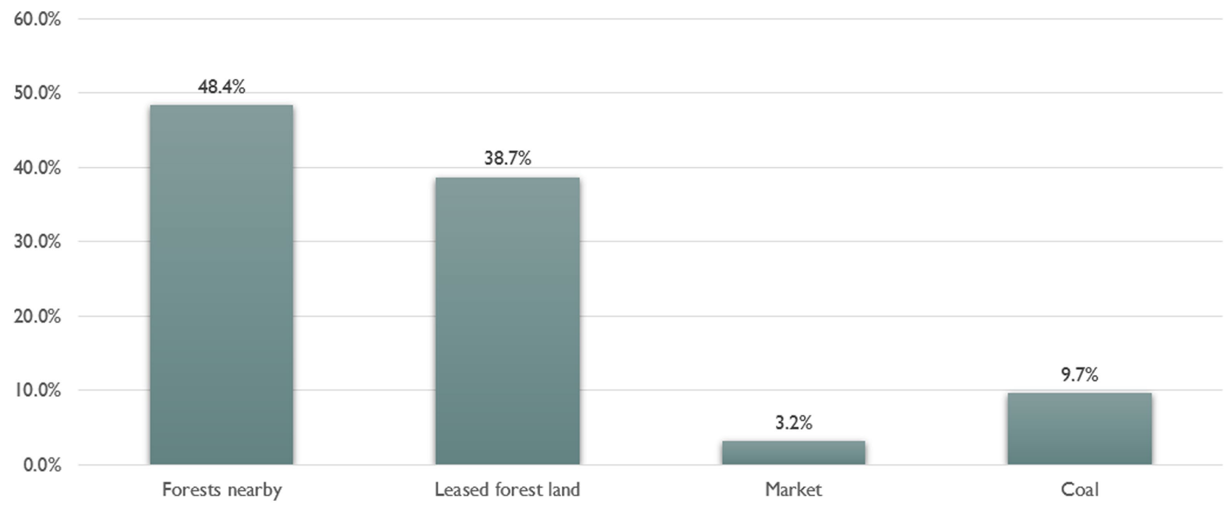

FIGURE 4
Main source of heat for households in forest users.

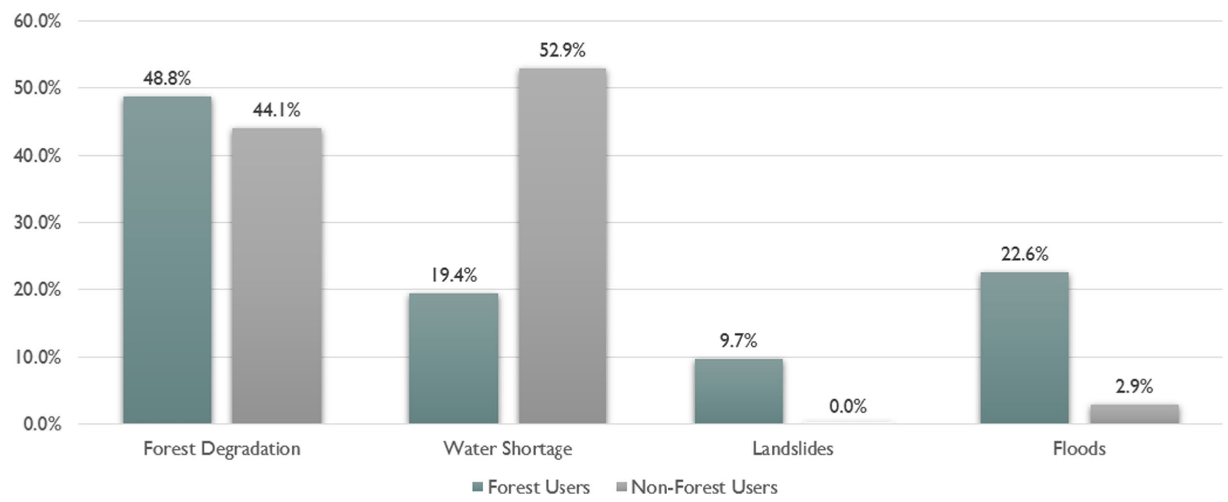

FIGURE 5
Perception of local environmental problems.

farmers that it would lead to the loss of valuable arable land and deprive the farmers of other subsistence agricultural opportunities. However, it was identified through the interview that 74.1% of households are willing to plant trees on their land in order to meet their own fuelwood demand if the government can support temporary fencing. Through the survey, it was also identified that local people are not aware of management through planting mixed tree species (Table 8). Most local communities during the interview showed their interest in learning if the government can provide capacity-building activities in order to enhance the protection status of natural forests and increase the income of dependent communities. Walnut and pistachio are commonly harvested from the forest; however, planting fast-growing tree species such as *Populus* can be integrated into farming systems around the forest, to provide alternative sources for fuelwood. Furthermore, fast-growing *Populus* mitigate the negative effects of droughts by absorbing groundwater with its deep roots (Djanibekov et al., 2013).

3.3.4 Satisfaction with the forest lease policy

In order to understand the challenges, through a satisfaction interview on forest lease policy, it was identified that 61.3% of respondents were satisfied, while 38.7% had various challenges, including changes in the forest lease policy (Table 9). A total of 62% of respondents indicated that the forest lease process became complicated, and 61.3% answered that they were not satisfied with the support from the government.

Following a series of interviews with local forest users, it became evident that the majority of community members are generally aware of the arrangements in place for accessing forest resources. However, recent policy changes have left many households unaware of their rights, including the legal and financial aspects, as well as the management of forest lands.

During interviews, several respondents expressed frustration with the allocation of forest lands by the *leshoz* (forest management

TABLE 8 Questions and responses to interview questions from forest-user household.

S/N	Interview question	Yes	No
1	Do you want to increase forest land through lease/tenure?	74.1	25.9
2	Do you know well borders of your leased land?	90.3	9.7
3	Do you collect wood from your leased land?	83.9	16.1
4	Do you use irrigation in your leased land?	35.5	64.5
5	Are you planning continuously lease land?	83.9	16.1
6	Do you know agroforestry/agrosilvopasture?	16.1	83.9

TABLE 9 Questions of satisfaction interview on forest lease policy.

S/N	Interview question	Yes	No
1	Are you satisfied with the forestland lease process?	61.3	38.7
2	Does forestland lease process simple?	38.0	62.0
3	Are you satisfied with the support from the government?	42.0	58.0
4	Are you satisfied with your land use rights?	38.7	61.3
5	Is forestland fee expensive?	71.0	29.0

authority) to individuals who do not reside in the study area and do not actively utilize the leased forest land, despite the significant reliance of local residents on these resources for their livelihoods. It was noted that *leshoz* management possesses considerable discretion in simplifying lease procedures, leading to dissatisfaction among the local population.

A total of 70% of respondents among central government officials acknowledged the necessity for a new or modified policy regarding forest land tenure. These observations align with broader economic expectations and highlight the need for further research to assess the impact of improved forest tenure. We anticipate similar findings from research in Toskool-Ata, providing additional insights into the effects of enhanced forest tenure policies.

Traditionally, in addition to livestock production, haymaking and walnut harvesting were the main sources of income in these walnut forests (Rehnus et al., 2013) Investigation through interviews showed that the economic performance of haymaking surpassed that of agricultural crops due to lower inputs such as labor costs.

Due to the lack of budget in the government, only 6.3% of all government officials indicated the possibility of fencing from the government (Table 10).

4 Discussion

4.1 Benefits and impact of forests on the socio-economy of the local people

Forest resources are recognized as readily available and valuable assets that can be harnessed in the battle against rural poverty (Fischman, 2012). The distinctive nut-fruit forests found in Toskool-Ata exemplify the multifunctional utilization of forests. These forests play a significant role in income generation, offering substantial benefits with low labor input and underscoring the importance of nut crops due to their high market value (FAO, 2016). For instance, walnut trees have the potential to significantly alleviate poverty, but it might take approximately 20 years to fully realize the benefits unless fast-growing tree species are planted alongside them (Hardy et al., 2018). Promoting the economic cultivation of local and endemic cultivars presents viable *in situ* conservation options for preserving genetic diversity as well.

TABLE 10 Percentage of response for possible incentives from the government.

What are the possible incentives? ($n = 16$)	Frequency	Percent (%)
Planting trees instead of lease fee	11	68.8
Provide loan for planting	1	6.3
Fencing	1	6.3
None	3	18.8

4.2 Perception of local people on local forest-related problems

Farmers who are non-forest users are not familiar with the range of ecosystem services provided and thus have a low perceived value of this land use. The crucial role of forests is mainly recognized by local farmers living near forests, and 48.8% of all respondents indicated that the main ecological problem in the area is deforestation. The lack of perception about various management possibilities in Kyrgyzstan can be explained by the fact that policies were shaped by Soviet Union strategies.

4.3 Willingness to grow fast-growing tree species in order to meet the demand for fuelwood through agroforestry or agrosilviculture

Planting trees provides goods and ecosystem services, offering numerous advantages not only at the household level but also at the village, country, regional, and global levels. These positive impacts are full-value public goods, contributing to food security and income of the rural population, and should be regarded as such Feyisa (2017). Farmers are willing to plant trees since they understand the added value of goods and services, which can increase their financial value (Fleming et al., 2019).

4.4 Challenges in long-term forest land lease/tenure

The majority of the people interviewed indicated that they not only feel insecure leasing the forest lands but there are also more cases of rich people who do not use forests or live nearby leasing forests for the long term. However, they do not do any activities on those lands. When farmers are certain about their land possession, they will be more willing to make long-term investments, such as agroforestry (Djanybekov et al., 2015).

Fencing around the seedling within pastures would prevent livestock from damaging forests and increase its chances of growing.

The lack of natural regeneration within pastures might cause a decrease in the overall tree cover and should therefore be brought to the attention of local authorities and organizations (Chamayou, 2011).

4.5 Possible supports or incentives for local communities

National and local incentives are essential to realizing the environmental and economic potential of agroforestry and its contribution to sustainable development in the Central Asian Countries (CACs). The role of the state is important in the development of multifunctional forest management (Fisher et al., 2004) in Kyrgyzstan. In particular, a change in the legal status of land rights is essential. Land and income tax exemptions may be considered to raise the financial attractiveness of agroforestry in the beginning (Norris, 2008). Local support is required to cover initial investments and attract farmers for such land use (Djanybekov et al., 2015). In order to make forestry healthy and sustainable, technical, financial, organizational, educational, and promotional activities are required.

This study posits that the development of incentive-based policies or enhancements to existing policies, such as offering incentives to forest land users for activities such as tree planting in lieu of lease fee payments, could lead to an increase in public goods derived from forest resources (Bruce et al., 2010). Given budget constraints, the government may face challenges in providing additional payments as incentives for local forest land users. However, offering the option to plant trees instead of paying lease or tenure fees could serve as a mutually beneficial solution for both local communities and the government.

5 Conclusion

The pivotal role of forests is widely acknowledged by local farmers residing in proximity to forested areas, with 48.8% of respondents highlighting deforestation as the primary ecological concern in the region. However, due to the absence of alternative energy sources, households are compelled to gather fuelwood from forests. Additionally, livestock grazing within forest lands contributes to vegetation depletion, exacerbating forest degradation. The reliance of local communities on natural vegetation for shelter further underscores their inability to fence leased land due to budget constraints.

This study examined constraints and opportunities for adopting agroforestry practices in the Kyrgyz Republic, utilizing a co-evolutionary socio-ecological systems framework. The framework facilitated the development of guiding questions, semi-structured interviews, and field observations. The findings regarding household characteristics revealed a significant dependence on local natural resources, particularly tree fruits. Given changing climatic conditions, the productivity of walnut forests will be pivotal for resilience.

These findings hold relevance within the global context, as there is increasing attention and support for adopting an ecosystem approach to rehabilitation and ecosystem restoration, particularly through tree planting. Moreover, this research sheds light on the necessity of addressing farmers' concerns when devising strategies to promote environmentally friendly practices.

The study underscores that farmers' awareness and satisfaction with forest land lease policies significantly influence their perceptions. Policymakers can leverage drivers of attitudes and perceived behavioral control to steer farmers' intentions and behaviors toward tree planting for forest restoration. Furthermore, the study reveals that secure land tenure positively impacts farmers' forest plantation activities and investments, suggesting that government incentives can motivate households to invest more in silviculture.

The preferred government incentive for tree planting, as indicated by respondents, is the fencing of forest lands. However, barriers such as past experiences, lack of knowledge, and inconsistent interpretation of legal rules hinder environmentally friendly behavior among farmers. To address these barriers, this study offers a novel contribution by elucidating the underlying values shaping perceptions of agroforestry.

Policy implications suggest that granting increased freehold land tenure could stimulate forest plantation establishment by farmers. Secured land rights conducive to plantation establishment may motivate more farmers to participate in such initiatives. Recognizing the values driving behavior is foundational for tailoring future approaches to increase agroforestry adoption.

In conclusion, this study suggests that under current conditions, market development for forest plantations can be successful and offer commodity benefits to landowners, consumers, and the country. Expanding similar analyses to other countries with varying political and economic circumstances would further validate the robustness of the study's findings.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Ethics statement

Ethical review and approval was not required for the study on human participants in accordance with the local legislation and institutional requirements. Written informed consent from the [patients/participants OR patients/participants legal guardian/next of kin] was not required to participate in this study in accordance with the national legislation and the institutional requirements.

Author contributions

AZ: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Resources, Visualization, Writing – original draft. HH: Conceptualization, Funding acquisition, Project administration, Supervision, Validation, Writing – review & editing. JY: Investigation, Methodology, Resources, Writing – original draft. PP: Supervision, Validation, Writing – review & editing. K-SK: Supervision, Validation, Writing – review & editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Supplementary material

The Supplementary material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2024.1349842/full#supplementary-material

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Temporal analysis of the state of the Gbele Resource Reserve in the Upper West Region, Ghana

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Introduction: This paper assessed the changes in the forest cover of the Gbele Resource Reserve from 1990 to 2020. This provides a basis for strengthening management decisions to protect the resources in the Gbele Resource Reserve effectively.

Methods: Landsat images for 1990, 2000, 2010 and 2020 were obtained from the United States Geological Service site. They were processed and classified in the System for Earth Observation Data Access, Processing, & Analysis for Land Monitoring (SEPAL), a web-based cloud computing platform. The accuracy of the images was assessed using 50 ground-truth points obtained from the 3-5 meter spatial and near-daily temporal resolution planet satellite images from Norway's International Climate and Forest Initiative (NICFI). Post classification change detection was used to analyse the changes in land cover from 1990 to 2000, 2000 to 2010 and from 2010 to 2020.

Results: The analysis revealed that the total forest area was 55273.2 ha. In 1990, 74.9 % of the reserve was open forest and 24.6 % was shrubs/grass. The open forest declined to 65.8 % in 2000 and further to 62.4 % in 2010 while the shrubs/grass cover increased to 35.7 in 2010. As of 2020, the forest increased to 73.6 % while the shrub/grass cover declined to 25.8 %.

Discussion: These changes could be attributed partly to widespread charcoal production in the fringe districts and rose wood harvesting in the early 2000s. charcoal production and rosewood logging have been livelihood sources for fringe communities. The ban on the harvest and exportation of rosewood after 2010 could partly explain the sharp increase in the open forest cover from 2010 to 2020. The changes in the extent of the reserve from 1990 to 2020 revealed that the reserve can vulnerable to excessive exploitation and can also be resilient if deliberate efforts are made to protect it. It is recommended that the fringe district and municipal Assemblies should strengthen the enforcement of the ban on the logging of the rosewood and trees in the reserve for the production of charcoal.

KEYWORDS
land cover, change detection, Sepal, image classification, open woodland

1 Introduction

Forests are important resources for the welfare of millions of the poor and the marginalized in Africa, and if they are effectively managed, they could improve livelihoods and the quality of life in the continent and facilitate the achievement of the Sustainable Development Goals (SDGs) (Center for International Forestry Research, 2005; Oyewole et al., 2019). Forests play a role pivotal in greenhouse gas emission reduction and climate change adaptation and mitigation and are central to the attainment of SDG 13 since 77% of the carbon stored in vegetation exists within forests, and 39% of the carbon stored in soil occurs underneath the forest cover (Oyewole et al., 2019).

Besides, forests contribute to poverty reduction, food security, improved healthcare and shelter (African Natural Resources Centre, 2018). Thus, the forests are capable of playing a significant role in achieving SDGs 1 and 8, which emphasize livelihood improvement, employment opportunities and poverty alleviation (Oyewole et al., 2019). This is because over 90% of people living in extreme poverty depend on forests for their livelihood (Oyewole et al., 2019). For instance, it is estimated that Uganda's forest reserves could generate income of more than US$ 100 million a year while the Kenya forests are estimated to provide subsistence worth more than $100 million a year to more than a quarter of the population (Emerton, 2001).

Although Ghana is making efforts toward achieving the environmental-related SDGs through policies such as Greening Ghana, Planting for Food and Jobs and the forest sector improvement policies, a lot of burden is exerted on existing forests and woodlands for livelihood adaptation against climate change. These have contributed to the poor performance of the country in the global environmental performance assessment. Ghana is ranked 170 out of 180 countries in the 2022 Environmental Performance Index (Wolf et al., 2022). Comparatively, Ghana was ranked 109 out of 163 countries in 2010 (MEST, 2012) and this shows that the country's environment has deteriorated compared to its global peers.

Amankwah (2012) observed that the northern part of Ghana is gradually experiencing degradation of vegetation cover, particularly areas where charcoal production is rampant. This has the potential of reversing the efforts of the Ghana government to achieve the environmental-related SDGs. These negative effects on the vegetation cover of the country have contributed to the deteriorating environmental situation in the country. Parts of the woodlands in the Upper West Region (UWR) of Ghana are gradually turning into grassland, which could result in desertification in the long term if the trend continues (Amankwah, 2012; Tengapoe et al., 2023). Afriyie et al. (2021) also observed that charcoal production has severe impacts on biodiversity in protected areas including the Gbele Resource Reserve and recommended that management planning should deliberately include mechanisms to detect the charcoal production activity and its effects on protected areas.

The Forestry Commission's 2021 inventory of rosewood across the country reported a 50% reduction in total stem numbers compared to the 2013 estimate. In the case of the rosewood, concerns were raised over excessive illegal logging, leading to intermittent bans on the felling and trade of rosewood starting in 2012. These raise questions about the impacts of these activities on the reserve and its real state, leading to arguments that management's efforts to conserve the flora of the Gbele Resource Reserve are ineffective.

A number of studies (Yahaya and Venkateswar, 2016; Osumanu and Atia, 2017; Afriyie et al., 2021) have addressed aspects of the bigger problem of the Gbele resource Reserve. Yahaya and Venkateswar (2016) questioned the effectiveness of modern institutions in conserving the flora of the Gbele Resource Reserve in the Upper West Region of Ghana and concluded that the institutions were ineffective in trying to accomplish the goal of nature conservation. Osumanu and Atia (2017) investigated the role of communities in the management of the Gbele Reserve and concluded that communities were constrained in participating in effectively managing resource reserves and were not involved in the sharing of direct economic benefits. Afriyie et al. (2021) assessed the management effectiveness of three protected areas in Ghana, including the Gbele Resource Reserve concluded that the selected protected areas faced intense external pressures and threats from human activities which were deeply influenced by the macro-economic and social environments of the country. While these aspects are relevant for sustaining the resource reserve, no long-term land cover analysis has been conducted on the reserve to ascertain the forest cover status and substantiate such arguments. This study assessed the state and changes in the land cover of the Gbele Resource Reserve from 1990 to 2020. This is necessary because the wild resources in the reserve depend on the vegetation for habitation and food. The assessment is equally necessary amid the increasing demand for charcoal, rosewood and fertile land for crop farming in the fringe communities of the reserve, which exerts pressure on the reserve. For instance, Jachmann et al. (2011) observed that the majority of the protected areas were becoming islands surrounded by people and farmlands, with average densities of 43 people/km^2 in the savanna areas. The reserve is also part of the network of corridor for the movement of wildlife from The Nazinga Reserve in Burikina Faso to the Mole National Park in Ghana. A degradation of the Gbele Resource Reserve would limit the movement of wildlife as they would be exposed to attacks.

2 Methodology

2.1 Description of the study area

The Gbele Resource Reserve (GRR) is located in the Upper West Region of Ghana and covers an area of 565 km^2. It is located between latitudes 10° 20' N and 10° 44' N and betwen longitudes 2° 18' W and 2° 10' W (Figure 1). It was acquired and gazetted in 1975 to protect plants and wildlife in the area (Acheapong, 2001). The GRR lies within three political administrative districts in the Upper West Region: Sissala West, Daffiama Bussie-Issa, and Wa East Districts (Afriyie et al., 2021).

It has a typical Guinea Savanna vegetation dominated by *Burkea africana, V. paradoxa, Parkia biglobosa, Terminalia* spp., *P. erinaceus*, and grasses such as *Hyparrhenia* spp. and *Pennisetum* spp. The common mammals are *Olive baboon Papio anubis, Patas monkey, King colobus Colobus polykomos, Roan antelope Hippotragus equinus, Bushbuck Tragelaphus scriptus, Waterbuck, Oribi, and Common warthog Phacochoerus africanus*. It also contains a number of endangered wildlife species namely, *Panthera leo, Loxodonta africana, Cercopithecus p., Syncerus caffer, Kobus* sp. *and Neotragus pygmaeus*.

The Gbele community is the only community that existed in the reserve at the time it was gazetted (Acheapong, 2001) but Bouché (2007) reported that two communities were in the reserve as of 2007. It is part of the protected areas network in northern Ghana and the Western Wildlife Corridor for the migratory route for elephants the from Nazinga Game Ranch in Burkina Faso through Gbele Resource Reserve to Mole National Park (World Bank, 2014).

Charcoal production and wood logging are brisk businesses in the fringe communities of the reserve. These activities could have a negative influence on the state of the forest and undermine the purpose for which it was created. In recent years, the reserve came under pressure as a source of rosewood (*Pterocarpus erinaceus*) for export and wood for charcoal production.

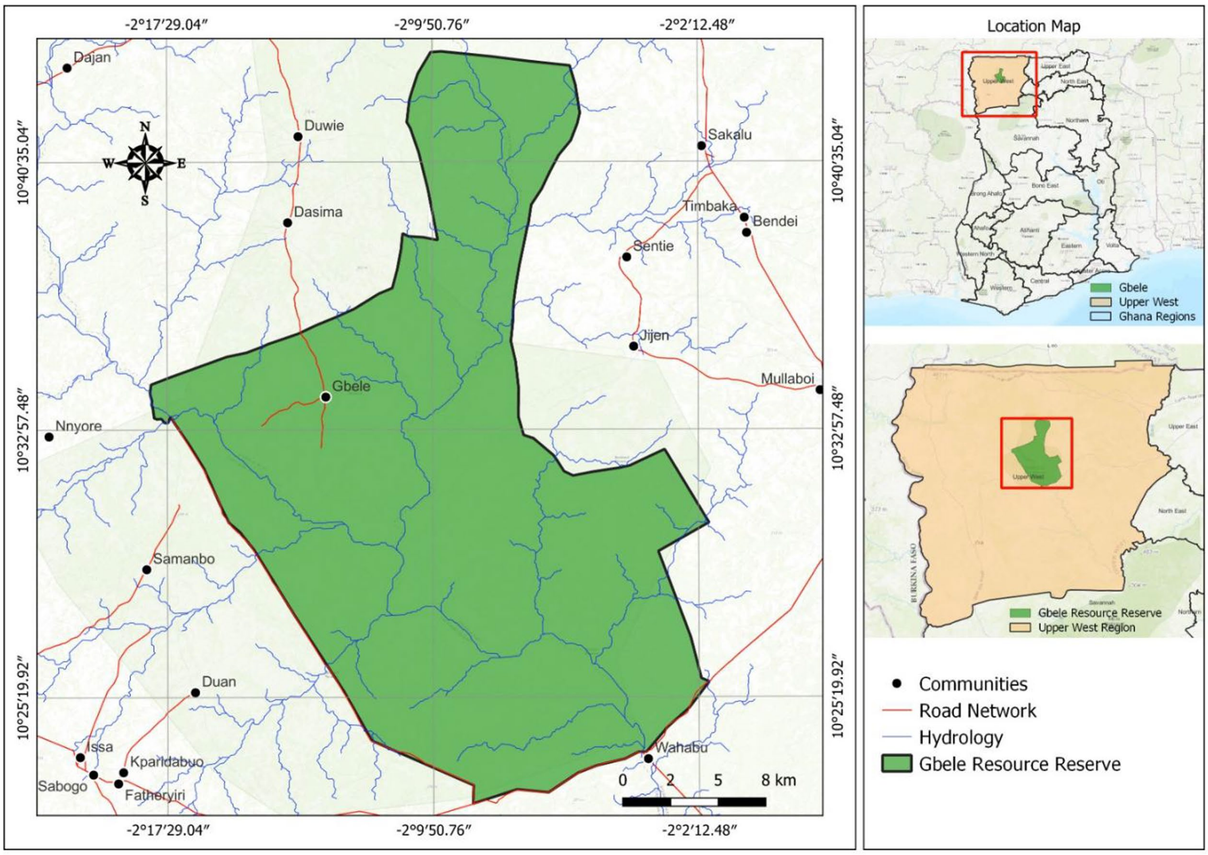

FIGURE 1
The Gbele Resource Reserve. Source: Authors' Construct, (2024) based on ESRI ArcGIS online Topographic Basemap.

TABLE 1 Satellite image characteristics and their date of acquisition.

Satellite	Sensor	Processing Level	Paths/ Rows	Image Date
Landsat 4–5	TM	L1TP	195/053	1990/10/12
Landsat 7	ETM+	L1TP	195/053	2000/10/31
Landsat 7	ETM+	L1TP	195/053	2010/11/12
Landsat 8	OLI	L1TP	195/053	2020/10/30

TABLE 2 Land cover description.

ID	Land cover	Description
1.	Open woodland	Natural woody vegetation with canopy cover ranging between 10 to 35% and canopy height exceeding 2.5 metres.
2.	Shrub/ Grassland	Vegetation is dominated by perennial grasses and shrubs.
3.	Bare Land	Areas with exposed soil surface
4.	Water Body	Areas holding surface water such as; streams and rivers.

2.2 Description of data

Cloud-free Landsat satellite images of the Gbele Resource Reserve for 1990, 2000, 2010, and 2020 were downloaded from the United States Geological Survey site (http://earthexplorer.usgs.gov). The reserve falls within row 053 and path 195 (see Table 1). To minimize the effect of seasonal variation in the different Landsat image tiles, all images were acquired for the period between October and November, the dry season.

Four land cover classes were considered to reflect the dominant land cover of the Savannah ecological zone where the study site is located following the IPCC, 2022 classification scheme. The four classes include open woodland, shrub/grassland, bare land and water body (Table 2).

2.3 Satellite image pre-processing, classification, and analysis

The image pre-processing and analysis were performed in the System for Earth Observation Data Access, Processing, and Analysis for Land Monitoring (SEPAL) environment, a web-based cloud computing platform. The satellite images were corrected for errors due to atmospheric and radiometry effects. Varying sun angles and surface reflectance changes were corrected in SEPAL. The radiometric corrections were performed using the "use scenes atmospherically corrected surface reflectance (SR)" and "correct for bidirectional reflectance distribution function (BRDF) effects" in SEPAL.

Fifty (50) ground truth points were collected from Norway's International Climate and Forest Initiative (NICFI) planet satellite base maps for each of the predetermined land cover classes. NICFI planet satellite base maps are of spatial resolution of 3-5 m and near-daily temporal resolution and facilitated the extraction of ground truth data. These points were used to train the Random Forest algorithm in SEPAL based on the spectral reflectance of Landsat satellite images, and the NICFI planet satellite base maps.

Random Forest Algorithm (RFA)was used to run a supervised classification of the images in SEPAL. It categorized and grouped pixels based on their likelihood of belonging to each of the land cover classes. The RFA is a machine learning technique that is increasingly used to classify images (Horning, 2010). RFA is regarded as one of the most extensively utilized algorithms for land cover classification (Jin et al., 2018). According to Xia et al. (2017), RFA offers several advantages, which include effective handling of outliers and noisy datasets, good performance with high-dimensional and multi-source datasets, higher accuracy compared to other popular classifiers, and enhanced processing speed by selecting important variables. The classified images were exported to ArcMap for the calculation of the extent of the various land cover types and the composure of the maps.

2.4 Accuracy assessment

Accuracy assessment is a tool used to evaluate the classification approach to determine whether the classified map conforms to reality based on the corresponding reference data from the ground control points (Manisha et al., 2012; Abbas and Jaber, 2020). Accuracy assessment was performed using the confusion matrix to compare the relationship between classified thematic and the reference points from ground-truthing. A total of 200 ground truth points were used to assess the accuracy of the classified image. The aspects of the accuracy assessed were producer accuracy, user accuracy, overall accuracy and kappa statistics.

2.5 Change detection

A two-date post-classification change detection was carried out to assess the landscape transition over time. Change detection analysis helps to reveal the inter-class changes between the various years and the land cover transitions. This assessment was done using the change detection algorithm in the Semi-Automatic Classification in QGIS. Final land cover maps and change maps were composed in the ArcMap version 10.8 environment.

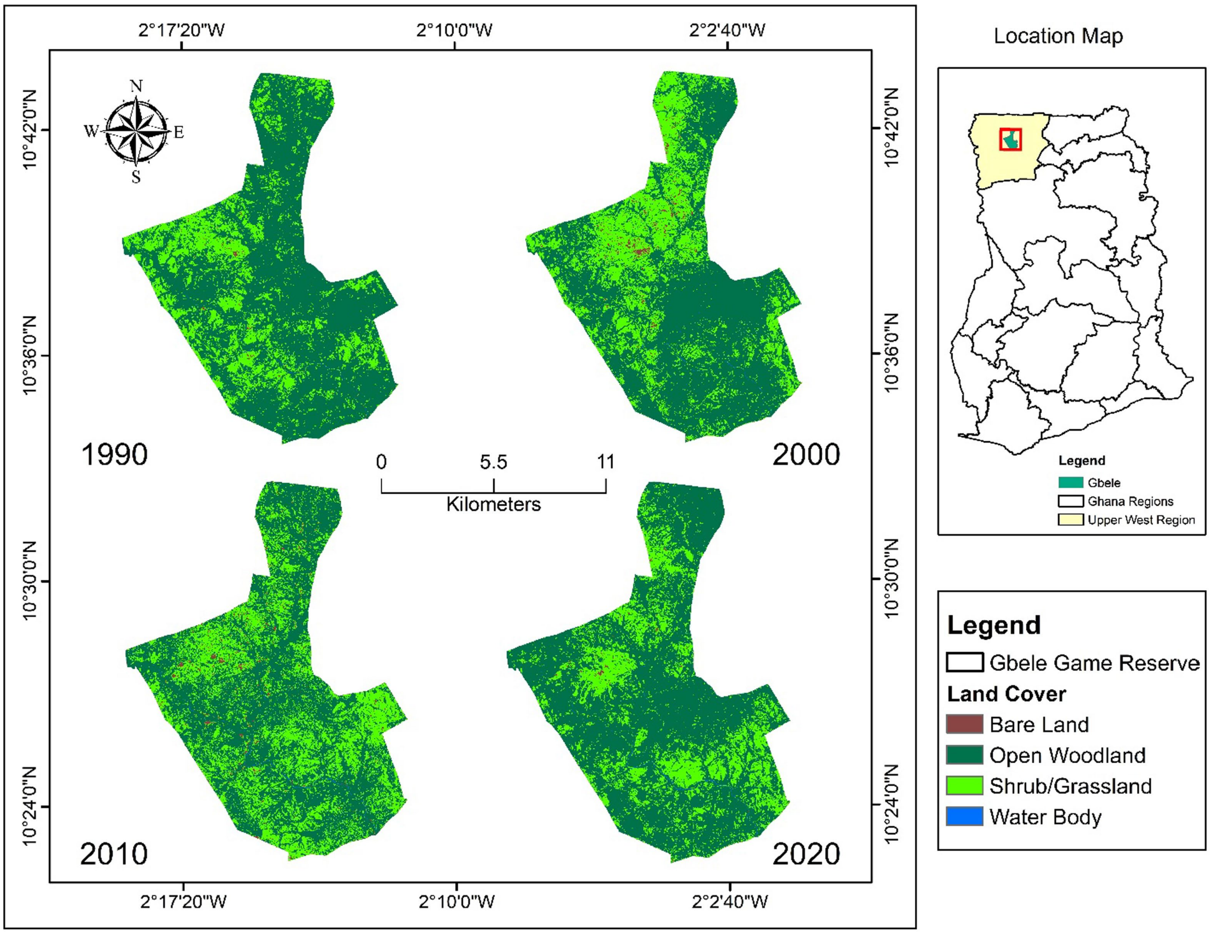

FIGURE 2
Spatial distribution of land cover types across the landscape of the Gbele Resource Reserve.

The gain, loss and swap for each land cover type were computed using Equations 1, 2 as in Huang et al. (2012) and Pontius et al. (2004) respectively.

$$g_j = C_{+j} - C_{jj} \quad [1]$$

$$l_i = C_{i+} - C_{ii} \quad [2]$$

Where g_j and l_i are the observed total gain and loss for land cover class *j* and *i*, respectively, C_{i+} and C_{+j} are the extent of land cover class *i* and *j* for the reference and current years, respectively, C_{ii} and C_{jj} are the no change area of class *i* and *j*, respectively.

3 Results and discussion

3.1 Land cover of the Gbele Resource Reserve

This section shows the vegetation cover that was present in the Gbele Resource Reserve specifically in the years 1990, 2000, 2010, and 2020. Figure 2 shows the spatial distribution of the land cover across the landscape. The open woodland dominated the north, north-eastern, eastern and south parts of the reserve in 1990, accounting for 75%. Shrubs and grass, which constituted 24.5%, dominated the western, north-western and south-western parts. By 2000, the shrubs had taken over the northern part with portions of the western parts recovered from shrubs to open woodland. Figure 3 shows the extent of the land cover classes in percentage. The woodland declined to 65.8% while the shrubs and grass area increased to 32.8% with noticeable bare land, covering about 1.4% of the reserve. In 2010, the open woodland not only declined in extent, but it was fragmented by shrubs and grass. The extent of open woodland reduced to 62.3% while the shrubs and grass increased to 35.7%. By 2020, the declining trend of the open woodland had reversed, and its extent appreciated to 73.6% while the shrubs and grass declined to 25.8%. The decline in the open woodland in the reserve from 1990 to 2010 is attributable to multiple factors, namely charcoal production in the fringe districts (Aabeyir et al., 2023), excess exploitation of rosewood in the middle and northern parts of the country and management issues (Yahaya and Venkateswar, 2016; Osumanu and Atia, 2017). For instance, Aabeyir et al. (2023) noted that the majority of the charcoal transported out of the region was produced at Hain, Zini, Jeffesi, Gwollu, which are fringe communities of the GRR and could have accounted for the decline in the woodland cover in the Reserve. As noted by Osumanu and Atia (2017), if the fringe communities do not

TABLE 3 Accuracy assessment results for the various classified images.

Accuracy Type	Classified images			
	1990	2000	2010	2020
Overall accuracy	90.9	92.9	89.7	91
Kappa coefficient	0.89	0.91	0.88	0.9

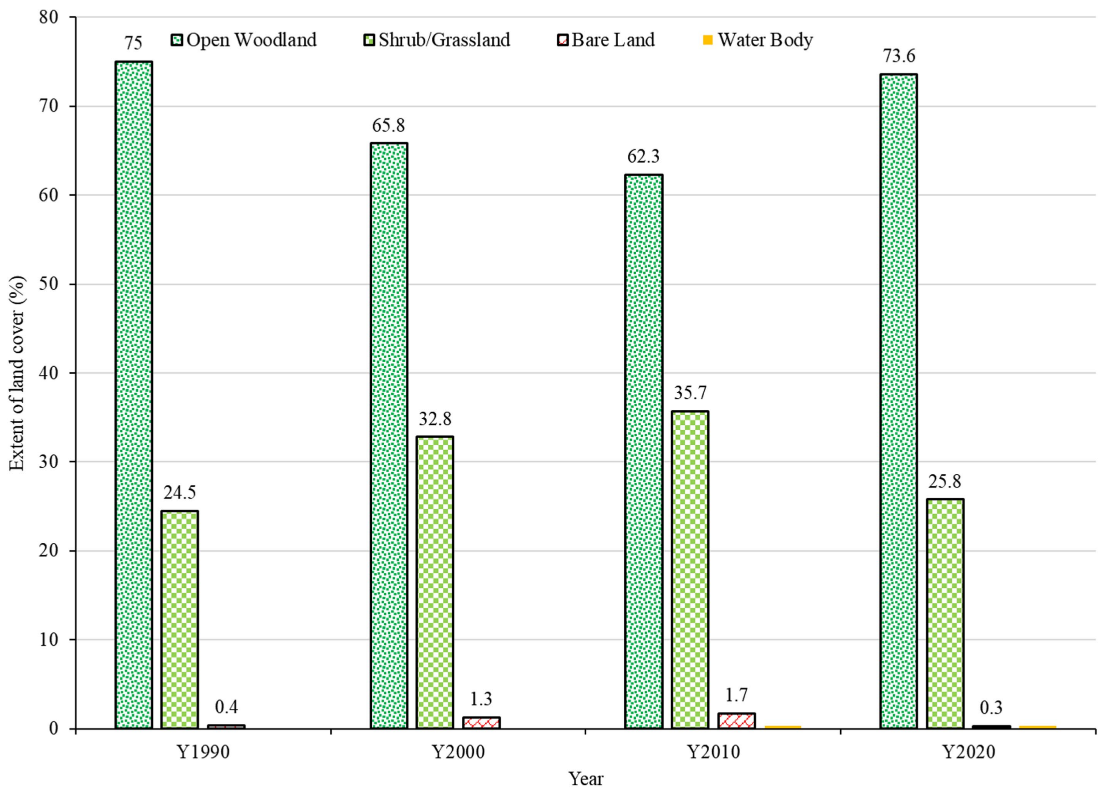

FIGURE 3
Changes in area of the land cover types from 1990 to 2020.

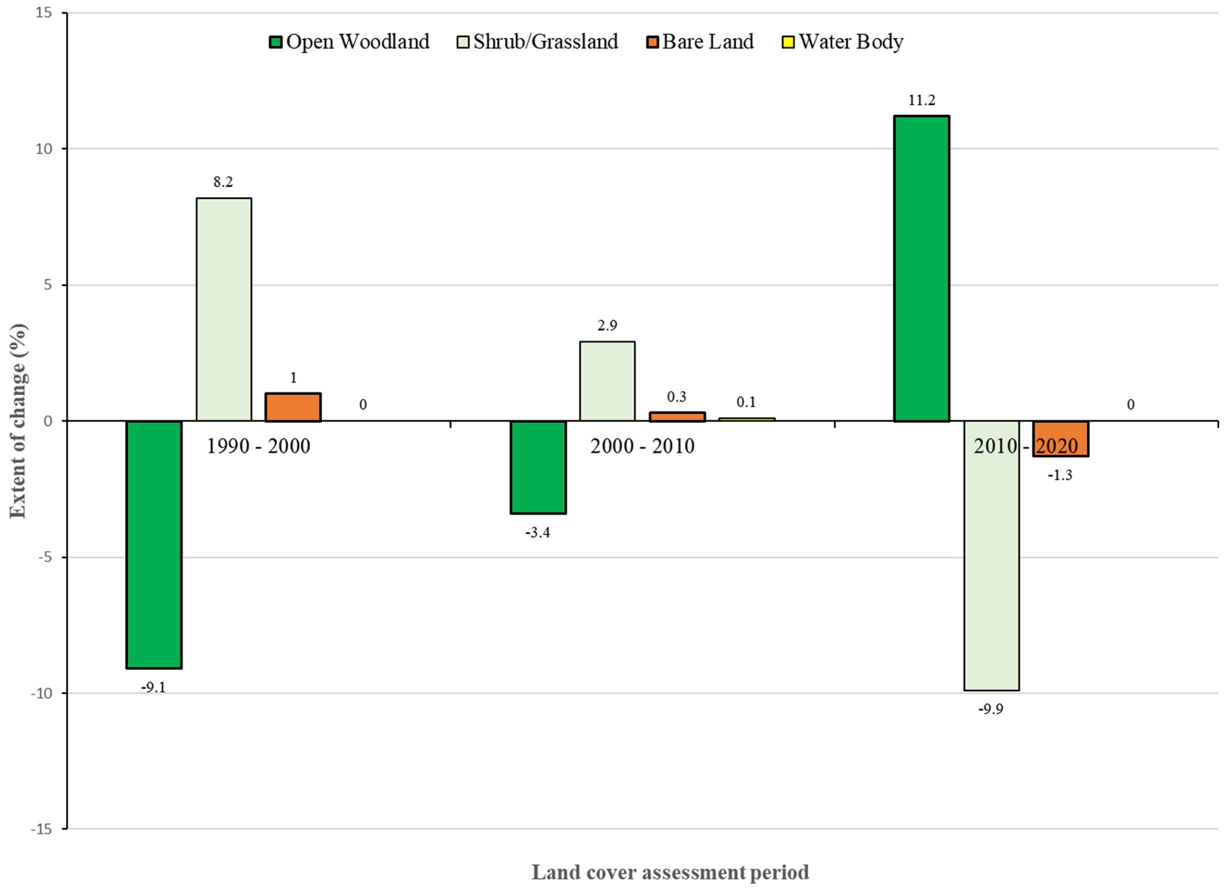

FIGURE 4
Land cover gain/loss in the Gbele Resource Reserve.

enjoy economic benefits that accrue from the management of the Reserve, they will find ways to seek livelihoods from it.

The recovery of the open woodland in the reserve could be due to the ban that was placed on the logging of rosewood by the Ghana Government in 2012. This is because, for a similar period (2011–2021), Tengapoe et al. (2023) reported a decline in woodland along the Black Volta corridor in the Nadowli District where there was no protection of the woodlands. The recovery is an indication that if the right management strategies are put in place, the vegetation of the Reserve can improve substantially. However, management of the Gbele Resource Reserve should not be complacent and neglect their responsibilities because forest fringe communities placed a greater premium on economic benefits than other benefits and, do not appreciate intangible benefits so much since they did not enhance their economic livelihood conditions (Favretto et al., 2021). This suggest that they could re-exert pressure on the reserve for their livelihood needs.

3.2 Accuracy of the maps

Post-classification accuracy assessment was performed to evaluate the relationship between classified land cover and the reference points from the ground truthing. The accuracy assessments were done using the confusion matrix to compute the overall accuracy, and kappa coefficient (Table 3). The kappa coefficient, also known as Cohen's kappa is a statistical measure to calculate the level of agreement or reliability of a land cover theme. The kappa coefficient has become a standard means of image classification accuracy assessment (Rwanga and Ndambuki, 2017). Kappa coefficient is interpreted as $0.8 < K < 1$ means there is perfect agreement; $0.4 < k < 0.80$ means there is moderate classification, and $K < 0.4$ means the agreement is not better than expected by chance (Emam, 1999; Jansen and Di Gregorio, 2004). This suggests that the classification results are satisfactory since the Kappa coefficient values are greater than 0.4.

3.3 Land cover gains and losses

The temporal analysis of the gain and loss in the various land cover types over the different decades from 1990 to 2020 revealed significant lost in open woodland cover and significant gain in the shrub and grassland covers for the first decade. The open forest lost about 9% of its cover between 1990 and 2010 while the shrub and grassland gained 8.2% of its cover in 1990 (Figure 4). The reverse occurred in the period from 2010 to 2020. The middle decade served as a recovery period for the open woodland. For the entire period from 1990 to 2020, the open woodland experienced a net loss of 1.3% while shrub and grassland experienced a net gain of 1.2% of its extent in 1990. This suggests as of 2020, the woodland

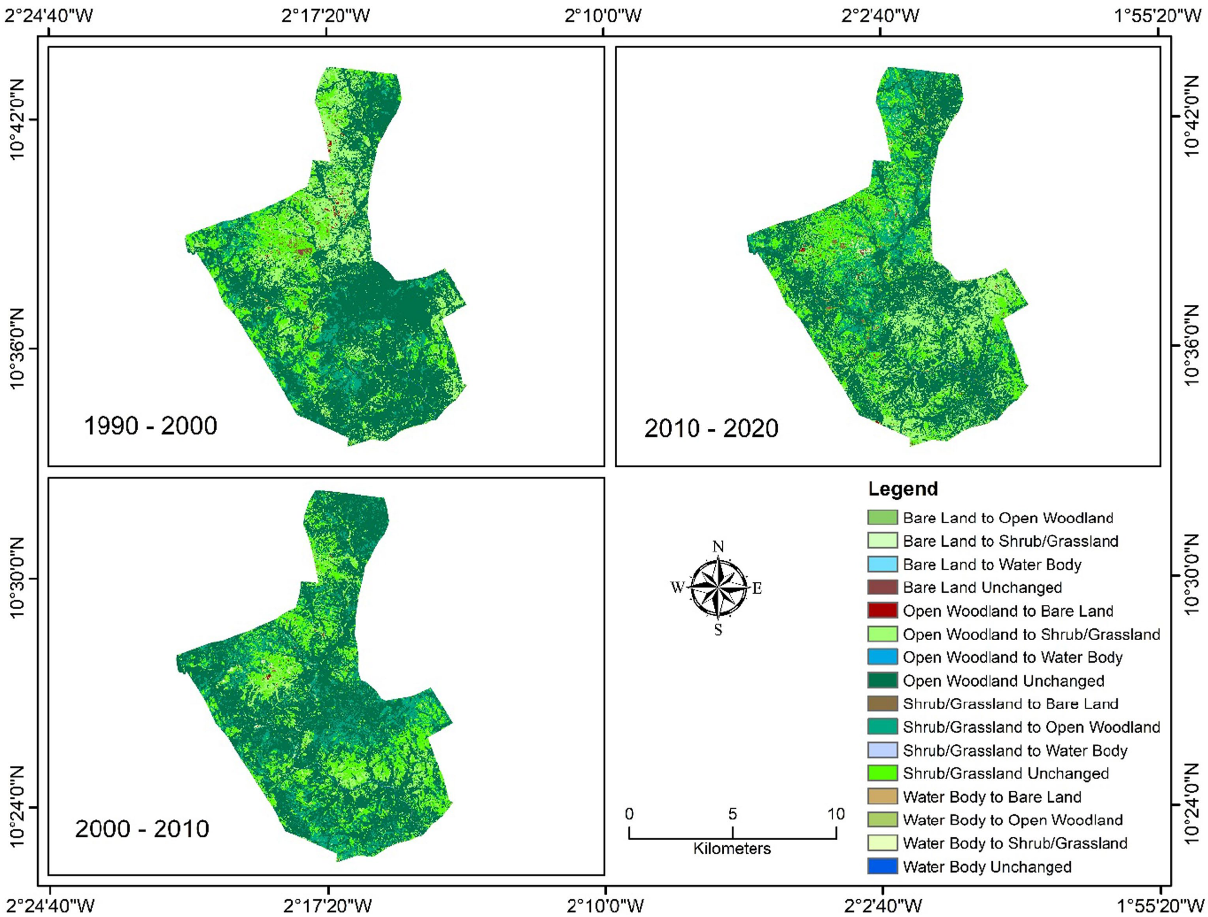

FIGURE 5
Spatial distribution of land cover changes across the landscape of the Gbele Resource Reserve.

did attain its cover in 1990 despite the significant gain it made from 2010 to 2020.

Negative land cover conversion has been identified as a serious threat to many protected areas and should be taken seriously in the management of protected areas. Farming and grazing contribute to the negative land cover conversion around the protected areas (UICN, 2010). In the Gbele Resource Reserve, Afriyie et al. (2021) identified logging, climate change, poaching, and grazing as the most threats to the management of the reserve. These threats to natural resource conservations still exist in the Gbele area could have contributed in no small way to decline in the overall open woodland cover in the resource reserve between 1990 and 2022. This is not surprising because the reserve is located in poverty prone area, where fringe communities resort to logging trees for charcoal production and for sale. For instance, the rosewood menace in the area is well underscored. Also, the reserve is situated in a fragile semi-arid area and the least stress on the reserve amplifies the effects of climate change on the reserve. Activities of nomadic fulbes do sometimes result in bushfires in an attempt to get fresh grass for their animals, the burn the dry grass in or around the reserve. Poachers sometimes burn the parts of the reserve so as to restrict wildlife in a particular area of the reserve where they can easily be poached. Group hunting, involving more than 50 hunters at a time, is a real threat as it takes only one or two people to initiate bush fires in reserve, which generally causes serious havoc to wildlife on an annual basis (Favretto et al., 2021).

As further noted by Afriyie et al. (2021), the effects of pressures and threats deepen with the lack of support from local communities, inadequate funding, and management resources. In the community resource protection, poverty and lack of transparency in benefit sharing are always contributor to lack of community support in resource reserve protection (see Osumanu and Atia, 2017).

3.4 Land cover transition within the Gbele Resource reserve from 1990 to 2020

The land cover transition maps are shown in Figure 5 while the extent of changes is presented in Table 4. In the period between 1990 and 2000 (light green), the main transition was the conversion of open woodland to shrub and grassland which occurred more in the northern part of the reserve.

4 Conclusion and recommendations

The paper assessed the dynamics in the land cover of the Gbele Resource Reserve from 1990 to 2020 to understand the influence of recent increases in the activities of charcoal production, rosewood logging and other anthropogenic pressures in the fringe districts on the Reserve. For the period from 1990 to 2010, the woodland cover

TABLE 4 Land cover transition matrix of the Gbele resource reserve from 1990 to 2000, 2000 to 2020 and from 2010 to 2020.

	2000						
	Land cover	**Open woodland**	**Shrub/grassland**	**Bare land**	**Water body**	**Total**	**Loss**
1990	Open woodland	56.8	17.9	0.3	0.0	75.0	18.2
	Shrub/grassland	8.9	14.7	0.9	0.0	24.5	9.8
	Bare land	0.0	0.2	0.2	0.0	0.4	0.2
	Water body	0.0	0.0	0.0	0.1	0.1	0.0
	Total	65.7	32.8	1.4	0.1	100.0	
	Gain	8.9	18.1	1.2	0.0		

	2010						
	Land cover	**Open woodland**	**Shrub/grassland**	**Bare land**	**Water body**	**Total**	**Loss**
2000	Open woodland	49.4	16.1	0.2	0.1	65.8	16.4
	Shrub/grassland	13.0	18.8	1.0	0.0	32.8	14.0
	Bare land	0.0	0.8	0.5	0.0	1.3	0.8
	Water body	0.0	0.0	0.0	0.1	0.1	0.0
	Total	62.4	35.7	1.7	0.2	100.0	
	Gain	13.0	16.9	1.2	0.1		

	2020						
	Land cover	**Open woodland**	**Shrub/grassland**	**Bare land**	**Water body**	**Total**	**Loss**
2010	Open woodland	55.0	7.3	0.0	0.0	62.3	7.3
	Shrub/grassland	18.2	17.4	0.1	0.0	35.7	18.3
	Bare land	0.4	1.1	0.2	0.0	1.7	1.5
	Water body	0.0	0.0	0.0	0.3	0.3	0.0
	Total	73.6	25.8	0.3	0.3	100.0	
	Gain	18.6	8.4	0.1	0.0		

decreased by 12.5% while the shrubs and grassland increased by 11.1%. However, the woodland cover increased by 11.2% from 2010 to 2020 and the shrubs and grassland decreased by 9.9%. This is an indication that the reserve is recovering from the pressures of illegal rosewood logging that the region experienced in the early 2000. It is also an indication that the ban on the logging of rosewood in the early 2010s could have positive effect of the state of the woodland cover. It is recommended that the fringe traditional authorities and the district assembles should tighten the enforcement of the ban on logging in the reserve.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

RA: Formal analysis, Methodology, Supervision, Writing – original draft. KP: Writing – review & editing. AA: Investigation, Writing – review & editing.

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Pathways to enhance the efficiency of forestry ecological conservation and restoration: empirical evidence from Heilongjiang Province, China

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This paper proposes a theoretical framework for assessing ecological protection and restoration from the perspective of ecological efficiency. We applied the super-efficiency Slack-based measure model to examine the social and economic impacts of ecological resource consumption transformation in Heilongjiang Province, China. Additionally, a convergence analysis was used to evaluate and test the impact of the standard deviation ellipse method on regional sustainability. The results indicated that the land use structure was unstable; the conversion rate of resource consumption was low; and the average Ecological efficiency was only 0.343 in terms of the land use structure. Funds for forest ecological restoration have a significant impact on the effectiveness of ecological resource transformation. Implementing the Chinese ecological restoration project improves the ecological efficiency level of the communities. The center of gravity of ecological efficiency moved greatly in the years when forestry investment increased. Technological transfer and diffusion, experience imitation in environmental regulation, and eventually convergent steady-state levels of the ecological efficiency of different regions are necessary to improve the economic and social development level of regions with low environmental quality efforts should be made to reduce resource consumption intensity, increase fund utilization efficiency, and form a comprehensive and systematic system of ecological environment governance through reasonable enhancement of regional environmental regulations, increased investment in technological advancement, and funds for ecological protection and restoration.

KEYWORDS

Chinese green economy, mixed ecosystems, ecological efficiency, Slack-based measure (SBM) model, standard deviation ellipse, spatial convergence

1 Introduction

Ecological restoration (ER) originated in the early 20th century in European and American nations (Higgs, 2003). The goal is to assist ecosystems in restoring their original structure and function after being damaged (Hira et al., 2023). ER involves assisting ecosystems to sustain their healthy historical development path (Gann et al., 2019). Land degradation can be reversed and biodiversity and ecosystem services can be improved through ecological restoration (ER). Global land degradation and environmental change are substantial risks to biodiversity and ecosystem services, as reported by Pandit et al. (2020). Wortley et al. (2013) state that these issues have been widely integrated into local and global natural resource management methods.

International organizations are currently focusing on the implementation of systematic protection and restoration principles in ecological restoration processes (Rey Benayas et al., 2009). These practices include the Brazilian Atlantic Tropical Forest Landscape Restoration (Chazdon and Uriarte, 2016), South Africa's National Parks Restoration Project (Moyo et al., 2021), and the wide-ranging restoration of riparian systems in the western United States (Oppenheimer et al., 2015). Forest restoration efforts in Nepal utilize the approaches mentioned by Laudari et al. (2022). Chinese governments have made significant investments and offered financial support in order to address the environmental crisis caused by massive land usage (Li et al., 2020; Bi et al., 2021; Yin and Cao, 2022). China has carried out 16 major ecological restoration initiatives since the 1980s, such as restoring farmland back into forests and safeguarding natural forests, resulting in substantial advancements (Qu et al., 2020; Worlanyo and Jiangfeng, 2021; Zhang et al., 2022). However, most countries find it challenging to simultaneously achieve rapid economic development and enhanced ecosystem services (Wang et al., 2021).

Previous studies on ecological restoration evaluation primarily focused on assessing vegetation structure, ecological processes, and biodiversity indicators, contributing to the limited universality and applicability of the ecosystem restoration evaluation index system (Yang et al., 2013). Hence, it is necessary to enhance research on multi-object and multi-scale ecological restoration in order to improve the evaluation system of ecological restoration effects. Researchers are turning their attention on the significance of evaluating ecological restoration in the field of ecology. Consequently, the obstacles encountered in ecological restoration evaluation research are gaining more recognition (Ding and Zhao, 2014). Researchers have conducted numerous ecological restoration evaluation studies regarding China's ecological protection and restoration projects. However, due to variations in research methodology, objectives, and perspectives, research areas encompass biodiversity, ecosystem services, climate change, land use, and ecological restoration theories and technologies. This diversity has led to a wide range of research outcomes and a scarcity of universally applicable summaries of ecological restoration experiences. Summary of a study on ecological restoration with general applicability (Gao and Yang, 2015).

Ecological attributes are the most widely utilized post-implementation indicators in China and other regions, which represent 94% of cases, but research on social and economic indicators is relatively limited at 3.5%. Ecological conservation and restoration research primarily emphasizes high-income countries over countries with significant deforestation rates (Wortley et al., 2013), leading to a notable scarcity of knowledge summaries for regions requiring extensive restoration efforts. Only a limited number of studies have endeavored to assess restoration from a social and economic perspective (Zheng and Zhuang, 2021). It must be emphasized that there is a significant absence of empirical information concerning the social and economic results of restoration (Pandit et al., 2020). Socio-economic factors should be included when assessing ecological conservation and restoration efforts.

Extensive research has been conducted on various ecosystems in China, such as rivers, wetlands, forests, grasslands, agricultural lands, and post-mining lands. However, there remains a need for a comprehensive understanding of ecological restoration practices across all terrestrial ecosystems (Cui et al., 2021). Although urbanization and ER conflict still persist during economic development (Wang et al., 2019), there are various issues that need to be addressed, including insufficient understanding of the systematicity and flowability of ecosystem elements, a lack of coordination between socio-economic and environmental factors, insufficient recognition of ecosystem nature, and a lack of long-term cooperation among departments (Xu et al., 2006; Qu et al., 2020). This has resulted in a lack of coordination, capacity, and adaptability among elements of the national spatial system. As research advances in different areas of ecological restoration, and as more ecologists reflect on large-scale ecosystem changes, ecological restoration programs should act as a bridge between complex and unpredictable changes, helping to restore meaningful and tangible relationships between humans and ecosystems (Ives et al., 2017). A new theory of progressive ecological restoration has also been proposed by scholar Liu (2020), emphasizing the importance of monitoring and dynamic assessment. In order to obtain the long-term effects of ecological restoration programs, ecological restoration projects must be integrated with some form of economic development. In order to optimize the effectiveness of ecological restoration efforts, it is essential to conduct research on evaluating ecological protection and restoration (Erbaugh et al., 2020), establish geographically specific attributes and management strategies (Edrisi and Abhilash, 2021), compile lessons learned, and formulate restoration models suitable for various scales.

China has made ecological conservation, restoration, and optimization national strategic priorities (Li et al., 2020). China proposed integrated restoration of mountain-river-forest-farmland-lake-grassland (MRFFLG) ecosystems in 2013 as an innovative approach to advancing territorial space protection using a systemic and holistic approach (Gao et al., 2022). It requires beginning by optimizing territorial space layout, adjusting land use structure and relationships, and conducting overall protection, systematic restoration, and comprehensive governance in accordance with the integrity, systematicity, and regularity of the ecosystem (Wang et al., 2022). China has implemented three batches (25 in total) of pilot projects since 2016 for the conservation and restoration of MRFFLG, which cover the majority of Chinese provinces (autonomous regions and municipalities). It is imperative to develop quantitative methods to determine whether or not projects are successful in order to rationalize and improve such methods. However, empirical research in ecological conservation and restoration lacks methods for analyzing social and economic outcomes and summarizing successful experiences. This has resulted in a lack of coordination, capacity, and adaptability among various elements in the country's spatial system. This has adversely affected the stability of the spatial structure.

There is no doubt that ER success is not simply determined by ecological progress but also by the ability to fully utilize its potential (Erbaugh et al., 2020), adapt to changes in the ecological environment, and improve the need for ER decision-making and policy (Wortley et al., 2013). Clearly, this promotes the development of an interdisciplinary research area aimed at assessing the effectiveness of restoration strategies from an interdisciplinary perspective and conducting a comprehensive analysis of both ecological and socioeconomic benefits (Abhilash, 2021).

It presents a theoretical framework for assessing biodiversity conservation and restoration based on multi-objective outputs and multi-factor inputs. Heilongjiang Province (HP) in China is used as an example for measuring and evaluating ecological efficiency (EE) of the MRFFLG ecosystem. EE is defined as "creating more goods and services through fewer resources and generating less waste and pollution" (Hitchens et al., 1998). This concept can also be used to describe "the effectiveness of ecological resources in meeting human needs" (Huang et al., 2023). The term EE refers to the efficiency with which MRFFLG natural resources are utilized to meet human needs. This creates more goods and services at a lower cost and less environmental pollution. It is intended to satisfy the multi-objective requirements of the ecosystem. In the context of eco-protection and socioeconomic growth, it is possible to achieve a win-win situation (Mao et al., 2019). Consequently, it is imperative that ecological resources are monitored and evaluated to ensure they are converted efficiently into economic and social outputs. This is to identify significant reasons for sluggish economic development, and to provide a basis for transforming natural resources into economic development opportunities.

This study aims to: (1) develop a framework for assessing the EE of the MRFFLG ecosystem from a social-ecological systems perspective, focusing on transforming ecological resources into economic and social development opportunities; (2) establish an indicator system that evaluates economic, social, and natural resources; (3) analyze the spatiotemporal evolution trend and convergence characteristics of the EE in the study area; and (4) offer policy recommendations based on the evaluation of changes in EE trends and spatial distribution patterns in regional areas.

2 Material

2.1 Study area

Heilongjiang Province (HP) is located in the northeastern forest zone of China and is the northernmost and easternmost province on mainland China (121°11′ W to 135°05′ E, 43°26′ S to 53°33′ N). It is also a key guarantee area for food safety and ecological security of China, with a total area of 473,000 km^2. It is divided into 12 prefecture-level cities and one prefecture, as well as 121 county-level administrative divisions (Figure 1). It is estimated that HP has 47.3% forest coverage and an average annual rainfall of 608.5 mm, with an average annual temperature of 4.2°C in 2021. The HP landscape is characterized by various types of ecosystems, which include five-types of mountains, one-type of water body, one-type of grass field, and three-types of farmlands, while the MRFFLG landscape is distributed over a broad area. A pilot project for "ecological protection and restoration of MRFFLG" is underway in this area. Since HP is a resource-rich province, it has great potential to expand ecological space and serves as an important carrier of ecological security on a national level (Liu et al., 2022). Historical development and the exploitation of natural resources have left the ecosystem with a lot of historical debt. The quality of the ecosystem has not yet been transformed qualitatively. In contrast, HP has lagged behind other provinces in China in economic development. There appears to be a contradiction between urgent economic growth and ecological protection and restoration. Economies and social development in the region must also be balanced with ecological protection and restoration. It is difficult to infer successful experiences and challenges faced by ecological protection and restoration as there is insufficient information regarding spatial changes in land use cover and ecosystem services in HP, the need for economic growth and economic development, and the mismatch between limited resources in quantity, space, and time. In light of the fact that natural resources can be converted into financial efficiency, it is worthwhile to explore the process. Research is challenging due to the variability of degradation types in ecosystems and the specificity of restoration goals. In addition to technical progress in ecological preservation and restoration, a comprehensive goal-guided framework is urgently required.

The measurement of the ecological value of the MRFFLG life community (LC) in HP and the identification of its spatial–temporal differences and evolution trends have great practical relevance for enhancing regional ecological value and promoting harmony between humans and nature.

2.2 Data sources

Specifically, this study examined data for 11 typical ecological cities in HP from 2006 to 2019. This included the 11th Five-Year Plan for National Economic and Social Development (2006–2010), the 12th Five-Year Plan (2011–2015), and the 13th Five-Year Plan (2016–2020). At this time, China's economic and social development entered an exciting phase. The contradiction between natural resources and the environment became more evident, indicating a significant gap between demand space and resource availability. The study uses statistical data collection and calculations based on geographical information as its data sources.

2.2.1 Statistical data

The urban gross domestic product (GDP) data is derived from the "China Urban Statistical Yearbook" (published by the China Statistical Publishing House); the public finance revenue and expenditure data (in ten thousand yuan), the pollution emissions data, the labor force population, and the area of green space *per capita* (in m^2) of urban roads and parks are all reported. All other economic data from 2006 to 2019 are derived from the "Heilongjiang Statistical Yearbook" (published by China Statistical Publishing House); urban CO_2 emissions data (in million tonnes) from 2006 to 2019 are derived from China city emission data in the Chinese Carbon Accounting Database[1] (Shan et al., 2022).

2.2.2 Geographic information data

In order to obtain the HP 2019 elevation data (Digital Elevation Model, DEM), the Geospatial Data Cloud website[2] sourced the SRTMDEM 90 M original elevation data; ESACCI (European Space Agency Climate Change Institute) provided the land use and land cover data from 2006 to 2019.

2.2.3 Extraction of geographic information data

In conjunction with the actual situation and research needs of HP, ArcGIS software was used to perform supervised classification and visual interpretation of remote sensing images. Land use types were reclassified and extracted based on the land classification standards of the third national land survey in China. Reorganizing "wetlands" as a primary land category alongside cultivated land, forest area, grassland and waters. For each prefecture-level city in HP,

1 http://www.ceads.net.cn/data/
2 https://www.gscloud.cn/

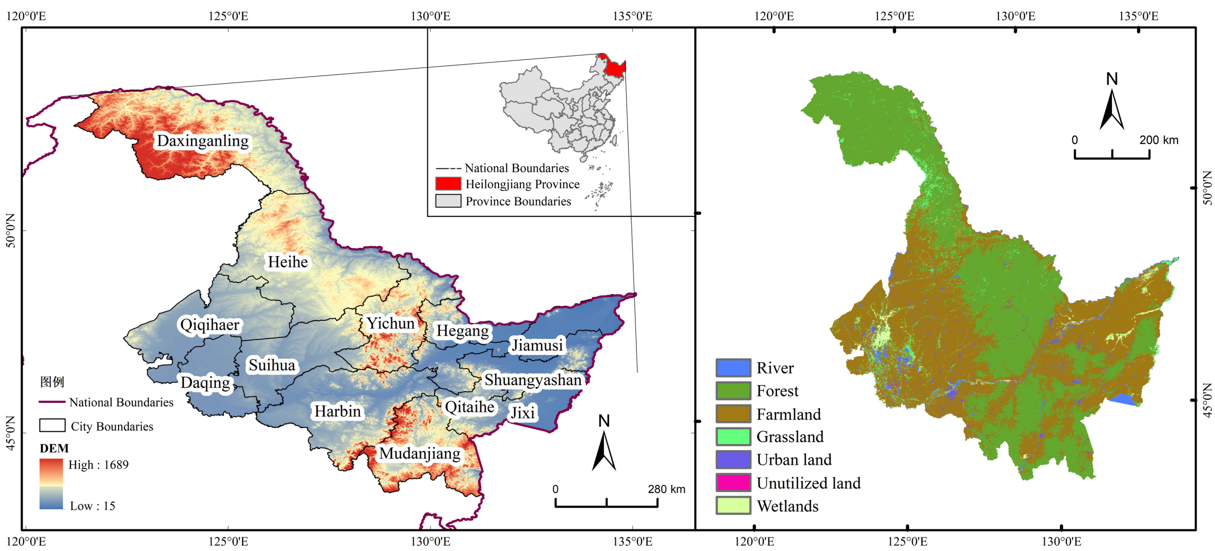

FIGURE 1
Study area in Heilongjiang Province (HP) in China.

TABLE 1 Criteria for reclassification of different land use types.

Primary land use type	Codes representing land-use types	Secondary land-use types
Water area	210	Rivers, Lakes, Reservoirs, Ponds, Canals, Water engineering land, Glaciers, and Permanent snow
Forest area	50/60/61/70/80/90/100/110/120/121/122	Deciduous forests, Bamboo forests, Shrub forests, and others
Cultivated land area	10/11/20/30/40	Rice fields, Irrigated lands, and Dry lands
Grassland area	130/140	Natural grassland, Artificial grassland, Marsh grassland, and Other grasslands
Construction land area	190	Urban and Rural areas
Wetland area	200/201/202	Mangrove forests, Forested swamps, Shrubby swamps, Salt flats, Marshland, and Tidal flats.
Unused land area	180	Unutilized bare areas

water area, forest area, cultivated land area, grassland area, construction land area, wetland area, and unused land area were calculated from 2006 to 2019. Specifically, forest types include deciduous forests, bamboo forests, shrub forests, and others. It consists of rivers, lakes, reservoirs, ponds, canals, water engineering land, glaciers, and permanent snow. Rice fields, irrigated lands, and dry lands constitute farmlands. Natural grassland, artificial grassland, marsh grassland, and other grasslands are mainly included in grassland. Construction land consists primarily of urban and rural areas. Unoccupied lands include fallow lands, terraced lands, and bare lands. Wetland habitats consist of mangrove forests, forested swamps, shrubby swamps, salt flats, marshland, and tidal flats. The specific classification criteria for the various types of land are shown in the Table 1. In cases where some land use types were found to be outside the main land use types expected to be classified during the actual classification process, they were corrected according to the actual utilization status, e.g., bare areas such as 200, 201, and 202 were classified as other types. The reclassification was done according to the above criteria through the ArcGIS Spatial Analysis tool, after which the area of each land type was extracted from the different areas in the area analysis module of the Spatial Analysis tool.

3 Research ideas and methods

3.1 The theoretical framework of research ideas

This article presents a detailed workflow (Figure 2). Assuming MRFFLG constitutes a LC from the perspective of EE, a theoretical framework is proposed that involves multi-objective outputs and multi-factor inputs in order to evaluate ecological protection and restoration.

3.2 Research methodology

3.2.1 Research methodology for measuring ecological efficiency in the community of life

The purpose of this paper is to measure EE in 11 cities in HP using the super-efficiency (SE) Slack-Based Measure (SBM) model, which has obvious improvements over traditional methods. In contrast to the radial SBM model, the SE-SBM model proposed by Tone (2001) integrates both the SE and SBM models, solving the problem that standard efficiency models cannot further compare the

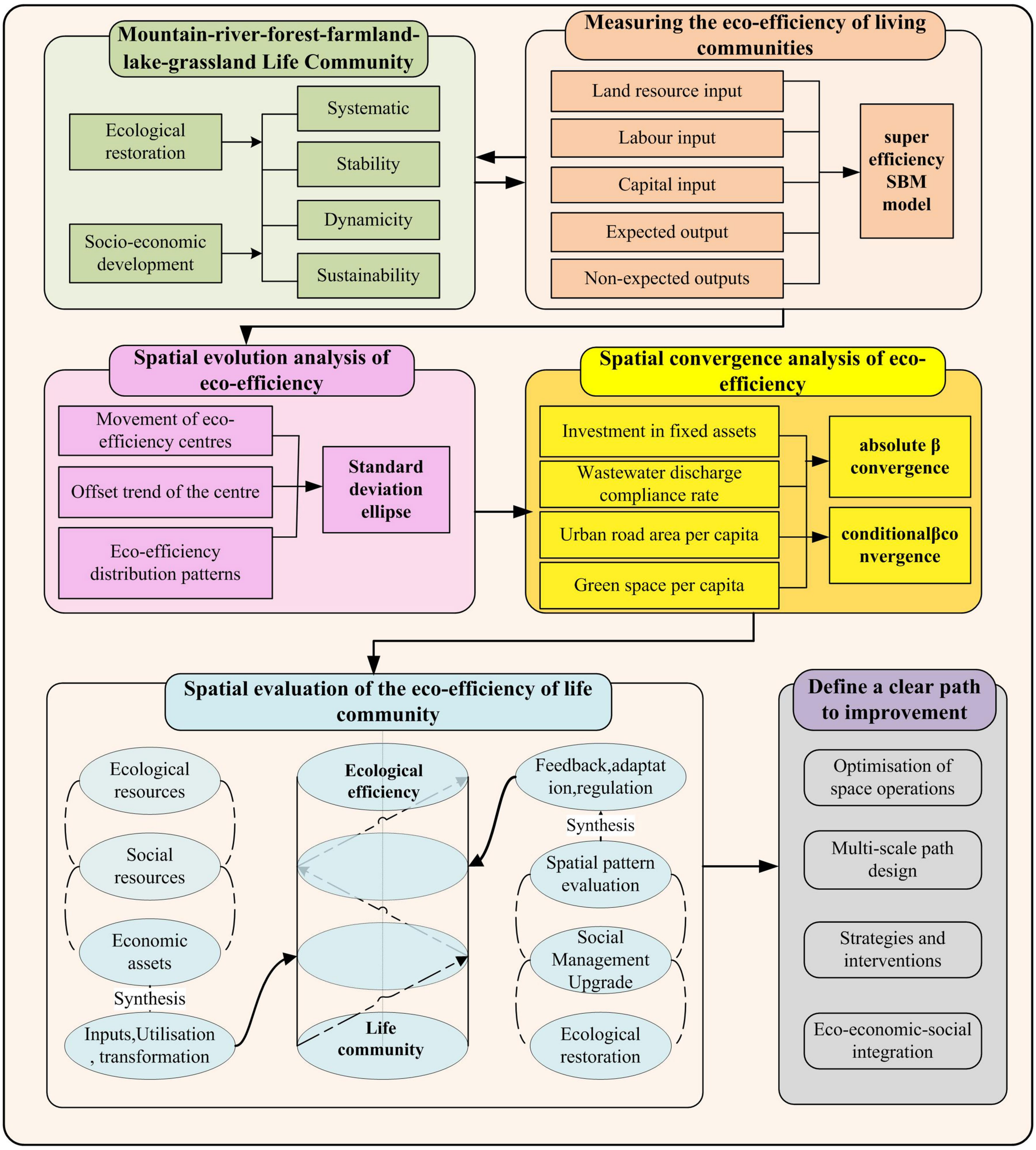

FIGURE 2
Framework for the evaluation of ecological protection and restoration that takes into account multi-objective outputs and multi-factor inputs.

efficiency of decision-making units when multiple decision-making units are efficient. Ultimately, this article selects the SE-SBM model with undesirable outputs for research because it has stronger practicality for measuring the EE within the MRFFLG LC. Equation (1) illustrates the construction of the SE-SBM model.

$$\theta^{*} = \min \frac{1-\left(\frac{1}{m}\right)\sum_{i=1}^{m}\frac{S_i^{-}}{X_{io}^{t}}}{1-\frac{1}{(q+h)\left(\sum_{r=1}^{q}\frac{S_r^{+}}{y_{ro}^{t}}+\sum_{k=1}^{h}\frac{S_k^{-}}{b_{ko}^{t}}\right)}} \quad (1)$$

$$X_{io}^{t} \geq \sum_{t=1}^{T}\sum_{j=1, j\neq 0}^{n} \lambda_j^t X_{ij}^{t} - S_i^{-}, i = 1, 2, \cdots, m.$$

$$b_{ko}^{t} \geq \sum_{t=1}^{T}\sum_{j=1, j\neq 0}^{n} \lambda_j^t b_{kj}^{t} - S_k^{-}, k = 1, 2, \cdots, h$$

$$\lambda_j^t \geq 0,\ j = 1, 2, \cdots, n$$

$$S_i^{-} \geq 0, i = 1, 2, \cdots, m$$

$$S_r^{+} \geq 0, r = 1, 2, \cdots, q$$

$$S_k^- \geq 0, k = 1, 2, \cdots, h$$

$$y_{ro}^t \leq \sum_{t=1}^{T} \sum_{j=1, j\neq 0}^{n} \lambda_j^t y_{rj}^t + S_r^+, r = 1, 2, \cdots, q$$

In the SE-SBM model, represents the optimal solution of the model, which is the EE value of the LC calculated by the model, with a value range of 0 to 1. i represents the number of input variables, which range from 1 to m, assuming there are m inputs. S_i^- represents the slack variables of inputs, indicating the meaning of reduction if inputs are excessive. X_{io} represents the o decision unit. Y_{ro} represents the output of the o decision unit. o is the evaluated object. λ is the weight variable. $\sum_{j=1}^{n} \lambda_j X_{ij}$ is a virtual effective decision unit that serves as a reference point for the optimal solution. Constraints on non-desired output b are added based on this reference point. The summation of λ_b represents the effective decision units constructed using different non-desired outputs, which is also a reference system.S_k^- represents the slack variable of unexpected output, which is the amount that unexpected output needs to be reduced; b_{ko}^b represents the k unexpected output of the o decision unit. For the o -th decision unit, if its unexpected output is to be effective, it needs to be reduced by S_k^-, and if its expected output is to be effective, it must increase output by. It is also required that both the slack variables of inputs and outputs must be greater than or equal to 0. $\lambda_j \geq 0$ represents the assumption of constant returns to scale.

3.2.2 Methodology for constructing an ecological efficiency measurement indicator system

The purpose of this paper is to construct an EE input–output evaluation indicator system for the MRFFLG LC in HP in four dimensions (Table 2). Firstly, the indicator system emphasizes the systematization and integrity of communities by using four dimensions of inputs, such as natural resources, social and economic inputs, and unanticipated and expected outputs. Secondly, natural resource elements involved in production and living in the research area have been included in the data selection, thereby enriching the indicator system for assessing the effectiveness of ecological conservation and restoration. From the perspective of a socio-ecological system, this study constructs an evaluation framework and focuses on the consumption of ecological resources during the transformation of ecological resources into economic resources and the challenges that are associated with regional sustainable development. As a function of output, the study considers how efficiently natural resources are utilized under conditions of economic growth.

A compelling argument is made by Kondo and Nakamura (2005) that the use of resources is the fundamental cause of environmental pressure and that policies designed to improve efficiency are often easier to adopt than policies that restrict economic activity levels. Although there are many input and output indicators for EE, few of these reflect the close relationship between economic, social, and natural ecological systems. It is difficult to directly apply the results of EE estimation to ecological system protection or restoration practices since indicator selection tends to focus more on the inputs and outputs of the social and economic system. GDP and CO_2 emissions are commonly used as output indicators (Grand, 2016). The scale of material inputs is increasingly becoming more important when it comes to issues related to waste generation and emissions (Behrens et al., 2007). As a result, unexpected CO_2 emissions have become an important indicator that cannot be ignored. Research on EE evaluation has shown that input indicators are diverse, with a greater emphasis on capital stock, labor quantity, and energy consumption being used as key input indicators for determining environmental efficiency (Kondo and Nakamura, 2005; Behrens et al., 2007; Yu et al., 2013). EE studies cannot be directly applied to ecological system governance if there is a lack of understanding of the local ecosystem and its close relationship with the natural ecological system (Hukkinen, 2001).

Several major resource types are considered in this article using a balanced multiple input approach, which includes natural elements such as MRFFLG and considers the importance of land demand in terms of ecological pressure for several major resource types. Combined with existing research on social and economic input indicators, CO_2 emissions are added as an unwelcome output indicator that identifies the quality of the environment. This indicator system is intended to represent the direction of regional EE development while incorporating the system thinking of the MRFFLG LC and also to respond to the spillover effects of differential policies on space through interregional correlation. Stock indicators are used to highlight the overall input of the MRFFLG LC resources for the purpose of designing indicators. Natural resource input stock indicators in HP include the supply of several major resource types such as MRFFLG while social and economic input indicators include labor input and public fiscal expenditure. In summary, this article provides a comprehensive system of inputs and outputs for evaluating EE.

TABLE 2 The ecological efficiency evaluation index system of the life community of mountain-river-forest-farmland-lake-grassland (MRFFLG) systems.

Dimension of indicator selection	Tier 1 indicators	Secondary indicators	Variable description
Fixed asset investment mountain-river-forest-farmland-lake-grassland (MRFFLG) life community natural resources input variables	Resource input indicators	Forest resources	Forestland area/m^2
		Farmland resources	Farmland area/m^2
		Grassland resources	Grassland area/m^2
		Water resources	River area /m^2
Socio-economic input variables for the life community of MRFFLG	Social input indicators	Workforce	Number of people employed in urban units at the end of the year/person
	Economic input indicators	Capital	Public finance expenditure /RMB million
Output variables of the community of life of MRFFLG	Non-desired outputs	Environmental pollution	Carbon dioxide emissions/million tonnes
	Desired output	Economic benefits	GNP *per capita*/yuan

3.2.3 Method for spatial evaluation of ecological efficiency

In this study, the standard deviation ellipse method and the convergence analysis method are used to analyze and evaluate the spatial variation of the EE centroid. They are also used to determine whether there is any spatial convergence of EE based on the MRFFLG LC EE measurements.

3.2.3.1 Evaluation method for the spatial migration model of ecological efficiency centroid

This study utilizes the standard deviation ellipse method to determine how the EE centroid of the MRFFLG LC has migrated and distributed over time in HP between 2006 and 2019. Its advantages over previous research methods are that it overcomes the disadvantages that resulted from the division of administrative regions in previous research methods. An integral part of spatial statistics, it quantitatively describes global and spatial properties, including centrality, dispersion, directionality, spatial morphology, and timely spatial evolution processes within the distribution of the research object (Zhang et al., 2022). Among its advantages is its ability to compare spatial patterns over time within the same dimension and to visualize research indicators for each year in a manner that accurately and intuitively illustrates the main regions and relative trends in the spatial distribution of geographical elements over time.

We calculated the main parameters of the standard deviation ellipse of the EE of the MRFFLG LC in HP from 2006 to 2019 using the ArcGIS measure geographic distribution module to better understand its spatial evolution characteristics and accurately describe its spatial distribution trend. The EE is shown as an ellipse with the EE as the weight, and the general characteristics of the spatial distribution can be observed through changes in the parameters such as centroid, azimuth, major axis, and minor axis (Equation 2). Here is the formula for calculating each of the above parameters:

$$\text{Center of Gravity}:\left(\bar{X},\bar{Y}\right)=\left(\frac{\sum_{i=1}^{n}w_i x_i}{\sum_{i=1}^{n}w_i},\frac{\sum_{i=1}^{n}w_i y_i}{\sum_{i=1}^{n}w_i}\right) \quad (2)$$

Azimuth:

$$\tan\theta=\frac{\left\{\left(\sum_{i=1}^{n}w_i^2\tilde{x}_i^2-\sum_{i=1}^{n}w_i^2\tilde{y}_i^2\right)+\sqrt{\left(\sum_{i=1}^{n}w_i^2\tilde{x}_i^2-\sum_{i=1}^{n}w_i^2\tilde{x}_i^2\tilde{y}_i^2\right)^2-4\sum_{i=1}^{n}w_i^2\tilde{x}_i^2\tilde{y}_i^2}\right\}}{2\sum_{i=1}^{n}w_i^2\tilde{x}_i\tilde{y}_i}$$

$$\text{X-axis standard deviation:}\sigma_x=\sqrt{\frac{\sum_{i=1}^{n}\left(w_i\tilde{x}_i\cos\theta-w_i\tilde{y}_i\sin\theta\right)^2}{\sum_{i=1}^{n}w_i^2}}$$

$$\text{Y-axis standard deviation:}\sigma_y=\sqrt{\frac{\sum_{i=1}^{n}\left(w_i\tilde{x}_i\sin\theta-w_i\tilde{y}_i\cos\theta\right)^2}{\sum_{i=1}^{n}w_i^2}}$$

$\bar{x}$ and $\bar{y}$ are the average centroid coordinates of the elements; w_i is the weight of the study unit; θ is the orientation angle of the ellipse; x_i and y_i are the central coordinates of the elements in each study area; $\tilde{x}_i$ and $\tilde{y}_i$ are the coordinate deviations of the centroid from the central coordinates in each study unit; σ_x and σ_y are the standard deviations along the X and Y axes, respectively; and n is the total number of elements.

3.2.3.2 Evaluation methods for eco-efficient spatial convergence analysis models

This article employs the β-convergence and β-absolute convergence methods to examine EE convergence in a LC. Spatial convergence models are widely used to examine whether *per capita* GDP of countries or regions converges or diverges. This approach can also be used to test for EE convergence and divergence across different regions during the sample period. The application of the β-convergence concept is more widely adopted and has stronger policy implications in empirical analysis. Neoclassical growth models assume that all economies have the same technology, but their initial factor endowments differ (Yu et al., 2013). Green spaces have been found to enhance human health and well-being (Wheeler et al., 2015). The control variables in this study included the area of park green spaces *per capita*, the area of urban roads *per capita*, the compliance rate with pollution discharges, and the fixed asset investment as variables that impact EE.

β convergence refers to the phenomenon where the rate of EE growth in regions with low EE levels is higher than that in regions with high EE levels. Absolute β convergence requires strict assumptions, such as uniform environmental conditions across evaluation units, including the same fixed asset input and *per capita* green area. Over time, different regions' EE will converge to the same level. The testing model for absolute beta convergence is as follows (Equation 3):

$$\ln\left(\frac{x_{it}}{x_{io}}\right)=T\alpha+T\beta\ln\left(x_{io}\right)+T\varepsilon \quad (3)$$

T represents the research period, x_{io} represents the EE of the ecosystem of MRFFLG in the $^{\text{i-th}}$ area at the beginning and end of the research, α represents the constant term, β represents the regression coefficient, and ε represents the random disturbance term. When β <0 and is significant, it indicates that the area with lower EE of the ecosystem of MRFFLG has a larger EE growth rate than the area with higher EE.

If the condition of β-convergence holds, the conditional assumption is eliminated, and taking into account the differences between cities in population, income levels, fixed asset investment, *per capita* green area, and other aspects, the growth rate of the EE of the LC will converge to their respective steady-state levels over time through different growth paths. The testing model for β-convergence is as follows (Equation 4):

$$\ln\left(\frac{x_{it}}{x_{io}}\right)=T\alpha+T\beta\ln\left(x_{it}\right)+T\sum_{j=1}^{J}\gamma_j x_{it}+T\varepsilon \quad (4)$$

T represents the research period, x_{io} represents the EE of the MRFFLG LC in the $^{\text{i-th}}$ region at the beginning and end of the study, α represents the constant term, β represents the regression coefficient, ε represents the random disturbance term, and γ_j represents the regression coefficient of the added $^{\text{j-th}}$ control variable. When β<0 and is significant, it indicates that the EE of the MRFFLG LC in the study area will converge to their respective stable levels based on their own conditional characteristics, and the difference in EE between different regions will persist in the long term.

Finally, this paper selects the Spatial Durbin Model to test β absolute convergence and β conditional convergence. Considering the spatial correlation of the study area, a spatial econometric model for the convergence of EE growth in the MRFFLG LC is constructed. The empirical test and analysis of the EE of the MRFFLG LC in HP are carried out, and the model is presented in Equation (5).

$$\ln\frac{AEE_{i,t+1}}{AEE_{i,t}} = \alpha + \rho\sum_{j=1}^{n} w_{ij}\ln\frac{AEE_{i,t+1}}{AEE_{i,t}} + \beta ln\left(AEE_{i,t}\right) + \sum_{k=1}^{n}\theta_k \ln X_{k,i,t} + \sum_{j=1,k=1}^{n} w_{ij}\phi_k \ln X_{k,i,t} + \varepsilon_{i,t} \tag{5}$$

In the formula, $\ln\left(AEE_{i,t+1} / AEE_{i,t}\right)$ represents the natural logarithm of the EE growth rate of the MRFFLG LC in the study area i from t to $t+1$. α and ρ are the parameters to be estimated, $\beta < 0$ represents absolute convergence. w_{ij} is an element in the spatial weight matrix, while ρ and λ are spatial correlation coefficients, $\rho$and λ reflects the influence of adjacent areas on the attribute values of the local area. ϕ_i represents the regression coefficient of the spatial interaction effect, and $\varepsilon_{i,t}$ is the random error term. Based on previous research, this paper selects fixed asset investment, the rate of wastewater discharge compliance, *per capita* urban road area, and *per capita* park green area as control variables to determine absolute convergence and conditional convergence. If the coefficient θ_k of each control variable is 0, the above model is absolutely convergent, and if θ_k is not 0, it is conditionally convergent. The paper examines the level of environmental regulation, the intensity of government macro-control, the level of urban development, and the degree of urban greening.

4 Results

4.1 Results of EE level measurement under the time dimension

Based on the existing social and natural resource inputs, the average EE of the MRFFLG from 2006 to 2019 was only 0.34, suggesting there is still room for improvement. The input–output balance has reached the optimal level when EE is 1, ensuring an effective balance between environmental protection and resource conservation. When the EE is close to 0, it indicates a lack of coordination between economic development and ecological protection because of the imbalance between input and output. HP has a relatively low level of EE. Figure 3 shows the average EE for the MRFFLG in HP during the period 2006 to 2019.

The overall EE level of Heilongjiang Province in 2006–2019 showed an M-shaped fluctuating change, and despite the large level of fluctuation, on the whole, the EE level in recent years has still shown an increase. The level of EE fluctuates substantially under the existing social and natural resource inputs, and 2009, 2011, and 2014 are the 3 years that have seen the greatest growth in EE. Throughout the entire study period, the main years with large changes were 2009, 2011, and 2014, in which a complete cessation of commercial logging of natural forests was implemented in the key state-owned forest areas of HP in 2014, and a pilot project for ecological protection and restoration of MRFFLG was launched in HP after 2018, The EE level in 2019 was about 0.07 higher than that in 2016, and the fluctuation of EE has gradually become smaller since 2016, with the overall level of change showing a steady increase. While around 2011 was the period of implementation of the second phase of the natural forest protection project. It can be seen that with the gradual improvement of China's ecological protection and restoration actions, the quality of the regional ecological environment has subsequently been improved.

4.2 Evaluation results of EE levels under the spatial dimension

The results show that the highest level of EE is located in the key state-owned forest areas, indicating that the level of EE is relatively higher in forest resource-rich areas. Table 3 shows the characteristics of the spatial distribution of the EE. It is clear from the center of gravity coordinates that the movement direction of the geographic center of gravity of HP's biotic community in each year indicates the "high density" areas of HP, which vary between 128.75°E to 130.05°E and 46.10°N to 47.08°N, which indicate that HP is a key state-owned forest area. During the study period, there were mainly two stages of significant movement directions among the EE in HP. In particular, the trend was to the northwest during 2006–2008, 2009–2011, and 2015–2018, while it was to the southeast during 2011–2012, 2014–2015, and 2018–2019. Between 2010 and 2013, the EE in HP shifted to the northwest direction from the perspective of the development direction of EE, indicating a greater degree of improvement in EE in the northwest region of HP than in the southeast region during this period.

The MRFFLG of HP during the study period is not stable enough. The results show that the eastern and southern regions of HP have higher levels of EE. The years with the largest movement of the EE in terms of distance traveled are 2010, 2011, 2014, and 2015 (Figure 4). There was a noticeable shift in the EE between 2010 and 2011, with a transfer direction toward the northwest and a significant offset. In contrast, during the period of 2014–2015, there was a greater change in factors and a greater distance of center of gravity movement. In terms of distance and speed of movement, the EE shifted toward the southeast. In both the north–south and east–west directions, the center of gravity movement distance increased significantly, and the straight-line movement distance also increased significantly. Thus, the EE level in the eastern and southern regions of HP improved more significantly during this period than in the western and northern regions.

4.3 Results of standard deviation ellipse analysis of EE

The results from the standard deviation ellipse again verify that the spatial distribution pattern of EE levels is not stable enough. EE standard deviation ellipses in HP were mainly located in the southeastern part of the central region from 2006 to 2019 (Figure 5). This larger flattening ratio indicates that the development level of EE

FIGURE 3
Trends in the mean ecological efficiency in HP for the years 2006–2019.

TABLE 3 Direction and distance of the shifting gravity center of ecological efficiency of (MRFFLG) systems in Heilongjiang Province (HP), China.

Year	Barycentric coordinates	Angle of ellipse direction	The perimeter of an ellipse	Area of an ellipse	The coordinates of the midpoint x	The coordinates of the midpoint y	The length of the x-axis	The length of the y-axis	Flat rate
2006	(129.66°E,46.45°N)	95.68	16.89	16.58	129.66	46.45	3.69	1.43	0.61
2007	(129.65°E,46.50°N)	95.35	17.00	16.46	129.65	46.50	3.74	1.40	0.63
2008	(129.29°E,46.65°N)	93.79	18.41	19.31	129.29	46.65	4.05	1.52	0.63
2009	(129.61°E,46.10°N)	96.78	16.54	17.93	129.61	46.10	3.45	1.65	0.52
2010	(129.38°E,46.50°N)	93.74	18.18	17.78	129.38	46.50	4.06	1.39	0.66
2011	(128.91°E,47.08°N)	99.44	18.16	21.00	128.91	47.08	3.84	1.74	0.55
2012	(129.32°E,46.82°N)	97.16	19.26	21.95	129.32	46.82	4.19	1.67	0.60
2013	(128.75°E,46.88°N)	96.34	19.73	23.77	128.75	46.88	4.24	1.78	0.58
2014	(128.90°E,46.92°N)	99.93	19.29	23.65	128.90	46.92	4.08	1.84	0.55
2015	(130.05°E,46.47°N)	99.09	17.88	18.01	130.05	46.47	3.95	1.45	0.63
2016	(129.57°E,46.69°N)	97.76	17.79	19.16	129.57	46.69	3.84	1.59	0.59
2017	(129.67°E,46.67°N)	97.98	17.22	18.14	129.67	46.67	3.70	1.56	0.58
2018	(129.51°E,46.60°N)	96.18	17.68	18.41	129.51	46.60	3.85	1.52	0.60
2019	(129.52°E,46.60°N)	96.72	17.56	18.24	129.52	46.60	3.82	1.52	0.60

is unevenly distributed throughout the region. It was 2010 that had the most directional distribution of the ellipse, while other years did not exhibit much change in the flattening ratio. From 2006 to 2019, the area of the standard deviation ellipse generally increased first and then decreased. Over the years 2006 to 2014, the circumference and area of the standard deviation ellipse gradually increased and then gradually decreased. Additionally, the variation in the long and short semiaxes was also increasing in the beginning and then decreasing in the end, indicating that spatial agglomeration characteristics have gradually become more evident in the east–west direction since 2014, while the degree of spatial agglomeration has been gradually increasing in the north–south direction. In 2013 and 2014, the ellipse coverage area was significantly larger than in previous years, while in 2006 and 2007, the ellipse coverage area was the smallest. EE ranges in 2018 and 2019 were higher than those in 2006 and 2007. The rotation of the azimuth angle showed a large variation. In 2011 and 2014, the azimuth angle increased by 5.70° and 3.59°, respectively. This indicates that the spatial distribution pattern of EE is deviating in a noticeable manner. The spatial distribution pattern of EE in HP shifted from southeast to northwest by 5.70° in 2011, and from northwest to southeast by 3.59° in 2014 and 2015.

In the MRFFLG in southern cities of HP, the average EE of the LC is higher than that in northern cities based on the spatial distribution pattern and movement path of the EE center of the LC during the research period. In general, the spatial distribution of EE in the MRFFLG of HP exhibits a northwest-to-southeast pattern and a

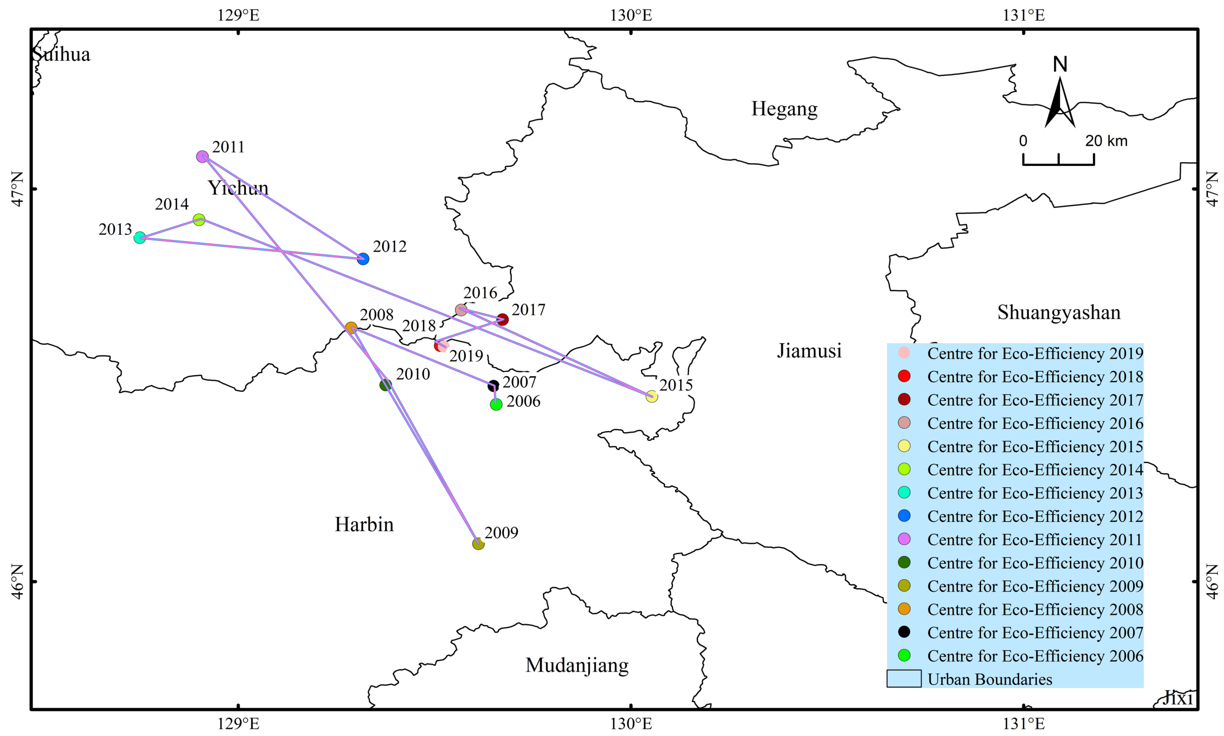

FIGURE 4
Migration trajectory of the ecological efficiency gravity center of (MRFFLG) systems from 2006 to 2019.

tendency to shift toward the southeast. Initially, the length of the X semi-axis decreases from the length of the major axis of the standard deviation ellipse, then increases, decreases again, and increases again, indicating that the EE of the LC in the MRFFLG of HP continues to vary. It appears that dispersion first occurs, followed by agglomeration, then dispersion again, followed by agglomeration. It can be seen that the length of the Y semi-axis increases and decreases over time, suggesting that the spatial distribution pattern of the EE center of the LC in the MRFFLG of HP during the study period is not stable enough, which indicates that the spatial spillover effect has not yet become apparent.

4.4 Results of spatial convergence analysis

The existence of spatial convergence pattern of EE of MRFFLG in HP was verified by convergence analysis method. First, the convergence analysis indicates that EE has reached absolute convergence in the MRFFLG research area of HP, which means that the levels of EE in the various research areas will eventually reach a steady state level. The second effect is that constraints on fixed asset investments and an increase in investments in environmental optimization have the potential to enable regions with lower EE to achieve relatively higher growth rates. As a result, the EE gap between regions is likely to gradually decrease in the future.

As shown in Table 4, all convergence models for AEE_{it} have significant coefficients of negative convergence, and most of them pass the significance level test at 1%. The conditional β-convergence results in Table 4 indicate that overall, EE in the MRFFLG region of HP exhibits significant conditional convergence under the joint influence of environmental regulation, government macro-control, urban development level, and resource endowment level. In other words, EE will show a trend of annual decline with an increase in environmental regulations, an improvement in government macro-controls, an acceleration in urbanization, and changes in the endowment of resources *per capita*. It is worth mentioning that the amount of investment in fixed assets passed the negative significance test, indicating that the amount of investment in fixed assets has a negative impact on the improvement of EE. Among the items of fixed asset investment, the top ones according to the proportion from high to low are manufacturing industry, real estate industry, transportation, storage and postal industry and so on. It can be seen that some inputs that need to rely on consuming a lot of resources to promote industrial development may have a negative impact on EE. On the other hand, the increase of parks and green areas *per capita* has a promoting effect on the regional EE, indicating that the optimization of the ecological environment and the increase of public green areas can help to improve the Eco-output.

5 Discussion

5.1 The EE levels from a socio-economic perspective

The problem of high evaluation indexes is corrected by the super-efficiency SBM model, and the results show that the overall EE level of HP is low, and Analyzing the overall situation, the average value of the ecological efficiency of MRFFLG in the 11 cities of HP during the period of 2006–2019 is 0.34, which means that there is still a lot of

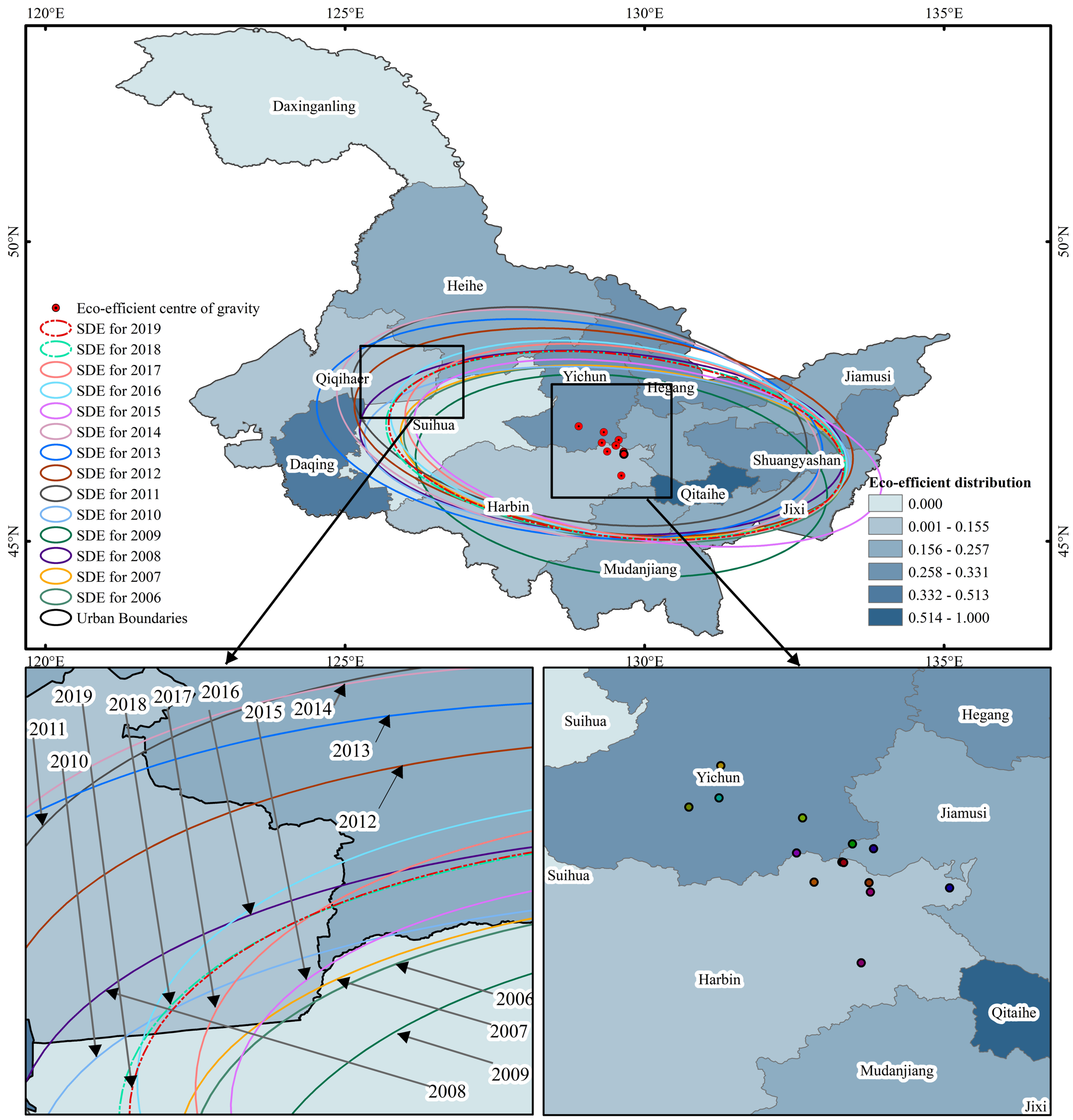

FIGURE 5
Frame diagram of the spatial–temporal evolution of ecological efficiency in HP, China.

room for improvement of the ecological efficiency with the existing inputs of social and natural resources, which indicates that the imbalance of inputs and outputs has led to the lack of a better coordinated development between economic development and ecological protection, and the EE of many cities in different periods still has obvious differences and shows certain spatial heterogeneity.

From the time distribution of the mean value of ecological efficiency of MRFFLG in HP, the overall fluctuating trend shows a large fluctuation trend, and the years in which changes in ecological efficiency have been substantially improved are 2009 (0.351), 2011 (0.532) and 2014 (0.483). A circular economy perspective reveals that the EE of an ecosystem reflects a separation between economic growth and environmental pressure (De Pascale et al., 2021; Liu et al., 2022). HP environmental pressures are still relatively high and cannot fully accommodate economic growth. Moreover, it falls within the area of ineffective EE in the ecosystem. The highest value recorded in previous years was only 0.523, indicating that ecological resources are not adequately converted into tangible advantages. Sustainable development in the region is threatened by ecological resource loss. Despite significant improvements in EE levels over the past 2 years (2011 and 2014), significant fluctuations have been observed in the overall trend. The reason is that China's constraints on resource protection have been alleviated by some forestry subsidy funds. This has temporarily improved the HP ecosystem's ecological efficiency. With time, policy investment and implementation will inevitably reach a point of fatigue, and EE will decrease until a new round of

TABLE 4 Spatial correlation test of ecological efficiency (MRFFLG) systems in HP, China.

Test parameters	Spatial absolute β convergence	Spatial condition β convergence
$lnAEE_{i,t}$	−0.372*** (0.068)	−0.517*** (0.073)
Fixed asset investment		−0.041*** (0.015)
Wastewater discharge compliance rate		−0.077 (0.051)
Urban road area *per capita*		0.137 (0.085)
Green space *per capita*		0.130** (0.056)
$W^*lnAEE_{i,t}$	−0.347*** (0.107)	−0.201 (0.149)
W*Fixed asset investment		0.028 (0.043)
W*Wastewater discharge compliance rate		−0.180 (0.117)
W*Urban road area *per capita*		−0.002 (0.125)
W*Green space *per capita*		0.075 (0.101)
ρ	−0.307*** (0.105)	−0.291*** (0.109)
R^2	0.152	0.330
N	143	143

*, **, and *** indicate the significance differences at the 10, 5, and 1% levels, respectively; values in brackets are standard errors.

financial investment or strong policy constraints will once again lead to a significant increase in HP's ecosystem's EE in the short term. Clearly, this illustrates the dependence on government regulation and fund investment, which are unpredictable.

5.2 Spatial variation in the center of gravity of EE

From the coordinates of the geographic center of gravity of the EE on the map, the movement of the geographic center of gravity in each year points to the "high-density" part of the ecological efficiency, which varies between 128.75°E ~ 130.05°E, 46.10°N ~ 47.08°N. Although the center of gravity shows an irregular trajectory, its main moving range is located in the key state-owned forest area of HP. In terms of moving distance, the years with larger moving amplitude were in 2010, 2011, 2014, and 2015, which corresponded to the end of the first phase of the Practice and Prospect for Natural Forest Protection Projects, the start of the second phase of the natural forest protection project, and the implementation of the policy of stopping commercial logging of natural forests in the key state-owned forest areas of HP, respectively. The center of gravity moved the largest distance and the fastest speed from 2014 to 2015, and the magnitude of elemental changes was more obvious, indicating that the full cessation of commercial logging of natural forests in key state-owned forest areas of HP produced a relatively large spatial difference in the EE of the community of life of MRFFLG.

Over time, EE levels and ER investment funds are related. EE levels in the ecosystem show similar trends due to national investments in ecological protection and restoration funds. China is expected to increase its investment in ecological protection and restoration funds for forestry. This will directly affect the MRFFLG ecosystem. It is anticipated that this will result in an increase in EE levels. Consequently, the ecosystem of MRFFLG will be improved, making it more sustainable. The above results indicate that the government's centralized investment in ecological protection and restoration has achieved positive results. In addition, the implementation of ecological protection and restoration actions concentrated in localized areas has led to large spatial differences in the overall level of EE in the study sample areas. Although the level of EE in HP has continued to improve, cities with lower levels of EE face challenges such as high investment in ecological resources, insufficient expected outputs, and excessive unintended outputs. The spatial distribution pattern of regional EE centers in HP shows a northwest-southeast bias. The spatial spillover effect has not yet appeared.

5.3 Characteristics of spatial convergence of EE

The MRFFLGs LCs in HP will eventually reach a steady-state level of EE, and regions with lower EE will experience relatively faster growth rates. It has been confirmed that the EE of the MRFFLGs LC in HP has passed the convergence test. Even though there are significant regional differences in the EE level of the LC, fixed asset investment has passed the negative significance test. This indicates that fixed asset investment negatively affects EE. Increasing *per capita* park and green space area through fixed asset investments promotes regional-EE. It is expected that the EE gap between regions will gradually narrow in the future.

Nevertheless, if fixed asset investments are increased in industries with high resource and pollution consumption, regional EE may be reduced although the overall level of EE in MRFFLGs LC in HP is low, places with significant levels of EE can provide experiences and references for places with lower levels of EE, including better government regulations and pollution limits.

According to the theory of technology transfer and diffusion, the greater the technological gap between different regions, the greater their technological reserves, and the diffusion of new products or technologies as well as cooperative innovations will lead to the rapid diffusion of technological advances and related experiences among different regions. This kind of experience imitation is less costly than spontaneous innovation, and the possibility of successful realization is higher. Therefore, regions with backward level of eco-efficiency will accumulate more experience imitation of eco-efficiency progress, and thus show faster improvement in practice, and can rapidly improve the eco-efficiency of their own ecological community of life in the MRFFLG by imitating the policies or measures of the neighboring regions at a low cost. Moreover, the ability of these regions to emulate and learn from each other is likely to grow over time, reducing the cost of borrowing more complex technologies, and may itself create "social capacity" in the development process, thus maintaining the level of eco-efficiency up to a certain level of sustainability and stability.

5.4 Research outlook

In the existing research on ecological restoration, the number of social science-based research fields is far lower than that of Science Technology, Physical Sciences and Life Sciences Bio-medicine research heat. As restoration ecologists increasingly respond to changes in large-scale ecosystems, a broader range of stakeholders will be included in the ecological restoration process, advancing the need for interdisciplinary communication. In the process of integrated ecosystem management there are bound to be conflicting contradictions between development and utilization and resource protection, and determining ecological protection and restoration management goals is a process that requires trade-offs. Assessing and managing new ecosystems from a more holistic perspective and considering socioeconomic factors that may have an impact on them will be worthwhile research in the future. Integrated ecosystem management inevitably involves conflicts between resource development and utilization and ecological protection and restoration. Managing both economic development and ecological protection is a multi-objective decision-making process, and different land use types have different ecosystem service functions. Currently, most research is focused on the relationship between changes in resource utilization types and ecosystem service functions, with little attention paid to the impact of alternative land use combinations. Research can be conducted in the future to investigate the relationship between the proportion of different resource utilization types and the improvement of ecosystem services.

6 Conclusion

In this study, the level of EE in HP was measured and evaluated. It presents a theoretical framework for evaluating ER based on multi-objective production and multi-factor inputs. EE indicators for ecological and socioeconomic aspects are included in the framework for assessing life communities' EE. To provide a reference for constructing an evaluation index system of ecological restoration efficiency that also includes socioeconomic dimensions. The research findings that: (1) The overall EE level is generally low, with the highest value of 0.523 in all the years, and most of the time Heilongjiang Province still belongs to the EE non-effective region, but in recent years it has begun to show a steady upward trend. (2) The spatial distribution of the EE in HP demonstrates a northwest-southeast pattern. The pattern was not stable enough during the research period, and spatial spillover effects were not significant. (3) The implementation and enforcement of policies related to ecological protection and restoration in China have significantly promoted the improvement of EE. The migration trajectory of the center of gravity of ecological efficiency shows that its main moving range is located in the key state-owned forest area of HP, and during the implementation of the natural forest protection project. (4) There are obvious differences in the EE levels of different types of cities, with resource-mature cities generally higher. (5) There is spatial convergence of EE levels in HP. In the future, regions with lower EE levels will have relatively higher growth rates, so that the EE gap between regions will gradually narrow in the future.

Research results have important implications for regional ecological protection and restoration, enrich theoretical frameworks and feasible paths for assessing the operational efficiency of the MRFFLG system both theoretically and practically, and serve as valuable references for the practice of protecting, restoring, and governing ecological systems. Most of the current research focuses on the impact of resource utilization type shift on the relationship between ecosystem service functions, while there are fewer studies on the impact of the combination of different land categories, and in the future, we can consider increasing the empirical research on the relationship between the enhancement of ecosystem service functions under the study of the weight of different resource utilization types. On the other hand, due to the limitations of data availability and the study area and period, the effects of some factors are not yet significant; with the deepening of the ecological protection and restoration process, a longer-term study can be carried out through continuous follow-up surveys.

Data availability statement

Publicly available datasets were analyzed in this study. This data can be found here: In order to obtain the HP 2019 elevation data (Digital Elevation Model, DEM), the Geospatial Data Cloud website (https://www.gscloud.cn/) sourced the SRTMDEM 90M original elevation data; ESACCI (European Space Agency Climate Change Institute) provided the land use and land cover data from 2006 to 2019).

Author contributions

YR: Data curation, Funding acquisition, Investigation, Methodology, Software, Visualization, Writing – original draft, Formal analysis, Resources, Validation, Writing – review & editing. MA: Formal analysis, Methodology, Supervision, Validation, Writing – review & editing. YC: Conceptualization, Formal analysis, Funding acquisition, Methodology, Project administration, Supervision, Validation, Writing – review & editing. SZ: Data curation, Methodology, Software, Supervision, Validation, Writing – original draft, Writing – review & editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Host–parasite interaction: an insight into the growth and physiological responses of sandalwood and associated host species

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Introduction: Sandalwood (*Santalum album* L.) is categorized as vulnerable in the IUCN Red list and is also an industrially important tree species valued for its heartwood and aromatic oil. Sandalwood is a semi-root parasite tree that relies on its host plants for its water and nutrient requirements. Therefore, there is need to understand the growth and physiological interactions between sandalwood and its hosts.

Methods: Sandalwood were planted with ten different host species viz., Syzygium cumini, Punica granatum, Phyllanthus emblica, Melia dubia, Leucaena leucocephala, Dalbergia sissoo, Casuarina equisetifolia, Citrus aurantium, Azadirachta indica and Acacia ampliceps to assess the interactive effect on the change in growth and physiology of both sandalwood and host tree species.

Results: The findings revealed that sandalwood grown with hosts *D. sissoo* and *C. equisetifolia* showed higher growth performance, while among hosts, *S. cumini*, followed by *C. aurantium* and *L. leucocephala*, showed better growth and physiobiochemical traits. The stepwise regression analysis and trait modeling indicated that the six traits, namely, plant height, photosynthetic rate, relative water content, water potential, intercellular CO_2 concentration, and total soluble protein, contributed greater growth in the sandalwood, while four traits, namely, water potential, osmotic potential, leaf area, and total soluble protein, contributed greater growth in the host species. The traits modeling study predicted greater growth of sandalwood with the hosts *D. sissoo* and *C. equisetifolia*, whereas among host species, prediction revealed greater growth of *S. cumini* and *C. aurantium*.

Discussion: The study concluded that host–parasite interaction modulated the growth and physiological processes in both sandalwood and hosts and sandalwood plantations can be successfully developed with the hosts *D. sissoo* and *C. equisetifolia*.

KEYWORDS

sandalwood, host species, plant-water relation, physiological processes, osmoprotectants

1 Introduction

Sandalwood (*Santalum album* L.) is the world's second most expensive tree that holds immense cultural, religious, and economic importance across the Asian sub-continent. Globally, a total of 18 species of sandalwood have been documented that belong to the genus *Santalum,* of which Indian sandalwood (*Santalum album* L.) is the commercially most valuable species, which is well known for aromatic oil derived from its heartwood. The demand for sandalwood products, including oil, is increasing annually in both the international and domestic markets (Kumar et al., 2012; Ramanan Suresh et al., 2020), and the future projection indicates an increase in the global demand for sandalwood to 6,000 and 7,000 metric tons/annum (Viswanath and Chakraborty, 2022). Moreover, at present, a significant disparity exists between the demand and supply of sandalwood, which creates tremendous pressure on harvesting of this species from natural stands. Simultaneously, illegal felling, smuggling, poor seedling establishment due to its parasitic nature, lack of knowledge on the host–parasite relationship, and other abiotic and biotic factors, etc., have greatly affected the natural plantation of sandalwood, resulting in species being classified as Vulnerable and included in IUCN Red list category (Arunkumar et al., 2019). Therefore, to fulfill the global demand for sandalwood products and to preserve its precious natural reserves, gradually more area needs to be brought under the sandalwood plantations.

In the past, sandalwood plantations were limited to natural forests; however, high demand-fostered greater price of sandalwood has led to species farming gaining huge popularity among the farming communities across Asia, Africa, and Australia, especially in India (Mishra et al., 2018). Since the last two decades, a huge expansion in area under sandalwood cultivation has been reported due to its high demand across the globe. Species can thrive in diverse climatic conditions and can be integrated into agroforestry as they can provide higher economic returns and conserve the natural environment (Mishra et al., 2018; Srikantaprasad et al., 2021; Kumar et al., 2022a,b; Verma et al., 2023a,b). However, the cultivation of semi-parasitic sandalwood is more challenging compared to monoculture plantations (Radomiljac et al., 1998). It obtains water, nutrients, and minerals from the host plant via haustorium (Kuijt, 1969). Haustorium is a complex set of physiological and structural linkages that connects the conductive system of phloem, xylem, or both, assisting parasitic plants by supplying water and minerals from the host (Yoshida et al., 2016). Moreover, the growth performance of sandalwood is influenced by the physiological activity of its hosts. Therefore, it is crucial to have a comprehensive understanding of parasitism ecology; particularly, the understanding between parasite and host is paramount to identify the suitable host, which is the most important strategy for the successful establishment of sandalwood plantations (Surendran et al., 1998).

Sandalwood has been found parasitizing on a wide range of plants (300 species) ranging from grasses to trees (Nagaveni and Vijayalakshmi, 2003). Among the host species, *Azadirachta indica* (Nagaveni and Vijayalakshmi, 2003), *Dalbergia latifolia* and *Syzygium cumini* (Guleria, 2013), *Acacia nilotica* and *Melia dubia* (Padmanabha et al., 1988), *Leucaena leucocephala* (Guleria, 2013), *Casuarina equisetifolia* (Nagaveni and Vijayalakshmi, 2003; Rocha et al., 2014), and *Acacia hemignosta*, *A. ampliceps*, and *Melia azedarach* (Radomiljac et al., 1998) were observed to be the most suitable host tree species in terms of maintaining the growth as well as biomass accumulation of sandalwood, but there is a dearth of knowledge regarding the physiological responses of sandalwood to diverse host species. Simultaneously, during the parasitism process, several growth and physiological alterations occur in the host species (Rocha et al., 2014), and at present, information about such is completely lacking. Moreover, parasite sandalwood induces several growth and physiological alterations in host species, which is one of the main hindrances in the successful establishment of sandalwood plantations; however, no systematic information on such interaction is available. Interaction inducing a favorable change in both sandalwood and host could be helpful in devising cultivation and management practices as well as could provide insight into the potential host trees that could support sandalwood throughout its lifespan (Lion, 2017; Rocha and Santhoshkumar, 2022).

Most of the previous studies were limited to screening of host species and assessing the influence of diverse tree host species on the growth and morphology of sandalwood. However, rarely any investigation is available that has reported the physiological response of both sandalwood and host species during their interaction process. We hypothesized that the parasite network process induces several changes in the growth and physiological response of hosts, thereby potentially influencing the growth and physiological processes of the sandalwood. The specific objectives of the study were (i) to systematically evaluate the response of growth and physiological processes involved in the complex interactions between host species and sandalwood and (ii) to identify potential traits that can greatly affect the performance of both sandalwood and host. Overall, this investigation aimed to explore host-specific compatibility by analysing alterations in growth, biochemical, and physiological traits of both host plants and sandalwood, which will aid in devising the best possible cultivation and management practices for the sandalwood.

2 Materials and methods

2.1 Experimental location and trial establishment

The present experiment was carried out at the ICAR-CSSRI in Karnal, Haryana, India (29° 42` 30`` N and 76° 57` 12`` E, with an elevation of 282 m above mean sea level). The study area exhibits a semi-arid, subtropical, monsoonal climate marked by significant temperature variations that range between 32.7°C and 42.8°C during summer and 3.4°C and 10.8°C during winter. The region receives annual rainfall ranging between 700 and 800 mm. The seeds of sandalwood were obtained from the Institute of Wood Science and Technology (IWST), Bangalore, and sown in germination beds during May 2020. The germinating seedlings were pricked from germination beds and transferred to polybags (6″ × 3″) containing soil: FYM: sand in the ratio of 1:1:1. The planting material of host species was brought from the local private nurseries. The healthy plants of both sandalwood and host species were considered for the present investigation. Furthermore, 6-month-old seedlings of sandalwood were planted with ten host species, i.e., *Syzygium cumini, Punica granatum, Phyllanthus emblica, Melia dubia*, *Leucaena leucocephala*, *Dalbergia sissoo*, *Casuarina equisetifolia*, *Citrus aurantium*,

TABLE 1 Information about the host species considered for the present study.

Host	English name	Family	References
Acacia ampliceps	Salt wattle	Fabaceae	Radomiljac et al. (1998), Guleria (2013), Padmanabha et al. (1988), and Brand et al. (2000)
Azadirachta indica	Neem	Meliaceae	Nagaveni and Vijayalakshmi (2003) and Parthasarathi et al. (1974)
Citrus aurantium L.	Bitter orange	Rutaceae	Singh et al. (2018)
Casuarina equisetifolia	Whistling pine	Casuarinaceae	Nagaveni and Vijayalakshmi (2003), Padmanabha et al. (1988), Rocha et al. (2014, 2017), and Singh et al. (2018)
Dalbergia sissoo	Shisham	Fabaceae	Guleria (2013), Padmanabha et al. (1988), and Ouyang et al. (2016)
Leucaena leucocephala	Suabul	Fabaceae	Guleria (2013)
Melia dubia	Malabar Neem	Meliaceae	Padmanabha et al. (1988)
Phyllanthus emblica	Indian gooseberry	Meliaceae	Nagaveni and Vijayalakshmi (2003)
Punica granatum	Pomegranate	Lythraceae	Singh et al. (2018)
Syzygium cumini	Jamun	Myrtaceae	Parthasarathi et al. (1974)

Azadirachta indica and *Acacia ampliceps*, and control (sandalwood alone) during October 2020. The selection of the host species in the study was evidenced from the previous empirical investigations, indicating their efficacy as suitable hosts for sandalwood (Table 1). After 2 months, the nursery host was completely removed to allow haustorial connections of sandalwood with the hosts except for the control. The experiment was conducted under pothouse conditions to exclude the effect of rainfall. Pothouse has optimum growing conditions with light intensity (PAR) of 600 flux, relative humidity >60%, temperature of 25–30°C, and CO_2 concentration of 400 ppm. Surprisingly, except for control (sandalwood alone), sandalwood grown with selected host plants had a 100% survival rate.

2.2 Experiment detail

In the present experiment, 50 plants each of sandalwood and 10 different host species were transplanted in 15-kg capacity plastic pots during October 2020. The growing media of plastic pots contained Soil: FYM: sand in the ratio of 6:3:1. A spacing of 10 cm was maintained between the host plant and sandalwood to allow the formation of haustorial connections. During the nursery stage, sandalwood necessitates a primary host, transitioning to a long-term secondary host under field conditions. Therefore, while transplanting, nursery host *Alternanthera* spp. was also planted with secondary hosts to ensure proper establishment of the sandalwood; 500 mL of irrigation water was provided to pots every day during October–November and February–March and alternatively during December–January. The irrigation requirement was calculated based on the soil volume and characteristics, and uniform irrigation was provided to all the plants during the entire experiment period. A standard dose of fertilizers (1 g of NPK per plant) along with a Hoagland solution for micronutrients was given to maintain the growth of plants (Bose et al., 2022). The weeding operations were carried out weekly. The suitability of the host for sandalwood was assessed by comparing the growth and physiological attributes of sandalwood as well as the host species. The data were recorded during March 2021, i.e., after 6 months of the transplantation.

2.3 Observations recorded

2.3.1 Plant growth

The plant height (cm) of both the sandalwood and host plants was recorded from the base to the apical shoot using a measurement scale. The collar diameter of both sandalwood and host plants was assessed using a vernier caliper. Leaf area measurements were conducted using a "Portable Laser leaf area meter—CI-202" (Verma et al., 2023a).

2.3.2 Plant–water relation

The plant–water relation parameters, namely, osmotic potential, water potential, and relative water content (RWC), were assessed in both sandalwood and host plants. For estimation of the RWC, fully expanded leaves derived from the middle portion of the plant were harvested during the morning timeframe between 9:00 am and 10:00 am and were then promptly conveyed to the laboratory in a sealed polythene bag. The leaves were cut into five-leaf disks of 1 cm diameter each, and fresh leaf weight (FLW) was recorded. After that, leaf disks underwent a 4-h immersion in distilled water to estimate the turgid leaf weight (TLW). After this hydration period, the leaf dry leaf weight (DLW) was recorded post-drying and RWC was executed using the formula (Turner, 1981):

$$\mathrm{RWC} = \frac{\mathrm{FLW} - \mathrm{DLW}}{\mathrm{TLW} - \mathrm{DLW}} \times 100$$

For the determination of water potential (ψ_w), the fresh leaves (1 g) were finely chopped and ψ_w was measured on WP4C Dewpoint Potentiometer (METER Group, Inc., United States) (Haghverdi et al., 2020) and expressed as –MPa. Furthermore, the osmotic potential (ψ_s) was ascertained utilizing the methodology outlined by Cuin et al. (2009), which measures osmolality (c) using the Vapor Pressure Osmometer (Model 5,600, ELITech Group, Belgium). The fresh leaves (1 g) were frozen at −20°C, crushed and squeezed to extract the sap. A 5 µL aliquot of the sap was taken to measure the osmolality on the osmometer and subsequently transformed to osmotic potential using the Van't Hoff equation (Hessini et al., 2019).

$$\psi s\,(\mathrm{MPa}) = -\mathrm{c} \times 2.58 \times 10^{-3}$$

2.3.3 Physiological processes

For the determination of chlorophyll content, a 200 mg of leaf sample was placed in a test tube containing 10 mL acetone and incubated overnight, and on the subsequent day, the optical density was measured at 645 and 663 nm utilizing UV spectrophotometer (double beam) (Yoshida et al., 1976) and expressed in milligrams per gram fresh weight (FW). The LI-6800 portable photosynthesis system with a standard 6 cm^2 leaf chamber (LI-COR, Inc., Lincoln, NE, United States) was used for the measurement of net rates of leaf photosynthesis, intercellular CO_2 concentration, transpiration, stomatal conductance, and vapor pressure deficit (VPD, KPa) during 09:00 am to 11:00 am on 2 consecutive sunny days. Cuvette conditions were regulated at an ambient CO_2 concentration of 400 ppm, relative humidity >60%, a photosynthetic photon flux density of 1,000 $\mu mol\, m^{-2}\, s^{-1}$, and leaf temperature of 25°C (Kumar et al., 2018, 2019). The uniform and fully expanded leaves of host plants and sandalwood were considered for measuring the gas exchange attributes.

2.3.4 Osmoprotectants

Various osmoprotectants, including total soluble sugars (TSS; Yemn and Willis, 1954), proline content (Bates et al., 1973), and protein content (Bradford, 1976), were measured. For estimation of TSS, a 100 mg sample was extracted in 2.5 mL of 80% ethanol, followed by centrifugation at 10,000 rpm for 10 min at 4°C. The resulting supernatant (10 µL) was mixed with anthrone reagent (5 mL), incubated at 100° C for 10 min and absorbance (using a UV spectrophotometer) was recorded at 620 nm. For proline determination, 200 mg of fresh leaves were homogenized in 3% sulphosalicylic acid (5 mL) and centrifuged (10,000 rpm for 10 min at 4°C). Subsequent supernatant mixed with glacial acetic acid and acid ninhydrin reagent (2 mL each), incubated (100°C for 60 min), and subjected to cooling after which toluene (4 mL) was added and vortexed and absorbance (using upper phase on UV spectrophotometer) recorded at 520 nm. Using a standard curve drawn with various L-proline concentrations, the proline content was determined. For protein estimation, the 100 µL supernatant was taken from TSS estimation and added to Bradford reagent (3 mL) taken in a test tube. The sample was then mixed thoroughly by Vortex and absorbance (using a spectrophotometer) was recorded at 595 nm (Bradford, 1976).

2.4 Statistical analysis

The present study was conducted in a randomized complete block design with five replicates to find out the best suitable host plant species for sandalwood. The normality of each variable was assessed through the Shapiro–Wilk (W) test using Q-Q plots of residuals. Variables found to deviate from normality underwent appropriate transformations. Subsequently, multiple comparison analysis was performed using Tukey's HSD test to discern the significant differences in various parameters of sandalwood and hosts at a 5% significance level. Crucial morpho-physiological traits were individually prioritized for both hosts as well as sandalwood, and predicted responses of growth diameter of host–sandalwood associations and host species were modeled through a stepwise regression approach (backward elimination) in STAR statistical software (IRRI, 2022).

$$Y = \alpha + \beta_1 X_1 + \beta_2 X_2 + \beta_3 X_3 \ldots\ldots\ldots\ldots\ldots + \beta_k X_k$$

where α is indicate the intercept.

β_i, ($i = 1, \ldots k$), = partial regression coefficients.

Thus, in this multiple linear functional equation with k independent variables (traits), the presence of β_i, (i.e., $\neq 0$) indicates the dependence of Y on X_i. The test of significance of each β_i for respective variables was performed through a *t*-test (Gomez and Gomez, 1984). The modeled equations were fitted to select the best host–parasite interaction, in which regression coefficients (βs) of individual traits significantly associated with higher diameter growth of sandalwood diameter were considered as weighted coefficients.

3 Results

3.1 Growth attributes of sandalwood and host species

The result demonstrated that the host species induced variability in the growth of sandalwood. Sandalwood exhibited significant ($p \leq 0.05$) differences in plant height, collar diameter, and leaf area of the sandalwood. Results explained that the diameter increment of sandalwood was recorded maximum ($p \leq 0.05$) with host *D. sissoo* (4.04 mm), followed by *C. equisetifolia* (4.01 mm) and *S. cumini* (3.64 mm) hosts, and minimum with the host *C. aurantium* (3.02 mm) (Figure 1A). However, among the host plant species, the highest ($p \leq 0.05$) collar diameter was recorded in *S. cumini* (8.64 mm), followed by *C. aurantium* (6.83 mm), whereas the lowest collar diameter was recorded in *C. equisetifolia* (4.18 mm). Results further showed the significant ($p \leq 0.05$) effect of host plant species on plant height in sandalwood which ranged between 42.25 cm (*A. ampliceps*) and 61.95 cm (*P. emblica*) (Figure 1B). Among the host species (Figure 1B), the maximum ($p \leq 0.05$) plant height was observed in *D. sissoo* (97.95 cm) followed by *P. granatum* (94.00 cm), *L. leucocephala* (90.50 cm), and the minimum plant height was recorded in *M. dubia* (43.55 cm). Results further explained the non-significant ($p \leq 0.05$) effect of different host species on the leaf area of sandalwood, whereas host plant species displayed significant variations ($p \leq 0.05$) in leaf area, which ranged between 0.48 (*C. equisetifolia*) and 27.70 cm^2 (*S. cumini*) with a mean value of 10.52 cm^2 (Figure 1C).

3.2 Plant water relation of sandalwood and host

The result showed that the host species induced differences in plant water relation of sandalwood. The highest ($p \leq 0.05$) relative water content (RWC) in sandalwood was recorded with host *L. leucocephala* (89.57), whereas its lowest value was observed with host *A. ampliceps* (73.85) (Table 2). Among the host species, *A. indica* (89.00) possessed the highest ($p \leq 0.05$) RWC, whereas *C. aurantium* (71.49) and *C. equisetifolia* (67.30) possessed the lowest value to RWC. In contrast, the host did not display any significant ($p \geq 0.05$) impact on the water potential (ψw) of sandalwood while the water potential of host species varied from −0.97 MPa (*A. ampliceps*) to

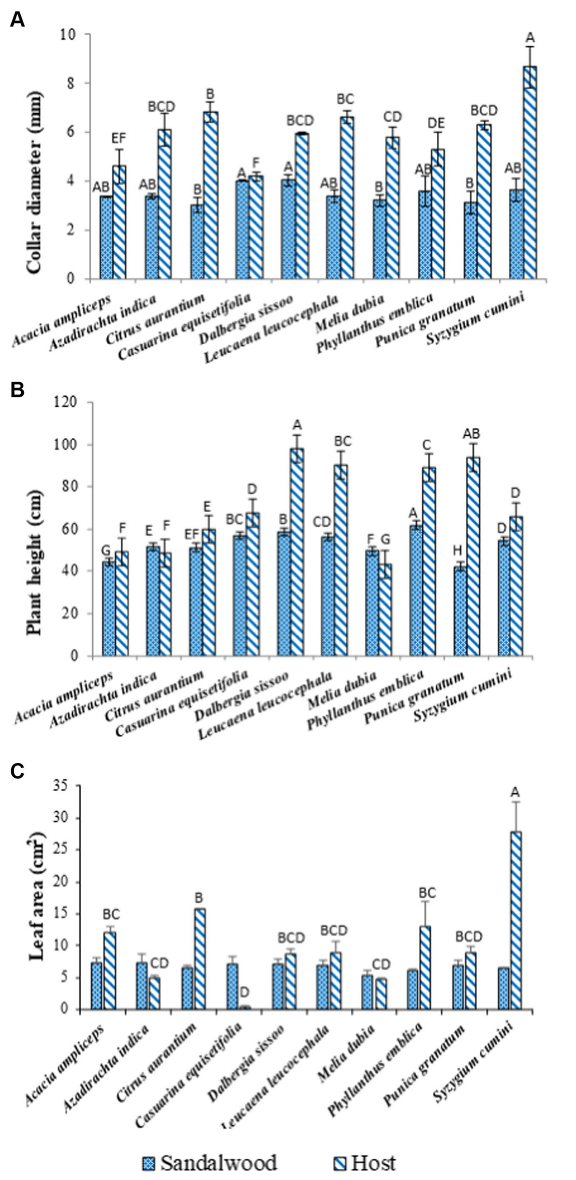

FIGURE 1
Collar diameter **(A)**, plant height **(B)**, and leaf area **(C)** of sandalwood and host species. Bars followed by different letters ($^{A,B,C...}$ etc.) indicate statistically significant differences.

−1.26 MPa (*A. indica*). The osmotic potential (ψs) of sandalwood is affected by different host species was ranged from a maximum of −1.78 MPa (*P. granatum*) to a minimum of −1.30 MPa (*C. aurantium*), whereas in host plants the maximum value of ψs was observed in *P. emblica* (−1.57 MPa) and minimum in *D. sissoo* (−1.08 MPa). Furthermore, host plants affected ($p \leq 0.05$) the vapor pressure deficit (VPD) in the sandalwood, which was observed highest with host *P. granatum* (1.39 MPa) followed by *S. cumini* (1.36 MPa), *P. emblica* (1.33 MPa), *L. leucocephala* (1.15 MPa), and *C. equisetifolia* (1.09 MPa) and minimum with host *D. sissoo* (0.12 MPa) (Table 2). Similarly, in host plant species, significantly ($p \leq 0.05$) highest VPD was noted in *C. equisetifolia* (3.43 MPa) followed by *C. aurantium* (3.33 MPa), *L. leucocephala* (3.16 MPa), and *S. cumini* (3.08 MPa), and minimum VPD was observed in *D. sissoo* (2.00 MPa).

3.3 Chlorophyll and gas exchange attributes of sandalwood and host

The findings indicated that the host species induced variations in chlorophyll and gas exchange characteristics of sandalwood while no change in photosynthetic rate and stomatal conductance of sandalwood were observed. Particularly, there were significant ($p \leq 0.05$) differences in the chlorophyll content in sandalwood leaves ranging from 0.91 to 1.43 mg g^{-1} (Table 3), which was observed in the order of *C. aurantium* > *D. sissoo* > *S. cumini* > *P. granatum* > *C. equisetifolia* > *M. dubia* = *P. emblica* > *A. ampliceps* > *A. indica*. Similarly, chlorophyll content in host plant leaves was recorded maximum ($p \leq 0.05$) in *C. aurantium* (2.04 mg g^{-1}), followed by *C. equisetifolia* (1.94 mg g^{-1}), *P. granatum* (1.81 mg g^{-1}), *P. emblica* (1.78 mg g^{-1}), *A. indica* (1.78 mg g^{-1}), and *S. cumini* (1.70 mg g^{-1}) host species and minimum in *D. sissoo* (0.92 mg g^{-1}). Furthermore, the present results explained that the host plant exerted a non-significant ($p \geq 0.05$) impact on the stomatal conductance and photosynthetic rate of sandalwood (Table 3). In contrast, host plant species displayed a significant ($p \leq 0.05$) difference in photosynthetic rate, which was recorded highest in *A. ampliceps* (8.56 µmol m^{-2} s^{-1}) followed by *A. indica*, *M. dubia*, and *S. cumini* and lowest in *C. equisetifolia* (1.69 µmol m^{-2} s^{-1}). Similarly, host *D. sissoo* showed maximum ($p \leq 0.05$) stomatal conductance of 9.27 mol H_2O m^{-2} s^{-1} followed by *A. ampliceps*, *M. dubia*, *L. leucocephala*, *A. indica*, *S. cumini*, and *P. granatum*, in decreasing order. Results further showed that the host plant contributed significantly to variations in the transpiration rate of sandalwood, which was observed at maximum ($p \leq 0.05$) with host *D. sissoo* (4.09 mmol m^{-2} s^{-1}) and *P. granatum* (4.09 mmol m^{-2} s^{-1}) and minimum ($p \leq 0.05$) with *P. emblica* (2.47 mmol m^{-2} s^{-1}) (Table 3). In host plants, the transpiration rate remained non-significant ($p \geq 0.05$) ranging from 0.70 mmol m^{-2} s^{-1} (*D. sissoo*) to 2.73 mmol m^{-2} s^{-1} (*M. dubia*). Similarly, the intercellular CO_2 concentration in sandalwood plants was observed to be highest ($p \leq 0.05$) with *A. ampliceps* (328.51 µmol mol^{-1}), followed by *A. indica* (307.32 µmol mol^{-1}), *C. aurantium* (296.41 µmol mol^{-1}), *D. sissoo* (273.17 µmol mol^{-1}), *P. emblica* (263.01 µmol mol^{-1}), *M. dubia* (255.06 µmol mol^{-1}) hosts, and minimum with host *S. cumini* (249.89 µmol mol^{-1}). In host plant species, the intercellular CO_2 concentration was recorded as maximum ($p \leq 0.05$) in *C. aurantium* (338.66 µmol mol^{-1}) and minimum ($p \leq 0.05$) in *S. cumini* (222.62 µmol mol^{-1}).

3.4 Osmolyte accumulation in sandalwood and host

The results revealed that the host species induced significant ($p \leq 0.05$) variations in proline content accumulation in sandalwood among the different osmolytes studied. Sandalwood showed non-significantly ($p \geq 0.05$) maximum total soluble sugar with host species *A. indica* (1.37 mg g^{-1}) whereas minimum with *A. ampliceps* (0.81 mg g^{-1}). Conversely among the host species, significantly ($p \leq 0.05$) higher total soluble sugar was found in *A. ampliceps* (1.63 mg g^{-1}), followed by *C. aurantium* (1.21 mg g^{-1}) and *P. granatum* (1.21 mg g^{-1}) and minimum in *C. equisetifolia* (0.72 mg g^{-1}). Furthermore, the proline content in sandalwood leaves was observed highest ($p \leq 0.05$) with host *L. leucocephala*

TABLE 2 Plants–water relations of sandalwood and host species.

Host species	Relative water content		Water potential (MPa)		Osmotic potential (MPa)		Vapor pressure deficit (MPa)	
	S	H	S	H	S	H	S	H
Acacia ampliceps	73.85^{B}	80.84^{ABC}	−1.13	−0.97^{A}	−1.32^{A}	−1.09^{A}	0.41^{DE}	2.92^{C}
Azadirachta indica	87.40^{A}	89.00^{A}	−1.33	−1.26^{C}	−1.54^{BC}	−1.34^{BCD}	0.57^{CD}	2.56^{D}
Citrus aurantium	87.18^{A}	71.49^{BC}	−0.96	−1.09^{ABC}	−1.30^{A}	−1.25^{B}	0.53^{CD}	3.33^{AB}
Casuarina equisetifolia	86.64^{A}	67.30^{C}	−1.20	−1.23^{C}	−1.50^{B}	−1.42^{D}	1.09^{AB}	3.43^{A}
Dalbergia sissoo	87.49^{A}	77.33^{ABC}	−1.33	−0.99^{AB}	−1.60^{CD}	−1.08^{A}	0.12^{E}	2.00^{E}
Leucaena leucocephala	89.57^{A}	79.69^{ABC}	−1.20	−1.25^{C}	−1.72^{EF}	−1.42^{D}	1.15^{AB}	3.16^{ABC}
Melia dubia	86.00^{A}	83.94^{AB}	−1.18	−1.14^{ABC}	−1.35^{A}	−1.25^{BC}	0.84^{BC}	2.07^{E}
Phyllanthus emblica	88.05^{A}	79.94^{ABC}	−1.23	−1.22^{C}	−1.54^{BC}	−1.57^{E}	1.33^{A}	2.30^{DE}
Punica granatum	89.01^{A}	82.42^{ABC}	−1.33	−1.23^{C}	−1.78^{F}	−1.38^{BCD}	1.39^{A}	2.96^{C}
Syzygium cumini	88.42^{A}	73.60^{ABC}	−1.31	−1.17^{BC}	−1.67^{DE}	−1.38^{CD}	1.36^{A}	3.08^{BC}
Mean	86.36	78.56	−1.22	−1.15	−1.53	−1.32	0.88	2.78
$CD_{(0.05)}$	8.44	16.55	NS	0.19	0.08	0.13	0.36	0.34

Mean followed by different letters ($^{A,B,C...}$ etc.) indicate statistically significant differences (S: Sandalwood; H: Host; NS: non-significant).

TABLE 3 Gas exchange attributes of sandalwood and host species.

Host species	Chlorophyll content (mg g^{-1})		Photosynthetic rate (µmol m^{-2} s^{-1})		Stomatal conductance (mol H_2O m^{-2} s^{-1})		Transpiration rate (mmol m^{-2} s^{-1})		Intercellular CO_2 concentration (µmol mol^{-1})	
	S	H	S	H	S	H	S	H	S	H
Acacia ampliceps	0.91^{BC}	1.46^{C}	3.76	8.56^{A}	3.30	9.06^{AB}	2.52^{A}	2.73	328.51^{A}	315.17^{AB}
Azadirachta indica	0.80^{C}	1.78^{AB}	4.29	6.44^{AB}	6.56	6.72^{ABCD}	2.66^{A}	1.64	307.32^{A}	247.98^{CDE}
Citrus aurantium	1.43^{A}	2.04^{A}	3.03	2.05^{C}	5.63	3.79^{BCD}	2.73^{A}	1.75	296.41^{AB}	338.66^{A}
Casuarina equisetifolia	1.06^{BC}	1.94^{AB}	3.30	1.69^{C}	3.48	3.35^{CD}	3.33^{A}	1.80	217.05^{BC}	227.74^{DE}
Dalbergia sissoo	1.24^{AB}	0.92^{D}	4.22	3.83^{BC}	3.06	9.27^{A}	4.09^{A}	0.70	273.17^{AB}	305.14^{ABC}
Leucaena leucocephala	0.80^{C}	0.97^{D}	5.30	3.91^{BC}	6.16	7.60^{ABCD}	3.05^{A}	2.47	220.60^{BC}	265.49^{BCDE}
Melia dubia	0.95^{BC}	1.46^{C}	4.53	4.83^{BC}	5.96	8.29^{ABC}	2.48^{A}	1.96	255.06^{ABC}	287.68^{ABCD}
Phyllanthus emblica	0.95^{BC}	1.78^{AB}	3.87	2.51^{C}	3.46	2.91^{D}	2.47^{A}	1.01	263.01^{ABC}	318.85^{AB}
Punica granatum	1.18^{AB}	1.81^{AB}	4.39	3.39^{BC}	3.71	5.39^{ABCD}	4.09^{A}	1.90	188.59^{C}	274.95^{BCDE}
Syzygium cumini	1.22^{AB}	1.70^{BC}	3.18	4.12^{BC}	3.36	5.63^{ABCD}	4.07^{A}	2.06	249.89^{ABC}	222.62^{E}
Mean	1.05	1.59	3.99	4.13	4.47	6.20	3.15	1.80	259.96	280.43
$CD_{(0.05)}$	0.36	0.29	NS	3.42	NS	5.38	1.72	NS	79.57	63.34

Means followed by different letters ($^{A,B,C...}$ etc.) indicate statistically significant differences (S: sandalwood; H: host; NS: non-significant).

(37.49 µg g^{-1}) followed by *M. dubia* (34.57 µg g^{-1}) and lowest ($p \leq 0.05$) with host *C. equisetifolia* (14.93 µg g^{-1}). Similarly, among the host plant species, *M. dubia* (24.73 µg g^{-1}) possessed the highest ($p \leq 0.05$) proline content followed by *C. aurantium* (17.54 µg g^{-1}) and *C. equisetifolia* (16.22 µg g^{-1}), and the lowest ($p \leq 0.05$) proline was recorded in *P. emblica* (7.1 µg g^{-1}). On the other hand, host plants did not ($p \geq 0.05$) affect protein content in sandalwood, which ranged from 7.70 mg g^{-1} (*P. emblica*) to 12.85 mg g^{-1} (*A. indica*), whereas in host plant the significantly ($p \leq 0.05$) highest protein content was recorded in *A. ampliceps* (15.75 mg g^{-1}) and minimum with *P. emblica* (6.86 mg g^{-1}) (Table 4).

3.5 Traits modeling in sandalwood and host plants through multiple regression approach

Plant water traits, gas exchange attributes, and osmolytes play a significant role in determining the host response to sandalwood. Magnitude of traits toward the host–parasitic relationship was identified through multiple regression approach. Analysis showed that six traits (plant height, total soluble protein, intercellular CO_2 concentration, relative water content, photosynthetic rate, and water potential) of sandalwood; while, four traits (water potential, osmotic potential, leaf area, and total soluble protein) of host plants induced significant

TABLE 4 Osmolyte accumulation in sandalwood and host species.

Host species	Soluble sugars (mg g^{-1})		Proline content (μg g^{-1})		Protein content (mg g^{-1})	
	S	H	S	H	S	H
Acacia ampliceps	0.81	1.63^{A}	21.16^{C}	7.70^{E}	12.61	15.75^{A}
Azadirachta indica	1.37	1.16^{B}	22.33^{BC}	9.43^{DE}	12.85	12.15^{AB}
Citrus aurantium	1.31	1.21^{B}	22.87^{BC}	17.54^{B}	9.33	9.85^{AB}
Casuarina equisetifolia	1.09	0.72^{C}	14.93^{C}	16.22^{BC}	12.30	11.37^{AB}
Dalbergia sissoo	1.06	0.90^{BC}	18.91^{C}	10.33^{CDE}	12.53	11.02^{AB}
Leucaena leucocephala	1.10	1.12^{B}	37.49^{A}	9.08^{DE}	11.50	9.78^{AB}
Melia dubia	1.21	0.97^{BC}	34.57^{AB}	24.73^{A}	13.34	11.98^{AB}
Phyllanthus emblica	1.27	0.90^{BC}	18.71^{C}	7.10^{E}	7.70	6.86^{B}
Punica granatum	1.01	1.21^{B}	22.27^{BC}	12.27^{BCDE}	8.23	7.56^{B}
Syzygium cumini	1.05	0.99^{BC}	19.56^{C}	14.55^{BCD}	8.36	8.88^{AB}
Mean	1.13	1.08	23.28	12.90	10.88	10.52
$CD_{(0.05)}$	NS	0.35	12.68	6.77	NS	7.99

Means followed by different letters ($^{A,B,C...}$ etc.) indicate statistically significant differences (S: sandalwood; H: host; NS: non-significant).

TABLE 5 Significance of traits magnitude toward diameter growth of sandalwood associated with different host plants.

Variables	Regression coefficients (β_s)	Std. error	*t* value	Pr(>|t|)
(Constant)	4.305	1.759	2.447	0.029
PH	0.047	0.011	4.330	0.001
WP	−1.378	0.480	−2.867	0.013
RWC	−0.042	0.018	−2.383	0.033
Pn	−0.204	0.070	−2.898	0.012
Ci	−0.005	0.002	−2.903	0.012
TSP	0.070	0.023	3.020	0.010
	Model fitted: DBH ~ 4.305+ 0.047 PH +(−1.378) WP+ (−0.042) RWC + (−0.204) Pn + (−0.005) Ci + 0.070 TSP			

TABLE 6 Significance of traits magnitude toward diameter growth of host species.

Variables	Regression coefficients (β_s)	Std. error	*t* value	Pr(>|t|)
(Constant)	3.768	2.288	1.647	0.120
WP	−11.757	2.757	−4.264	0.001
OP	8.460	2.332	3.628	0.0002
LA	0.144	0.021	6.942	0.000
TSP	−0.159	0.072	−2.196	0.044
	Model fitted: DBH ~ 3.768+ (−11.757) WP + (8.46) OP+ (0.144) LA + (−0.159) TSP			

differences in the growth performance of sandalwood (Tables 5, 6). Consequently, these traits were included in sandalwood diameter growth modeling to predict the sandalwood response with respect to host species in different host plant associations. Furthermore, on the basis of diameter growth modeling, host species were prioritized on the basis of predicted growth response of growth diameters.

Overall, results showed that *Casuarina equisetifolia* followed by *Dalbergia sissoo* is the best host for sandalwood. However, among the host species *Syzygium cumini* followed by *Citrus aurantium* is best with respect to their growth performance (Tables 7, 8).

4 Discussion

Sandalwood is a semi-root parasite plant that depends on the host plants for water and nutrient requirements. Consequently, a large number of growth and physiological process occurring in both sandalwood and host species govern their performance under a particular set of conditions. The present results revealed significant differences in plant height, collar diameter, and leaf area of the sandalwood. The highest growth of sandalwood with *P. emblica* followed by *D. sissoo* and *C. equisetifolia* might have resulted from the better association of sandalwood with these hosts. The varied growth pattern of sandalwood with different host have been extensively studied (Rocha et al., 2014; Doddabasawa et al., 2020; Sahu et al., 2021), which suggested that the performance of sandalwood is governed by the characteristics of the different host plants (translocation of mineral nutrients and water, slow growth rate, and lateral root system) and the competition for above-ground resources, including light (Rocha et al., 2014). Specifically, the sandalwood roots not only absorb water and nutrients from the host plants, but also control the host plant roots and effectively regulate the supply of water and nutrients (Verma et al., 2023a,b), thereby regulating various growth processes in the sandalwood. Simultaneously, host requirement and the growth stage of host introduction have also a substantial impact on the development of sandalwood (Ramya, 2010), which indicates that sandalwood growth and development are regulated by the selected host species.

In the current investigation, the plant water traits, such as RWC, osmotic potential (ψs), and vapor pressure deficit (VPD) of sandalwood, were higher with hosts *L. leucocephala* and *P. granatum*. The host-wise variation in plant water traits of sandalwood suggests the difference in the water absorption by both sandalwood and host plants. Rocha et al.

10.3389/ffgc.2024.1395394

TABLE 7 Prioritization of host–sandalwood associations through the predicted response of sandalwood diameter growth.

Sandalwood–host association	Constant (α)	0.047× PH	−1.378 × WP	−0.042 × RWC	−0.204 × Pn	−0.005 × Ci	0.07 × TSP	Predicted diameter	Rank	Actual diameter	Rank
Acacia ampliceps	4.305	2.09	1.55	−3.10	−0.77	−1.64	0.88	3.32	8	3.36	7
Azadirachta indica		2.43	1.83	−3.67	−0.88	−1.54	0.90	3.38	6	3.37	6
Citrus aurantium		2.41	1.32	−3.66	−0.62	−1.48	0.65	2.92	10	3.02	10
Casuarina equisetifolia		2.67	1.65	−3.64	−0.67	−1.09	0.86	4.10	1	4.01	2
Dalbergia sissoo		2.76	1.83	−3.67	−0.86	−1.37	0.88	3.86	2	4.04	1
Leucaena leucocephala		2.65	1.65	−3.76	−1.08	−1.10	0.81	3.46	5	3.38	5
Melia dubia		2.33	1.62	−3.61	−0.92	−1.28	0.93	3.38	7	3.21	8
Phyllanthus emblica		2.91	1.69	−3.70	−0.79	−1.32	0.54	3.65	3	3.57	4
Punica granatum		1.99	1.83	−3.74	−0.89	−0.94	0.58	3.12	9	3.13	9
Syzygium cumini		2.56	1.81	−3.71	−0.65	−1.25	0.59	3.64	4	3.64	3
Model fitted: DBH ~ 4.305 + 0.047 PH + (−1.378) WP + (−0.042) RWC + (−0.204) Pn + (−0.005) Ci + 0.070 TSP											

Where DBH, collar diameter; PH, photosynthetic rate; RWC, relative water content; WP, water potential; C_i intercellular CO_2 concentration; TSP, total soluble protein.

(2014) also reported that host species induced variation in plant water of sandalwood, and they showed that the host *Casuarina equisetifolia* has a higher water potential than the sandalwood (without host). However, sandalwood could adjust its water and osmotic potential as well as relative water content either by accumulating osmotically active chemicals or by maintaining the xylem-to-xylem connection with host plants. Such processes could lead to the enhancement in turgor to facilitate the water uptake in plants (Dhaniklal, 2006; Rocha et al., 2014; Kumar et al., 2021). Results further revealed that among different host species, *A. indica*, *L. leucocephala,* and *C. equisetifolia* showed the highest value for RWC, ψs, and VPD, respectively. The species-specific differences in morphology of total number of leave and roots biomass, etc., might be responsible for variation in plant–water relations in these species. Mielke et al. (2005), Johnson et al. (2009), Klein (2014), and Leuschner et al. (2019) have provided substantial evidence supporting the existence of species-specific variations in plant–water relations, which indicates that plant–water relations in sandalwood are governed by the type of host species as latter is responsible for maintaining the plant water status of the former.

Furthermore, results revealed that, *C. aurantium* as a host of sandalwood as well as individually showed the maximum chlorophyll content in the leaves, indicating that citrus might be absorbing a greater quantity of minerals responsible for the formation of higher chlorophyll content on its own as well as in sandalwood leaves (Rocha et al., 2014; Doddabasawa et al., 2020). The host-wise difference in chlorophyll content of sandalwood suggests species-wise differences in the absorption of substances responsible for initiating the synthesis of chlorophyll in the leaves (Coste et al., 2010; Li et al., 2018; Leuschner et al., 2019). Furthermore, the non-significant ($p \geq 0.05$) effect of host plant on both the photosynthetic rate and stomatal conductance of sandalwood suggests species ability to maintain both these processes equally, irrespective of types of host species. Moreover, despite influence of the hosts, sandalwood can adjust activities of chlorophyll, RuBPase, and stomata under the prevailing environmental conditions (Balasubramanian et al., 2021). Furthermore, Rocha et al. (2014) showed that the lower content of nitrogen and chlorophyll and the altered RuBPase activities were responsible for decline in the photosynthetic rate of sandalwood. The species (host) wise differences in light utilization efficiency, stomata closure, and CO_2 absorption in chloroplasts were major causes of differences in their photosynthetic rate (Kumar et al., 2016; Sheoran et al., 2021; Soni et al., 2021).

The host plant also induced differences in transpiration rate of sandalwood, which was observed maximum ($p \leq 0.05$) with host *D. sissoo*, suggesting that despite providing uniform irrigation to all plants, the host induced differences in the transpiration rate of sandalwood, which can be attributed to the substantial disparity in driving forces responsible for the movement of water from the soil-to-leaf system and leaf–atmosphere interface. Moreover, from the physiological point of view, vaporpressure is sought to play a key role in the rate of water flow and transpiration (Zhang et al., 2017). It has also been reported that sandalwood for its survival produces high transpiration rates through uncontrolled stomata to maximize water loss and generate a water potential gradient and sink strength even larger than the host to extract the crucial resources from it (Liu et al., 2003; Grewell, 2008). Furthermore, the 'resistance of host to xylem solute transfer, as well as its stomatal response, also influences the maintenance of solute flow from host to parasite (Jiang et al., 2003). The highest intercellular CO_2 concentration in

TABLE 8 Prioritization of host species through the predicted response of the diameter growth of the host.

Sandalwood host	Constant (α)	−11.757 × WP	8.46 × OP	−0.144 × LA	−0.159 × TSP	Predicted diameter	Rank	Actual diameter	Rank
Acacia ampliceps	3.768	11.40	−9.22	1.72	−2.50	5.17	9	4.59	9
Azadirachta indica		14.76	−11.34	0.71	−1.93	5.97	5	6.10	5
Citrus aurantium		12.76	−10.53	2.28	−1.57	6.70	2	6.83	2
Casuarina equisetifolia		14.40	−11.97	0.07	−1.81	4.46	10	4.18	10
Dalbergia sissoo		11.64	−9.14	1.26	−1.75	5.78	6	5.94	6
Leucaena leucocephala		14.70	−11.97	1.29	−1.55	6.23	4	6.62	3
Melia dubia		13.40	−10.58	0.68	−1.90	5.37	8	5.78	7
Phyllanthus emblica		14.28	−13.28	1.87	−1.09	5.55	7	5.30	8
Punica granatum		14.40	−11.63	1.29	−1.20	6.62	3	6.29	4
Syzygium cumini		13.76	−11.67	3.99	−1.41	8.43	1	8.64	1
Model fitted: DBH ~ 3.768+ (−11.757) WP + 8.46 OP +0.144 LA + (−0.159) TSP									

Where DBH, collar diameter; WP, water potential; OP, osmotic potential; LA, leaf area; TSP, total soluble protein.

sandalwood plants with the host *A. ampliceps* and individually in *C. aurantium* resulted from species-specific differences in the absorption of CO_2. Similarly, Annapurna et al. (2006) also observed host species effects on physiological processes of sandalwood and reported that hosts *D. sissoo* maintained higher RWC along with the higher photosynthetic rate, stomatal conductance, and transpiration, compared to rest of the host species, which might have contributed to the higher growth of sandalwood. Therefore, the type of relationship between parasite and host is one of the most important aspects in terms of intake and translocation of mineral nutrient, as well as maintenance of physiological efficiency (Shen et al., 2006; Kumar et al., 2016).

Moreover, in the current study, sandalwood showed maximum ($p \leq 0.05$) production of total soluble sugar with host *A. ampliceps*. The greater production of specific organic solutes (osmolytes) aids in maintaining the plant water status and photosynthesis as well as safeguarding the cellular machinery against harmful substances (Lata et al., 2019; Kumar et al., 2021). Moreover, during the transfer of nutrients from host to the parasite, potentially hazardous chemicals and disease infections may pass through the haustoria, causing stress in the parasitic plants that contribute to an increase in the production of osmolytes (Zagorchev et al., 2021). Among the host species, *A. indica* showed maximum total soluble sugar, suggesting an increase in total sugar might contribute to greater production of biomass in the species. Similarly, Zhou et al. (2021) reported that the soluble carbohydrate accumulation acts as a source of energy for branch sprouting, elongation, and thickening. Moreover, the soluble carbohydrate is sought to have a key role in the acquisition of solute by reducing the ψ_w of hemi- parasite and facilitating the flow of salute from the host that might have substantial ramifications for the host–parasite system. Furthermore, these might help to counterbalance the excessive inorganic ion concentrations in vacuoles and improve the stability of membranes and enzymes (Karakas et al., 1997; Pooja Nandwal et al., 2019). The higher soluble sugar concentrations have been reported to also alter the nutrient availability in hemiparasitic plants (Gomes and Adnyana, 2017).

Proline accumulation is a key physiological indicator of signaling regulatory molecules in plants that trigger a variety of responses during the adaptation process. Our result suggested that sandalwood accumulates higher proline content with host *L. leucocephala,* compared to the rest of the host species, indicating the better adaptability of sandalwood with this host. Moreover, an increase in proline content will eventually promote the synthesis of protein and the development of cells and will provide nitrogen and carbon needed for them to expand and energy use (Bell and Adams, 2011; Christgen and Becker, 2019). However, results showed non-significant influence of host plants on the protein content in sandalwood. This was due to the greater change in cellular soluble protein content occurring only under stress conditions. Moreover, the increased protein accumulation indicates the preservation of nontoxic useable forms of cellular N that aids in the re-establishment of C and N balance by utilizing carbon from photosynthesis and glycolysis (Chen et al., 2017). Physiologically, changes in metabolite concentrations typically occur in trees before the appearance of any visible symptoms, and the changes in nitrogen metabolism are inextricably linked to changes in carbon metabolism (Minocha et al., 2019).

5 Conclusion

The current investigation provided detailed insight into the simultaneous changes in the growth and physiology of both sandalwood and different host species during the host–parasite interaction process, thereby enhancing our understanding of the physiological and biochemical aspects involved in this complex interaction. In conclusion, sandalwood exhibited higher growth performance when grown with hosts *D. sissoo* and *C. equisetifolia*, while among host species, *S. cumini*, *C. aurantium*, and *L. leucocephala* showed superior growth and physiobiochemical traits. Moreover, plant height, water potential, relative water content, photosynthetic rate, intercellular CO_2 concentration, and total soluble protein were the major traits influencing the growth of the sandalwood while only four traits, namely, water potential, osmotic potential, leaf area, and total soluble protein favored the growth of the host plants. The present finding will aid the manipulation of physio-biochemical traits during host–parasite interaction and based on the performance of traits during the interaction process, the site-specific cultivation and management practices could be devised to successfully establish and manage the sandalwood plantations across the globe.

Data availability statement

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding authors.

Author contributions

KV: Writing – original draft, Visualization, Resources, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. AK: Writing – original draft, Validation, Supervision, Software, Project administration, Data curation, Conceptualization. RK: Writing – original draft, Validation, Supervision, Software, Project administration, Data curation, Conceptualization. AB: Writing – review & editing, Validation, Supervision, Formal analysis. SD: Writing – review & editing, Visualization, Validation, Formal analysis. AS: Writing – review & editing, Visualization, Resources, Investigation. PS: Writing – review & editing, Visualization, Resources, Funding acquisition, Formal analysis, Data curation.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Monitoring vegetation degradation using remote sensing and machine learning over India – a multi-sensor, multi-temporal and multi-scale approach

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Vegetation cover degradation is often a complex phenomenon, exhibiting strong correlation with climatic variation and anthropogenic actions. Conservation of biodiversity is important because millions of people are directly and indirectly dependent on vegetation (forest and crop) and its associated secondary products. United Nations Sustainable Development Goals (SDGs) propose to quantify the proportion of vegetation as a proportion of total land area of all countries. Satellite images form as one of the main sources of accurate information to capture the fine seasonal changes so that long-term vegetation degradation can be assessed accurately. In the present study, Multi-Sensor, Multi-Temporal and Multi-Scale (MMM) approach was used to estimate vulnerability of vegetation degradation. Open source Cloud computing system Google Earth Engine (GEE) was used to systematically monitor vegetation degradation and evaluate the potential of multiple satellite data with variable spatial resolutions. Hotspots were demarcated using machine learning techniques to identify the greening and the browning effect of vegetation using coarse resolution Normalized Difference Vegetation Index (NDVI) of MODIS. Rainfall datasets of Climate Hazards Group InfraRed Precipitation with Station data (CHIRPS) for the period 2000–2022 were also used to find rainfall anomaly in the region. Furthermore, hotspot areas were identified using high-resolution datasets in major vegetation degradation areas based on long-term vegetation and rainfall analysis to understand and verify the cause of change whether anthropogenic or climatic in nature. This study is important for several State/Central Government user departments, Universities, and NGOs to lay out managerial plans for the protection of vegetation/forests in India.

KEYWORDS

vegetation, land degradation, MMM approach, remote sensing, sustainability

1 Introduction

Land is referred as an entity with zero consumption rate (Thepade and Chaudhari, 2021). Therefore, it is one of the most important resources to maintain the functioning of the socioeconomic system perfectly (Xie et al., 2022). Degradation of land is based on the continuous or long-term loss of natural resources. World's 40% of land area is currently degraded and is increasing over time and directly threatening (Shao et al., 2024). Among all landscapes, forests are critical strongholds for environmental services and can help for ecological restoration (Fa et al., 2020; Sharma et al., 2022) apart from forest monitoring, natural resources such as agriculture monitoring is also important for understanding the consequence of a mismatch between land suitability and land use (Kılıç et al., 2024). According to the Forest Resource Assessment by Food and Agriculture Organization (FAO), 4 billion hectares of global land come under forest, half of which is found in the tropics and subtropics (Forneri et al., 2006; Singh et al., 2022; Nabuurs et al., 2023). Among all land uses, quantifying forest extent and change in forest is most important not only because forest protects climatic condition but also it supplies most essential flow of ecosystem services, such as fiber, energy, supports biodiversity, carbon storage, flux, and water (Coulston et al., 2014; Ferreira et al., 2023). Due to massive importance of forests, United Nations Framework Convention on Climate Change (UNFCCC) has shown significant interest on the protection of forest patches; the convention outlined that forest degradation contributes to total global carbon emission to a large extent, leading to global warming, and therefore, forest degradation needs immediate protection (Gao et al., 2020; Liang and Gamarra, 2020).

Tropical forests have always been critically acclaimed for rainfall generation because forests in these areas produce rainfall twice more than in any other latitude (Doughty et al., 2023); therefore, forest degradation in these regions can cause a reverse convective cloud cover and disturb the global rainfall circle (Watson et al., 2018; King et al., 2024). Tropical forest is niche of several species (Noulèkoun et al., 2024); therefore, deforestation and degradation of forest can lead to the extinction of biological diversity by habitat destruction and isolation of formerly contiguous forest and significant physical and biological consequences at the fringe or boundary zone between forest and deforested areas (Gascon et al. 2000; Rani et al., 2024). In the past, tropical deforestation globally reported approximately 1.1–2.2 PgC/year release (Gullison et al. 2007; Sasaki and Putz, 2009, Pujar et al., 2024) due to continuous increase in threats and pressure on forest resources; therefore, it is believed that demand for quantitative measurement, timely and accurately, can help in sustainable management of forests.

Earlier studies have often suggested that forest degradation is one of the major reasons for the discrepancy of forest cover maps (Sexton et al., 2015; Qin et al., 2024). To overcome this draw back, UNFCCC has defined forest as an area of more than 0.5–1.0 ha with tree crowning covering 10–30% (UNFCCC, 2002; Romijn et al., 2013; Asante et al., 2017). Furthermore, the convention has recognized forest habitats as important factors influencing the decline in forest-dependent fauna and flora (Rurangwa et al., 2021); therefore, it is important to understand forest degradation for forest restoration work (Roy et al., 2013; Jashimuddin et al., 2024). Southeast Asian countries have highest rate of tropical deforestation globally (Miettinen et al., 2011; Chen et al., 2023; Stan et al., 2024). In this region, food insecurity and poverty are the main concern enduring undue pressure on agricultural and forested land, and therefore, reconciliation of these lands is the main priority (Carrasco et al., 2016; Gonzalez-Redin et al., 2024). India is one of the mega-biodiversity nations lying in junction between three major biogeographic realms, namely, the Indo-Malayan, the Eurasian, and the Afro-tropical (Reddy et al., 2015). According to Forest Survey of India (FSI) reports, the country's forest covers 7, 13,789 km^2 (Forest Survey of India, 2021), which represents 21% of the total geographical area of the country. Although change in forest cover is highly debatable due to seasonal inference in countries such as India, Pasha and Dadhwal, 2024 claimed that forest cover is declining at the rate of 2.43 ha across the country.

In recent decades, remote sensing has gained its popularity due to its unique capability for providing imageries of the Earth's surface in such a way that it allows easy identification of features, location, and characteristics in accordance to different time span (Singh et al., 2022; Chauhan et al., 2024). Remote sensing technologies use spectral properties to reveal information regarding the health of the canopies (Houborg et al., 2015; Wen et al., 2024). The vegetation dynamics of any land mass have always been treated as a significant indicator that can be used to quantitatively detect ecosystem processes at different scales by remote sensing experts (Sur et al., 2018). Absorption of leaf signatures is prominent due to leaf pigmentation (such as chlorophylls, carotenoids, and xanthophylls) in the visible spectral region (400–700 nm), with moderate absorptions by water in the shortwave-infrared (SWIR, 1300–2,100 nm) and only slight absorption by leaves in the near-infrared (NIR, 700–1,300 nm) region (Huete, 2012; Santana et al., 2024). To add further, in recent years, the increased availability of computing power through openly available cloud platforms (e.g., Google Earth Engine) and readily accessible machine learning algorithms from open source software tools (e.g., Python Scikit-learn) has enabled the processing of large volumes of satellite imagery over increasingly large geographical areas with reduced complexity and time (Gorelick et al., 2017; Sur et al., 2021; Verma et al., 2023; Singh et al., 2024). Therefore, this study is designed to make a spatial framework in GEE to compute forest degradation using a special Multi-Sensor, Multi-Temporal and Multi-Scale (MMM) approach. This approach is unique of its own because it will help to assess vegetation degradation over any selected period. This particular study is based on the temporal window 2000–2022 in a single frame capturing three distinct seasonal variations (summer, monsoon, and winter) using machine learning techniques for entire India. Furthermore, it demonstrates the distinctive utility of satellite images in GEE platform to identify the hotspots and evaluate the causes of vegetation degradation and take sustainable combating measures.

2 Study area

The study area (Figure 1) is focused on Indian landmass comprising of 28 states and 8 union territories. Since 1990, after economic liberalization of India, socio-economic development lead to considerable stress on the natural ecosystem (Visser, 2017), which needs immediate sustainable management strategies. Therefore, the study concentrates on vegetation including 14 major forest types of India based on multi-season satellite images, biogeography, climate, and elevation. Wet evergreen and semi evergreen forests are mainly located in the high-rainfall areas of Western Ghats, Andaman and Nicobar Islands, and north eastern region. Central India is mainly dominated by dry and moist deciduous forests. Mangroves are found

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FIGURE 1
Study area.

in the coastal region of West Bengal, Odisha, Andhra Pradesh, Tamil Nadu, Maharashtra, Goa, and Gujarat. Montane, sub-alpine, and sub-tropical pine forests are distributed in the high-altitude Himalayan and north-east regions (Chakraborty et al., 2018). India has a typical climatic condition, often affected by heat waves, cold waves, fog, snowfall, floods and droughts, monsoon depressions, and cyclones (Dash et al., 2007). Combined action of physical factors and socio-economic factors makes Indian landmass more vulnerable and fragile.

3 Methodology

The overall methodology adopted for the study is shown in Figure 2. Google Earth Engine (GEE) is an open source platform for geospatial data analysis. Its interface used in this study uses Java script and Python for data processing. The entire methodology can be divided into four broad structural domains: (i) selection of database for analysis, (ii) technical algorithm for understanding greening and browning, (iii) technical algorithm for understanding anomaly leading to wet and dry regions, (iv) visualization of degraded vegetation in high resolution.

3.1 Selection of database for analysis

3.1.1 Vegetation index datasets

The vegetation index product MOD13Q1 (V6.1) (Didan et al., 2015) is provided by Terra Moderate Resolution Imaging Spectroradiometer (MODIS). This study uses this product from 2000 to 2022 for analysis. It consists of two layers (a) Normalized Difference Vegetation Index (NDVI) and (b) Enhanced Vegetation Index (EVI) per pixel basis. The EVI layer minimizes canopy background variations and maintains sensitivity over dense vegetation conditions. The EVI also uses the blue band to remove residual atmosphere contamination caused by smoke and sub-pixel thin cloud. The EVI products are computed from atmospherically corrected bi-directional surface reflectance that has been masked for water, clouds, heavy aerosols, and cloud shadows. This product provides dataset of 250-m spatial resolution.

3.1.2 Rainfall datasets

Dataset from Climate Hazards Group Infra Red Precipitation with Station data (CHIRPS) is a quasi-global rainfall dataset (Funk et al., 2012). The CHIRPS dataset incorporates 0.05° resolution satellite imagery with *in-situ* station data to create gridded rainfall time series for trend analysis and seasonal monitoring of rainfall. In this study, monsoon datasets for the period 2000–2022 were utilized to compute the anomaly of the rainfall distribution.

3.1.3 Landsat datasets

Landsat is a joint program of the USGS and NASA, and since 1972 it has been observing the Earth continuously. The Landsat satellites image the entire surface of the Earth at a 30-m resolution approximately once every 2 weeks, including multispectral and thermal data. Three bands (Red, Green, and Blue) were used for monitoring vegetation degradation between 2000 and 2022.

3.1.4 Technical algorithm for understanding greening and browning

Theil–Sen slope and Mann–Kendall test were used to analyze long term (2000–2022) vegetation trend in a time-series of each pixel (Fensholt et al., 2012). Theil–Sen slope algorithm was preferred because of its efficiency to handle huge discrete dataset, and on the other hand, Mann–Kendall test was used along with Theil–Sen slope due to its advantage of not requiring samples to confirm a specific distribution and is free from the effect of outliers' interference (Feng et al., 2023). Therefore, this study uses Theil– Sen slope and Mann–Kendall test to estimate the greening and browning trend of the time-series over the forest cover.

3.1.5 Technical algorithm for understanding rainfall anomaly

The impact of rainfall variability on vegetation in Indian landscape was characterized based on the analysis of time-series anomalies. The response of vegetation to moisture availability was analyzed, especially in dry, normal, and wet years. The spatiotemporal variability in rainfall anomalies was produced using the annual mean rainfall from 2000 to 2022. Equation (1) is the standard anomaly equation which is generally used in statistics to estimate the Z-score (Atkinson et al., 2011). Since the study covers a vast region, rainfall quantity is different in dry and wet regions. Thus, we have to standardize rainfall in order to compare the relative variability across the country.

$$RA_{(i,x,y)} = \frac{Rainfall_{(i,x,y)} - Mean\left[Rainfall_{(2000-2022),(i,x,y)}\right]}{STD\left[Rainfall_{(2000-2022),(i,x,y)}\right]} \quad (1)$$

where RA (i, x, y) represents the annual standardized rainfall anomaly in the ith year at the pixel location (x, y) and rainfall (i, x, y) represents

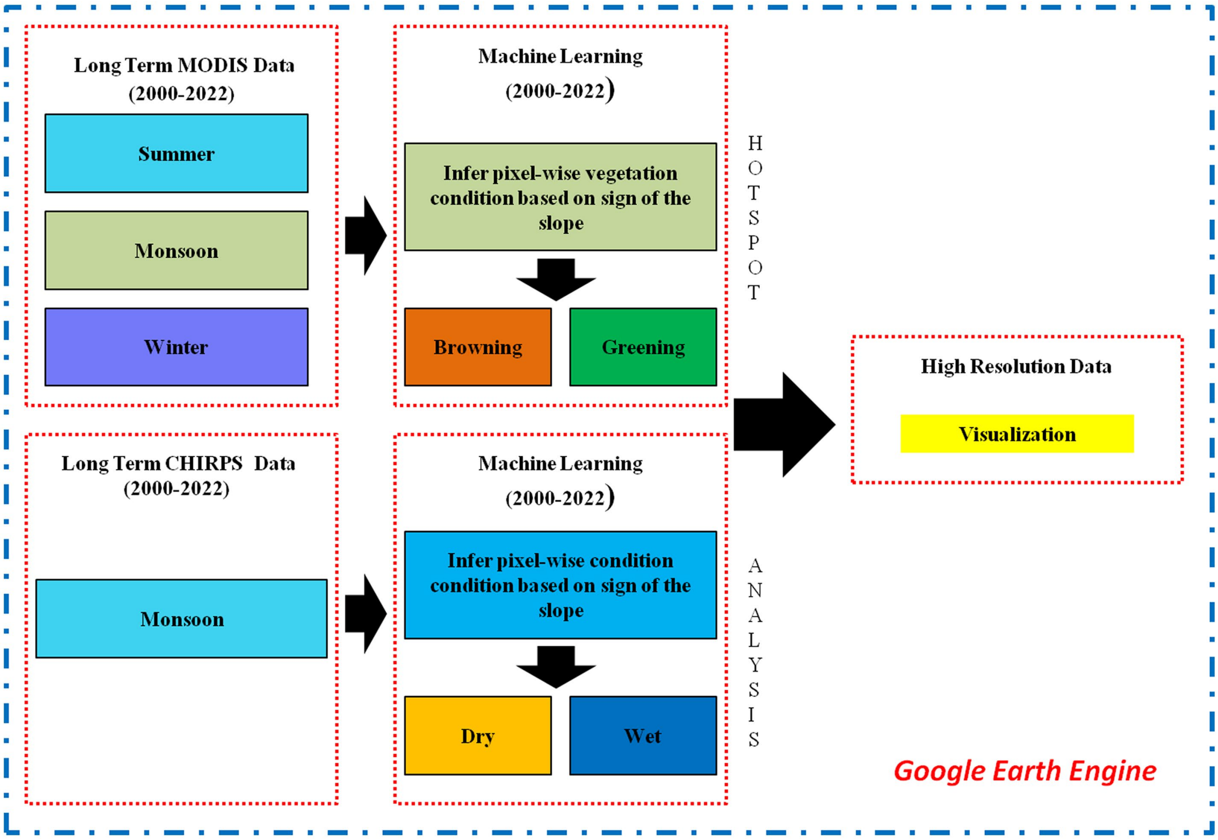

FIGURE 2
Schematic diagram of the methodology.

the annual rainfall of ith year being processed. Mean [rainfall 2000–2022 (i, x, y)] represents the long-term mean annual rainfall from 2000 to 2022. STD [rainfall 2000–2022 (i, x, y)] represents the standard deviation of annual rainfall from 2000 to 2022.

3.1.6 Visualization in high resolution

Places with high browning trend in all three seasons and high rainfall anomaly were further observed in high-resolution datasets to infer the actual cause of the vegetation degradation.

4 Results and analysis

The main approach of this study was based on Multi-Sensor, Multi-Temporal and Multi-Scale (MMM) approach in an automotive way using cloud computing system Google Earth Engine (GEE) to identify vegetation degradation easily. The distinct patterns in three seasons (summer, monsoon, and winter) clearly demonstrated the role of climate variation in the region. In Figure 3, monsoon has the most greening effect with respect to the other two seasons, summer shows maximum browning effect, and winter shows moderate browning effect in the region. It is carefully observed that mostly degraded vegetation boundaries are clearly found in extreme eastern part of the country in the Seven Sisters states, Punjab, and Haryana foot hill region and a small portion of Jharkhand and Odisha.

The bar graph in Figure 4 and Table 1 shows the browning and greening effect of summer season in India. The browning effect is most prominent in Madhya Pradesh due to the presence of large forest area in the state, which is highly affected in summer. Approximately 131271.577 km^2 is under browning in this region, whereas 177416.1116 km^2 remains green even in summer. Among Union territories, the most affected region is Delhi during summer, wherein approximately 516.049 km^2 is affected by browning.

Maximum rainfall in India is received in monsoon season during July and mid-November. The bar graph in Figure 5 and Table 1 shows the browning and greening effects of monsoon season in India. The greening effect is most prominent in monsoon season due to the availability of water in the country from rainfall. Rajasthan, the driest state of India, also shows profound greening effect on monsoon (319787.27 km^2) because this is the only season in which vegetation and crop growth in Rajasthan is abundantly observed in the area. Apart from Punjab and Haryana, the two most agriculturally dominating states show a high concentration of greening area during Monsoon. Punjab state shows 47522.15 km^2 under greening and 3550.737 km^2 as browning, whereas Haryana state shows 40634.58 km^2 under greening and 4131.61 km^2 under browning.

Winter season in India is very crucial because in most regions, it offer a chance for dual cropping, but on the other hand, deciduous forest in India suffer from browning effect and leaf shedding during this period, except for rain forests. Considering

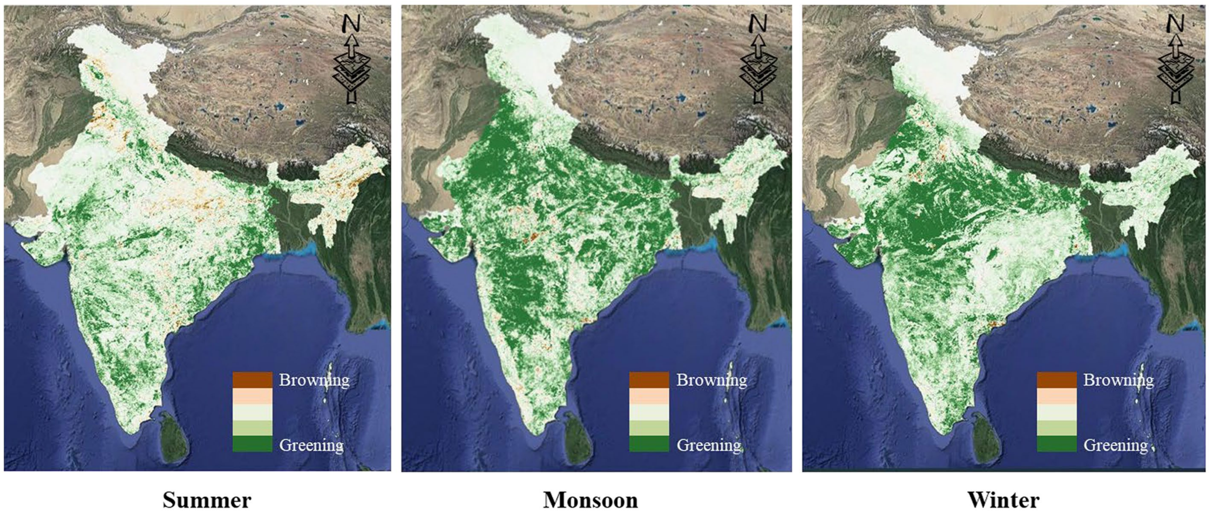

FIGURE 3
Greening and browning trend of NDVI for the time span 2000–2022.

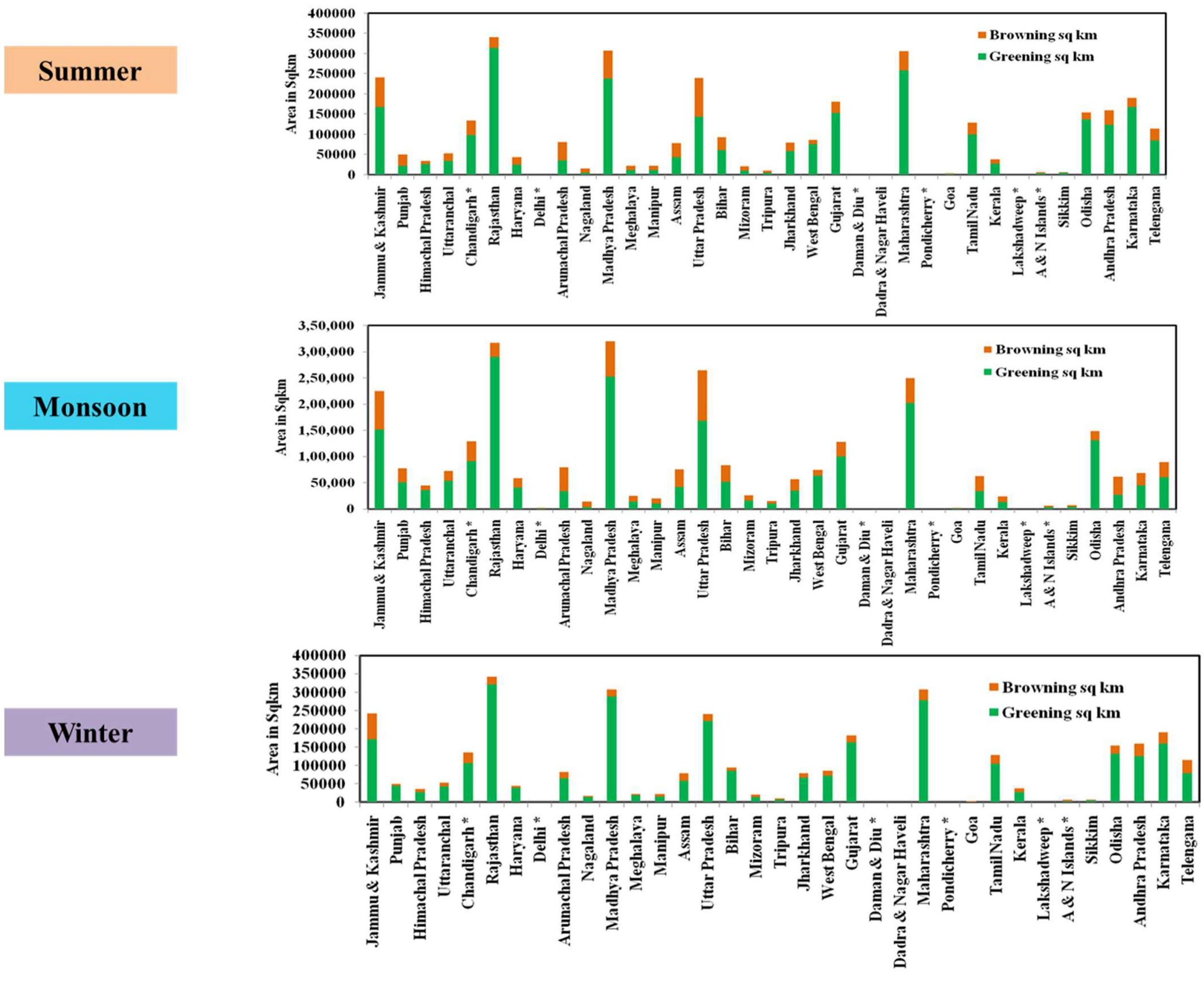

FIGURE 4
Bar Graph showing greening and browning trend of vegetation for the time span 2000–2022 for 29 states of India and 6 Union territories in summer season.

TABLE 1 Long-term dynamics of vegetation (greening and browning) for 29 states of India and 6 Union territories.

		Summer		Monsoon		Winter	
Sl no	State	Browning sq. km	Greening sq. km	Browning sq. km	Greening sq. km	Browning sq. km	Greening sq. km
1	Jammu & Kashmir	66013.486	176876.5686	35929.861	206920.8158	56466.583	186193.9775
2	Punjab	22508.028	28565.8836	3550.737	47522.15783	7344.427	43768.66949
3	Himachal Pradesh	15462.6	20567.9476	8130.016	27896.22283	7076.757	28971.59049
4	Uttaranchal	21678.594	32656.7856	9378.011	44956.74183	10298.69	44044.87849
5	Chandigarh*	60929.281	75186.5716	25315.498	110804.4798	23379.589	112688.2965
6	Rajasthan	55987.347	286978.2266	23243.451	319787.2778	32301.089	310662.8115
7	Haryana	9739.95	35029.8156	4131.619	40634.58383	6603.677	38203.26349
8	Delhi*	516.049	1650.2746	562.095	1600.501829	579.53	1626.116486
9	Arunachal Pradesh	43534.45	39209.3466	26164.26	56573.16683	19591.843	63103.60949
10	Nagaland	9994.642	7261.1916	7314.766	9936.067829	2620.257	14668.06549
11	Madhya Pradesh	131271.577	177416.1116	68656.221	240045.0338	17012.002	291667.6935
12	Meghalaya	12200.123	10902.8306	11348.144	11748.95783	4386.73	18744.75049
13	Manipur	10165.275	12797.5396	8151.392	14806.29583	5560.574	17423.17349
14	Assam	33869.441	45240.3386	29608.607	49495.31483	20016.481	59065.13449
15	Uttar Pradesh	106645.535	134672.9376	26652.777	214686.8948	29904.965	211442.9805
16	Bihar	35123.207	59593.1386	11348.569	83371.00283	9616.885	85129.45349
17	Mizoram	10409.014	11346.5716	8554.759	13196.21583	7330.816	14443.01349
18	Tripura	5570.793	5528.8246	4126.637	6968.527829	3171.655	7956.244486
19	Jharkhand	34847.713	45686.6136	9948.728	70589.14483	10201.733	70346.60549
20	West Bengal	17289.127	70204.7786	16371.478	71114.03483	16258.81	71207.99049
21	Gujarat	86634.994	95392.2306	29945.085	152090.6838	17100.133	164865.9235
22	Daman & Diu*	48.34	707.7386	47.989	703.6768286	13.586	782.6464857
23	Dadra & Nagar Haveli	341.612	814.8826	114.657	1037.602829	70.889	1125.399486
24	Maharashtra	83469.906	224397.7726	41188.338	266684.0588	24637.307	283181.5905
25	Pondicherry*	168.227	992.7776	125.209	1032.237829	190.869	1009.474486
26	Goa	1803.121	2481.2346	1217.836	3062.502829	447.244	3873.245486
27	Tamil Nadu	31548.307	98796.6416	32504.04	97837.53683	22773.762	107465.5355
28	Kerala	16631.058	22713.1226	17556.248	21780.94083	13975.62	25381.80349
29	Lakshadweep*	17.148	671.4906	13.357	671.0888286	15.678	712.8414857
30	A & N Islands*	3016.06	4160.0956	2459.623	4711.066829	1900.724	5287.202486
31	Sikkim	2803.369	4991.1926	1675.059	6115.818829	1333.003	6493.378486
32	Odisha	38156.318	117461.0556	21436.843	134180.1768	19145.658	136486.5045
33	Andhra Pradesh	48212.1	112289.9836	44853.119	115655.9258	30182.601	130308.8615
34	Karnataka	44213.755	147715.4386	34269.637	157664.8398	35351.268	156560.6435
35	Telengana	37537.912	77946.5866	18037.988	97448.74883	20946.948	94561.24749
	TGA	3,287,263		3,287,263		3,287,263	

*Union Territory.

this reason, our analysis also brings out the fact that northern part of Himalayas remains green even in winter, while the degradation in vegetation is clearly observed as we come down to lower Himalayas or southern part of the region. The dynamics of vegetation degradation is exactly opposite to that of summers. The greening of vegetation is maximum in Madhya Pradesh during winter, wherein greening and browning is 291667.69 km^2 and 17,012 km^2, respectively. Apart from Madhya Pradesh, greening is also observed more in states of Gujarat and Maharashtra.

From the bar graphs shown in Figures 4–6, it is clear that almost all the states are undergoing browning effect in India. Alarming proportions are observed in Jammu and Kashmir, Odisha, and Andhra Pradesh, apart from the North-Eastern states such as Assam, Nagaland, Mizoram, Tripura, Manipur, Arunachal Pradesh, and Meghalaya. The proportion of

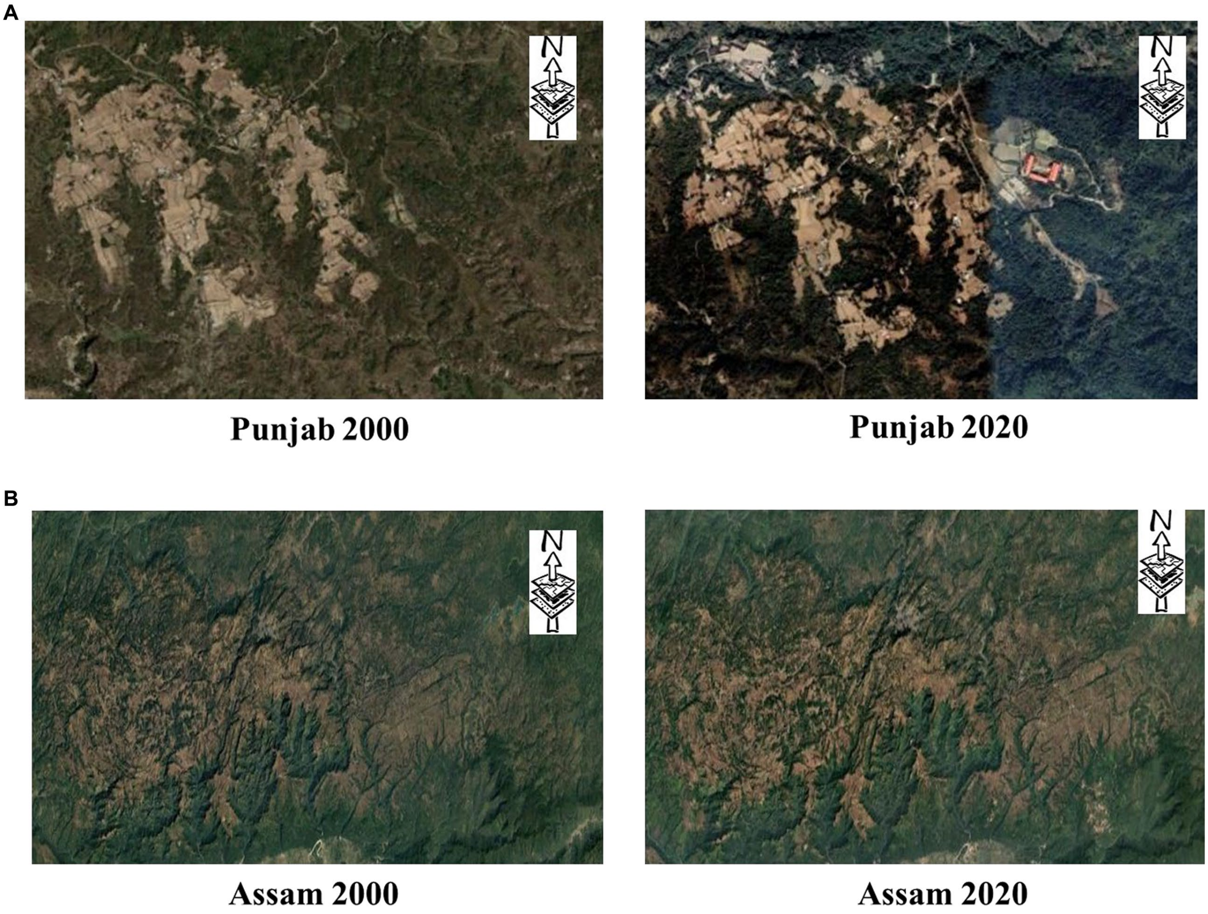

FIGURE 5
Examples of Vegetation Degradation Hotspots.

forest cover in North Eastern India and Jammu and Kashmir is high because less agricultural land is available and the terrain is composed of hard and rocky mountains. Therefore, these regions are very vulnerable and needs proper management of plans over time by central and local government bodies.

Since rainfall in India is predominantly dependent on the monsoon, the rainfall datasets of monsoon months from July to October were considered. According to Figure 5, it is clear that anomaly of rainfall is high in the regions, which significantly shows red in color during 2000–2022, indicating them as dry zone, whereas on the other hand, the region which is almost black indicates that mostly the rainfall is constant over the selected years during monsoon season. Resultantly, the rainfall variation might affect the vegetation/forest cover in India because rainfall has significant effect on soil moisture and other climatic phenomena.

After the greening and browning analysis, the vegetation degradation hotspots were clearly visible; therefore, Natural Color Composite (RGB) was built using the Landsat archive datasets to clearly visualize the changes in forest cover in GEE. Two hot spot zones, namely, (a) Punjab and (b) Assam are shown as representatives of the entire study area. Two years (2000–2022) RGB datasets of Punjab clearly demonstrate the encroachment/anthropogenic pressure leading to the degradation of vegetation cover, and Assam forest patch has been cleared for agricultural purposes.

5 Conclusion

Remote sensing-based vegetation indices and rainfall datasets for 22 years were used to investigate the response of forest cover over Indian landmass. The research revealed a complex spatial pattern of diverse vegetation responses to seasonal variation, which is expressed through long-term trends. Indian vegetation condition showed resistance to dry spells and water scarcity despite abrupt rainfall trends. This means that the productivity of vegetation has been sustained even during dry conditions in different years. Nevertheless, it was found that rainfall impacted vegetation growth to some extent: negatively in summer years (2002) and positively in monsoon years. The research presented here used a new technique, named as MMM technique, using Multi-Sensor, Multi-Temporal and Multi-Scale datasets, which helped to compute in a single platform, for a unique visualization for the first time so as to manage sustainable vegetation cover from further degradation. It is recommended to set up a network of rainfall stations, runoff measurements, phenocams, and eddy covariance towers, to quantify the impacts of climate change and assess vegetation degradation vulnerability relative to future changes/developments across India. Such a network may also help to understand the exchange of carbon and water fluxes, water use efficiency, and the impact of drought at the microlevel.

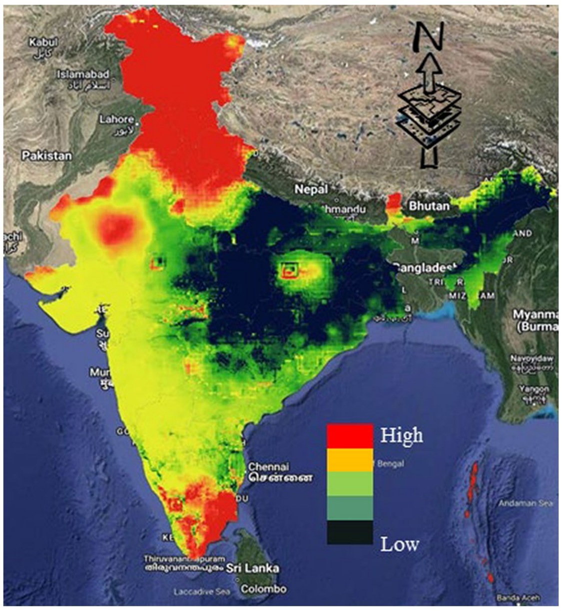

FIGURE 6
Rainfall Anomaly from 2000-2020.

Data availability statement

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author.

Author contributions

KS: Conceptualization, Data curation, Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Resources, Software, Supervision, Validation, Visualization, Writing – original draft, Writing – review & editing. VV: Data curation, Formal analysis, Funding acquisition, Methodology, Project administration, Resources, Software, Supervision, Validation, Visualization, Writing – review & editing, Writing – original draft. PP: Funding acquisition, Project administration, Resources, Supervision, Visualization, Writing – review & editing. GS: Funding acquisition, Project administration, Resources, Supervision, Visualization, Writing – review & editing. SC: Funding acquisition, Project administration, Resources, Supervision, Writing – review & editing. AN: Funding acquisition, Project administration, Resources, Supervision, Visualization, Writing – review & editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Plant distribution, ecological traits and diversity patterns of vegetation in subtropical managed forests as guidelines for forest management policy

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Forest vegetation is an important component of forest ecosystems, contributing to terrestrial plant diversity while also providing a variety of ecological services. In managed landscapes, plantations emerge as dominant kinds after stand-replacing disturbances. However, the dynamics of vegetation cover, diversity, and composition in plantation forests remains poorly understood in the subtropical region. Our study recorded a rich floral diversity with 173 angiosperm species, characterized by varying life forms and distinct flowering phenology. The uneven distribution of species across families demonstrated the complexity of the ecosystem, with Poaceae being dominant. Diversity patterns among different plantation types varied, with *Dalbergia sissoo* and *Populus nigra* plantations exhibiting higher species richness and diversity. Conversely, *Eucalyptus camaldulensis* and *Morus alba* plantations displayed lower diversity, emphasizing the influence of plantation type on biodiversity. Non-metric multidimensional scaling (nMDS) and PERMANOVA analyses revealed significant dissimilarity patterns of vegetation composition. Indicator species analysis identified unique compositions within each plantation type, emphasizing the importance of conserving specific types to protect indicator species and maintained ecological distinctiveness. Canonical Correspondence Analysis (CCA) demonstrated that road accessibility, stem cutting, and fire significantly influenced plant distribution patterns. The present research underscored the importance of considering plantation type in forest management for biodiversity conservation and highlighted the environmental variables' influence on the formation of plant communities. These results provided major implications for sustainable forest management and conservation efforts in tropical regions.

KEYWORDS

species diversity, plantations, forest management, subtropical, Pakistan

1 Introduction

Forest management as an important ecological indicator produces a considerable influence on plant species diversity (Ćosović et al., 2020; Haq et al., 2024a,b). Understanding how different forest management approaches affect plant species diversity is essential for achieving ecologically sustainable forest management (Oettel and Lapin, 2021). However, majority of the forest clearing already takes place in subtropical forests, and future deforestation is predicted to be concentrated in the tropics (Wright, 2010). To maintain biodiversity, wise and rigorous forestry management practices are urgently required to address diversity loss and the resultant degradation of ecosystem performance (Shin et al., 2022). Reforestation activities are supported all across the Earth to accommodate the expanding requirement for forest products, particularly in developing countries (Haq et al., 2023a,b). Subtropical forests thus encourage the preservation of forest cover in the region, as well as the development of ecosystem services that support local populations' livelihoods (Borma et al., 2022). Understanding vegetation dynamics and ecological aspects is crucial in managed forests, where human activity influences vegetation composition and structure (Ivanova et al., 2022).

The composition of vegetation within managed forests is influenced by a myriad of factors, ranging from climatic conditions and soil characteristics to human activities such as silvicultural practices and land management (Ameray et al., 2021). Different tree species have varying growth rates, competitive abilities, and interactions with understory vegetation, resulting in distinct ecological traits that influence ecosystem dynamics (Balandier et al., 2022). The success of these plantations is measured not only in terms of economic productivity but also in terms of their ability to maintain or enhance ecological integrity (Karr et al., 2022). For instance, fast-growing pioneer species may establish dominance in early successional stages, altering nutrient cycling and microclimate conditions (Poorter et al., 2021). On the other hand, slower-growing, shade-tolerant species may shape the later successional stages, influencing habitat complexity and biodiversity (Khoja et al., 2022; Forrester et al., 2023).

In subtropical regions, the subtleties of climatic variability, including seasonal changes in temperature and precipitation, further complicate the dynamics of managed forests (Devi et al., 2023). Consequently, different plantation types may display varying responses to these climatic fluctuations, which can subsequently influence the interactions between plantation species and native vegetation (Liu et al., 2023). Furthermore, the establishment of plantations may involve the removal of native vegetation or alteration of natural hydrological regimes, potentially leading modulations in soil characteristics, water availability, and nutrient flux (Zhou et al., 2020). Such modifications can have cascading impacts on the overarching ecosystem structure and function, necessitating a thorough understanding of how vegetation composition and ecological traits interact within these managed landscapes (Brown et al., 2023). Furthermore, the impacts of different plantation types extend beyond the vegetation composition alone. They encompass a spectrum of ecological processes that influence nutrient cycling, water availability, microclimate conditions, and habitat suitability (Piotto et al., 2021). Native species may respond differently to these alterations, potentially leading to shifts in community structure and diversity (Williams and Newbold, 2020).

The implications of these changes extend to various trophic levels, affecting wildlife habitat, pollinator interactions, and overall ecosystem resilience (Murphy et al., 2020; Theodorou, 2022). Therefore, evaluating the sustainability and multifunctionality of managed forests in subtropical temperatures necessitates a thorough examination of the complex connections between vegetation composition, ecological traits, and ecosystem processes. Remedying this scholarly gap, our investigation was undertaken. in the Chichawatni subtropical managed forests with the specific objectives (1) assessing vegetation composition, growth forms, life span, phenology, nativity, and taxonomic groups of major forest types in the region, (2) exploring conceivable variances in diversity and Spatial arrangements across these forests, (iii) how do anthropogenic factors influence the diversity patterns of vegetation composition and communities assemblage in different forest types? The results would directly affect management plans and decision-making procedures to improve cooperative effectiveness in achieving global goals for preserving biodiversity. Moreover, conservation strategies targeted at the species level played a pivotal role in enhancing the resilience and sustainability of ecosystems.

2 Materials and methods

2.1 Study area

The Chichawatni forest is situated in the southern zone of Punjab, Pakistan, encompassing a geographical range from approximately 30°-29′-32.91″N to 30°-33′-45.84″N in latitude and 72°36′00.25″E to 72°46′48.65″E in longitude (Figure 1). This forest plantation lies at an elevation that varies between 153.6 and 163.7 meters above sea level. Its establishment as a man-made forest dates back to 1913, with the initial allocation of 4,726.73 hectares officially designated as the Chichawatni Reserved Forest. As of the present day, the total area of this plantation stands at 4,666.8 hectares, with a net available planting stock area spanning 3,823.20 hectares. Originally, the Chichawatni plantation exemplified a typical dry tropical forest ecosystem, characterized by its indigenous flora, including species like *Salvadora oleoides*, *Tamarix aphylla*, *Prosopis cineraria*, and *Capparis aphylla*. Irrigation for this forest was facilitated by the Lower Bari Doab Canal, with an extensive network of water courses channeling water from canal to the forest. In its current state, the plantation primarily composed of *Morus alba*, *Eucalyptus camaldulensis*, *Populus nigra*, *Bombax ceiba*, and *Dalbergia sissoo*, both in pure stands and mixed forms. Additionally, there are naturally occurring *Neltuma juliflora* trees interspersed throughout. Other scattered species found within the plantation boundaries comprised of *Capparis aphylla, Tamarix aphylla, Ziziphus mauritiana, Prosopis cineraria, Albizia lebbeck* and *Albizia procera*.

2.2 Data collection

The comprehensive survey of the entire forest was conducted during the period spanning from 2022 to 2023. Based on a range of factors including plantation type, physiognomy, levels of disturbance, grazing pressure, stem cutting, and the age of the plantation, the study area was categorized into five distinct plantation types to analyze vegetation. A systematic sampling approach was employed to survey the vegetation within each of these designated plantation types. The investigation into vegetation composition was targeted at delineating five distinct plantation types, comprising four monoculture plantations and a singular plantation

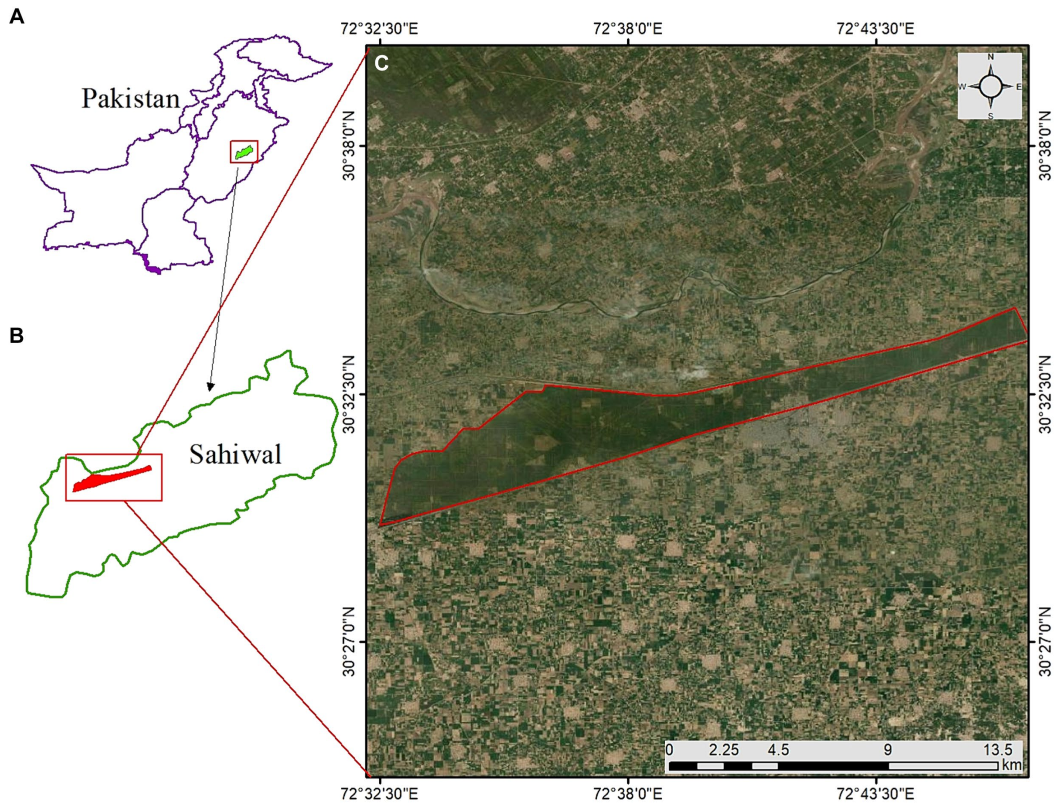

FIGURE 1
Study area map **(A)** Pakistan, **(B)** Sahiwal, **(C)** showing the location of Chichawatni forest.

characterized by a mixed assembly of tree species, denoted as the Mixed Tree Plantation (MTP). The monoculture plantations under scrutiny encompassed a diverse array of tree species, namely the *Populus nigra* plantation (PNP), *Dalbergia sissoo* Plantation (DSP), *Eucalyptus camaldulensis* plantation (ECP), and *Morus alba* plantation (MAP). Within each plantation type, 10 plots measuring 30 m × 30 m were randomly allocated based on vegetation abundance. These plots were then further subdivided into five 5 m × 5 m quadrats for the collection of phytosociological data about shrub species. In addition, for herbaceous plants, 10 quadrats of 1 m × 1 m size were systematically placed within the selected plots. Primary vegetation data, encompassing Density, Frequency, and Cover, were systematically documented for each species at every site, by established standard methodologies outlined by Mueller-Dombois and Ellenberg (1974). To accurately identify the collected plant specimens, we relied on a combination of available literature and flora, including references such as the Flora of Pakistan, e-flora of China, Flowers of India, as well as works by Nasir and Ali (1970–1985), Stewart (1972), Ali And Qaiser (1986), Qazi et al. (2023) and online resources such as Kew online.[1] To conduct a thorough assessment of the biological spectrum of plants, incorporating diverse life forms and leaf characteristics, Raunkiaer's (1934) classification methodology was adopted. By employing such systematic framework, we were able to classify and examined the wide range of life forms and leaf characteristics displayed by the plant species under investigation. Additionally, to quantify the ecological importance of each species, we computed the Importance Value Index. The index was calculated by aggregating the relative values of Relative Density, Relative Frequency, and Relative Canopy Cover, following the approach outlined by Böcher and Bentzon (1958). Furthermore, the assessment of human disturbance was visually conducted for each sample plot. Degrees of grazing, stem cutting, fire, road accessibility, and biological invasion were appraised on a four-tiered scale. These visual assessments categorized the intensities of disturbance as follows: 0 for absent, 1 for low, 2 for moderate, and 3 for strong. These categories were established following methodologies outlined in prior studies by Abbas et al. (2017) and Nowak-Olejnik et al. (2020).

2.3 Data analysis

To scrutinize significant variations in species composition between different plantation types, a robust PERMANOVA analysis was conducted, bolstered by 999 permutations. Subsequently, SIMPER was employed to pinpoint the key species responsible for discriminating between the plantation types, elucidating the drivers behind diversity differences. All these analyses were executed using PAST software (version 4.12), as previously documented (Waheed et al., 2022a,b; Haq et al., 2024a). The study systematically analyzed four key diversity metrics in the study area, included the Shannon index (H), measuring species richness and evenness, and the Simpson index (1-D) calculating species dominance. The study compared such measures to determine

1 https://powo.science.kew.org

TABLE 1 Eigenvalues and explained variation of species distribution and environmental variables on the first four canonical correspondence analysis (CCA) ordination axes.

Statistic values	Axis 1	Axis 2	Axis 3	Axis 4
Eigenvalues	0.6738	0.383	0.2599	0.2217
Explained variation (cumulative)	7.17	11.25	14.02	16.38
Pseudo-canonical correlation	0.9505	0.9213	0.8551	0.8425
Explained fitted variation (cumulative)	39.46	51.89	77.12	90.1

whether vegetation diversity was due to a few dominant species (Simpson index) or many equally abundant species (Shannon diversity). The Dominance Index and Evenness were added to the analysis to better understand community structure. Canonical Correspondence Analysis (CCA), performed with CANOCO version 5.0 software, was employed to explore the complex interplay between plantation and type of disturbance. Alongside CCA, we strategically applied the Non-Metric Multi-Dimensional Scaling (NMDS) technique to address the high dimensionality of the dataset, a commonly utilized approach for assessing plant community similarity. The NMDS and chord diagrams were created using the Origin Pro software version 10.

3 Results

3.1 Floristic composition and ecological traits

The Chichawatni Forest boasted a rich floral diversity, harboring a total of 173 angiosperm species belonging to 42 different families, as exhibited in Supplementary Table S1. However, such distribution across families was not uniform, with six families representing half of the total species count, while the remaining 37 families shared the other half. Eighteen families contained just a single species each. The Chichawatni Forest stands out as a unique ecosystem within the broader irrigated region, characterized by its fertile terrain. That particular environment offered a stable range of climatic conditions and other variables, which in turn supported a remarkable diversity of species within a relatively compact area. Among the families, Poaceae occupied the largest, with 25 species (14.4%) of the total species count, followed by the Fabaceae family (12%), Asteraceae (9%), and Amaranthaceae (7.5%) of species (Supplementary Table S1).

Of the total number of species recorded, 49% ($N=85$) were forbs, 19% ($N=33$) trees, 7% ($N=24$) trees, 14% ($N=25$) graminoids, 8% ($N=14$) shrubs, 5% ($N=9$) climbers, and 4% ($N=7$) sedges (Table 1). Most species were annuals (68%, $N=117$) followed by perennials (27%, $N=47$) and biennials (5%, $N=9$) (Supplementary Table S1). The predominant tree species inhabiting the forest were deciduous (e.g., *Dalbergia sissoo, Populus nigra, Bauhinia variegata, Morus alba, Morus nigra, Bombax ceiba, Cassia fistula,* and *Prosopis cineraria*). The evergreen trees and shrubs were also frequent including *Terminalia arjuna, Nerium oleander, Eucalyptus camaldulensis, Callistemon lanceolatus, Corymbia citriodora,* and *Salix tetrasperma,* etc. The Raunkiaer biological spectrum analysis revealed that therophytes comprised the dominant life form, accounting for 44% ($N=76$), followed by hemicryptophytes at 31% ($N=54$), Phanerophytes 12% ($N=21$), Chamaephytes 8% ($N=14$), and Geophytes 5% ($N=8$) (Figure 2). The most common leaf spectrum was microphyll (48%, $N=83$) followed by mesophyll (23%, $N=40$), nanophyll (20%, $N=35$), leptophyll (5%, $N=9$), macrophyll (9%, $N=29$) megaphyll (2%, $N=4$), and aphyllous (1%, $N=2$) (Figure 3).

3.2 Pattern of alien species

Out of the 173 species, 48 species were alien species in all forest types, the ECP (41 species), followed by the PNP (36 species), MAP (25 species), DSP (23 species), and MTF (17 species) (Figure 4). *Lantana camara*, *Parthenium hysterophorus, Neltuma glandulosa, Malvastrum coromandelianum, Abutilon theophrasti, Imperata cylindrica*, and *Datura inoxia* were discovered in five forest types examined. *Neltuma* juliflora was spread widely over the ECP and MAP. In MTP and DSP, *Ipomoea hederacea* and *Ageratum conyzoides* were widely distributed.

3.3 Pattern of flowering phenology

Plants were stratified into four clusters, while months were categorized into three groups according to their flowering phenology (Figure 5). The phenological spectrum of flora primarily encompasses the flowering duration of each species. Notably, the current investigation delineated two principal flowering periods within the flora. The primary flowering period extended from May to August in which about ($N=93$, 54%) of plant species like (*Poa annua, Euphorbia helioscopia, Achyranthes aspera, Digera muricata, Chenopodium album, Lactuca serriola, Cirsium arvense, Heliotropium strigosum, Cleome viscosa, Ipomoea carnea, Xanthium strumarium,* and *Citrullus colocynthis*) were noted during the flowering stage. The subsequent flowering phase was lasted from March–April, in which a total of (N- 58, 34%) plant species (e.g., *Cocculus pendulus, Conyza canadensis, Oxalis corniculata, Verbascum thapsus, Mentha longifolia, Galium aparine, Withania somnifera, Lantana camara,* and *Polygonum plebejum*) appeared in full bloom. A significant portion of the variability in flower timing was linked to the growth habit, deciduous nature, and size of the plant species. Additionally, certain species initiated flowering even during the winter season. Notably, species such as *Sida spinosa*, *Malva neglecta*, and *Sisymbrium irio* exhibited flowering activity in February.

3.4 Diversity pattern

During examination of various plantation types, the species richness typically in a range of 50 to 72, with 61 in the *Dalbergia sissoo* plantation (DSP) and 59 in the *Populus nigra* plantation (PNP) was observed. The Shannon diversity indices, provided insights into the

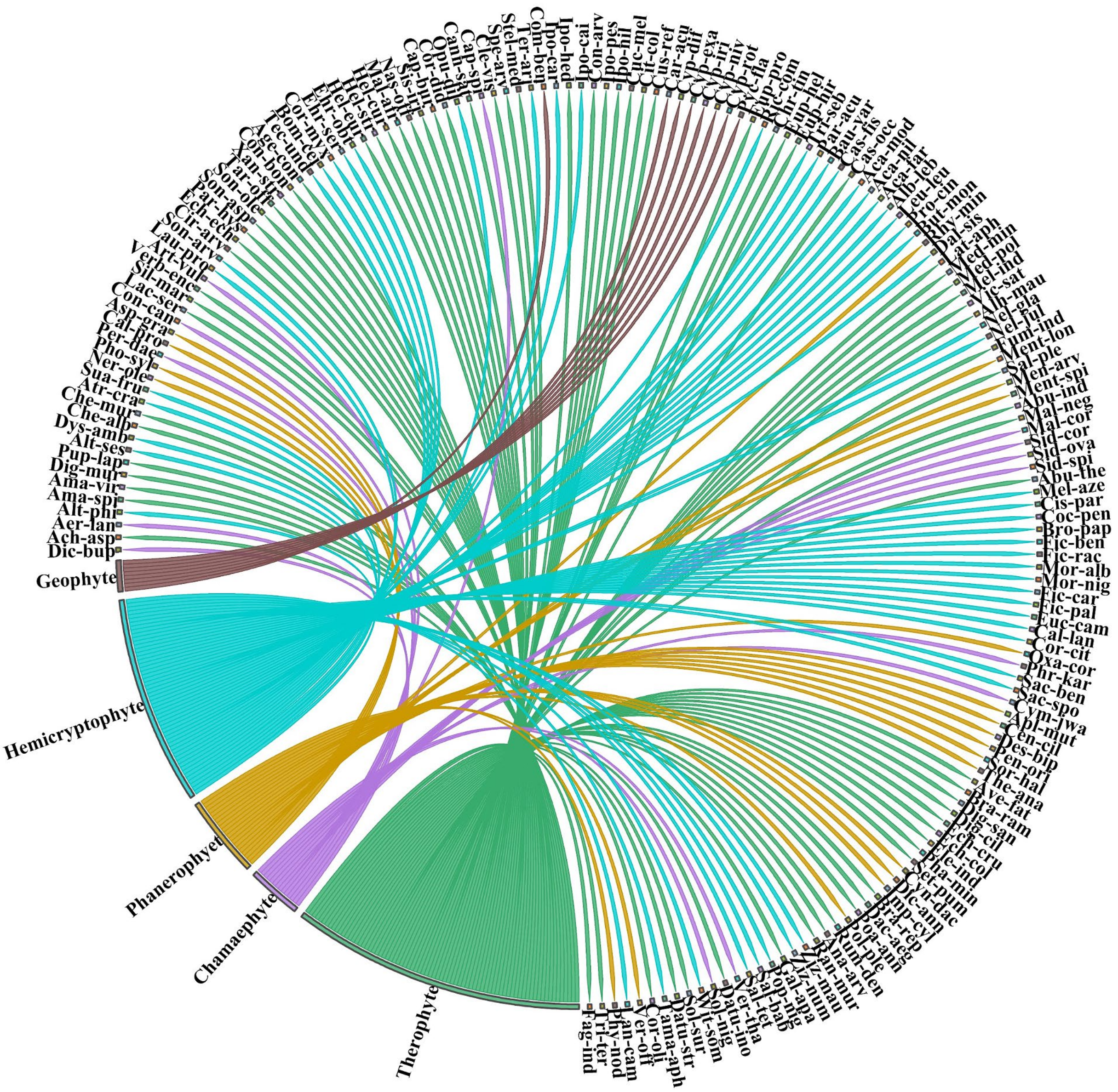

FIGURE 2
Distribution of species in Chichawatni Forest is classified using the Raunkiaer life form categorization. The direction of the lines in a graphical depiction reflects the association of each species with certain life form categories, while the thickness of each bar represents the number of species within each individual life form category. Table 1 contains a complete list of species names related to their life form types. For full species names see Supplementary Table S1.

diversity of species within these plantations, displayed significant variation, ranging from a maximum of 2.4 at the *Dalbergia sissoo* plantation (DSP) to a minimum of 0.61 at the *Morus alba* plantation (MAP). These diversity indices exhibited noteworthy differences between different plantation types, as depicted in Figure 6. The highest levels of Shannon and Simpson index values were recorded in the *Dalbergia sissoo* plantation (DSP) and *Populus nigra* plantation (PNP), suggesting a greater diversity of species in these areas. Conversely, the *Eucalyptus camaldulensis* plantation (ECP) and *Morus alba* plantation (MAP) exhibited significantly lower Shannon index values. Furthermore, when it came to Pielou's evenness, the *Dalbergia sissoo* plantation (DSP) stood out with the highest value, while the mix trees plantation (MTP) had the lowest. In terms of dominance index, it was noteworthy that the *Morus alba* plantation (MAP) and mix trees plantation (MTP) displayed the highest levels of dominance, indicating a prevalence of certain species within these plantations. Conversely, the *Populus nigra* plantation (PNP) exhibited the lowest dominance index, signifying a more balanced distribution of species. The findings underscored the varying ecological dynamics and biodiversity levels among different plantation types.

3.5 Similarities between plantation types

The nMDS ordination provided compelling evidence of distinct dissimilarity patterns among different plantation types. The analysis

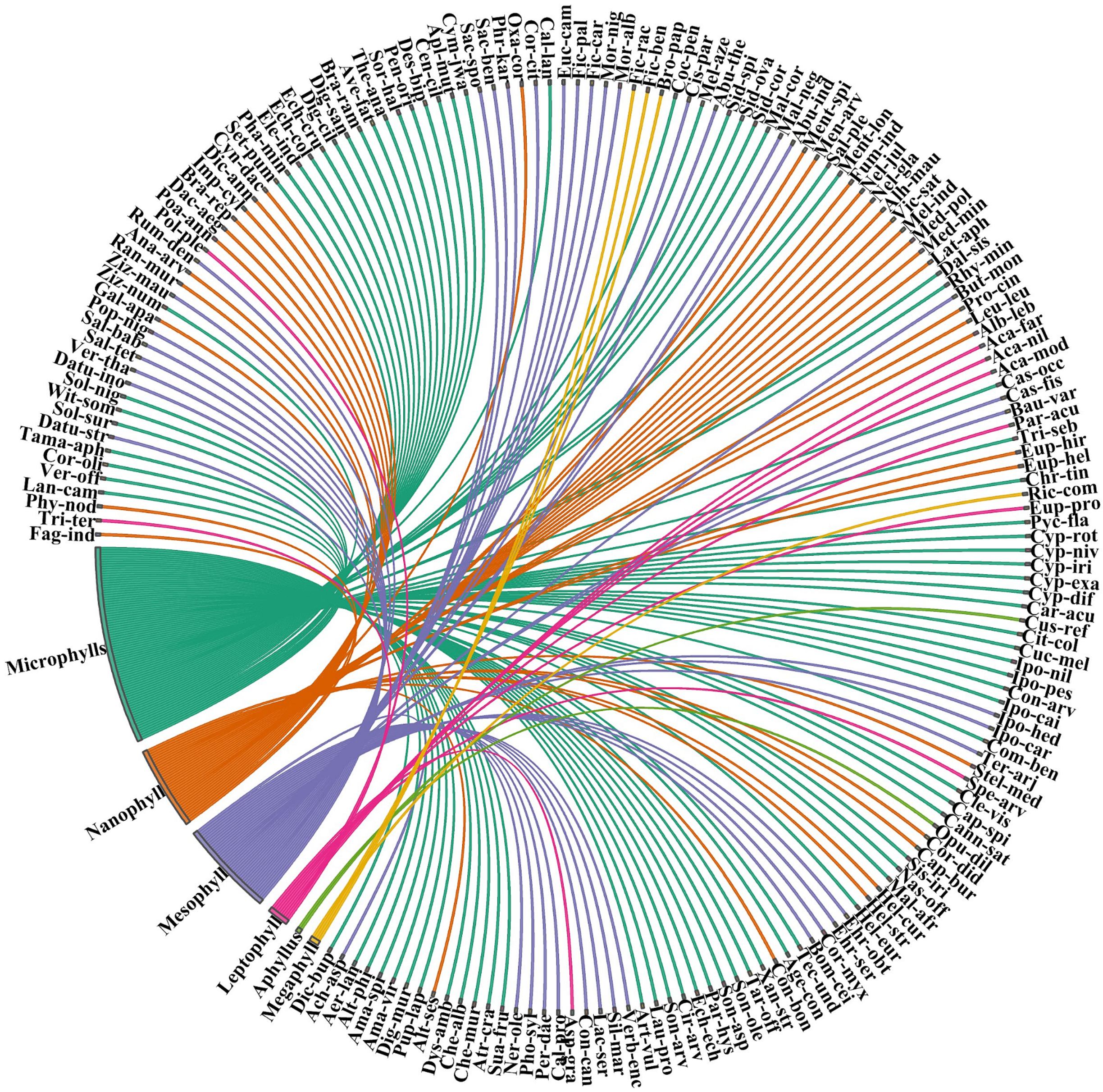

FIGURE 3
Distribution of species in Chichawatni Forest is classified using the leaf spectra categorization. The direction of the lines in a graphical depiction reflects the association of each species with certain leaf spectra categories, while the thickness of each bar represents the number of species within each individual life form category. Supplementary Table S1 contains a complete list of species names related to their leaf spectrum types. For full species names see Supplementary Table S1.

involved the examination of data collected from 173 plant species sampled across various plantation types (Figure 7). The nMDS plot revealed that groups positioned closely or along the same axis exhibited a positive correlation, while those situated far apart or on different axes displayed a negative correlation. This spatial representation of the data visually conveyed the variations in plant composition based on altitude.

Furthermore, the PERMANOVA analysis yielded significant results, indicating marked differences in plant composition between sites within the *Populus nigra* plantation (PNP) and *Eucalyptus camaldulensis* plantation (ECP) in comparison to the *Morus alba* plantation (MAP) and mix trees plantation (MTP). Bray–Curtis dissimilarities were the main factor influencing the observed differences. Additionally, the study highlighted substantial differences in species composition across the different kinds of plantations, as shown by significant findings for the Jaccard ($F = 17.59$, $p < 0.01$) and Bray–Curtis ($F = 27.74$; $p < 0.01$) indexes. These outcomes emphasized the influence of plantation type on species richness and relative abundances, contributing significantly to variations in species composition among the plantation types under investigation. The study unveiled a substantial divergence in plant species composition among the PNP, ECP, DSP, MAP, and MTP, with all five plantation types displaying noteworthy distinctions in their species makeup. Out of the 173 recorded species, 86 were recognized as noteworthy contributors to the differences in plant composition seen in the various forest plantations.

Remarkably, the key species collectively accounted for a substantial 75% of the overall contribution, highlighting their pivotal contribution to the formation of plant communities at varying plantation types. The close association of the *Populus nigra* plantation (PNP) and *Dalbergia sissoo* plantation (DSP) groups in the nMDS plot suggests a high degree of species composition similarity between these two types of plantations. This spatial clustering suggests that the ecological characteristics and environmental conditions of DSP and PNP foster similar species assemblages.

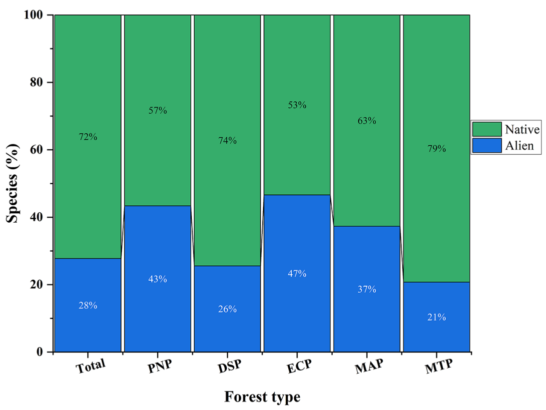

FIGURE 4
Status of native and alien species in different forest types of the study area.

3.6 Indicator species

Through a detailed SIMPER analysis, the investigation unveiled that a specific set of 86 plant species played a pivotal role in accounting for a significant 75% of the dissimilarity observed among the five plantation types under study, namely the *Populus nigra* plantation (PNP), *Dalbergia sissoo* plantation (DSP), *Morus alba* plantation (MAP), *Eucalyptus camaldulensis* plantation (ECP), and mix trees plantation (MTP). To further elucidate the distinctiveness of each plantation type, an indicator species approach was applied, which led to the identification of 24 indicator species distributed across these plantations.

The results from the indicator species analysis painted a clear picture of the unique compositions within each plantation types. In the *Populus nigra* plantation (PNP), the significant indicator species

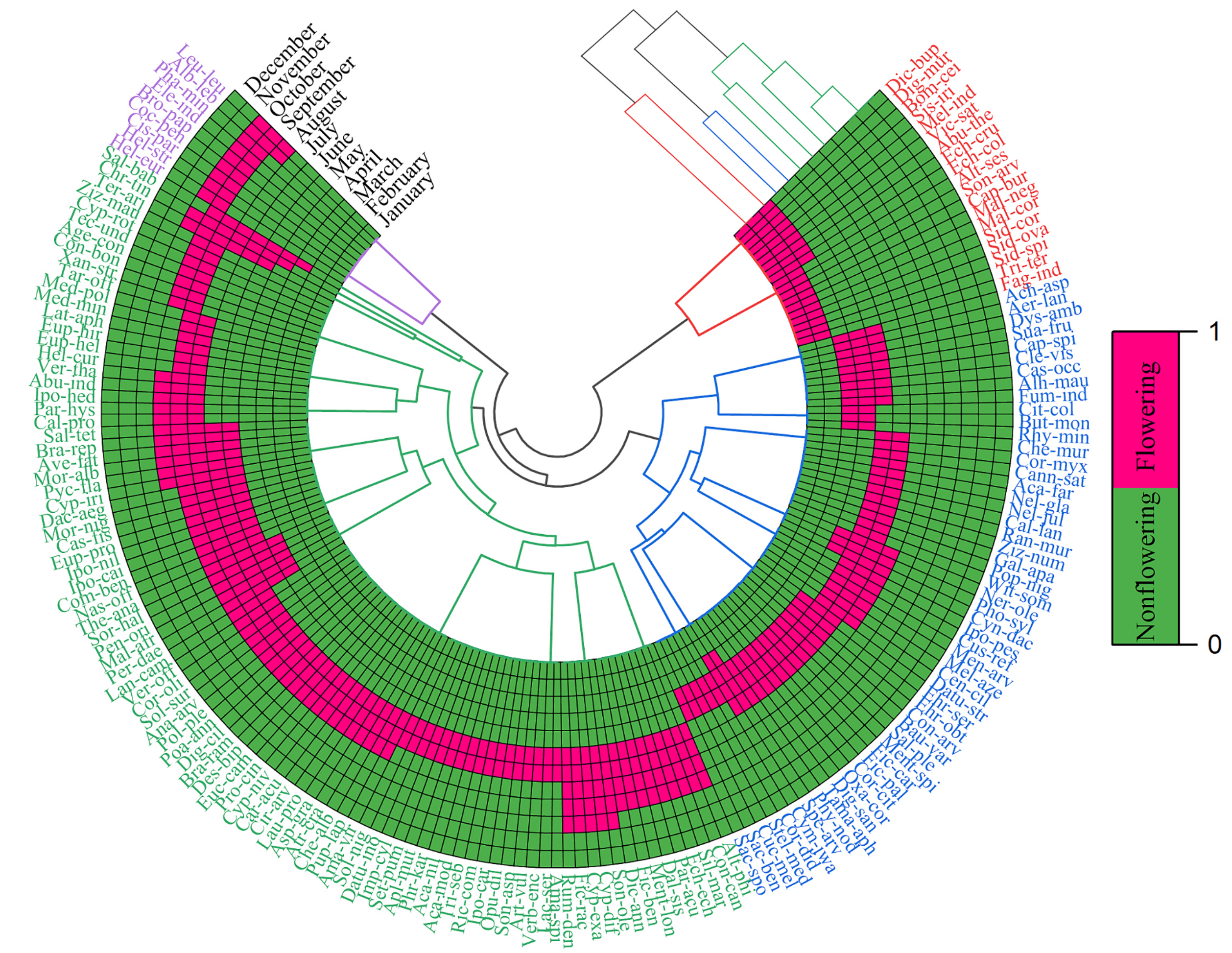

FIGURE 5
Polar cluster heat map showing the species and moth blue color indicates non-flowering and yellow for flowing stage. For full species names see Supplementary Table S1.

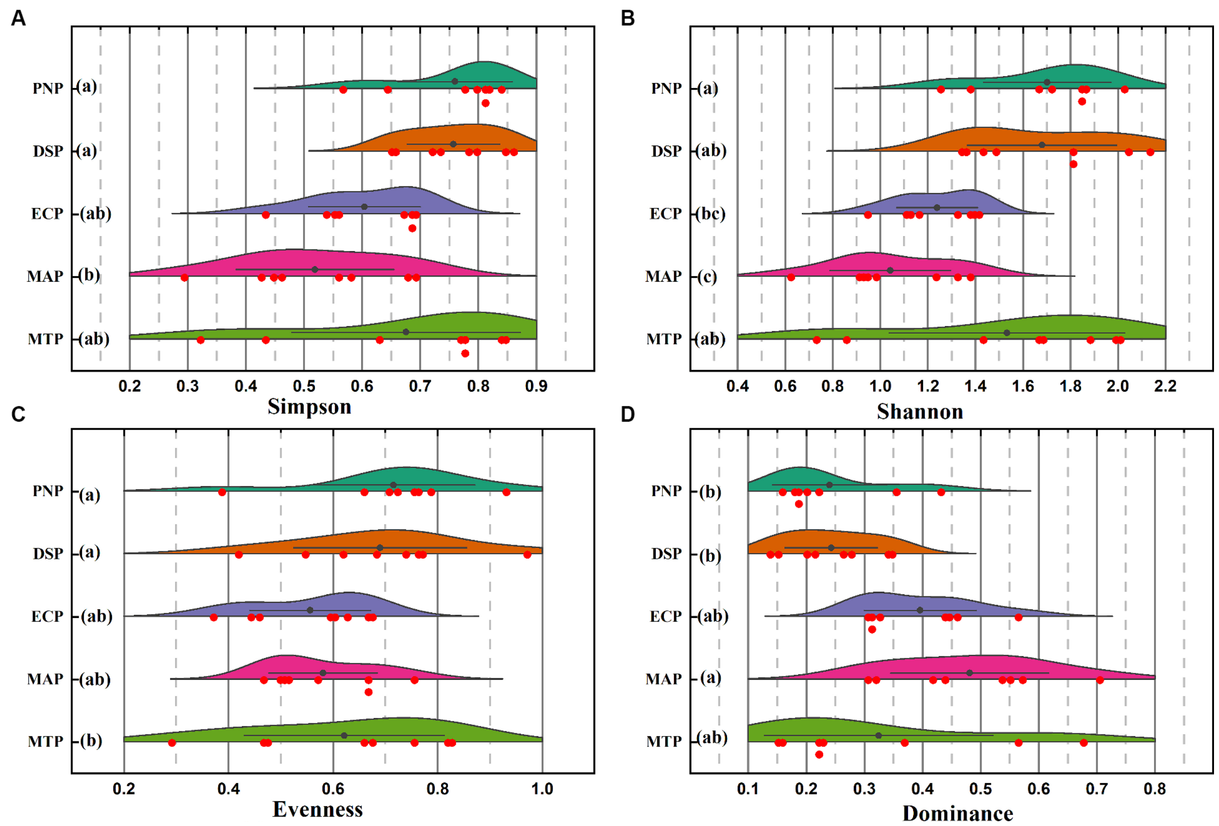

FIGURE 6
Diversity profiles of the plantation type within the Chichawatni Forest in Pakistan are illustrated in the ridgeline diagram. In this diagram, the sampling sites are represented, and lower-case letters (a, b) are used to denote significant differences between various plantation types, as determined by the Tukey test. **(A)** Simpson index, **(B)** Shannon index, **(C)** Evenness, and **(D)** Dominance Index.

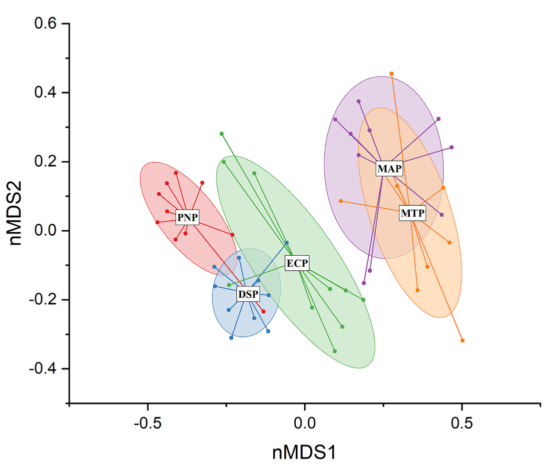

FIGURE 7
Representation of sampling sites at the alpine region, concerning the plantation types [*Populus nigra* plantation (PNP), *Dalbergia sissoo* plantation (DSP), *Morus alba* plantation (MAP), *Eucalyptus camaldulensis* plantation (ECP), and mix trees plantation (MTP)], was achieved using non-metric multidimensional scaling (NMDS). The 95% confidence intervals are displayed through ellipses.

included *Dicliptera bupleuroides, Xanthium strumarium, Bombax ceiba, Capsella bursa-pastoris, Cuscuta reflexa,* and *Ziziphus nummularia*, each displaying a noteworthy indicator value ($p \leq 0.05$).

Meanwhile, in *Dalbergia sissoo* plantation (DSP), the indicator species were comprised of *Acacia farnesiana, Chenopodium album, Broussonetia papyrifera*, and *Xanthium strumarium*, all marked by significant *p*-values. In the *Eucalyptus camaldulensis* plantation (ECP), the indicator plants emerged were *Acacia nilotica, Ehretia serrata, Cleome viscosa, Euphorbia helioscopia, Melilotus indica*, and *Neltuma glandulosa*, once again featuring significant indicator values. In *Morus alba* plantation (MAP), the significant indicator species were *Cannabis sativa, Cirsium arvense, Lantana camara*, and *Saccharum bengalense* while in mix trees plantation (MTP), the species possessing significant indicator values were *Neltuma juliflora, Euphorbia hirta, Parthenium hysterophorus*, and *Ehretia serrata* (Figure 8). It's noteworthy that *Xanthium strumarium* emerged as a common indicator species in *Populus nigra* plantation (PNP) and *Dalbergia sissoo* plantation (DSP), while *Lantana camara* is was common indicator species in *Morus alba* plantation (MAP) and *Eucalyptus camaldulensis* plantation (ECP).

3.7 Impact of anthropogenic disturbance on vegetation

Collectively, the first and second axes in the Canonical Correspondence Analysis (CCA) jointly elucidated 18.42% of the total variation in the data, and impressively, they accounted for a substantial 91.35% of the fitted variation. More specifically, CCA axis 1 unveiled 7.17% of the total variation, corresponding to 39.46% of the fitted variance, while axis 2 contributed 11.25% of

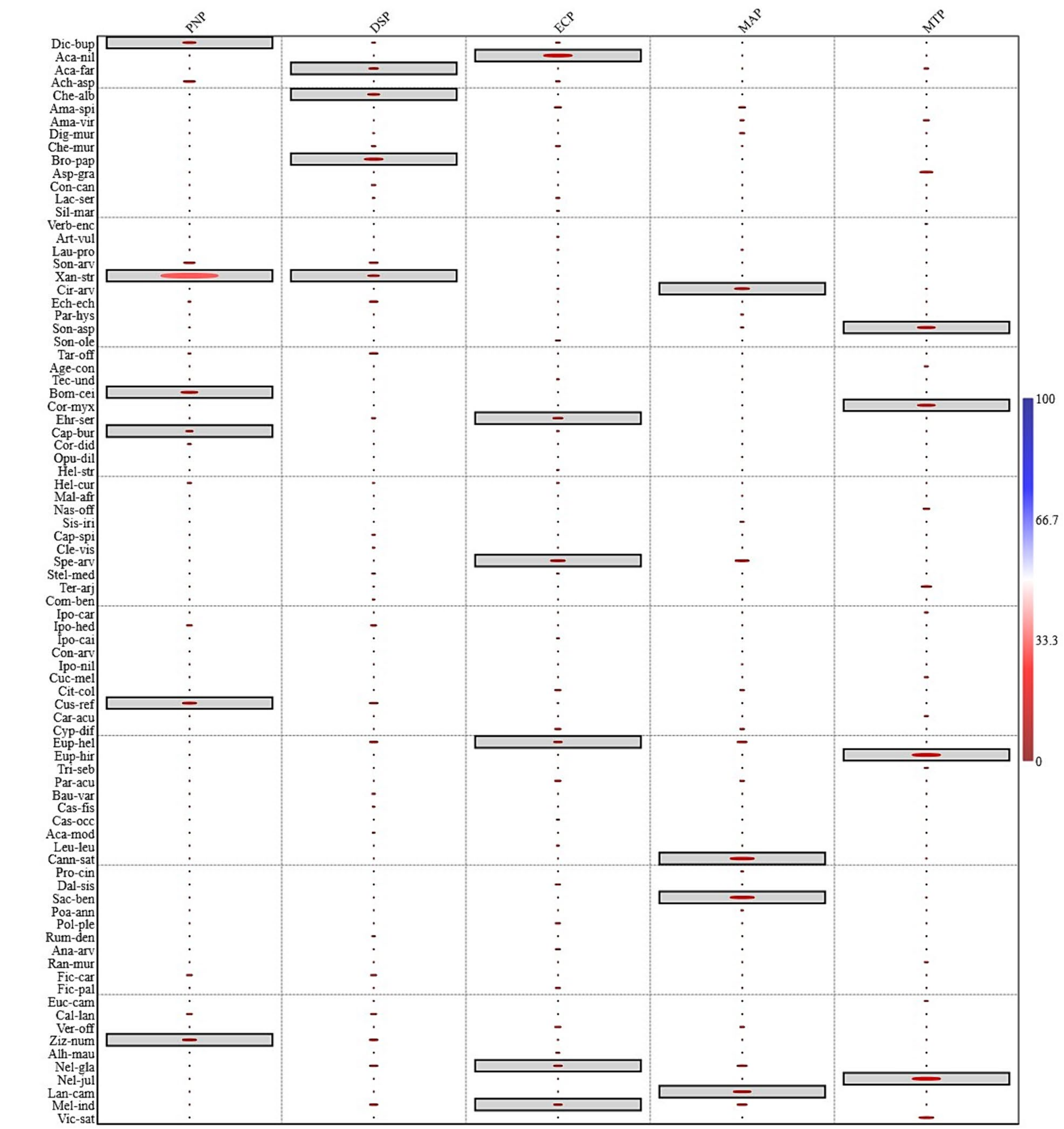

FIGURE 8
Assessment of indicator species among plants from different plantation types in the Chichawatni Forest to enlighten the noteworthy species ($p \leq 0.05$) specific to each plantation types. These significant species were identified and specially marked within their respective boxes. For full species names see Supplementary Table S1.

the total variation, capturing 51.89% of the fitted variance (as detailed in Table 1). The impact of various parameters on the composition of plant species was found to be highly significant ($p < 0.01$). The CCA results underscored that certain factors, notably road accessibility, stem cutting, and fire, played pivotal roles in explaining the variance observed in plant distribution patterns ($p < 0.05$), while factors such as biological invasion and grazing contributed less to this observed variation. Notably, the percentage contribution was statistically significant for only three parameters in elucidating the variance in vegetation patterns ($p < 0.05$). Road accessibility emerged as the most influential, accounting for 34.9% of the observed variation, followed by stem cutting at 20.5%, and fire at 18% among these key factors (as presented in Table 2). Road accessibility, stem cutting, and fire emerged as the dominant drivers behind the observed variation in vegetation patterns, shedding light on their crucial roles in shaping the plant composition and distribution within the studied plantations (Figure 9).

TABLE 2 Effects of variables on plantation types.

Name	Explains %	Contribution %	pseudo-*F*	*P*	*P(adj)*
Road accessibility	6.3	34.9	3.3	0.002	0.01
Stem cutting	3.7	20.5	2	0.002	0.01
Fire	3.3	18	1.7	0.002	0.01
Biological invasion	2.7	14.8	1.4	0.012	0.06
Grazing	2.1	11.8	1.2	0.21	1

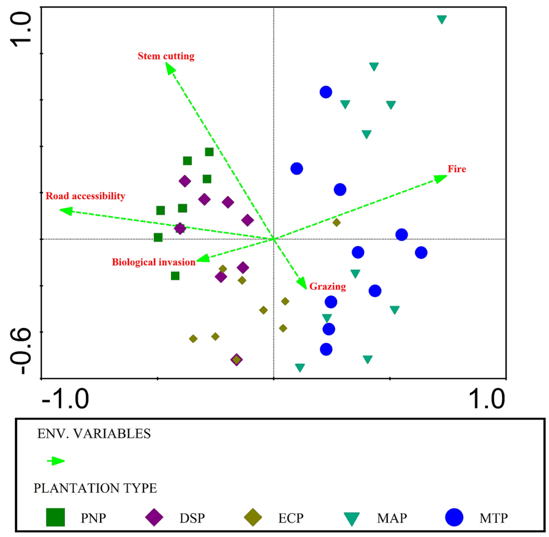

FIGURE 9
Species ordination using CCA (canonical correspondence analysis) and five disturbance variables. The direction and length of the arrow reflect the link between the factor and the ordination axes, while the various shapes and color symbolizes the plantation types.

4 Discussion

Vegetation structural complexity as an essential ecological characteristic referred to the three-dimensional distribution of plants within an ecosystem (LaRue et al., 2023). One certainty in the modern anthropogenic epoch biological communities exhibited dynamic instability and underwent fluctuations over both short and long temporal scales (Smith et al., 2009). Predicting the effects of ongoing habitat and biodiversity declined on ecosystem functions and services required an understanding of the role diversity plays in ecosystem stability (Dolezal et al., 2020). One of the primary objectives of the study was to assess the biodiversity and ecological traits of plant species within the Chichawatni forest plantation. The results revealed a remarkable diversity of angiosperm species, with 173 species belonging to 42 different families. This high diversity is a testament to the unique ecological conditions prevailing in the subtropical region of the India–Pakistan subcontinent. The dominance of the Poaceae family, with 25 species, is noteworthy. This family's prevalence is indicative of the ecological importance of grasses in subtropical ecosystems, which often play vital roles in soil stabilization, nutrient cycling, and as habitat for various wildlife (Altaf et al., 2022; Waheed et al., 2022a,b, 2023a; de Oliveira et al., 2023; Joshi et al., 2023). Other prominent families, such as Fabaceae, Asteraceae, and Amaranthaceae, also contributed significantly to the overall species count. Comparable findings were documented by Arshad et al. (2023) and Waheed et al. (2023a,b) from the subtropical region of Pakistan. Fabaceae was recorded as the dominant family by Haq et al. (2023a,b) from Shawilks Mountain Range, Western Himalayas, while Mehraj et al. (2018) recorded Asteraceae as the dominant family from Himalayas. The research insights of Haq et al. (2019, 2021) demonstrated the remarkable adaptive capacity of the Asteraceae family across various regions and climates. The abundance of graminoids, forbs, and deciduous tree species, including *Dalbergia sissoo* and *Morus alba*, indicated the adaptability of these plants to the region's distinct climate, characterized by seasonal changes in temperature and precipitation. This diversity mirrored the results of studies in other subtropical forests such as the Western Ghats in India (Subashree et al., 2021) and the Kabal valley, Swat, Pakistan (Ilyas et al., 2018). However, while the number of species is impressive, the uneven distribution across families is a common trend in forest ecosystems globally (Singh et al., 2020). The prevalence of the Poaceae family is consistent with its ubiquitous presence in many forest ecosystems (Behera and Misra, 2006; Joshi et al., 2021).

The presence of therophytes as the dominant life form aligns with the region's semi-arid climate, where these annual plants have evolved strategies to complete their life cycles within the limited wet period (Moro et al., 2015; Rashid et al., 2021; Manan et al., 2022). Hemicryptophytes, with their capacity to persist through unfavorable conditions, also exhibited a substantial role. These life forms reflected the resilience of plant species in subtropical climates, where they must cope with both dry periods and monsoon rains (Pfadenhauer and Klötzli, 2020). The prevalence of microphyll and mesophyll leaf spectrum types revealed the need for efficient water use in subtropical climates, where water availability could be a limiting factor (Amjad et al., 2017; Rahman et al., 2021). Understanding the prevalence of these ecological traits is crucial for predicting how plant communities may respond to future climate fluctuations in the region (Alexander et al., 2018). These traits are crucial for the survival of plant species during extended dry spells and highlighted the adaptability of the Chichawatni forest flora. The identification of two major flowering periods in the Chichawatni Forest is a noteworthy finding, as it showcases the adaptability of plant species to seasonal changes. Such phenological variations are consistent with the findings of studies in other subtropical and temperate forests, where species exhibited distinct flowering periods in response to local climatic conditions (Pei et al., 2015; Du et al., 2019; Song et al., 2021). However, the study also reported flowering during the winter months, a phenomenon observed less frequently in subtropical forests but documented in temperate ecosystems (Negi et al., 2022). This suggests that the Chichawatni Forest may be subjected to unique climatic influences, further highlighting the subtropical region's ecological complexity.

The study indicated the variations in plant richness and diversity metrics across various plantation types within the forest. Such variations in diversity are common in managed forests globally (Goded et al., 2019; Poudyal et al., 2019). The higher diversity observed in *Dalbergia sissoo* and *Populus nigra* plantations were aligned with findings in various plantation studies, where the species-rich plantations tended to support a more diverse range of understory species (Ramovs and Roberts, 2003; Bremer and Farley, 2010). Conversely, *Eucalyptus camaldulensis* and *Morus alba* plantations exhibited lower diversity, a pattern often associated with *Eucalyptus camaldulensis* plantations (Alem and Pavlis, 2012; Bekele and Abebe, 2018). The findings underscored the importance of considering plantation type in forest management decisions to promote biodiversity (Liu et al., 2019; Blondeel et al., 2021). Such finding implied that particular tree species exerted a more favorable impact on the diversity of the understory vegetation. The outcomes aligned with prior research had shown the importance of tree species selection in plantation design (Barlow et al., 2007). The significance of plantation type was extended beyond species richness and diversity. The study also unraveled varying levels of evenness and dominance within different plantation types. The *Dalbergia sissoo* plantation displayed the highest evenness, indicating a more balanced distribution of species, while the mixed trees plantation had the lowest evenness, suggesting that certain species dominated the plantation. Understanding such patterns of evenness and dominance could have practical implications for forest managers. For instance, enhancing evenness within plantations could contribute more stable and resilient ecosystems, less susceptible to the dominance of a few species (Sanderson et al., 2004). The findings had direct implications for forest management strategies in subtropical regions like India and Pakistan. Sustainable forestry practices should consider not only the economic value of timber but also the ecological consequences of tree species selection. The positive relationship between certain plantation types and higher biodiversity suggested that careful species selection could promote both economic goals and the preservation of native biodiversity.

The nMDS ordination results offer compelling evidence of distinct dissimilarity patterns among different plantation types within the Chichawatni forest. The analysis involved data collected from 173 plant species sampled across various plantation types, namely the *Populus nigra* plantation (PNP), *Dalbergia sissoo* plantation (DSP), *Morus alba* plantation (MAP), *Eucalyptus camaldulensis* plantation (ECP), and mix trees plantation (MTP). The nMDS plot visually represented these dissimilarity patterns by placing groups of plant communities closer or farther apart based on their similarity in species composition (Clarke et al., 2006). Conversely, groups situated far apart or on different axes display a negative correlation, indicating greater dissimilarity in species composition. The dissimilarity patterns revealed by the nMDS analysis were not just abstract representations but carried significant ecological implications (Legendre and Anderson, 1999). The patterns highlighted the varying ecological dynamics and species compositions among different plantation types, which were influenced by factors such as tree species selection, silvicultural practices, and disturbance regimes (Paillet et al., 2010). For example, the proximity of the *Dalbergia sissoo* plantation (DSP) and *Populus nigra* plantation (PNP) groups in the nMDS plot suggested a higher degree of similarity in species composition between these two plantation types. Such observation aligned with the previously discussed findings of higher species richness and diversity indices in such plantations (Wang et al., 2019). It indicated that certain species within these plantations might share ecological niches or exhibited similar responses to environmental conditions, leading to a more homogeneous plant community (Hamann and Wang, 2006). Conversely, the *Morus alba* plantation (MAP) and mix trees plantation (MTP) groups appeared farther apart in the nMDS plot, indicating greater dissimilarity in species composition. Such divergence revealed the distinct ecological dynamics within the plantation types. Factors such as tree growth patterns, competitive interactions, and responses to disturbances might contribute to the dissimilarities (Lloret et al., 2011). Understanding differences is crucial for tailoring management strategies that promote both economic objectives and ecological health (Moldan et al., 2012). The PERMANOVA analysis showed significant plant composition differences between plantation types. Notable dissimilarities were identified between the *Populus nigra* plantation (PNP) and the *Eucalyptus camaldulensis* plantation (ECP) compared to the *Morus alba plantation* (MAP) and the mixed tree plantation (MTP). This emphasizes the crucial significance of individual species in generating different plant communities across various plantation types (Waheed et al., 2020, 2022b; Haq et al., 2023a).

The indicator species identified played critical ecological roles within their respective plantation types in the subtropical forest. The reported species possessed specific ecological traits that made them well-suited to their environments, and understanding their traits enhanced our comprehension of the broader ecosystem dynamics (Dufrêne and Legendre, 1997). For instance, in the *Populus nigra* plantation, species like *Dicliptera bupleuroides* and *Xanthium strumarium*, characterized by high indicator values, are often associated with open habitats and pioneer vegetation. Those might serve as early colonizers, contributing to the initial stages of ecological succession (McLane et al., 2012). *Bombax ceiba*, a prominent indicator species, was a deciduous tree known for its ability to tolerate seasonal water fluctuations, potentially influencing the hydrological regime of the plantation (Maiti et al., 2021). *Capsella bursa-pastoris*, a common indicator, as an annual herb thrived in disturbed areas, reflecting the ecological conditions within the *Populus nigra* plantation (Petersen et al., 2021). In the *Dalbergia sissoo* plantation (DSP), indicator species like *Acacia farnesiana* and *Chenopodium album* were often associated with arid and semi-arid regions (Waheed et al., 2023a,b), reflecting the specific moisture and soil conditions of the DSP. *Broussonetia papyrifera*, another DSP indicator, was valued for its role in stabilizing soils and providing shade, highlighting its importance in shaping the microenvironment (Agyeman et al., 2016). *Xanthium strumarium*, shared with the PNP, was an adaptive species with broad ecological tolerance, indicative of its capacity to thrive in different environments (Ullah et al., 2022). In the *Eucalyptus camaldulensis* plantation (ECP), species like *Acacia nilotica* and *Lantana camara*, identified as indicators, often found in disturbed areas and were known for their allelopathic effects, which could influence plant interactions (Zhou et al., 2019).

The species might be significantly involved in shaping the competitive dynamics and understory composition. *Neltuma juliflora*, another indicator, was adapted to arid environments, reflecting the specific ecological conditions of the ECP. Understanding these ecological traits of indicator species provides insights into the interactions and processes shaping the *Eucalyptus camaldulensis* plantation (ECP) (Dondofema et al., 2023). In the *Morus alba* plantation (MAP) and mix trees plantation (MTP), indicator species *viz.*; *Cannabis sativa* and *Cirsium arvense* were often associated with ruderal habitats and might thrive in areas subjected to disturbances like fire and grazing (Vakhlamova et al., 2022). *Lantana camara*, shared with the ECP, was known for its invasive potential, influencing plant community composition. Understanding these ecological roles is essential for managing the dynamics of these plantation types and addressing issues related to disturbance and invasiveness (Meyer et al., 2021; Haq et al., 2022; Rai, 2022).

The Canonical Correspondence Analysis (CCA) elucidates the significant influence of factors like road accessibility, stem cutting, and fire on plant distribution patterns within various plantation types, aligned with a broad body of literature documenting the impacts of disturbance on vegetation and species composition in managed and natural forest ecosystems (Haq et al., 2019; Kutnar et al., 2019; Alam et al., 2023). Understanding such impacts was vital for informed forest management and conservation strategies (Haq et al., 2021). The impact of anthropogenic disturbances, such as road accessibility, stem cutting, and fire, on plant distribution patterns reaffirmed their significance in shaping forest ecosystems. Those findings held significant implications for forest management and conservation strategies in subtropical regions, emphasizing the necessity for sustainable practices aimed at alleviating the adverse effects of disturbances while promoting biodiversity. Future research in the Chichawatni Forest and similar subtropical ecosystems should focus on long-term monitoring to assess the resilience of plant communities for ongoing environmental changes and disturbances. Road accessibility emerged as a significant driver of vegetation change, with road construction fragmenting habitats and potentially facilitating the spread of invasive plants (Ramprasad et al., 2020). Stem cutting, often associated with resource extraction, had discernible impacts on specific sites within the plantation, highlighting the need for sustainable harvesting practices (Vogel et al., 2021). Fire, another disturbance factor, can disrupt ecosystem processes and negatively affect plant communities (Miller and Safford, 2020). These findings underscore the importance of managing those disturbances carefully to maintain the ecological integrity of managed forest ecosystems.

The Chichawatni forest plantation serves as a microcosm of subtropical managed forests in the broader India-Pakistan region. These forests are invaluable not only for timber production but also for their role in regulating water resources, conserving biodiversity, and providing ecosystem services to local communities (Ghafoor et al., 2022). As such, the results of the present investigation inferred regional significance. Such findings regionally provided valuable insights into how plantation design, ecological traits, and disturbances intersect in subtropical ecosystems. The study's results had implications for conservation and forest management in the subtropical region. Forest managers could use the insights gained to inform tree species selection and plantation design, aiming for a balance between economic productivity and ecological health. Conservation efforts could be significant from a better understanding of how subtropical ecosystems respond to human activities and climate change, facilitating the preservation of these vital ecosystems (Haq et al., 2024a,b). The research provided a comprehensive understanding of the intricate relationships between biodiversity, ecological traits, plantation types, and disturbances in a subtropical managed forest in the region. The findings contributed to our knowledge of subtropical ecosystems and offered practical guidance for sustainable forest management and conservation in the ecologically diverse and economically significant area.

4.1 Implications for forest management

The Forest ecosystem management is hailed as the answer to conservation issues. Ecologists are inferring that natural reserves alone would not be sufficient to effectively preserve biological diversity. Because of the current reserve network's limitations, the likelihood of a significant extension, and movement constraints, species found in reserves are particularly sensitive to the effects of climate change. Thus, present studies devised a mechanism of supplementing the natural forest reserves with a managed forest matrix that utilized ecological principles to manage for both commodity production and species variety conservation. Forty-eight species as alien species from various forest types, and the maximum (41 species) in the *Eucalyptus camaldulensis* plantation, and minimum of (17 alien species) in the Mix trees plantation were reported. Moreover, alien species posed a significant challenge to vegetation management due to their invasive nature. Hence, considering the ecological traits of species within each forest type was imperative. The current investigation brought attention to the phenomenon observed in *Populus* and *Eucalyptus* forest stands characterized by a prevalence of alien species. The findings highlighted that the introduction of exotic plantations significantly facilitated the expansion of those alien species when contrasted with their native counterparts. Thus, it was vehemently opposed the introduction of non-native species into the managed forests, and further suggested that the natural forest should be used as reference system to offer guidance on the plantation in the management forests that would be dedicated to both wood production and conservation of biodiversity.

Present findings indicated that native *Dalbergia sissoo* plantations and mixed tree plantations harbor greater richness in vascular plant species, particularly featuring a higher proportion of native taxa compared to exotic tree plantations. That highlighted the distinct effects of exotic tree plantations on understory plant communities and biodiversity. The study underscored the importance of selecting native tree species for plantation in biodiversity preservation efforts, which had often been overlooked in large-scale forest restoration projects worldwide. Given the increasing prevalence of exotic plantations in Pakistan and other regions, urgent modifications to reforestation management practices were needed to enhance stand structure, understory vegetation, and biodiversity conservation. The present analysis recommended prioritizing native or mixed tree plantation strategies for clear-cut areas in Pakistan subtropical region to better preserve indigenous plant diversity in managed forests. Finally, it was suggested that the lessons learnt from the current study would be useful for design and effectively manage such multipurpose landscape.

5 Conclusion

The present study provides valuable insights into the intricate relationships between vegetation composition, ecological traits, plantation types, and disturbances within the Chichawatni forest plantation, a subtropical managed forest type. The Chichawatni forest is enriched with a diverse floral composition comprising of 173 angiosperm species from 42 families. Such diversity reflected the region's subtropical climate, fertile terrain, and varied ecological niches. The prevalence of therophytes, deciduous trees, and specific leaf spectrum types underscored the adaptability of plant species to the region's seasonal variations. Different plantation types within the forest exhibited varying impacts on biodiversity and vegetation composition. Notably, *Dalbergia sissoo* and *Populus nigra* plantations displayed higher species richness and diversity indices, highlighting the role of plantation type in shaping ecological dynamics. Those findings had important implications for sustainable forest management practices tailored to specific plantation types. Disturbances, such as road accessibility, stem cutting, and fire, significantly influenced plant distribution patterns within the forest. Road construction could fragment habitats, stem cutting impacted certain sites, and fire could disrupt ecosystem processes. Understanding the impacts of those disturbances was critical for effective forest conservation and management. The study's findings aligned with broader ecological patterns observed from the subtropical region. Subtropical ecosystems from the area were characterized by unique climatic conditions and rich biodiversity. The study contributed to the understanding of how subtropical forests responded to various ecological factors, providing insights that could monitor and create conservation and management strategies for the ecologically diverse region. Forest managers, policymakers, and conservationists in the subtropical region could be benefited from the study's insights. Tailoring forest management approaches to different plantation types, considering ecological traits and responses to disturbances, also helped to strike a balance between economic interests and ecological health in managed forests.

Data availability statement

The original contributions presented in the study are included in the article/Supplementary material, further inquiries can be directed to the corresponding author.

Author contributions

MW: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Writing – original draft, Writing – review & editing. SH: Data curation, Project administration, Validation, Writing – review & editing. FA: Project administration, Supervision, Validation, Writing – review & editing. RB: Project administration, Writing – review & editing. AH: Funding acquisition, Writing – review & editing. EFA: Funding acquisition, Writing – review & editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

Supplementary material

The Supplementary material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2024.1406075/full#supplementary-material

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Evaluation of forest loss data using fractal algorithms: case study Eastern Carpathians–Romania

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Logging causes the fragmentation of areas with direct implications for hydrological processes, landslides, or habitats. The assessment of this fragmentation process plays an important role in the planning of future logging, reconstruction, and protection measures for the whole ecosystem. The methodology used includes imaging techniques applying two fractal indices: the Fractal Fragmentation Index (FFI) and the Fractal Fragmentation and Disorder Index (FFDI). The results showed the annual evolution and disposition of deforested areas. Only 3% of deforestation resulted in the fragmentation and splitting of forest plots. The remaining 97% resulted in the reduction of existing compact stands without fragmentation. The method has many advantages in quantifying the spatial evolution of forests, estimating the capture of carbon emissions and establishing sustainability of bird and animal habitats. The analysis took place in the Eastern Carpathians, in Romania, in the time period of 2001–2022.

KEYWORDS

deforestation, forest management, imaging technique, fractal analysis, fractal algorithms

1 Introduction

Deforestation and forest fragmentation are interrelated processes that have significant impacts on ecosystems. Logging reduces forest cover, while fragmentation divides the remaining forest into smaller, isolated patches, altering the landscape and affecting biodiversity (Taubert et al., 2018; Liu et al., 2019). Deforestation and selective logging significantly increase the ratio of forest edge to area and the number of forest fragments, with edge effects extending deep into the remaining forest areas, leading to ecological consequences such as reduced genetic diversity, altered pollinator interactions, and different environmental conditions (Broadbent et al., 2008). Fragmentation affects the genetic and demographic structure of plant populations. It can lead to reduced genetic diversity, altered pollinator interactions and changes in environmental conditions due to edge effects (Honnay et al., 2005). Remote sensing technologies, which provide critical insight into the conservation status of habitats and structural changes within forested landscapes over time, have become a key tool in the analysis of forest fragmentation (Wulder, 1998). Remote sensing data was used by different researchers for the analysis of deforestation and forest fragmentation (Saunders et al., 1991; Nagendra et al., 2003; Biradar et al., 2005; Echeverría et al., 2011; Sahana et al., 2015, 2018). It allows for the efficient mapping of large areas by providing a precise and relevant collection of digital data at multiple scales. Forest fragmentation often occurs over large areas that are difficult to monitor using field-based methods alone (García-Gigorro and Saura, 2005; Carranza et al., 2014; Ganivet and Bloomberg, 2019). Forest fragmentation caused by deforestation is a major problem in natural landscapes, with significant consequences for terrestrial eco-systems (Echeverria et al., 2006; Taubert et al., 2018; Fischer et al., 2021; Slattery and Fenner, 2021; Ma et al., 2023). The relationship between deforestation and flooding is a critical area of study with significant implications for environmental management, economic activity, and human well-being (Angelsen and Kaimowitz, 1999). Previous studies have shown a significant correlation between deforestation and flood risk in affected regions (Bradshaw et al., 2007; Danáčová et al., 2020; Fazel-Rastgar, 2020; Peptenatu et al., 2020; Posada-Marín and Salazar, 2022; Hou et al., 2023). The process of forest fragmentation alters the structure and functions of these ecosystems, reducing their ability to regulate water flows and prevent flooding (Cramer and Hobbs, 2002; National Research Council, 2008; Hurtado-Pidal et al., 2022). Deforestation can affect hydrological services such as flood control, which is essential for preventing natural disasters and maintaining ecological balance (Bradshaw et al., 2007). As a result, forest fragmentation increases land vulnerability to flooding and amplify its impacts on human communities and infrastructure (Randhir and Erol, 2013; Renó et al., 2016). In developing countries, where the economic and human toll of floods can be devastating, this relationship is particularly pronounced. Loss of forested areas impacts liquid runoff on slopes (rapid flooding, accelerated land erosion, etc.) as well as loss of biodiversity. Such a study can contribute to the development of the institutional and regulatory framework for forest sector activity and sustainable management of forest resources. The preponderance of evidence supports the importance of forest conservation in flood risk management, although there is some debate about the impact of deforestation on major flood events (Chang et al., 2009; Dijk et al., 2009; Brookhuis and Hein, 2016; Tan-Soo et al., 2016).

Short-winged birds are particularly vulnerable to the effects of forest fragmentation. Because of their limited ability to move through gaps between woodlots in search of mates and food, many species accustomed to living deep in forests are exposed to predators in fragmented forests (Weeks et al., 2023). Conflicts between humans and wildlife intensify when the migration routes of certain species (wild boar, bears) are destroyed. Wild animals see human-altered landscapes, such as farms and towns, as another part of their original natural habitat (Betts et al., 2022).

The loss of carbon sequestration capacity through deforestation is a major contributor to accelerating climate change. Trees absorb carbon dioxide from the air and release it when they are felled (Tong et al., 2020). In addition, evapotranspiration, the process by which water is recycled to the atmosphere through leaf evaporation and plant root transpiration, is decreased when forests are cleared with a direct consequence through increased air temperature (Li et al., 2020).

The EU Deforestation Regulation (EUDR), which aims to tackle the urgent problems of greenhouse gas emissions and biodiversity loss, confirms the desire of governments to adopt a more cautious attitude towards forests Regulation (EU) 2023/1115 of the European Parliament and of the Council of 31 May 2023 on the making available on the Union market and the export from the Union of certain commodities and products associated with deforestation and forest degradation and repealing Regulation (EU) No 995/2010. Green Deal: EU agrees law to fight global deforestation and forest degradation driven by EU production and consumption (2022, December 6). European Commission. In this research we aim to analyze the dynamics of forest fragmentation through fractal imaging and examine the spatial processes of deforestation using the Fractal Fragmentation Index (FFI) (Andronache et al., 2016) and Fractal Fragmentation and Disorder Index (FFDI) indices (Peptenatu et al., 2023). These indices have demonstrated their utility in previous studies analyzing the evolution of forested areas (Einzmann et al., 2017; Andronache et al., 2019; Diaconu et al., 2019).

2 Materials and methods

2.1 Study area and data

The Eastern Carpathians (Figure 1) are the eastern part of the large mountain segments of the Carpathian Mountains range, with a wide variety of geological rocks, geophysical, geological, and morphological features, elevations, and degree of forestation.

On the Romanian territory, they are generally oriented NW–SE, representing 55% of the country's mountainous area. They are fragmented by numerous watercourses and depressions. Their present appearance dates to the second part of the Quaternary, with the warming of the climate that produced the melting of the glaciers on the highest peaks.

Petrographically, these mountains have as geological bedrocks a strip of crystalline rocks in the central part, sedimentary rocks in the eastern part and volcanic rocks in the west. However, the average altitude is a little lower, about 950 m, the maximum altitude being 2,303 m. The Rodna Mountains, the most striking feature of the relief is the parallelism of the peaks.

The climate is zoned by both altitude and the length and width of the mountain range. The average annual air temperature is between +6°C and −2°C and rainfall ranges from 800 mm to 1,200–1,400 mm per year.

The Eastern Carpathians are distinguished by a large extension of spruce and mixed forests (fir, spruce, and beech) representing 40% of Romania's forest heritage. This area was chosen because many local communities engage in logging activities, floods caused by rivers that drain the mountainous area and the existence of fauna biodiversity that is pressured by human intervention (Paveluc et al., 2021; Crişan et al., 2023; Andronache, 2024).

2.2 Methodology

Fractals are conceptual objects displaying structures at all spatial scales, with a scale-dependent self-similarity (Mandelbrot, 1983; Al Saadi and Badr, 2024). The shape of fractals cannot be rectified, consisting of an infinite sequence of clusters within clusters. In rectifiable objects, increasingly accurate measurements based upon

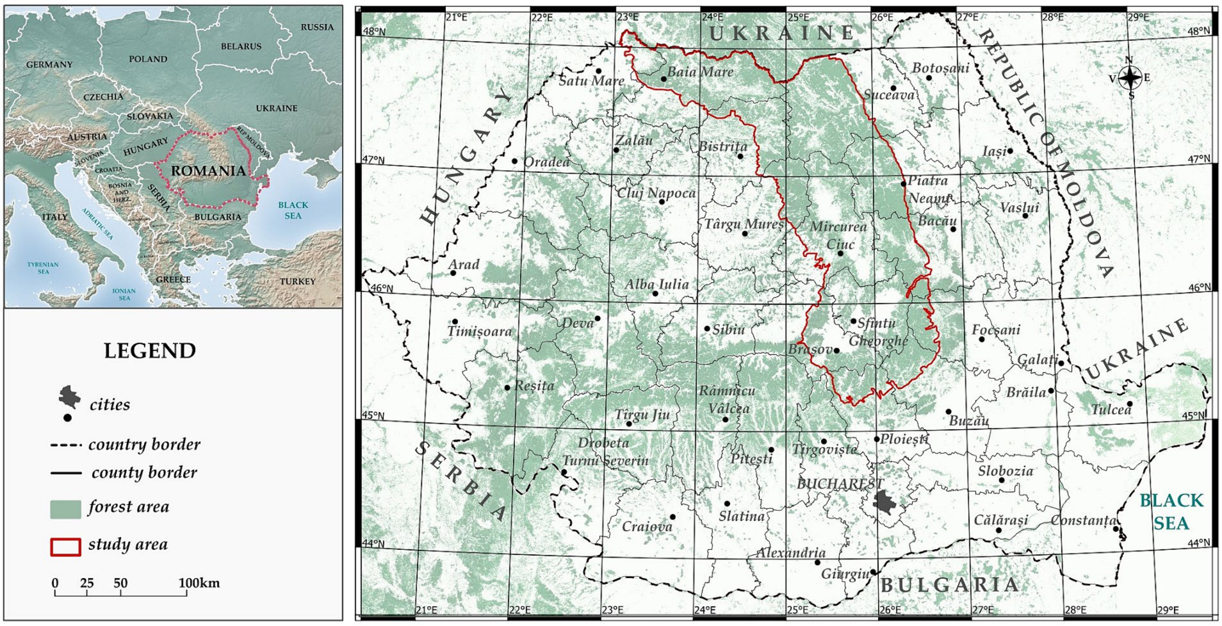

FIGURE 1
Location of the research area.

successive scale reductions lead to a series converging to a limit: the true extent of the object.

The fractal concept is also useful for characterizing certain aspects of patch dynamics. The research architecture used in this article is summarized in Figure 2.

The final phase of the research, fractal analysis using the FFI and FFDI index, is also structured as follows (Figure 3).

2.2.1 Image preprocessing

The Global Forest Change 2000–2014 database provided by the University of Maryland Department of Geographic Sciences was used to assess the area of deforestation. This database was created by analyzing 654,178 LSTM+ images characterizing global forest surface from 2000 to 2022 (Hansen et al., 2013).

The forest and deforestation maps were extracted in .tiff format with the open-source software QGIS at a resolution of 2,480 × 3,507 pixels. This resolution was chosen because it provides a sufficient level of detail and covers the entire study area for fractal analysis. The analysis started with the reprojection of the Hansen_GFC into the national coordinate reference system Pulkovo 1942(58)/Stereo 70, EPSG: 3844 to obtain metric results. All of the deforestation areas have been extracted using the Raster layer unique values report tool from the QGIS Processing toolbox. This algorithm returns the number and the area of each unique value in each raster layer. In this study it has returned the number of all the pixels and the area in square meters. For the entire Romanian Eastern Carpathians, forested and deforested areas over the 22-year period were analyzed.

The images from 2001 to 2022 and the tree cover image from 2000 (Hansen et al., 2013) were converted to binary using Fiji/ImageJ2 2.14.0/1.54f java 1.8.0_322 [64-bit] (Schindelin et al., 2012; Schneider et al., 2012; Schindelin et al., 2015): 'Image – Adjust – Threshold' function with a range of [0, >0]. The binary images of cumulative loss were obtained by adding the loss area image from 2002 to the loss area

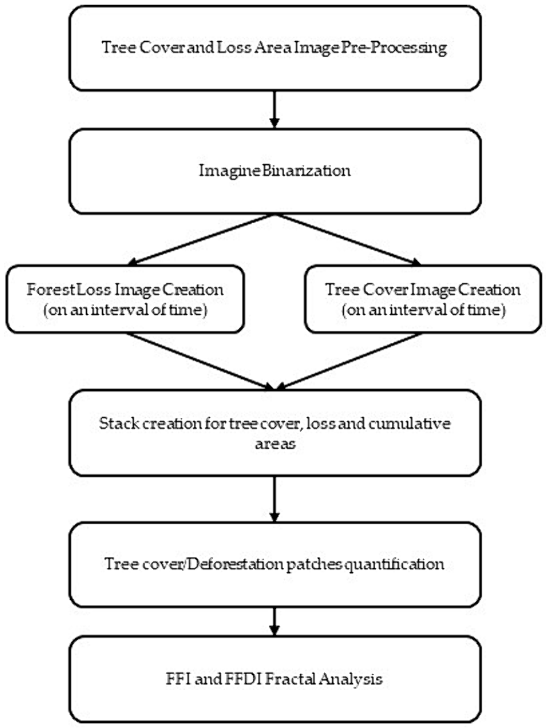

FIGURE 2
Research flowchart.

image from 2001 for the 2001–2002 image, using Image Calculator from Fiji/ImageJ2. Similarly, the deforested area image from 2003 was added to the cumulative loss image from 2001 to 2002 for the 2001–2003 image. This iterative process continued, and the deforested area

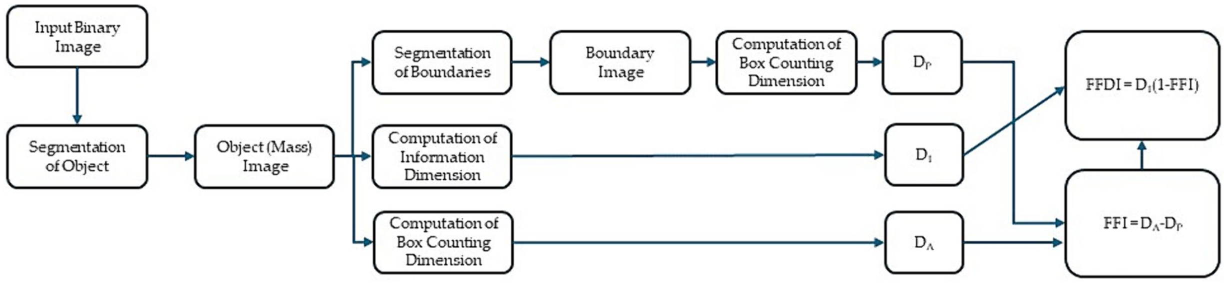

FIGURE 3
FFI and FFDI research flowchart.

image from 2022 was added to the cumulative loss image from 2020 to 2021 for the 2001–2022 period (Figure 4).

The tree cover image for 2001 was extracted from the tree cover image of 2000, using the loss for 2001. The tree cover image for 2002 was obtained by extracting the cumulative loss image for 2001–2002 from the tree cover image of 2001. This iterative process continued until the tree cover image for 2022 was obtained by extracting the cumulative loss image for 2001–2021 from the tree cover image for 2021 (Figure 5).

Particle analysis is a technique used to count and measure the characteristics of objects or particles in a digital image. It can determine the number, size, shape, distribution, and other properties of particles present in the image. Fiji/ImageJ2 software was used for this analysis.

Fractal fragmentation index (FFI). The Fractal Fragmentation Index (FFI) is a fractal index that estimates the degree of fragmentation or compaction of objects filling a space, as well as the deviation of each object's shape from a geometrical Euclidean shape (Andronache et al., 2016). The FFI is calculated using multi-scale fractal techniques and the following Equation (1):

$$\mathrm{FFI} = D_A - D_P = \lim_{\varepsilon\to 0}\left(\frac{\log N(\varepsilon)}{\log\frac{1}{\varepsilon}}\right) - \lim_{\varepsilon\to 0}\left(\frac{\log N'(\varepsilon)}{\log\frac{1}{\varepsilon}}\right) \tag{1}$$

where, FFI the Fractal Fragmentation Index, D_A is box counting fractal dimension of areas (forest patches in this paper) and D_P is the box counting fractal dimension of the perimeters of those areas; ε represents the size of the boxes used in the analysis; log $N(\varepsilon)$ represents the number of contiguous and non-overlapping boxes required to cover the object area(s), and log $N'(\varepsilon)$ represents the number of contiguous and non-overlapping boxes required to cover only the object perimeter(s). According to the equation: FFI = 0, FFI is 0 ($D_A = D_P$) when the image is completely fragmented. FFI is 1 ($D_A = 2$, $D_P = 1$) when a single object is being analyzed and it has an Euclidian form (for example, a square). Therefore, the images with FFI = 0 are perfectly fragmented images and those with FFI = 1 are perfectly compact. In practice, those situations are rare, FFI having values situated between 0 and 1. As the image gets more fragmented, the FFI will start to tend towards 0. The reverse is true and as the image gets more compact, the FFI will start to tend towards 1. When the dynamics of a process or phenomena is analyzed, FFI helps with understanding the trend of fragmentation by reducing its value in time or compaction by raising its value in time the fractal entities being analyzed are very small, their contour cannot be extracted effectively, therefore $D_A = D_P$; ($0 > \mathrm{FFI} < 1$).

FFI provides a global view of fragmentation across the whole image, which allows a comprehensive analysis of the spatial distribution of fractal objects. This indicator is insensitive to small artifacts, such as isolated or residual pixels, which may occur during image binarization. This provides a more accurate and reliable analysis of fragmentation.

Fractal fragmentation and disorder index (FFDI). The Fractal Fragmentation and Disorder Index is a fractal index that uses multi-scale fractal techniques to estimate the degree of fragmentation/compaction and spatial disorder of objects that fill a space (Peptenatu et al., 2023). The FFDI is derived from both the FFI (Andronache et al., 2016) and the Information Dimension (D_1) (Baker and Gollub, 1996). Like the FFI, it differentiates spatial organization patterns. It is calculated using Equation (2).

$$\mathrm{FFDI} = D_1(1-\mathrm{FFI}) = \left(\lim_{\varepsilon\to 0}\sum_{i=1}^{N(\varepsilon)}\frac{m_i(\varepsilon)\log(m_i(\varepsilon))}{\log(\varepsilon)}\right)\left(1-\left(\lim_{\varepsilon\to 0}\left(\frac{\log N(\varepsilon)}{\log\frac{1}{\varepsilon}}\right)-\lim_{\varepsilon\to 0}\left(\frac{\log N'(\varepsilon)}{\log\frac{1}{\varepsilon}}\right)\right)\right) \tag{2}$$

where $m_i = M_i/M$, M_i is the number of points in the ith box, M is the total number of points of the object, ε is the size of the boxes, $N(\varepsilon)$ is the number of contiguous and non-overlapping boxes required to cover the area of the object(s) and $N'(\varepsilon)$ represents the number of contiguous and non-overlapping boxes required to cover only the perimeter of the object(s).

In this definition, 1-FFI is used because FFI = 0 indicates fragmentation and FFI = 1 indicates compaction (Andronache et al., 2016).

The FFDI range is between 0 and 2. The maximum value of FFDI approaches 2 when objects are strongly disordered and fragmented, whereas the lowest value approaches 0 when objects are weakly disordered and compact.

FFI can be used to discern patterns and identify thresholds in the spatial distribution of the objects under analysis, thus providing a better understanding of the structure and evolution of the system

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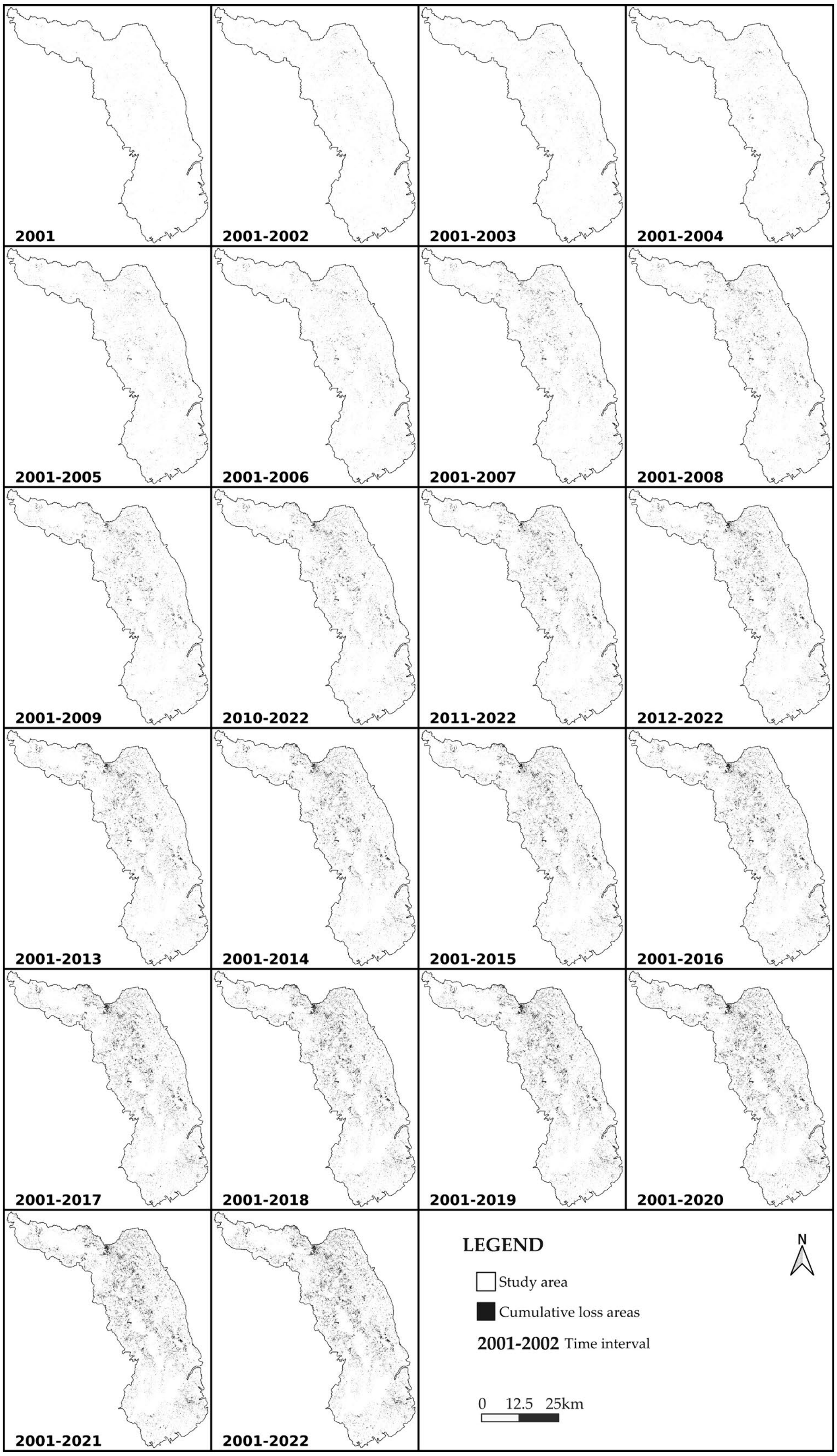

FIGURE 4
Dynamics of tree cumulative loss areas in the Eastern Carpathians from 2001 to 2022.

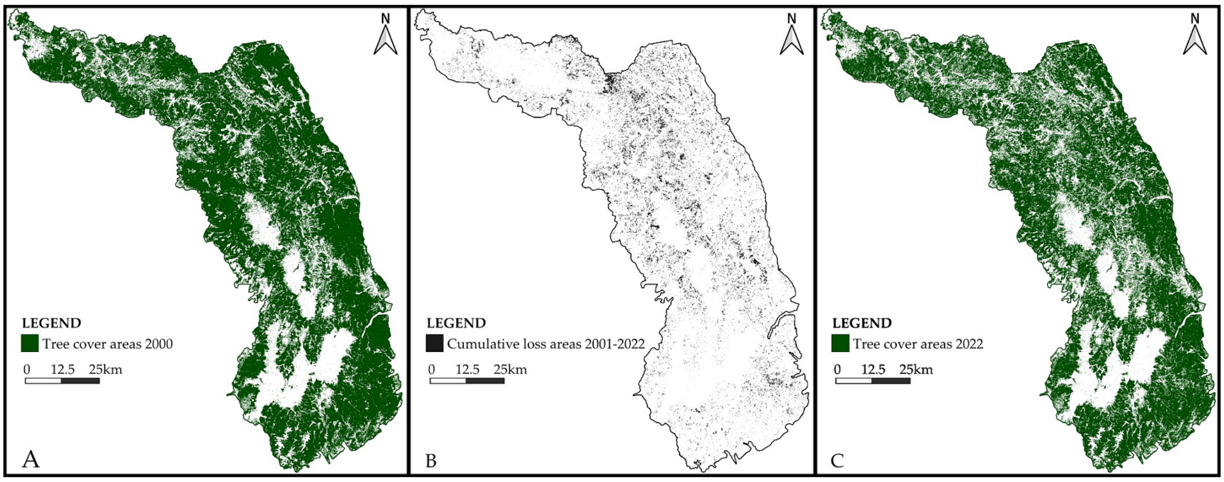

FIGURE 5
Dynamics of tree cover areas in the Eastern Carpathians from 2000 **(A)** to 2022 **(C)**, in green. The image on the center **(B)** displays in black the cumulative loss areas between 2001 and 2022.

under study. The index can be used to analyze the dynamics of fragmentation over time and the influence of different factors on it. This allows to monitor and evaluate changes in the spatial distribution of the analyzed objects in an efficient and detailed way.

FFI can provide a global picture of fragmentation when the analyzed image includes several objects, but can also provide a local picture when the analyzed image includes only one object. FFDI is only a global analysis.

FFI and FFDI have been implemented as an open-source plugin for ImageJ2/Fiji within the ComsystanJ package (Ahammer et al., 2023). This plugin can be downloaded from https://comsystan.github.io/comsystanj/ or from GitHub at https://github.com/comsystan/comsystanj/tags. FFI and FFDI were intended for use exclusively with binary images.

For the fractal analysis, three image stacks were initially created, one stack for loss areas with 22 images, each image corresponding to 1 year from 2001 to 2022. A second stack for cumulative loss areas also with 22 images for the same period and a third stack for tree cover with 23 images, each image corresponding to 1 year from 2000 to 2022. Each stack was imported into ComsystanJ, and the "Fractal Fragmentation Indices" plugin was applied. The maximum number of boxes was set to its maximum value (# boxes = 10), and the values for regression min and regression max were set to 1 and 10, respectively.

Conventional GIS methods and techniques for cartographic representation of information as well as Word and Excel packages for information representation and editing were used in the research and data representation.

3 Results

The analysis of the loss areas showed that the tree cover of the Eastern Carpathians decreased by 201,077.4218 ha, from 2,327,594.44 ha in 2000 to 2,126,517.01 ha in 2022, which is a decrease of 8.6%. The most intensive deforestation occurred in 2003 (3,117.05 ha) and 2007 (23,135.51 ha).

During the analyzed period, a total of 74,902 forest patches were cleared, with an average of 3,405 patches per year. The highest number of patches occurred in 2007 (6,679), while the lowest was in 2003 (1,461). Between 2001 and 2022, 74,902 forest patches were deforested, resulting in 31,710 cumulative patches. Of these loss forest patches, 58% disappeared due to merging, leading to the fragmentation and disorder of compact forest areas. During this period, 2,262 new forest patches appeared due to fragmentation, representing a 17% increase in patches (from 10,851 in 2000 to 13,113 in 2022). Only 3% of deforestation resulted in the creation of new forest patches. The remaining 97% led to the reduction of compact forest areas without fragmentation, by deforestation occurring in isolated forest patches.

The trends in tree cover and deforestation changes from 2001 to 2022 are shown in Figure 6.

From 2001 to 2012, there was a downward trend in the percentage of new patches of cumulative loss resulting from new loss events, as shown in Figure 6A. However, after 2012, although the downward trend continued, there were some years where the percentage of new patches of cumulative loss increased compared to the previous year, specifically in 2010, 2014–2015, 2020, and 2022. The most significant decrease was observed in 2003 compared to 2002 (−12%), while the most significant increase was observed in 2013 compared to 2012 (13%). Between 2002 and 2007, 50–89% of loss patches resulted in new patches of cumulative loss, indicating that deforestation mostly occurred in isolation. In contrast, deforestation occurred more in continuation of pre-existing losses in the years 2012, 2018–2019, and 2021–2022, as only between 17 and 25% of loss patches resulted in new patches of cumulative loss.

Furthermore, there is a trend towards an increase in the percentage of tree cover patches due to new patches of cumulative loss, which is more pronounced during the period of 2001–2012 (refer to Figure 6B). There were some exceptions where there were decreases in the years 2010, 2017, 2019, and 2022 compared to their preceding years. Like the previous situation, the largest increase was in 2018 compared to 2017 (5.32%), while the largest decrease was in 2013 compared to 2012 (−14%). Between 2001 and

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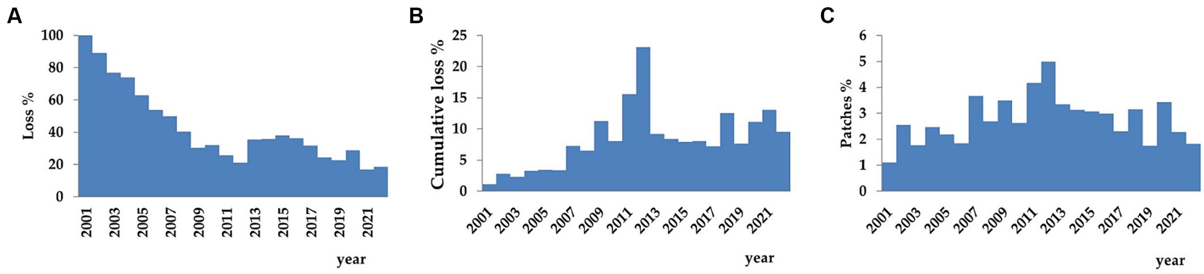

FIGURE 6
Analysis of change evolution in Eastern Carpathian Forests between 2001 and 2022: Trends in the percentage occupied by new loss **(A)**, new cumulative loss patches **(B)**, and tree cover patches **(C)**.

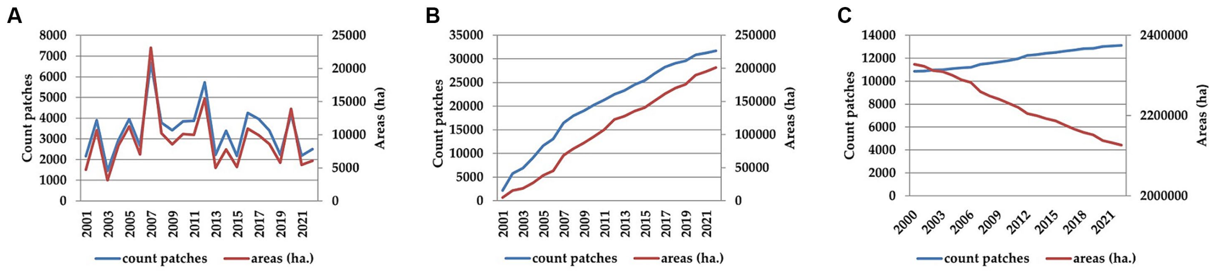

FIGURE 7
Relationship between area and number of patches for loss areas **(A)**, cumulative loss areas **(B)** and tree cover **(C)**.

2006, de-forestation only slightly fragmented the tree cover, with new patches of cumulative loss generating new patches of tree cover at a rate of 1.1–3.4%. However, in more recent years (2009, 2011–2012, 2018, and 2020–2022), as deforestation continued from pre-existing clearings, the fragmentation of tree cover into more patches increased significantly, with new patches of cumulative loss resulting in new patches of tree cover at a rate of 11.2–23.1%.

Regarding the percentage of tree cover patches resulting from new patches of cumulative loss (Figure 6C), there is a trend of yearly increase during the period 2001–2012, followed by a slight trend of decrease after 2013. Prior to 2012, deforestation led to a greater fragmentation of tree cover, with deforestation occurring in a more disorderly manner. However, after 2013, deforestation occurred more in continuation of previous clearings, but also over isolated small patches of tree cover. Two distinct patterns have been identified in the data. The largest increase occurred in 2007 compared to 2006 (1.82%), while the largest decrease occurred in 2013 compared to 2012 (−1.64%). Between 2001 and 2022, new loss patches generated new patches of tree cover in only 1.2–2.2% of cases, with higher values of 3.2–5% recorded in 2007, 2009, and 2011–2015, as well as in 2020.

In the case of loss areas, deforestation averaged between 2.13 ha/patch in 2003 and 3.46 ha/patch in 2007, while cumulative loss areas increased from 2.19 ha/patch in 2001 to 6.44 ha/patch in 2022. Conversely, tree cover areas decreased from 214.51 ha/patch in 2000 to 162.17 ha/patch (Figure 7).

Figure 7 illustrates the relationship between areas and number of patches for loss areas (Figure 7A), cumulative loss areas (Figure 7B) and tree cover (Figure 7C). Figures 7A,B show a dependence between deforested areas and the number of patches for loss and cumulative loss areas. However, as forest area decreases due to forest loss, it fragments, leading to an increase in the number of forest patches.

The analysis of loss and cumulative loss areas between 2001 and 2022 indicates that deforested patches are small and heterogeneously distributed spatially (Figures 8A,B).

Deforestation was slightly more compact in 2004, 2007, and 2020 (Figure 8A). The same pattern is observed for cumulative loss, with FFI increasing from 0 in 2001 to 0.003 from 2001 to 2022 (Figure 8B). A trend towards forest compaction was observed between 2001 and 2012, despite disorderly deforestation during the same period (Figure 8C). As deforestation occurs chaotically and in small, fragmented, rarely interconnected patches (Andronache et al., 2016), FFI values are very low, making this index unsuitable for differentiation between years. To tackle this issue, we employed FFDI, which measures the level of disorder in deforested areas (Figures 8D–F). FFDI peaked in 2007, a year marked by extensive, fragmented, and disorderly deforestation (Pintilii et al., 2017). The lowest FFDI value was observed in 2003, during a period of reduced deforestation and a much more orderly spatial distribution (Figure 8D). Over 22 years of continuous deforestation, the FFDI increased by 33%, indicating increasingly fragmented forests (Figure 8E). The analysis of deforestation effects on tree cover revealed continuous forest fragmentation, with FFI decreasing by 41% between 2000 and 2022 (Figure 8C). This ongoing forest fragmentation also led to spatial disorder of forest patches, as indicated by FFDI estimating a 4% increase in forest fragmentation and disorder (Figure 8F).

Figures 9A–C illustrate the correlation between the count of patches and FFDI. In all three analyses (loss, cumulative loss, and tree

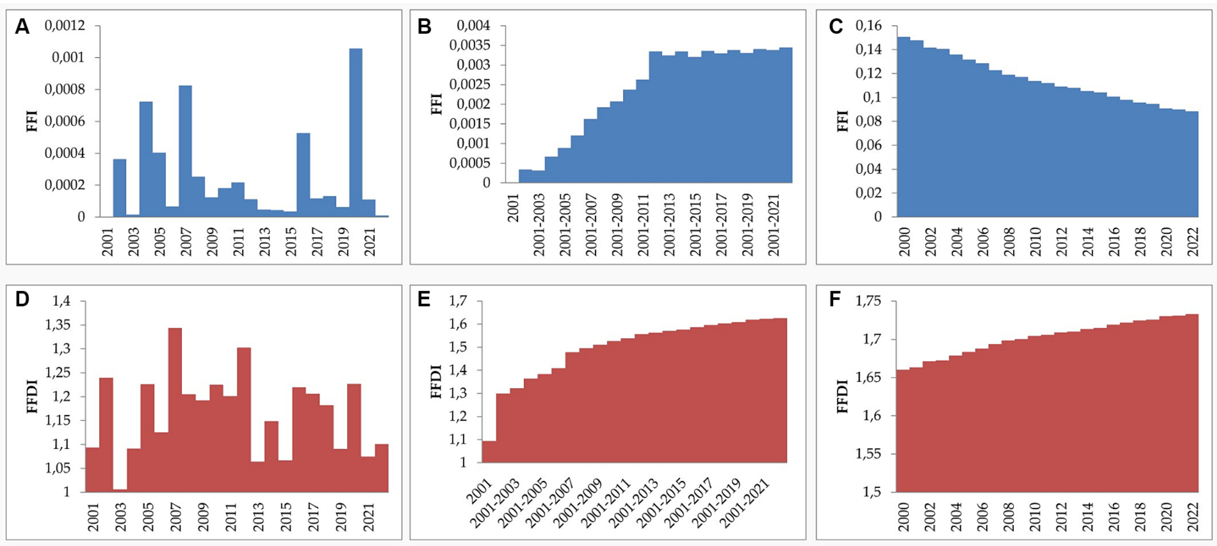

FIGURE 8
Dynamics of deforestation patterns and forest fragmentation in Eastern Carpathian forests from 2000 to 2022 **(A-C)** fractal fragmentation index of **(A)** loss, **(B)** cumulative loss, **(C)** tree cover; **(D-F)** fractal fragmentation and index of **(D)** loss, **(E)** cumulative loss, **(F)** tree cover.

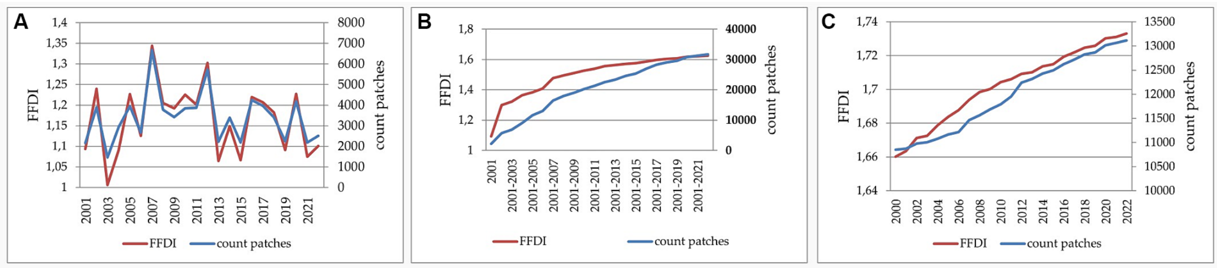

FIGURE 9
Correlation between count of patches and forest fragmentation disorder index (FFDI) in Eastern Carpathian forests, for: **(A)** loss, **(B)** cumulative loss and **(C)** tree cover patches.

cover), the fragmentation and disorder of these patches increase as the number of deforested patches increases.

The analysis showed that a significant amount of deforestation occurred during the studied period, resulting in a total loss of 76,205 forest patches, averaging 3,464 patches per year. The highest number of patches was observed in 2007, indicating significant deforestation activity, while the lowest was observed in 2003.

4 Discussion

The proposed methodology has proven to be relevant for identifying areas where major land use changes have occurred in a short time. The research is part of a series of analyses that looked at the structured impact of forest loss on the ecosystem (Andronache et al., 2016; Einzmann et al., 2017; Andronache et al., 2019; Diaconu et al., 2019; Peptenatu et al., 2023) and local economies (Pintilii et al., 2017).

The importance of this analysis methodology lies in the rapidity with which it provides information on how the woody mass on slopes has been exploited, and consequently on the potential impact on run-off coefficients on slopes and the degree of habitat fragmentation. The need for such research is highlighted in numerous studies (Clarke and Schweizer, 1991; Imaizumi et al., 2008; Zhang et al., 2020; Zhao et al., 2021).

FFDI revealed patterns of forest fragmentation and disorder over time, with FFDI values gradually increasing over the studied period, indicating increasingly fragmented forests. Figures 9A–C support this trend, indicating a positive correlation between the count of deforested patches and FFDI across different analyses (loss, cumulative loss, and tree cover).

The research has shown that the FFI–FFDI pair is advantageous for analyzing forest fragmentation. Both FFI and FFDI are useful for analyzing tree cover fragmentation and disorder. However, FFDI yielded better results for the analysis of loss and cumulative loss.

The results highlight the complex dynamics of deforestation and its implications for forest fragmentation and spatial disorder. Effective forest management strategies are important to mitigate fragmentation and preserve forest ecosystems. Detail analyses allow for better understanding of the factors that determine forest loss and are relevant methods to assist decision making in the silvic and biodiversity sectors. As such, silvic code could be completed with mechanisms that monitor resources and exploitation process of wood.

Our findings reveal ongoing forest fragmentation as a result of deforestation, accompanied by an increase in their spatial disorder. This fragmentation and disorder may contribute to heightened flood risk in the affected regions. Fragmented and disorderly forests may lose their ability to regulate water flows and mitigate flood risk, exacerbating land vulnerability to extreme precipitation events. Therefore, it is essential to understand and appropriately manage forest fragmentation and disorder processes to reduce the risk of flooding and protect human communities and infrastructure.

The evaluation methodology of forest fragmentation does not have a relevant correspondent that permits validating indices. It must be mentioned that analyzed image resolution and synoptic situations may lower the accuracy of data and generate errors. Another limitation of the FFI analysis is that it is calculated for the whole picture without providing information on regional variations. A scale limitation is another general problem in image analysis, with a single pixel as the smallest scale and the whole image size as the largest scale. In addition, natural objects are also intrinsically limited in scaling due to the finite size of their structural units. Our study has some limitations that must be addressed. The images used, with a spatial resolution of 30 m, allowed us to capture only a coarser picture of forest patterns. The use of more detailed images would resolve this restriction and thus improve FFI or LCFD accuracies.

The main disadvantage of FFI is that this index does not differentiate patterns where the component objects are extremely small so that the edges cannot be extracted, or where the objects completely occupy the space. The FFDI was also designed to overcome these limitations by further quantifying the information entropy in the image using the Information Dimension. FFDI provides a clearer differentiation in the categorization of synthetic images and in real applications.

Both FFI and FFDI have the disadvantage of only analyzing binary images. Initial preprocessing of binary images, such as binarization and noise removal, can influence the results of fractal analysis and must be carefully managed. Depending on the threshold chosen, binarization may introduce analytical bias; however, this effect can be further investigated and could be corrected by incorporating a scaling relationship or using FFI or FFDI as a relative index for comparing different patterns or states of the system, rather than as an absolute measure of fragmentation or fragmentation and de-ordering.

Furthermore, the interpretation of fractal analysis results can be influenced by the scale at which measurements are made, and an inappropriate choice of scale can lead to misinterpretations (Ciobotaru et al., 2019; Tarko et al., 2020).

In Romania, the economic pressure on the silvic fund has risen after 1990 at the same time when external commerce of logs became more permissive (Tronicke and Lück, 2023). The legislative regulations from that moment led to a rise in fragmentation due to authorized and unauthorized logging on surfaces covered in forest (Knorn et al., 2012; Iordăchescu and Vasile, 2022). In the field, this is identified with great difficulty, but when using remote sensing technologies this becomes more accessible and accurate.

Speaking of limitations, the resolution of satellite images significantly influences the calculated values of the Fractal Fragmentation Index (FFI) and Fractal Fragmentation and Disorder Index (FFDI). High resolution images provide more detailed information, which can lead to a more accurate fractal analysis of forest fragmentation. Lower resolution images may miss fine details, resulting in an underestimation of fragmentation metrics. In this study, to avoid this problem, all images analyzed were converted to the same resolution.

Converting grayscale images into binary images, a necessary step for fractal analysis, involves selecting a binarization threshold. An inappropriate threshold can either hide critical details or introduce noise, thus affecting the values of FFDI or other fractal metrics. It is important to note that the FFI is relatively insensitive to this bias, since isolated, noise-like pixels do not influence the final result. However, to minimize bias for FFDI or other binary fractal analysis metrics, future research should consider using adaptive binarization techniques that adjust based on local image features. In the situation of our study, there was no need for adaptive binarization because the images in the Global Forest Change database only provide information about the presence or absence of forests or deforestation.

The scale at which fractal analysis is performed can significantly affect the interpretation of FFI and FFDI. Analysis at different spatial scales can reveal different aspects of forest fragmentation. Analyses at small scales may reveal detailed patterns of fragmentation, while analyses at large scales may show broader trends. Future studies should conduct multi-scale analyses to capture both fine and coarse details of fragmentation. A hierarchical approach, analyzing data at different scales, can provide a comprehensive understanding of forest fragmentation.

5 Conclusion

Deforestation occurred through various mechanisms during this period. These mechanisms included the emergence of new loss areas, continuation of deforestation in previously lost areas, or the merging of two previously lost areas into a single loss area.

A notable finding is that 57% of loss patches disappeared due to merging, leading to the fragmentation and disorder of compact forest areas. This highlights the impact of deforestation on the spatial structure of forest landscapes.

The analysis identified the creation of new forest patches due to fragmentation, representing a 17% increase over the studied period. However, only 3% of deforestation resulted in the creation of new forest patches, indicating that most deforestation led to the reduction of compact forest areas without fragmentation.

Trends in tree cover changes and deforestation patterns were demonstrated. From 2001 to 2012, there was a downward trend in the percentage of new patches of cumulative loss, followed by fluctuations in subsequent years. Deforestation tended to occur more in continuation of pre-existing losses in recent years, rather than as isolated events.

The fractal indicators used offer the possibility to quantify the reduction in forest area in a fast and highly accurate way. If the limitations of the methodology are taken into account in this process, it can be widely used at least at the level of forested areas.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

DD: Conceptualization, Supervision, Validation, Writing – original draft, Writing – review & editing. IA: Conceptualization, Data curation, Formal analysis, Software, Validation, Writing – original draft. AG: Investigation, Writing – original draft. TB: Formal analysis, Writing – review & editing. AB: Investigation, Software, Validation, Writing – review & editing.

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Land degradation neutrality and carbon neutrality: approaches, synergies, and challenges

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Land is being degraded rapidly worldwide. United Nations Convention to Combat Desertification in 2015 has invited countries to formulate voluntary targets to achieve Land Degradation Neutrality (LDN). Under the Paris Agreement, a legally binding international treaty adopted in 2015, the world is transitioning toward Carbon Neutrality (CN) with more mitigation actions. This paper intended to review the concepts of land degradation, LDN along with CN emphasizing the degradation types, approaches, models available to analyze, synergies, economic aspects and challenges. The review explores approaches and models available for achieving LDN and CN which are both synergistic, economically efficient and could overcome the common challenges. Land degradation has to focus beyond the traditional definitions to incorporate more persistent and the difficult to restore degradation causes. Such complex land degradation requires specialized LDN approaches. The level of degradation and restoration progress could be analyzed using a variety of modeling approaches including economic models. Approaches for LDN and CN can bring significant synergies for each other. The approach proposed by the present study will provide a logical flow for decision-making while minimizing time and effort and avoiding a piecemeal approach. The approach therefore maximizes the output in relation to the inputs thus enhancing sustainability.

KEYWORDS
land degradation, climate change, land degradation neutrality, carbon neutrality, synergy

1 Introduction

The modern agricultural, industrial and urban development implies that the land is continuously being degraded and that degradation could sometimes stay as a permanent state. Land degradation is caused by a variety of reasons and traditionally associated with soil erosion (Chalise et al., 2019) and desertification (Briassoulis, 2019). However, increasing levels of chemicals and pollutants in the environment means that the degradation has wider spatial and temporal dimensions. The nature of the materials that are added to the environment, their behavior and sensitivity of the receiving environment will determine the severity of the degradation, and the possibility of recovery. The level of degradation could be analyzed using a variety of models ranging from simple map-based models to complex models that require multiple data. The feasibility of recovery or restoration could be assessed by various economic tools. Degradation that is difficult to recover, unrecoverable or unrestorable implies a large economic cost.

Emission of higher levels of greenhouse gases (GHGs) have increased the humanity's vulnerability to climate change. Global actions have proposed mechanisms for carbon neutrality (CN) which are nicely linked with the land degradation neutrality (LDN). Assessment and modeling of carbon neutralizing options, their feasibility from both technical and economic viewpoint will enable us to recognize cost minimizing synergies for achieving LDN and CN. Although land degradation is a local issue carbon dioxide is a global pollutant. Properly implemented LDN actions therefore generate global benefits. It is therefore worthwhile finding the matching pairs that could achieve both LDN and CN.

The study intends to understand the degradation types at a deeper level to recognize easy to remediate degradation types and more persistent degradation types. This knowledge is further enhanced by models that predict future outcomes resulting from the degradation. Among the options available for achieving LDN and CN, the ones that maximizes the economic efficiency can be chosen while understanding challenges. This approach will provide a logical flow in making the decision while minimizing the time and effort. The approach therefore maximizes the output in relation to the inputs thus enhancing sustainability.

2 Methodology

Literature was searched using variety of search engines such as google scholar and ScienceDirect with search words of the main topic including Land Degradation, Land Degradation Neutrality and Carbon Neutrality. Then for each term the following search words were added; definitions, global actions, approaches, models, economics, challenges. Then combinations of main topics including LDN and CN were searched with the word synergy. After screening the abstracts and contents of the articles, most significant sources of information were selected and reviewed for analysis and synthesis. Only the papers published in indexed peer-reviewed journals that are included in Scopus database and book chapters from recognized publishers were used. Eighty-four publications were found which were relevant. Non-English articles and articles that do not focus on the goal of the study were excluded. Then the results obtained from the above search were synthesized, edited and presented under each topic. Supplementary Figure S1 summarizes the review protocol.

3 Land degradation and its impact on nature and humans

Land degradation is defined as a "reduction or loss of biological or economic productivity and complexity of agroecological systems as a consequence of land use, or from one or more processes that may arise from human activities" (UNCCD, 2024). Land degradation exerts severe negative impacts on global and regional economic and social development and food security (Deng and Li, 2016; Chan et al., 2023). Highly degraded areas cover about 29% of global land area which is home to about 3.2 billion people (Le et al., 2016). Agricultural lands and natural habitats are degraded due to various forms of land degradation. For example, livelihood of two-third of the population of India, are vulnerable due to land degradation (Mythili and Goedecke, 2016) and land degradation hotspots cover about 51% of land area in Tanzania and 41% in Malawi (Kirui, 2016).

Analysis of causes of land degradation and their extents (Mythili and Goedecke, 2016) adopting multidimensional perspective (Prăvălie, 2021) help to design suitable policies to overcome the degradation. Traditionally, land degradation is explained in relation to erosion by water and by wind, salinization, and soil acidification and vegetation degradation. However, more severe and permanent land degradation could result from land pollution from a variety of chemicals and plastic. Land degradation due to chemicals is a complex phenomenon which requires understanding of system dynamics (Gunawardena, 2022).

Pesticides including insecticides, fungicides, herbicides, rodenticides can lead to sterile soils (UNCCD, 2017; Tang et al., 2021). These may also lead to depletion of soil biodiversity (Beaumelle et al., 2023) and overall biodiversity in the landscape due to impairment of pollination function (Hashimi et al., 2020). In addition, mixtures of insecticides and herbicides could bring significant synergistic ecotoxicological effects to the earthworms (Uwizeyimana et al., 2017) in the presence of heavy metals. Heavy metals could bring a variety of negative impacts on soil organisms (Liu et al., 2021) including earthworms (Morgan and Morgan, 1992; Fourie et al., 2007).

Persistent organic pollutants (POPs) resulting from industrial activities may be either directly dumped onto the land or land filled. Sludge applications to crops is another source of POPs resulting from industrial wastewaters (Arvaniti and Stasinakis, 2015). This may lead to contamination of ecosystems, food chains, and water. The damages resulting from landfills could be severe with large loads of contaminants with future risk that may last for centuries (Weber et al., 2011) due to their capacity to bioconcentrate, bioaccumulate, and biomagnify. In addition, electronic waste recycling sites can also generate complex mixtures of dioxin-related compounds which can contaminate surface soils (Manz et al., 2001).

Mining activities, for example, coal (Dhyani et al., 2023), mineral and metal mining cause land degradation. Increasing mining efforts lead to generation of higher waste quantities per unit of useful product which is disposed into tailing dumps (Slipenchuk et al., 2019). Adverse effects of plastics and microplastics in soils (Rillig, 2012) are likely due to dumping of disused or abandoned plastic, municipal wastewater effluents, landfilling with sewage sludge and plastic used in agricultural activities (Chae and An, 2018; Hale et al., 2020). Furthermore, microplastics can be bioaccumulated among organisms (Yang et al., 2023).

Deforestation leads to land degradation and water depletion due to the increased levels of soil erosion and associated nutrient depletion and sediment transport (Chan et al., 2023). Land degradation could result in significant loss of ecosystem services. Millennium ecosystem assessment report defines land degradation as the long-term loss of ecosystem services (Nkonya et al., 2016a).

4 International actions and multilateral environmental agreements related to land and climate change

The 1992 Earth Summit initiated three Rio Conventions on climate change, desertification, and biodiversity. The United Nations Framework Convention on Climate Change (UNFCCC) aims to prevent "dangerous" human interference with the climate system (UNFCCC, 2024). The United Nations Convention to Combat Desertification (UNCCD) is dedicated to combatting desertification and mitigating the impacts of drought in countries facing severe desertification or drought conditions (UNCCD, 1994).

Under the Paris Agreement, a legally binding international treaty adopted in 2015, the world is transitioning toward CN with more mitigation actions. In October 2015, the 12th Conference of the Parties (COP12) of the UNCCD proposed a definition for land degradation neutrality (LDN). In 2017, an LDN Scientific Conceptual Framework was developed and endorsed by UNCCD Member States (Cowie et al., 2018).

There have been several chemical-related multilateral environmental agreements (MEAs) that have a relevance to land degradation including the Basel Convention on the control of transboundary movements of hazardous wastes and their disposal, Rotterdam Convention on the prior informed consent procedure for certain hazardous chemicals and pesticides in international trade and Stockholm Convention on persistent organic pollutants. However, the linkages of these conventions with land degradation have been less obvious.

During the UN Summit for the adoption of the post-2015 development agenda, an agreement on set of 17 Sustainable Development Goals (SDGs) (UN, 2015) was made. Under SDG 15 Life on Land, target 15.3 intends to achieve a land degradation neutral world while target 2.4 of SDG 2 encourages resilient agricultural practices and gender equality over land resources is emphasized under Target 5.a. Achieving LDN increases ecosystem services and improves soil quality, contributing to several other SDGs, including SDG 3 (good health and wellbeing), SDG 5 (gender equality), SDG 6 (clean water and sanitation), SDG 11 (sustainable cities and communities), SDG 12 (responsible consumption and production), SDG 13 (climate action), SDG 14 (life below water), and SDG 15 (life on land) (Feng et al., 2022). It is important to note here that each SDG has synergies and trade-offs with other SDGs. On the other hand, the key MEAs related to land and climate change are reflected in several SDGs. For example, UNCCD is linked to seven SDGs, UNFCCC is linked to nine SDGs, and chemical conventions are linked to six SDGs (UNEP, 2016).

5 Land degradation neutrality

It is important to discuss the causes of land degradation and also how to neutralize the impacts. Land degradation will effectively reduce the useful amount of land available for ecosystems, biodiversity and other living beings. For example, built up land or urban infrastructure implies permanent loss of land and their ecosystem services (Maes et al., 2015) where the concept of LDN is not applicable or not achievable. Urban vegetation including vertical gardens and roof top gardens may bring some nature that will improve the greenery, but the loss of soil or permanent cover of soil will lead to permanent loss of primary ecosystem services such as soil formation (Jayakody et al., 2023). Increasing built-up land and pavements and disturbed landscapes will reduce the water that is infiltrated to the ground and increase the runoff and degrade further the quality of the water that is added to the waterways (Kriech and Osborn, 2022).

Other transformed landscapes, such as agricultural and plantation areas, when managed under organic conditions, imply higher ecosystem services. However, under chemically intensive conditions, they imply a permanent loss of ecosystem services (Kremen and Miles, 2012) and non-achievement of LDN, since such chemicals act as stock pollutants and tend to stay in the soil for a long period of time. When the soils are contaminated with POPs, such soils cannot be restored because effective removal of the POPs is extremely costly. Achieving LDN in such contexts is therefore a non-attainable goal. Certain pesticides that are classified under POPs could have contaminated large extents of tropical soils before their ban under the Stockholm Convention.

Materials considered as chemically inactive such as plastic will be detrimental to earthworms and other soil microorganisms. Microplastics and nanoplastics contaminated soil cannot be reversed to the original situation and currently most agricultural soils are faced with this problem due to the higher and higher use of plastic-based materials in agricultural areas (Serrano-Ruiz et al., 2021; Scopetani et al., 2022). Such soils will be in a permanent degraded state and any productivity will have to be achieved with very high level of externally supplied inputs yet with uncertainty in outcomes. Extremely high costs associated with of such type of LDN may not be cost effective given the limited resources available in low-income tropical countries. It is worth noting here that the investments in sustainable land management (SLM) have been low in the developing countries where the impact of land degradation are most crucial (Chen et al., 2022).

Even the most common type of land degradation caused by soil erosion is assumed to be recoverable when the replacement is done for the lost nutrients, organic matter and macro and microorganisms (Lal, 2015) known to occur in the original soil. However, due to uncertainty of such information, it may not be fully recovered. For example, information on types and numbers of soil microorganisms present in a soil is extremely difficult to find and the replacement will be incomplete.

6 Carbon neutrality, the need and approaches

Carbon neutrality, a state of net zero carbon emission is proposed to be achieved with decarbonization strategies. A hierarchical approach has been defined to achieve this. The first in the list is avoiding carbon intensive actions, the next is reduction of carbon emitting activities by efficiency improvements. Replacing carbon intensive activities with alternatives is the third option and finally, offsetting any leftover emissions that are unavoidable is suggested (Finkbeiner and Bach, 2021). Carbon neutrality can be defined for a country, company, product, activity, or at individual level and the total emissions emitted directly or indirectly has to be balanced by offset or removal mechanisms (UNFCCC, 2021). Carbon neutrality has to be achieved involving all sectors of the economy (Nkonya et al., 2016b). Figure 1 indicates options available for LDN and CN and their interrelations.

Proper forest management can effectively improve the carbon storage of the forests. Examples include different rotation periods (Hektor et al., 2016), use of different tree species especially those with high carbon storage potentials (de Morais et al., 2019), enriching with lianas and climbers (Shukla et al., 2020) and different forest management measures that could improve soil organic carbon (Ma et al., 2021). Agricultural crops when grown under organic conditions,

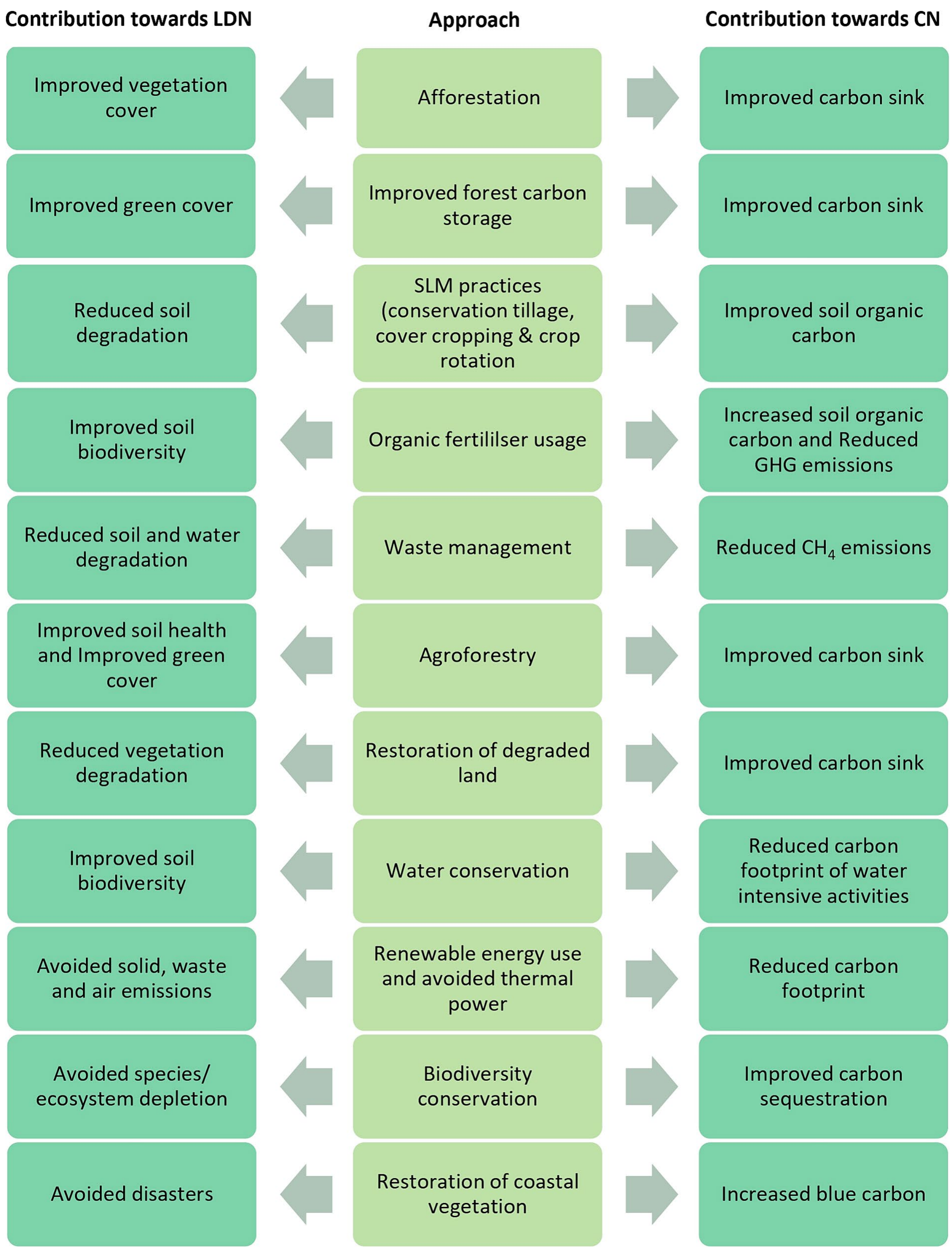

FIGURE 1
Approaches to achieve LDN and CN.

will bring both LDN and CN benefits. Options such as agroforestry will improve soil health, carbon sinks and bring additional benefits to the farming communities. Water conservation will improve soil biodiversity thus generating positive effects toward LDN while reducing carbon footprints. Use of renewable energy will also reduce the carbon footprint with lesser emissions deposited in land thus minimizing land pollution. Moreover, to achieve net-zero carbon emissions and sustainable development, sequestration in terrestrial and marine ecosystems must be promoted (Cheng, 2020). In addition, deploying negative-emission technologies at large scale, promoting regional low-carbon development and establishing a nationwide "green market" have been proposed for China (Liu et al., 2022). Carbon sink technology is another option for CN. Carbon sink refers to the process of absorbing carbon dioxide in the atmosphere through afforestation, vegetation restoration and other measures. It generally consists of terrestrial carbon sinks and ocean carbon sinks (Wu et al., 2022).

The above options indicate range of opportunities with differing efficiency in addressing LDN and CN. As such, a combination of approaches would be useful in addressing land degradation and carbon emission issues in a given socioeconomic and ecological landscape.

Increment of soil organic carbon (SOC) sequestration is considered as a possible solution to mitigate climate change as well as to reduce the land degradation. A study done by Minasny et al. (2017) reports SOC stocks from 20 regions in the world. The top 1 m layer of soil contains about 600 Gt of carbon and if SOC stocks are increased by 0.4%, it can mitigate about 30% of global GHG emission. The study reports mean SOC stocks for each country and how much SOC sequestration rate is required to achieve the 0.4% initiative and also it reports opportunities available to sequester more carbon. This finding has been translated into an initiative named "4 per mille Soils for Food Security and Climate" that was launched at the COP21 of the UNCCD to increase global soil organic matter by 4 per 1,000 (or 0.4%) per year. In order to encourage better management practices among farmers that sequester more carbon, economic incentives could be provided such as direct payments, tax concessions and emission trading tools.

7 Synergies between land degradation neutrality and carbon neutrality

There are clear synergies between LDN and CN as indicated in Figure 1. Synergistic implementation of the neutralities will reduce the total cost and the need for many different expertise. LDN actions will always result in multiple benefits including socioeconomic benefits. Soil organic carbon is an indicator for LDN and therefore many countries have established links between LDN and National Determined Contributions (NDCs). There are several synergistic sectoral impacts. Carbon neutrality will result in positive health outcomes, poverty alleviation, and improvements in national and global security and in the economy.

7.1 Carbon neutrality and health links

There are positive health outcomes from achieving CN. Achieving CN by decarbonizing energy sector, for example, could result in cleaner air which can bring large improvement to the human and ecosystem health. Achieving a net-zero status by the year 2050 will result in a decrease in pollutants like particulate matter (PM), ozone, PM precursors, nitrous oxides (NO_x), sulfur dioxide (SO_2), and other harmful air pollutants (Kerry and McCarthy, 2021).

7.2 Carbon neutrality and poverty link

Many of the climate extreme related costs are mostly borne by the low-income countries and low-income communities (EPA, 2021). Among thermal energy dependent countries, large amounts of NO_x and SO_2 are inevitable. Reductions of such pollutants will result in significant productivity increases of the workforce as a result of health improvements. It has been found that air pollutants can affect educational attainment and thus could result in lowered labor productivity (Zivin and Neidell, 2018). Extreme weather events can also bring disruption of critical health care and such impacts are mostly felt by low-income communities (Mach et al., 2019). Carbon neutrality achieved through LDN provides cost effective solutions for both.

7.3 Synergy between carbon neutrality and economy

Projections from the USA economy shows that avoided damages from fewer deaths, less damage to infrastructure, and fewer lost wages could be $49 billion/year in 2050 if 1.5°C-compatible scenarios have been adopted (Kerry and McCarthy, 2021). For India, net zero will result in net increase in employment opportunities, creating about 15 million jobs beyond a baseline scenario by 2047. Households could save as much as $9.7bn in energy costs by 2060 (ASPI, 2022).

7.4 Synergy between carbon neutrality and security

LDN and CN together could establish national and global security. Continuous disasters drain national financial and infrastructure resources leading to national financial insecurity. More frequent diversion of military assets and personnel to assist and recover the disaster affected regions could result in risks to the national security (Kerry and McCarthy, 2021). In addition, extreme climatic events could bring additional conflicts within the same community and between communities and between nations (Mach et al., 2019).

7.5 Enhancing synergies and minimizing negative feedbacks

In order to enhance the synergies, it is important to adopt actions at different levels. First, it is important to establish linkages within and between biophysical, biogeochemical, and socioeconomic interactions. Second, in order to identify the priority response actions and policy responses, vulnerabilities need to be identified. Thirdly, exchanging the knowledge among stakeholders at various levels and integrating different knowledge systems (e.g., indigenous, citizen science), and co-generating new knowledge, (Raymond et al., 2010; Reed et al., 2011) are essential in fast tracking the response strategies. Finally, innovation is needed to adapt with the changing circumstances (Webb et al., 2017).

When things operate in opposing directions, it implies negative feedback mechanisms. It is therefore important to understand any such negative feedbacks prior to proposing more resilient solutions. Impacts of climate change could lead to desertification and abandonment of lands. Climate change could accelerate land degradation. For example, more frequent droughts, changes in soil properties and vegetation growth can induce land degradation. Therefore, mitigating climate change will inevitably mitigate desertification, too (Reed and Stringer, 2015).

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In order to minimize land degradation and climate change negative feedbacks, four core multi-level actions could be adopted. These include, establishing links between land degradation and climate change impacts, identifying most vulnerable systems, improving knowledge and investigate policy options. Reducing emissions from forest degradation and deforestation (REDD+) projects offer a "triple-win," encompassing climate change mitigation, biodiversity conservation, and the well-being of local communities (Siril et al., 2022).

7.6 Synergies among multilateral environmental agreements related to land and climate change

In 2016, the Intergovernmental Panel on Climate Change (IPCC) agreed to create a special report on desertification, land degradation, and climate change, which would complement the Sixth Assessment Report (AR6). Coordination among the UNCCD, UNFCCC, and UN Convention on Biological Diversity (UNCBD) has been improved to identify and harness synergies in response to land degradation and climate change (Chotte et al., 2019). There is further need to integrate chemical related conventions to identify their linkages with land and climate related conventions.

8 Modeling approaches for land degradation neutrality and carbon neutrality

Analysis of a complex problem is easily done with models as they represent the reality in a manageable scale. Understanding the impacts of land degradation and effectiveness of LDN and CN is best done with modeling approaches since they are capable of characterizing impacts that span across much wider spatial and temporal scales. The simplest and the oldest type of models were soil erosion models which provide considerable amount of information toward land degradation from a conventional point of view.

The next type of models has been GIS based maps which are mostly supported by remotely sensed data which provides useful source of information on extents and the severity of land degradation at national and global levels. For example, degradation hotspots among major land cover types were identified using biomass productivity as an indicator of land degradation (Le et al., 2016). This type of information could be verified with ground-based measurements (Anderson and Johnson, 2016).

Recent IPCC reports have illustrated a variety of scenarios, pathways and models that explore future emissions, climate change related impacts and risks, and possible mitigation and adaptation strategies. Most common type of scenarios is Shared Socio-economic Pathways (SSPs) that cover a range of possible future development of anthropogenic drivers of climate change. Under this, the very high GHG emission scenarios (SSP5-8.5) assume CO_2 emissions that roughly double from current levels and the very low scenario (SSP1-1.9) assumes CO_2 emissions declining to net zero around 2050 (IPCC, 2023). In addition, Representative Concentration Pathways (RCPs) were used by the Working Group I and Working Group II of the IPCC to assess regional climate changes, impacts and risks. There have been several models of combining land degradation, LDN and CN scenarios around the world and Table 1 provides a summary.

Modeling approaches provide useful basis toward assessment of degradation and restoration effectiveness. In order to design restoration options, it is required to assess the status of degradation. In order to assess the effectiveness of restoration efforts and level of achievement of LDN and CN, there is the need for accurate measurements. The assessment of degradation is, however, still based on conventional approaches, taking mainly the soil erosion as the main cause of degradation. Soil pollution aspects have rarely been considered in modeling, which poses a greater challenge in restoring majority of landscapes in the coming years.

9 Economics of land degradation neutrality and carbon neutrality

Economics is about allocation of scarce resources with a view to maximize economic efficiency. Environmental and social concerns have become essential components of such analyses lately. Economic efficiency implies maximizing net benefits or minimizing the cost. In the context of LDN or CN, economic analysis becomes an important decision-making tool in allocating scarce resources toward land degradation and carbon emissions while selecting the most efficient option. The essential first step of such an analysis is to identify and to quantify the costs and benefits of each option and then estimate monetary values (Mishra and Rai, 2014). Whenever the full range of monetary estimates are not available, multicriteria analysis could be adopted to overcome issues associated with monetary estimates (Imbrenda et al., 2021). Cost benefit analysis is the most promising approach in evaluating various options applying monetary estimates since it can incorporate the temporal dimensions also to the analysis. Monetary estimates related to LDN and CN could provide a strong basis for implementing economic instruments such as taxes and subsidies.

A variety of estimates available on land degradation is summarized by Nkonya et al. (2016b) using various case studies across several countries. Those studies highlight that preventing land degradation in the first place is much cheaper than letting the damage happen and repairing it later. On average, one USD investment toward restoration of degraded land gives a return of five USD. This stand as a strong incentive for taking action against land degradation (Nkonya et al., 2016d).

10 Challenges

There are several types of challenges in achieving LDN and CN. The complexities associated with understanding the land degradation, uncertainties associated with carbon dynamics and vulnerable socioeconomic situations in developing countries bring the challenges that need to be addressed. First, identification of degradation is a challenge when the complex nature of the current degradation types is considered. For example, although the traditionally considered degradation such as soil erosion are prominent, and can be measured easily in the field context, most of the chemical pollution related degradation are not visible to the naked eye. Specific techniques, assessments and models are required to

TABLE 1 Examples of land degradation and climate change related models.

Source	Model/s	Types of scenarios	Findings
Chen et al. (2023)	Multi-objective land use and land cover (LULC) optimization coupled model with CN objective Integrated valuation of ecosystem services and trade-offs (InVEST) model with IPCC inventory methodology Non-dominated sorting genetic algorithm-II (NSGA-II) and patch-generating land use simulation (PLUS) model	Four LULC scenarios: natural development (ND), low carbon emissions (CE), high carbon storage (CS), and carbon neutrality (CN)	Compared to ND scenario, the LULC patterns within the CE, CS, and CN scenarios exhibit higher LULC values and contribute more to CN
Jones et al. (2023)	Integrated model—FABLE calculator	Four scenarios: status quo, improvements on current trends, land sparing and land sharing.	Land use and agricultural sector are net carbon sinks in both land sparing and land sharing pathways,
Li et al. (2023)	Linear programming model (LPM), Markov, future land use simulation (FLUS), emission coefficients and InVEST	Four scenarios; natural development, spatial planning, low-carbon emission, and high-carbon storage by 2035	Optimized land use patterns in the low-carbon scenarios will result in a greater reduction in carbon emissions and a larger increase in carbon sinks than the spatial planning scenario
Liu et al. (2023)	PLUS	Three land-management scenarios developed and simulated for 2020–2060	Protecting and regenerating forests are more effective than afforestation in lowland tropical areas for storing carbon
Wang et al. (2023)	Land use structure optimization (LUSO)	Carbon neutral scenario and baseline scenario	Under carbon neutral scenario, LULC is more moderate, aggregation degree of the overall landscape spreading degree is increased
Wang et al. (2022)	PLUS	Coupled SSP and RCP scenarios (SSP119, SSP245, and SSP585)	Zone-based management with LULC regulation lead to CN
Udayakumara and Gunawardena (2022)	InVEST sediment retention model; Digital elevation model	Three scenarios: status quo, three land-use intervention scenario with 3 Soil and water conservation (SWC) intervention:	SWC for the watershed reduces the soil erosion rate by 23%. Implementing SWC by farmers requires payment transfers from the fertilizer importing authority.
Williams et al. (2021)	Modeling the entire U.S. energy and industrial system with new analysis tools	Eight deep decarbonization scenarios: expand renewable capacity 3.5-fold, retire coal, maintain existing gas generating capacity, increase electric vehicle and heat pump sales	Actions required in all pathways were similar
Nkonya et al. (2016c)	Uses Landsat data to examine land use change and its impact on sediment loading in hydroelectric power plants	–	Determining the returns to SLM

determine the degradation status and this becomes a challenge in tropical developing countries.

Lack of relevant data could pose another challenge in modeling (Baumgartner and Cherlet, 2016). Correct projections of land degradation toward future requires a large amount of good quality data and these are largely not available or difficult to generate in developing countries. The most appropriate remedial measures are also outcomes of science and technology for the most part and hence expensive. This can be another challenge in identifying and prioritizing suitable remedial measures. Importance of local traditional knowledge can ease the situation to a certain extent, but the complex degradation causes such as chemical pollutants have not yet generated a set of traditional knowledge outcomes and cannot expect those to arise in the near future.

It is important to recognize clearly whether we are moving in the correct pathways in adopting remedial measures since some of remedial measures would involve longer time frames. One should be able to recognize the milestones that ensure the movement in the correct direction. In such contexts, modeling has a role to play and lack of expertise and quality data will pose a challenge for the developing countries.

There is an obligation toward implementation of NDCs under the Paris agreement. However, LDN is not obligatory. Among the NDCs also, there are voluntary components and non-voluntary components which will only be implemented with financial assistance. The larger the non-obligatory component, it is difficult to expect that land and carbon neutralities are priorities of national governments and hence may largely result in non-adoption. This is further exacerbated by the frequent risks associated with many tropical Asian developing countries for example, internal conflicts, financial crises, debt burdens, poor governance (Khan and Al Shoumik, 2022) and disasters which are often the result of climate change (Webb et al., 2017).

One of the biggest challenges is to establish and demonstrate links between LDN and other sectors of the economy. For example, improved LDN and CN may generate forest outputs, agricultural outputs, and better health outcomes as indicated under the Section 7. The main subsequent challenge is to establish quantitative links in physical terms and monetary terms between LDN measures and associated benefits and CN measures and associated benefits. For example, there is a monetary estimate available for social cost of carbon, indicating how much global damage is caused by a ton of carbon emitted to the atmosphere. Similarly, one unit of land degradation reversed could be associated with some x units of agricultural or y units of forest product improvements. Establishing such linkages for different land use and ecosystem types would be a next challenge. Establishing values for other sectors could help the decision making in identifying the best LDN or CN measure that contributes toward other overall of the economy.

11 Discussion

Assessment of degradation of land and proposing the level of intervention needed to restore requires information at various levels. First, the level of degradation and the type and level of restoration need to be assessed. Biophysical assessments are the first level of information which can also be model based. Secondly, it is required to assess whether the society is willing to pay to full cost of restoration by looking at full range of costs and benefits of the operation. Economic analysis is important in this respect. Developing countries in the tropical belt may be restrained by the information and the expertise in making such judgments and therefore LDN may remain as a distant prospect.

Carbon neutrality when achieved together with the LDN may present a win-win case for the poor countries. However, when it is not possible, countries may have to look for financial transfer mechanisms that could provide support for achieving NDCs. This may require a more detailed analysis with emphasis on each and every individual NDC being subjected to an in-depth analysis under the local socioeconomic and technical knowhow conditions.

It is worthwhile understanding the best approaches available both biophysical and policy contexts. Carbon stored by forest tree planting can generate carbon credits which can be sold in the Emissions Trading System (ETS) market (van der Gaast et al., 2018). This is further facilitated by carbon accounting standards that are available and global agreements that are encouraging carbon related payments. However, inclusion of forests in ETS schemes around the world has been complicated due to issues of carbon leakage, permanence and complexity of accounting. International initiatives such as 4 per mille will face challenges during their implementation due to lack of data, limitation of soil sinks and issues related to resource poor farmers and small land holders (Lal, 2016).

Agroforestry involves establishing trees, mostly forest trees in croplands and silvopastoral systems. It enhances carbon sequestration (Zomer et al., 2016) since number of trees in a unit area is higher and it utilizes vertical space. Successful implementation of agroforestry systems toward CN or LDN require careful selection of agroforestry species, monitoring of carbon dynamics and finding suitable financing mechanisms. In the city context, achieving both LDN and CN could be possible with urban forestry and green infrastructure. Maintaining forest cover in urban contexts provides variety of ecosystem services (Khan et al., 2022) including human health benefits.

Complex problems such as climate change and land degradation can be tackled by implementing a full spectrum of complementary policies across multiple sectors rather than relying on any single policy of single sector. Implementing LDN and CN simultaneously would bring multiple benefits since it minimizes the effort and resources required. In the future planning, LDN and CN can be inbuilt into single projects so that it will minimize the efforts of experts, resources and land requirement.

One of the limitations of the study is that majority of modeling studies were available from limited locations. For example, mainland China dominated the modeling studies in the literature. In order to overcome these issues, it is essential that studies are conducted covering a wider geographical context.

12 Summary and conclusion

This paper intended to review the concepts of land degradation, LDN along with CN emphasizing the degradation types, approaches, models available to analyze, synergies, economic aspects and challenges. Different degradation types result in different consequences including short- and long term and sometimes persistent impacts. Understanding degradation as well as monitoring the mitigation requires proper models that are built upon good quality physical data and supported by quantitative economic analysis. Efforts toward LDN and CN may generate multiple benefits across national and global scales. In order to synergize LDN and CN actions with current activities, such benefits need to be quantitatively linked with necessary policy instruments.

The study highlights the need to go beyond the traditional degradation types, the role of different approaches specially in reaching synergies between LDN and CN in order to minimize costs. Such synergies are best brought to light by meeting the challenge of establishing and demonstrating links between LDN and other sectors of the economy in a quantitative way using both physical terms and monetary terms. One unit of land degradation or carbon neutrality could be associated with some x units of another economic sector, be it agriculture, human health, or national security and there is a need to establish such linkages for different land use and ecosystem types in a move toward a carbon neutral and land degradation neutral earth.

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Supplementary material

The Supplementary material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2024.1398864/full#supplementary-material

SUPPLEMENTARY FIGURE S1
Review protocol.

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Seed ecology and seedling dynamics of western Himalayan treeline tree species

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Several high-elevation plant species would experience an increased risk of regional extinction due to various climatic and anthropogenic factors. Information about the effects of climate change is urgently needed for modeling vegetation dynamics because it influences the various seed parameters like seed germination, seed maturation, seed mass, and seed bank in the soil. The present study was conducted at an elevation of 3145–3560 m in the treeline area of the western Himalayan region of India. The change in seed color is correlated with other seed parameters such as seed moisture content, seed germination, seed mass, and seed fall density. A decline in moisture content in maturing seeds is closely related to seed maturity ($p < 0.05$). *Quercus semecarpifolia* contains the highest seed mass followed by *Abies spectabilis*. Reportedly, the species with higher seed mass have an advantage in light-restricted environments for seed germination and seedling development. In addition, the fruit mass was observed to be the highest for *Rhododendron campanulatum*, while both *Betula utilis* and *R. arboreum* had similar fruit mass. The seed fall density varied between 1.55 and 7.85 seeds m^{-2} and the maximum mortality of up to 32% of seedlings was observed during post-monsoon season from November to February. The potential disruption in the timing of seed fall, soil seed bank, and seed germination due to climatic irregularities has broader implications for forest ecosystems. Generally, the soil in treeline areas gets frozen during winter, resulting in seedlings facing severe water stress and a high rate of transpiration. The present study addresses the issue regarding the survival and proliferation of important treeline species in the western Himalayan region of India.

KEYWORDS
seed maturity, seedling dynamics, seed mass, soil seed bank, treeline

1 Introduction

Treeline forms one of the most prominent ecologically significant boundaries in the Himalayan arc that marks the upper limit of the forest vegetation and represents an ecotone between the closed canopy forest and the alpine zone (Maletha et al., 2020; Singh et al., 2023). In the recent past, treeline zones have been identified as sensitive areas to environmental change and could be effectively modeled and monitored as an indicator for climate changes at regional and global levels (Maletha et al., 2020; Singh et al., 2019a). In the higher elevation, the impact of climate change is considered the most pronounced and severe risk to species regeneration, survival, diversity, distribution, phenology, physiology, and seed ecology (Singh et al., 2019b). Although, several studies have been conducted on vegetation structure and composition, plant communities along altitudinal gradients, resource utilization patterns, anthropogenic pressure, and the impact of climate change on Himalayan vegetation (Maletha et al., 2023), only a little attention paid to investigating the seed-related studies of treeline species in the Himalayan region.

In recent years several high-elevation or alpine plant species experienced an increased risk of regional extinction due to climate change (Dirnböck et al., 2003; Singh et al., 2021). Species differently response to warming either immediately or after some years. With the recent warming, several plant species showed early fruit ripening, particularly in the spring season, which has a significant impact on plant recruitment and population dynamics (Tewari et al., 2019). Information on regeneration under climate change is urgently needed for modeling vegetation dynamics (Leishman et al., 1992; Ibáñez et al., 2007; Morin and Thuiller, 2009) because it influences seed germination via seed maturation and/or seed mass and/or seed bank in the soil. High and low temperatures and lower amounts of water in soil imposed on parents influence the phenotypic expression of maturation, viability, seed mass, seed longevity, dormancy, germination percentage of seeds, and early growth in tree species, over more than one subsequent generation (Kochanek et al., 2010).

Climate has a dominant influence on several life-history traits of plant species (Bernareggi et al., 2016). Among plant reproductive phases, seed germination and seedling establishment are probably the most sensitive to variation in climate conditions (Walck et al., 2011). In seasonal climates, characterized by cyclic variations in temperature and precipitation, seed germination is usually synchronized with the changes in environmental conditions, being delayed until a favorable period occurs (Fenner and Thompson, 2005; Baskin and Baskin, 2014). Seed development and maturation is a process comprising a series of morphological, physical, physiological, and biochemical changes that occur from ovule fertilization to the time when seeds become physiologically independent of the parent plant (Delouche, 1971; Sripathy and Groot, 2023). Physiological maturity is identified as maximum seed dry mass accumulation. Seed maturation is one of the main factors of seed quality and a prerequisite for successful germination and emergence (Sripathy and Groot, 2023). After fertilization, the moisture content of seeds increases throughout the early stages of development before starting to decrease until environmental conditions are in balance (Sripathy and Groot, 2023).

The strong correlation between climate and plant regeneration from seeds has resulted in the evolution of specific germination requirements across many species (Fenner and Thompson, 2005; Baskin et al., 2000), which play a key role in plant distribution and vegetation dynamics (Thuiller et al., 2008). The effects of climate change on the early life-history stages of plants from high altitude and high latitude environments have recently become a focus of researchers (Briceño et al., 2015). Historically, it was believed that sexual reproduction in these ecosystems was rare and less important than clonal reproduction because of the harsh environmental conditions (Billings and Mooney, 1968; Hoyle et al., 2013). However, recent studies have found important persistent soil seed banks (Hoyle et al., 2013; Venn and Morgan, 2010), high rates of natural seedling recruitment (Venn and Morgan, 2010; Forbis, 2003), and considerable gene flow among populations (Jonsson et al., 1996; Pluess and Stöcklin, 2004), which suggests that recruitment from seed is common and plays an essential role in high altitude and high latitude community dynamics (Briceño et al., 2015). This study aims to investigate the physiological maturity of fruits and seeds, soil seed banks, and seedling dynamics of five major treeline-forming species in the western Himalayan region which somewhere divulges directly or indirectly into the seed ecology-related issues of treeline species that are rarely investigated such as; how global warming affects treeline species germination and survival, their seed maturation timing, what temperatures are needed for seed development, dormancy break and how does precipitation affect germination.

2 Materials and methods

The present study was conducted at Tungnath treeline areas, situated in the western Indian Himalayas. The investigated site is located between 3145 and 3360 m above sea level at $30^{0}29'45{}^{1}/_{2}$N latitude and $79^{0}13'24{}^{1}/_{2}$E longitude and falls within the sub-alpine and alpine zones (Figure 1). The soil in these regions has a characteristic brown hue and possesses a sandy loam texture, characterized by a high concentration of sand and silt particles, and exhibits acidity, as indicated by pH values ranging from 4.0 to 5.0. The climate of the research region is impacted by the monsoon, which is characterized by extended periods of harsh winters and brief periods of mild summers. In the study sites, the monthly temperature varied between -6.02 ± 0.23 and 13.71 ± 1.03 C and the monthly precipitation was 13.0 ± 1.16 and 541.0 ± 4.37 mm during at study period between 2017 and 2020 (IHTP, 2021). The study was conducted in five major treeline-forming species, *A. spectabilis, B. utilis, Q. semecarpifolia, R. arboreum,* and *R. campanulatum.*

A designated area of 100×100 m was demarcated at the study site. Within this area, a total of 25 mature and healthy trees of each studied species were randomly selected and marked of approximately similar height and diameter, having a sufficient number of fruits and seeds. The fruits/seeds were directly collected manually from the previously marked trees. The color of fruits/seeds was manually observed and analyzed by Pantone color chart. The physical parameters of fruits and seeds, size (mm^2) were measured using a digital vernier caliper, weight, and mass (g) were measured using a digital electronic balance (Singh et al., 2023; Mittal et al., 2020). The moisture content of fruits/seeds was calculated by comparing their actual weight to their dry weight, dried at 103 ± 2°C for 16 ± 1 h (International Seed Testing Association, 1981). The germination experiment was conducted in a dual chamber seed germinator for each collection date (Singh et al., 2021). The petri-dish and germination paper were sterilized at a high temperature (130°C) for 4 h to make it free from fungal infection. The germination of seeds was carried out at 25 ± 1°C on the top of the seed germinator paper under dark conditions

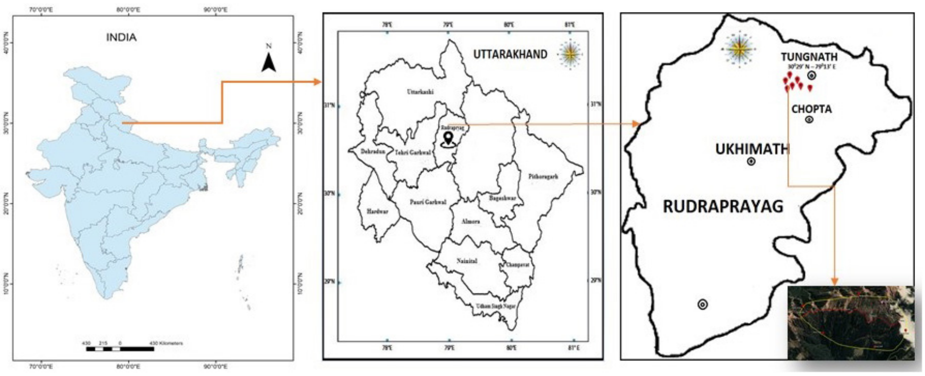

FIGURE 1
Map of the study site (Map of India source: Survey of India, Satellite Map of the study site source: Google Earth).

in the seed germinator. Water was added as required during the experiment. After completion of the experiment, germination percent was calculated as the total number of germinated seeds out of tested seeds within the test period. The seed fall density, soil seed bank, and seedling dynamics assessment were carried out in different locations such as, under the canopy of trees, between the canopy of trees, and inside the canopy of trees at the study sites (Singh, 2019). The seed fall density assessment was observed in 20 seed traps of 1 × 1 m size from the different selected locations at the study site. Further, any loss of fruits/seeds due to any reason was neglected at the study site. For seed fall density assessment, the seeds were collected from the trap every week (*Q. semecarpifolia* and *B. utilis*) and fortnightly (*A. spectabilis, R. arboreum,* and *R. campanulatum*) during the seed fall season (Mittal, 2018), the fallen seeds were counted to calculate for seed fall density. For assessment of the soil seed bank, five soil samples from each 1 m^2 in each transect sample, soil monoliths 25 × 25 × 15 cm^3 were extracted from three depth classes 0–5 cm, 5–10 cm, and 10–15 cm in 20 quadrats of 5 × 5 m with a sub-plot of 1 m^2 located in the middle of each quadrate from different locations at the study sites, after the seed fall season. From each soil depth class larger-sized seeds could be separated manually from the soil and soil excluding large seeds was kept depth-wise in a germination tray. To assess the seed density per m^2 seed germination experiment was conducted. Newly germinated seedlings were identified and recorded (Mittal et al., 2021). To assess the seedling dynamics, newly recruited seedlings were tagged individually in the field on a 1 m^2 plot from 20 different locations at the study sites, and the survival/mortality of the tagged seedling was subsequently monitored (Singh et al., 2021). The data were subjected to analysis of variance with a 95% confidence level using SPSS version 2016. The correlation coefficient (r) was used to express the strength of the relationship between variables.

3 Results and discussion

The study emphasizes the unique ecological context of the Himalayan treeline region, where seed maturation is synchronized with the onset of the monsoon. This synchronization is critical for the reproductive success of many forest tree species in the region. The relationship between climate and the maturation process of seeds and fruits is a critical aspect of plant biology and ecological dynamics. Climate change, characterized by shifts in temperature and precipitation patterns, has profound effects on the timing and success of plant reproduction (Totland, 1997). Various mature and immature fruits/seeds can be distinguished in various ways e.g., by color difference, increased firmness, decreased moisture content, specific gravity, and change in physical dimensions (Tamta and Singh, 2018; Jyotsna et al., 2020). Distinct fruit/seed color changes have been associated with seed maturity in several species (Singh et al., 2020a). In this study, the color change from greenish brown to brown of *A. spectabilis* seeds, from dark green to brown of *B. utilis* catkins, from light green to dark greenish brown of *Q. semecarpifolia* acorn and from green to dark green of *R. arboreum* and *R. campanulatum* capsules serves as a reliable indicator of maturity (Table 1). The changes in fruit/seed color from its appearance to maturity showed a significant relationship with other seed parameters such as seed moisture content, seed germination (Tewari et al., 2019; Tamta and Singh, 2018; Jyotsna et al., 2020), seed mass, and seed fall density. Seeds may mature earlier or later than usual in response to changes in temperature, impacting the overall reproductive success of plant species.

Besides color, the other physical parameter that is interrelated with maturity is moisture content. Changes in precipitation patterns, including alterations in the frequency and intensity of rainfall, can directly affect plant water availability. Adequate water is crucial for seed ripening and development, and variations in precipitation can influence seed moisture content (Khaeim et al., 2022). The maximum germination rates under laboratory conditions were observed for *Q. semecarpifolia* followed by *A. spectabilis, R. arboreum, R. campanulatum* and *B. utilis* (Table 1). Generally, fleshy fruits typically increase in moisture content during ripening, while dry fruits experience a decline. The decline in moisture content in maturing seeds in the study serves as a strong indicator of ripeness and maturity (Singh et al., 2023; Singh et al., 2021; Tewari et al., 2019; Tamta and Singh, 2018; Jyotsna et al., 2020; Singh et al., 2020a). The observation from the present study suggests a negative correlation between seed germination and moisture content of *A. spectabilis* (r = −0.13), *B. utilis* (r = −0.71), *Q. semecarpifolia* (r = −0.88), *R. arboreum* (r = −0.93) and *R. campanulatum* (r = −0.74) at 0.05% significant level

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TABLE 1 Physical parameters of fruits/seeds, seed fall density (m^{-2}), soil seed bank (m^{-2}), and seedling survival (m^{-2}) of studied treeline species.

Parameters	*A. spectabilis*	*B. utilis*	*Q. semecarpifolia*	*R. arboreum*	*R. campanulatum*
Fruits/seeds color during ripening	Greenish brown	Dark green	Light green	Green	Green
Fruits/seeds color during the end of maturation	Brown	Brown	Dark greenish brown	Dark green	Dark green
Fruits/seeds moisture content during ripening (%)	35.31 ± .032	61.63 ± 0.34	71.81 ± 0.04	72.32 ± 0.19	70.51 ± 0.88
Fruits/seeds moisture content during the end of maturation (%)	30.64 ± 0.27	36.92 ± 0.42	48.75 ± 0.54	22.65 ± 0.11	28.82 ± 0.52
Fruits/seeds moisture content during maximum germination (%)	30.90 ± 0.47	35.06 ± 0.55	42.31 ± 0.44	25.90 ± 0.14	29.19 ± 0.37
Highest germination of seeds (%)	46.33 ± 0.21	25.67 ± 0.12	63.29 ± 0.41	40.00 ± 0.42	39.57 ± 0.37
Fruits/seeds size during ripening (mm^2)	47.44 ± 1.34	60.24 ± 1.11	133.71 ± 3.02	76.81 ± 1.21	63.71 ± 0.35
Fruits/seeds size during the end of maximum germination (mm^2)	51.50 ± 2.12	114.9 ± 2.15	546.14 ± 5.09	224.24 ± 1.54	235.00 ± 3.21
Fruits/seeds size during the end of maturation (mm^2)	51.50 ± 2.17	114.90 ± 1.62	438.06 ± 2.25	199.75 ± 1.49	235.00 ± 1.53
Fruits/seeds mass during ripening (g)	2.18 ± 0.09	16.30 ± 0.21	116.67 ± 1.21	21.23 ± 0.54	19.13 ± 0.22
Fruits/seeds mass during the end of maximum germination (g)	6.22 ± 0.05	33.27 ± 0.44	531.81 ± 2.65	33.43 ± 0.13	71.15 ± 0.98
Fruits/seeds mass during the end of maturation (g)	6.59 ± 1.11	33.12 ± 0.77	663.30 ± 4.97	32.20 ± 1.23	68.08 ± 2.43
Fruits/seeds fall density during low activity period (m^{-2})	3.75 ± 0.09	1.55 ± 0.05	4.58 ± 0.12	2.56 ± 0.11	2.89 ± 0.09
Fruits/seeds fall density during peak activity period (m^{-2})	7.85 ± 0.81	6.12 ± 0.76	6.54 ± 0.23	4.58 ± 0.21	5.48 ± 0.41
Soil seed bank at 0–5 cm soil depths (m^{-2})	0.78 ± 0.02	2.0 ± 0.04	1.33 ± 0.08	1.67 ± 0.07	1.67 ± 0.09
Soil seed bank at 5–10 cm soil depths (m^{-2})	2.33 ± 0.09	5.67 ± 0.07	0.00 ± 0.00	4.11 ± 0.13	4.0 ± 0.14
Soil seed bank at 10–15 cm soil depths (m^{-2})	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
The mean number of tagged seedlings (m^{-2})	4.4 ± 0.02	4.2 ± 0.05	5.7 ± 0.04	8.3 ± 0.03	7.3 ± 0.04
The mean number of survived seedlings after 24 months (m^{-2})	0.70 ± 0.01	0.30 ± 0.01	0.40 ± 0.02	3.3 ± 0.03	3.7 ± 0.02

($p < 0.05$). This suggests that as seeds mature and moisture content decreases, germination potential increases. This relationship is important for understanding the optimal conditions for seed germination and how moisture content serves as a regulatory factor. In the Himalayan treeline region, there are many forest tree species in which seed maturation is synchronized with the commencement of monsoon, and their seed viability is very low (Tewari et al., 2019). Climatic irregularities like rising temperatures and irregular patterns of rainfall may impact the synchronization between the timing of seed fall and monsoon rains, particularly in *Q. semecarpifolia* which is a viviparous species and coincides with its seed maturation with monsoon rains (Tewari et al., 2019).

The physical parameters of fruit/seed such as size, weight, number, and mass have also been associated with maturation time (Singh et al., 2023). The study showed that the fruits/seed's size and weight continuously increased from ripening to maturation. During maturation, the mean fruits/seeds size and the mean mass of 100 fruits/seeds were calculated for seeds of *A. spectabilis*, catkins of *B. utilis,* acorn of *Q. semecarpifolia,* and capsules of *R. arboreum* and *R. campanulatum* (Table 1). Species attain maximum weight and mass at the time the end of maturation and increase seed quality, including seed longevity (Ramtekey et al., 2022). The continuous increase in fruit and seed size and weight from ripening to maturation, suggests a strategic allocation of resources by plants. Larger and heavier seeds may have advantages in terms of nutrient reserves, potentially increasing the chances of successful germination and seedling establishment (Domic et al., 2020). Among the studied species, *Q. semecarpifolia* showed a higher seed size and seed mass (Table 1). The species that contain higher seed mass may be better equipped to withstand certain environmental stresses and provide a competitive advantage in light-restricted environments for seed germination and seedling development and also seeds with higher quality and longevity may have a better chance of persistence in the soil seed bank, contributing to the long-term resilience of plant populations (Chazdon, 2013). On the other hand, species like *B. utilis, R. arboreum*, and *R. campanulatum*, with smaller and minute seeds, may face challenges during the transition from germination to seedling emergence. Small-seeded

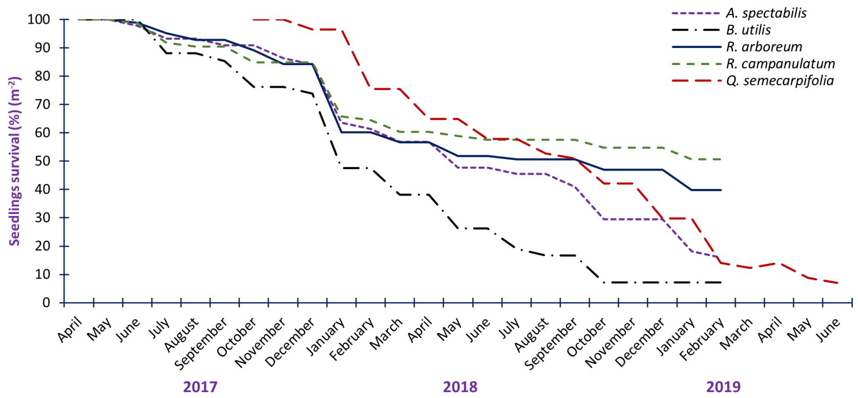

FIGURE 2
Seedling survival percent of studied treeline species during the study period.

species often experience higher mortality rates in this critical stage (Singh et al., 2020a).

Successful management of natural regeneration must use the natural patterns and timing of seed production to maximize the available seed supply to a given site during the period of greatest site receptivity (Nixon and Worrell, 1998). Seed production can vary considerably as a result of tree species, tree age, woodland management, and climate conditions in a particular area (Broome et al., 2016). In this study, the fruits/seeds fall density varied between 3.75 ± 0.09 and 7.85 ± 0.81 seeds m^{-2} in *A. spectabilis* and 1.55 ± 0.05 to 6.12 ± 0.76 seeds m^{-2} in *Q. semecarpifolia,* 4.58 ± 0.12 to 6.54 ± 0.23 catkin m^{-2} in *B. utilis*, 2.56 ± 0.11 to 4.58 ± 0.21 capsule m^{-2} in *R. arboreum* and 2.89 ± 0.09 to 5.48 ± 0.41 capsule m^{-2} in *R. campanulatum* (Table 1). The seed fall density of these species was found very low, as a report on *Q. semecarpifolia* suggests that the species can produce a good seed crop after a few years of gap (Verma et al., 2015). Further, a study on conifers reported the first large cone crop at the age of 15 to 30 years (Broome et al., 2016), as *A. spectabilis* is also a conifer species. While *B. utilis, R. arboreum,* and *R. campanulatum* produce very minute seeds, the germination rate of minute seeds becomes very low (Martínez-Garza et al., 2013). Climate change-induced irregularities, such as rising temperatures and unpredictable rainfall patterns, pose a threat to the synchronization between seed maturation and monsoon rains. The delay or prolonged breaks in monsoons can adversely affect the timing of seed fall and, consequently, the regeneration ecology of forests in monsoonal climates (Negi and Rawal, 2019). Climatic factors such as temperature and precipitation, along with local variations in elevation, aspect, and exposure, play a crucial role in seed production. Favorable climatic conditions support the development of seeds and contribute to higher seed viability. At higher altitudes, cones tend to be smaller, with fewer and lighter seeds that have lower average viability levels. Increasing elevation can also influence the periodicity of good seed years with the intervals between good seed years being longer and more irregular at higher elevations compared with lower elevations (Nixon and Worrell, 1998). The potential disruption in the timing of seed fall and germination due to climatic irregularities has broader implications for the overall dynamics of forest ecosystems, whereas, lower seed fall density and irregular seed production patterns can pose challenges for natural regeneration. These challenges may lead to fluctuations in the abundance and distribution of plant species within forest ecosystems.

A soil seed bank is a major initiator of regeneration in a natural forest, especially in treeline areas where several anthropogenic and grazing pressures are commonly observed (Singh et al., 2019a). Across the treeline sites and species, the number of viable seeds of all species declined with increasing soil depth. The maximum number of viable seeds of all species were present in the topsoil layer while it was completely absent on 10–15 cm soil depths of all species (Table 1). The topsoil layer contains the highest number of viable seeds because the topsoil layer has the maximum number of seeds of the current year and they maintain their viability while at the deeper layer, they lose their viability with time. The overall, soil seed bank of studied species was very low and ranged between 0.78 ± 0.02 and 2.33 ± 0.09 seed m^{-2} of *A. spectabilis*, 2.0 ± 0.04 and 5.67 ± 0.07 seed m^{-2} of *B. utilis*, 1.33 ± 0.08 seed m^{-2} of *Q. semecarpifolia,* 1.67 ± 0.07 and 4.11 ± 0.13 seed m^{-2} of *R. arboreum* and 1.67 ± 0.09 and 4.0 ± 0.14 seed m^{-2} of *R. campanulatum* (Table 1). Several factors such as low density of trees, immature or over mature trees, low seed production, gap in the mast seed year, collection of fruit/seed from the tree, lopping, and grazing could be responsible for minimal soil seed banks. Increased frequency and intensity of extreme weather events, such as storms or droughts, can disturb the soil and affect seed viability and germination.

The regeneration of a forest is a vital process in which old trees die and are replaced by young ones in perpetuity (Malik and Bhatt, 2016). Harsh climatic conditions in the alpine zone might restrict the regeneration and survival of treeline tree species (Tewari et al., 2018). Changes in winter temperatures may impact the freezing and thawing of soil, affecting seedling survival. Over the study period, the seedlings number continuously declined from initial to final observation. The seedling survival percentage (m^{-2}) after 24 months of observation, was 15.91% seedling m^{-2}, 7.14% seedling m^{-2}, 7.02% seedling m^{-2}, 39.76% seedling m^{-2} and 50.68% seedling m^{-2} respectively of *A. spectabilis, B. utilis, Q. semecarpifolia, R. arboreum* and in *R. campanulatum* (Figure 2). The transformation from seedlings to adults is important and therefore the regeneration dynamics is a major thrust area of the study in terms of regeneration and management of forests (Miranda et al., 2018). The maximum 20–32% mortality of seedlings was observed during the winter season from November to February (Figure 2). Generally, the soil in treeline areas gets frozen during winter and the transpiration rate of seedlings becomes high and faces severe water stress this may be the main cause of maximum mortality during winter (Singh et al., 2019b). Periods of low water or drought, intensified by climate change, can induce stress on plants, potentially affecting seed development and maturation. Drought stress might lead to premature maturation, impacting the quality, viability, and survival of seeds and seedlings (Zhang et al., 2009). Severe biotic and high anthropogenic pressure was also responsible for high seedling mortality in treeline areas (Singh et al., 2020b). Across all the study sites heavy grazing was observed during snow snow-free period from May to October during the daytime which can directly or indirectly impact the overall life cycle of tree species.

4 Conclusion

Temperature and precipitation play a significant role in various aspects of plant reproductive and vegetative processes, including seed maturation, seed fall, soil seed banks, and seedling survival in treeline areas. The visible indicators such as color changes in fruits and seeds highlight the complexity of ecological processes influenced by climatic factors. The challenges observed in seedling survival are exacerbated by heavy anthropogenic pressure, particularly grazing, and stressful winter conditions. The low regeneration of seeds and seedling survival of treeline species emphasize the need for adaptive management approaches that consider both natural and human-induced factors. Knowledge of the exact time of seed maturation is essential that help land managers prescribe site treatments that produce desired vegetation conditions for future multiplication. Furthermore, to authenticate the above conclusion, a more extensive period will be required.

Data availability statement

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding authors.

Author contributions

NS: Writing – review and editing, Writing – original draft, Visualization, Validation, Supervision, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. AT: Writing – review and editing, Writing – original draft, Validation, Supervision, Methodology, Conceptualization. AM: Writing – review and editing, Validation, Methodology, Data curation, Conceptualization. SS: Writing – review and editing, Validation, Methodology. MB: Writing – review and editing. SS: Validation, Writing – review and editing, Funding acquisition. MA: Validation, Writing – review and editing. AS: Validation, Writing – review and editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Declining interest in afforestation under the common agricultural policy. Evidence from Poland and Lithuania

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Land afforestation is an important aspect of forested land development. Increasing the area of forest areas through the reforestation of uncultivated, abandoned or agriculturally unsuitable land is considered an important way to diversify economic activities in order to reduce dependence on agricultural activities and improve environmental conditions in rural areas. The main objective of the study is to identify the factors affecting the afforestation of agricultural land carried out in the years 2004–2020 by farmers under the individual financial perspectives of the Rural Development Programme (RDP) in Poland and Lithuania. The study included a review of Polish and Lithuanian regulations aimed at providing financial support for afforestation under the RDP. Moreover, a comparative analysis of the rules and criteria for financial support for afforestation in relation to selected socio-economic indicators of the two countries was carried out. Based on the study results, it can be clearly stated that in both Poland and Lithuania, the support for afforestation under the RDP fails to meet the beneficiaries' expectations. It would, therefore, be advisable to adapt the Programme to the changing economic conditions and keep the afforested land under the RDP under technical supervision. Support for afforestation should be continued to ensure the improvement in land use and the enhancement of the prospects for long-term economic activity in rural areas as well as to implement the assumptions of the green economy.

KEYWORDS

agricultural land, afforestation, common agricultural policy, forest policy, green economy, rural development programme

1 Introduction

In Central Europe, a clear human influence on the forest became evident around the 5th millennium BC with the spread of agriculture and a settled lifestyle (Krawczyk et al., 2021). Currently, forests and other wooded areas cover more than 43.5% of the EU's land area and are essential for human health and wellbeing. Forests are vitally important to us because of their impact on the air we breathe and the water we drink, and due to their rich biodiversity and unique natural system, they provide home and habitat to most species found on land around the world. Not only do they provide a place where humans can feel close to nature and enhance their physical and mental health, but they are also essential to maintaining dynamic and thriving rural areas (New EU Forest Strategy for 2030, 2021).

Currently, the greatest concerns for all users of space worldwide include the overexploitation of forest resources and the trend towards decreasing the area of forests. Forests have long served an extremely important role in the country's economy and are vital to society (they provide job creation, food, medicines, materials, clean water, etc.). For centuries, forests have been a thriving centre of cultural heritage as well as craftsmanship, tradition and innovation. In the past, the forests were important, and they are of crucial importance for our future. Forests are naturally conducive to adapting to and combating climate change and will contribute significantly to Europe becoming the first climate-neutral continent by 2050 (New EU Forest Strategy for 2030, 2021). Kaliszewski (2018) also draws attention to the "state forest policy" that has recently been most focused on the current European forest policy priorities.

The policy of increasing the forest cover at different levels in individual countries is determined by the forest cover of the particular territory, land use traditions, links between forms of ownership, legal practice, administration, geographical features of territories, as well as other factors. In countries with few forests (e.g., United Kingdom, Iceland), programmes for increasing forest cover are being implemented more at a regional level. As the expansion of urbanised areas in Western Europe continues, increasing attention is being paid to the planning of forests and green spaces in these areas (Konijnendijk, 2001). In comparison with the neighbouring countries (Latvia, Estonia, Belarus, Sweden, Finland and Germany), the forest cover of Poland and Lithuania is one of the smallest (Riepšas, 2002). It is worth noting that Lithuania's forest cover is 3% higher than that of Poland. Changes in the economic and land ownership systems in the Baltic States (mainly Latvia, Lithuania and Estonia), from the centrally planned economy to the Soviet Union to the free market and private ownership of modern, newly independent states, have had a considerable impact on land use, especially the balance between forestry and agriculture. In all of the Baltic States, large areas of agricultural land have been abandoned and made available for afforestation over the past decades (Jõgiste et al., 2015).

Recently, there has been much controversy in Poland, as well as in other EU countries, over the European Green Deal (EGD), under which the forest policy is one of the key policies of the EU's environmental reform package. The EGD's main emphasis is on the afforestation of agricultural land, especially soils of low valuation classes, characterised by low suitability for crop production. The EGD will enable Europe to become climate-neutral by 2050. Therefore, one of the priorities of the EU's environmental policy is to promote afforestation, i.e., the establishment of forest plantations on non-forest land, areas unsuitable for agricultural production, or uncultivated land. This takes on particular importance at this time of progressing climate warming and its evident irreversible consequences. Another important document is the New EU Forest Strategy for 2030 (NFS, 2021), which is part of the European Green Deal (EGD), and builds on the Biodiversity Strategy for 2030 (European Commission, 2020), aims to increase afforestation and improve forest health and resilience, as well as exploit the potential of forests which play an important role in the ecosystem. This will be implemented through, *inter alia*, soil protection (mainly against erosion), a reduction in air pollution, involvement in the hydrological cycle, and work for the benefit of the climate (especially through carbon storage). Sierota and Miścicki (2022) predict that the new EU Biodiversity Strategy for 2030 will implement a programme of planting one billion trees with appropriate consideration given to the potential of forests, based on sustainable management principles and the assumption that the future climate will be neutral.

The Common Agricultural Policy (CAP) is the EU's largest programme that distributes approx. 40% of the EU budget to problem areas. Since its introduction in 1962, the CAP has aimed to provide subsidies and programmes to develop agriculture and rural areas. In various periods, different EU Member States had uncommon political priorities. Some of them were more focused on sustainable agriculture, environmental protection or biodiversity, which could have led to less afforestation on new land. EU grants also varied from country to country, affecting the motivation to implement afforestation. Afforestation of agricultural land is one way to develop land of marginal importance for agriculture and is a key measure to achieve the objectives of the National Forest Cover Augmentation Programme. The goal of this program is to increase the forest cover in Poland and optimise land use in accordance with the diverse needs and possibilities of individual regions of the country (Sioma, 2019).

Poland pinned high hopes on increasing its forest cover to 30% by 2020. In the first years of EU membership, there was indeed a great deal of interest in afforestation programmes, which, however, declined in subsequent years. The situation was somewhat different in Lithuania. The European Union comprises countries with very diverse geographical and climatic conditions, which affects their capability to carry out afforestation. For example, Scandinavian countries have a naturally high forest cover but do not allocate additional land for new forests. In contrast, southern European countries such as Spain or Greece may have limited opportunities due to natural and climatic conditions (Mason et al., 2022). Therefore, two neighbouring countries, Poland and Lithuania, were chosen as the study area. It is also worth noting that both countries share similar historical, natural and climatic conditions and joined the EU at the same time.

After 20 years of Poland's and Lithuania's membership in the European Union, it is worth summarising the measures taken to date to increase the forest cover levels of both countries. Therefore, the identification of the factors affecting the afforestation of agricultural land since 2004 by farmers under the different financial perspectives of the Rural Development Programme in Poland and Lithuania, i.e., in the years 2004–2006, 2007–2013 and 2014–2020, was adopted as the main objective of the study.

2 The implementation of afforestation in Poland and Lithuania to date

Land use in Poland and Lithuania has changed over time. Agricultural land has always been the dominant form of land management. In Poland, moderately fertile and poor soils prevail, which may favour a change of land use to a non-agricultural one. Lithuania is located in a forest zone, where the natural state of the

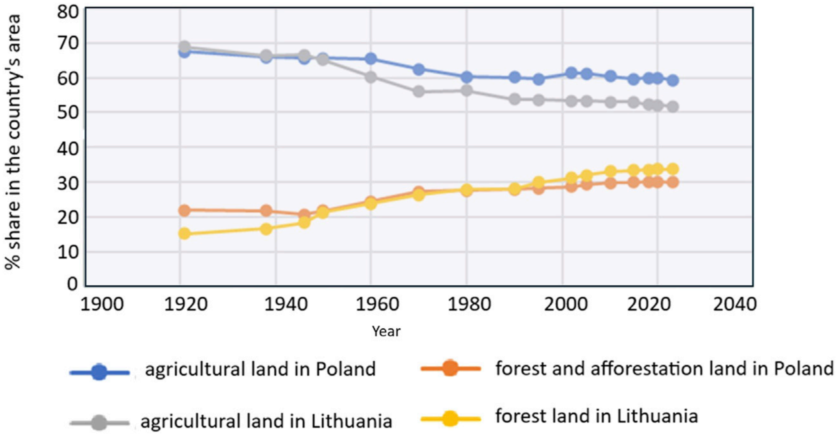

FIGURE 1
The percentage of agricultural land and forests in the total area of the country in Poland and Lithuania in the years 1921–2022.

territory is the forest. However, over time, as a result of human economic activities, Lithuania's forest cover has decreased, and a very uneven forest cover has been established in the country's individual regions, which is related to the fertility of agricultural land. Changes in the percentage of the total area of agricultural land as well as forests and woodlots in the individual years are provided in the graph below (Figure 1).

In Poland, in the past, due to socio-economic processes, mainly the expansion of land for agricultural use, the forest cover decreased to 38% in 1820 and to 20% in 1938. In 1945, the forest cover in Poland accounted for 20.8% (National Forest Cover Augmentation Programme, 2003). Compared to other European countries in which the forest cover was approx. 30%, an increase in the forest cover in Poland has become an objective necessity (Smykala, 1990). In the years 1945–2000, the area of forests and land associated with forest management increased from 6,470 thousand ha to 9,059 thousand hectares, i.e., by 40.0%. During the period, the country's forest cover increased from 20.8% to 28.4%. The greatest volume of afforestation works was noted in the 1960s (with up to 60 thousand ha afforested per annum) (Biczkowski and Rudnicki, 2013; Kurowska et al., 2014). Afforestation carried out on such a huge scale was not solely determined by the agricultural unprofitability of the land under consideration. At the time, the allocation of specific areas for afforestation was determined by the huge supply of undeveloped land (Sobczak, 1996). In the 1980s, interest in afforestation declined as a result of the development of a stable basis for agricultural policy and the equal treatment of all agricultural sectors in Poland (Smykala, 1990). As demonstrated by Szujecki (2003), approx. 30% of Polish forests grow on land that was deforested and then used for agricultural purposes or left fallow. In Poland, there is a large variation in the forest cover between regions, ranging from 20% to 50% (Wysocka-Fijorek et al., 2020a). Biczkowski et al. (2024) also draw attention to the large variation in the needs of wooded areas in the country.

The literature on the subject has repeatedly emphasised the development of forest protective functions in the historical context, both in Poland and in other European countries (Parviainen and Frank, 2003; Referowska-Chodak and Kornatowska, 2021). An example here is an analysis by Zajączkowski (2003), who discusses the importance of forest management principles in developing the sustainable multifunctionality of Polish forests and forestry. The author points out that forests play an important role in protecting biodiversity, regulating the climate, and preventing soil degradation (Zajączkowski, 2003). Klocek (2005) draws attention to the economic aspects of managing multifunctional forests and indicates the difficulties in reconciling different functions of forests, e.g., timber production, nature conservation and recreation. The author emphasises that the development of multifunctional forests requires trade-offs between these functions which involve economic challenges. On the other hand, Wiśniewski (2015), in a study on the anti-erosion function of soil-protective forests, draws attention to the crucial role of forests in preventing soil erosion. The author points out that soil-protective forests provide important ecological functions that are essential for maintaining the health of ecosystems and protecting water and soil resources.

In the present century, the increase in the area of forests in Lithuania has been modest and is largely due to the afforestation of agricultural land. More detailed forest cover indicators for Lithuania were analysed by Professor P. Matulionis (1930), Lukinas (1968), Karčiauskas (1971), Eitmanavičienė (1976), Karazija (1979), Karazija (1988), Pauliukevičius (1982), Pauliukevičius and Kenstavičius (1995), and others. According to the State Forests data, in 1956, the forest cover in Lithuania was 19.7%, in 1966–22.6%, 24.6%, and in 1983–27.9%. Before Lithuania regained its independence in 1990, the country's forest cover was 28.5%. Later, after regaining independence in the year 2000, the forest cover showed an increasing trend. In 2017, it was already at a level of 30.9% (Ivavičiūtė, 2018; Tiškutė-Memgaudienė and Tiškutė-Memgaudienė, 2021). According to the data of the Land Fund of the Republic of Lithuania, in 2023, the country's forest cover was as high as 35.7%.

In 1995, Poland adopted the document National Forest Cover Augmentation Programme, which aimed to increase the area of forests in the country. The programme was updated in 2003 and 2014. According to its assumptions, Poland's forest cover was to increase to 30% in 2020 and to 33% by 2050. It can be said that Poland has been consistently meeting its targets, yet the prospect of

33% in 2050 poses an enormous challenge (Daniłowska, 2019; Kurowska et al., 2020; Kaliszewski and Jabłoński, 2022).

Before its accession to the European Union, Poland had a diverse system of financing afforestation works. State-owned land was afforested by the State Forests with the State Forests National Forest Holding's own funds. The extent of afforestation under implementation depended largely on the amount of funds allocated annually for this purpose in accordance with the adopted forest management policy. In addition to subsidies from the budget, afforestation of private land was supported by funds from Voivodeship Funds for Environmental Protection and Water Management and, to a small extent, by funds from the State Forests in the form of free allocation of seedlings. Between 2002 and 2004, a new system of financing afforestation of agricultural land was applied pursuant to the Act on the allocation of agricultural land for afforestation (Kurowska et al., 2014; Kurowska and Kryszk, 2017).

In Poland, under the RDP, the largest areas of land that were afforested were located in the areas where State Agricultural Enterprises predominated (with a large amount of potential land eligible for afforestation) (Kurowska and Kryszk, 2017). After 2006, there was a sharp decline in the area of afforestation in Poland as a result of a change in the criteria for allocating private agricultural land for afforestation and competition from agricultural subsidies. An equally considerable decline in the volume of afforestation was observed in the State Forests, which was due to a reduction in the area of former agricultural land and wasteland allocated for afforestation by the Agricultural Property Agency (currently the National Centre for Agricultural Support) (Banach et al., 2017; Wysocka-Fijorek et al., 2020b).

Lithuania's forest policy is developed in accordance with the Constitution of the Republic of Lithuania and other legislation. One of the most important tasks of forest policy in Lithuania is to increase the forest cover, which is determined by a combination of legal, organisational, socio-economic and ecological-environmental factors. In Lithuania, before its accession to the European Union, significant areas of agricultural land not used for agricultural purposes were a major problem. In recent decades, the area of abandoned and uncultivated agricultural land has been increasing. Such a trend emerged mainly as a result of the land reform and the adaptation of the agricultural sector to free market conditions. As a result, some agricultural land unsuitable for cultivation has been abandoned, and rapid, uncontrolled renaturalisation processes have commenced on it (Ribokas and Rukas, 2006; Ribokas and Milius, 2001). The process of agricultural land renaturalisation is most often observed in areas less favourable for agriculture, where the largest area of unproductive land is located, and hilly and naturally sensitive areas are predominant. In these areas, the efficiency of agricultural production is significantly lower than the national average.

One of the major factors leading to an increase in forest cover in both Poland and Lithuania has been the integration with the European Union and the possibility of obtaining support from the EU's structural funds. In Lithuania, the main factors determining the extent of afforestation under the RDP include the large area of unproductive abandoned and degraded land, the need to improve ecological and environmental conditions, and the creation of new jobs (Lietuvos kaimo plėtros 2004–2006 metų planas, 2004). By 2004, the largest areas were afforested by the forestry authorities (Lithuanian State Forests). Since 2004, EU funds have been allocated for afforestation. In Lithuania, the establishment of new broad-leaved and mixed forests is the priority (Šepetienė et al., 2014). A forest scenario modelling study and a qualitative analysis of users' needs in Lithuania (Jukneliėnė et al., 2024) confirmed the diverse perspectives, wishes, visions and intentions of key Lithuanian entities involved in forestry with regard to the goals, tasks and core functionality of forest scenario modelling tools.

Increasing the country's forest cover is a complex process, and without the afforestation of marginal land owned by farmers, it will be difficult to meet the Polish objectives of the National Forest Cover Augmentation Plan by 2050 and implement Lithuania's afforestation policy of achieving 38% in 2050. It is, therefore, advisable to adapt the existing programmes and search for support mechanisms that respond to the needs of the national policy implementation. The structure of forest ownership in Poland and Lithuania is not without significance. It is worth noting that in Lithuania, private ownership of forests accounts for approx. 50% (Jukneliėnė et al., 2015), whereas in Poland, it accounts for less than 20% (Źróbek-Różańska et al., 2014).

3 Methodology

After 20 years of the membership of Poland and Lithuania in the European Union, it is worth summarising the measures taken to date to increase the countries' forest cover indicators. Therefore, the identification of the factors affecting the afforestation of agricultural land since 2004 by farmers under the individual financial perspectives of the Rural Development Programme in Poland and Lithuania, i.e., in the years 2004–2006, 2007–2013, and 2014–2020, was adopted as the main objective of the study. In order to fulfil this objective, the following specific objectives were envisaged:

- A comparative analysis of the criteria for applying for afforestation support in Poland and Lithuania.
- A comparative analysis of the financial conditions of afforestation under the individual RPD financial perspectives in Poland and Lithuania.
- A balance of completed afforestation operations under the RDP, taking into account completed afforestation operations (the area by country in particular years) and the support received for afforestation (new commitments as well as commitments from the previous perspective – maintenance premium and afforestation premium).
- The identification of the internal factors (national legislation, development strategies) and the external factors (EU legislation and EU membership obligations).
- Financial and statistical analysis of the level of expenditure on afforestation in the years 2004–2022 in Poland and Lithuania, as compared to the EU.
- Recommendations aimed at increasing the volume of afforestation works, especially on land of little use for agricultural production.

TABLE 1 A comparative analysis of agricultural land afforestation under the particular financial perspectives in Poland and Lithuania.

Specification	Poland	Lithuania
RDP 2004–2006		
Title of the action	Afforestation of agricultural land and of non-agricultural land	Agricultural land afforestation
Afforestation payment period	20 years	20 years
Land eligible for afforestation	land used as arable land, permanent meadows, pastures or orchards	Agricultural land in good agricultural condition
Beneficiaries	Farmers – natural persons	Farmers or associations, and others
Minimum/maximum area requirements	A minimum area of 0.3 ha; no maximum area restrictions	A minimum area of 1 ha; an exception where newly afforested land borders an existing forest, no area-related restrictions
Payment types *	Support for afforestation (a one-off payment) Maintenance premium (5 years) Afforestation premium (20 years)*	Afforestation allowance (a one-off payment) Maintenance and protection allowance (5 years) Compensation for loss of income (20 years)*
Total budget	EUR 84.7 million	EUR 24.05 million
RDP 2007–2013		
Title of the action	Afforestation of agricultural land and of non-agricultural land	The first afforestation of agricultural land and the first afforestation of non-agricultural land
Afforestation payment period	15 years	15 years
Land eligible for afforestation	Land used as arable land and orchards, located outside Nature 2000 sites	land in good agricultural and environmental condition
Beneficiaries	Natural or legal person, or a group of natural or legal persons	Natural or legal persons owning agricultural land and state forest land managers
Minimum/maximum area requirements	A minimum area of 0.5 ha; a maximum area of 100 ha	None
Payment types *	Support for afforestation (a one-off payment) Maintenance premium (5 years) Afforestation premium (15 years)*	Afforestation allowance is paid in the first or second year after afforestation (a one-off payment) Annual allowance for the maintenance and protection of a newly planted forest (5 years) Annual allowance per hectare to compensate for loss of agricultural income following afforestation (15 years)*
Additional criteria, division into schemes	Yes Scheme I - afforestation of agricultural land Scheme II - afforestation of non-agricultural land	Yes 1. Afforestation of agricultural land afforestation of non-agricultural land 2. Afforestation of non-agricultural land - Short-rotation plantations from 6 to max. 15 years
Total budget	EUR 84.7 million	EUR 24.05 million
RDP 2014–2020		
Title of the action	Support for afforestation and establishment of afforested areas RDP 2014–2020	Investments in the development of forest areas and improvement of forest vitality
RDP 2014–2020		
Afforestation payment period	12 years	12 years
Land eligible for afforestation	land registered as agricultural areas or woodland/bushland on agricultural areas	Agricultural land in good agricultural condition
Beneficiaries	Farmers – land owners, local government units (LGU)	Farmers
Minimum/maximum area requirements	A minimum area of 0.1 ha; a maximum area of 20 ha	None

(Continued on following page)

TABLE 1 (*Continued*) A comparative analysis of agricultural land afforestation under the particular financial perspectives in Poland and Lithuania.

Specification	Poland	Lithuania
Payment types *	Support for afforestation (a one-off payment) Maintenance premium (5 years) Afforestation premium (12 years)*	Afforestation allowance is paid in the first or second year after afforestation (a one-off payment) Annual allowance for the maintenance and protection of a newly planted forest (5 years) Annual allowance per hectare to compensate for loss of agricultural income following afforestation (12 years)*
Additional criteria	Criteria for the selection of afforestation operations – a minimum of 6 points	Criteria for the selection of afforestation operations – a minimum of 30 points
Total budget	EUR 301 million	EUR 81.6 million

The analysis covered the afforestation measures taken to date in Poland and Lithuania. In order to define recommendations for future action aimed at implementing afforestation on marginal land that is not very suitable for agricultural production, a comparative analysis of the criteria and conditions for financial support was carried out. This is important in terms of both implementing the national policies of Poland and Lithuania and fulfilling the established objective of achieving climate neutrality by 2050 in the European Union countries.

The analysis was based on the data obtained from the Department for Direct Payments of the Ministry of Agriculture and Rural Development in Poland and Lithuania. The data covered the payments disbursed to beneficiaries that implemented the measure called Afforestation of agricultural and other agricultural lands within the frameworks of the RDP as of 31 December 2022. The Regional Databank data made available by the Central Statistical Office and also those contained in the yearbooks prepared by that Office were used for analyses.

4 Results and discussion

4.1 A comparative analysis of the land afforestation programme and financial conditions under the RDP in Poland and Lithuania

Poland and Lithuania are two countries with similar forest cover. However, the structure and dynamics of changes in the forest resources of both countries show some differences. Below is a detailed comparison based on the data from Table 1.

Since 2004, Polish and Lithuanian farmers have been able to afforest agricultural land, for which they receive support from EU funds under the Rural Development Programme. Over the years, the formal requirements (especially those concerning the minimum/maximum area and land eligibility for afforestation), as well as the rules for financial support, have changed. Basic information on the eligibility rules for afforestation entities and afforestation activities in Poland and Lithuania is provided in Table 1 below.

In Poland, the most favourable conditions were in the first financial perspective, i.e. 2004–2006. The most appealing component of the system was the compensatory premium, as it guaranteed income for 20 years. Similar opinions were also expressed in other countries, such as Spain (Vadell et al., 2019; Segura et al., 2021). In subsequent years, the afforestation premium payment period was reduced from 20 to 15 years. In the years 2014–2020, this payment period was reduced to 12 years (Kurowska and Kryszk, 2017). The actual area that could potentially have been used for afforestation is not without significance. In the years 2004–2006, the requirements were the most liberal. At that time in Poland, there were no area-related restrictions on the area under afforestation (Plan, 2004). Subsequent perspectives introduced area-related restrictions for one beneficiary for up to 100 ha and then for up to 20 ha. The Polish state, having noted the lack of interest in afforestation with area-related limits of up to 20 ha, responded by raising this limit again to 100 ha. In the years 2014–2020, financial support could be provided to afforested areas of up to 20 ha. An additional restriction was the introduction of point-based criteria for qualifying land for afforestation. The applications eligible for the afforestation procedure were those for which the land scored a minimum of 6 points. It is worth noting that the point-based criteria focused on environmental aspects (e.g., areas to be afforested located in ecological corridors, adjacent to surface waters, and bordering the existing forests). As a result, relatively few afforestation works were carried out in Poland under the financial perspective of 2014–2020, compared to previous periods. It is worth noting that economic conditions also changed to the detriment during this period, including, *inter alia*, an increase in the value of agricultural property, including that of poor quality (Klepacka, 2020), an increase in the price of forestry work services, and an increase in the price of planting material) (Kurowska and Kryszk, 2017).

In Lithuania, private landowners also started afforestation works in 2005, after the launch of the Rural Development Programme for the years 2004–2006. In the following years, i.e., from 2007 onwards, Lithuanian forestry received significant support from the European Union under the Lithuanian Rural Development Programme for the years 2007–2013 and 2014–2020. Thanks to the utilisation of the above-mentioned support funds, several thousand hectares of new forests were planted every year in Lithuania. The owners received a fixed payment for the planted forest and could choose whether to carry out the forest planting works by themselves or contract them with others. Compared to 2004–2006, in subsequent payment periods under the programme, the financial support also increased significantly. According to the Ministry of Agriculture data, afforestation of agricultural, non-agricultural and agriculturally abandoned land has increased by as much as six times, having received support from European Union programmes. In Lithuania, as in Poland, a point-based assessment was also introduced to qualify

TABLE 2 Forms of financial support for afforestation and payment rates in Poland under the individual financial perspectives.

Financial perspective	2004–2006		2007–2013		2014–2020	
1 ha per year EUR						
Form of support	Coniferous tree species	Broad-leaved tree species	Coniferous tree species	Broad-leaved tree species	Coniferous tree species	Broad-leaved tree species
Support for afforestation	1000.0	1163.0	1074.0	1218.0	1524.0	1664.0
Maintenance premium	98.0		98.0		98.0	
Protection against game (2 m high metal mesh fencing)	560.0		603.0		2.0 EUR/running metre	
Afforestation premium	- an agricultural producer receiving at least 20% of their income from agriculture – 326.0 - an agricultural producer receiving less than 20% of their income from agriculture – 84.0		- an agricultural producer receiving at least 25% of their income from agriculture – 367.0		283.0 + SAP	

TABLE 3 Forms of financial support for afforestation and payment rates in Lithuania under the individual financial perspectives.

Financial perspective	2004–2006		2007–2013		2014–2020	
In EUR per ha per year						
Form of support	Coniferous tree species	Broad-leaved tree species	Coniferous tree species	Broad-leaved tree species	Coniferous tree species	Broad-leaved tree species
Forest planting allowance	1009.0	1548.0	A min of 1360.8	A max. of 4082.4	A min of 1370.0	A max. of 3796.0
Maintenance and protection allowance for an established forest	On average, 250.0		On average, 500.0		On average, 280.0	
Compensation for loss of income	For farmers and associations 72.40–147.7 For other private individuals or legal persons: 18.10–36.92		for farmers: 111.0, for other applicants: 25.0		for farmers: 171.0	

land for afforestation. However, Lithuanian assessments focused on both the species structure of newly established forests and the allocation of agricultural land for afforestation, where there were difficulties in agricultural cultivation (e.g., agricultural land with steep slopes, supplementation of forest enclaves, and priority given to communes where the afforestation rate was lower than the national average of 33.3%).

During the first period in Lithuania, the amount of support for the forestry sector was about 10% of the total support for the development of rural areas. In the subsequent years, financial support for planting new forests was reduced. Between 2014 and 2020, the proportion of support for forestry decreased by 24% and accounted for 6.6% of the rural development budget (in the years 2007–2013, it accounted for 7.5%). In Poland, the expenditure for afforestation in the RDP structure was considerably lower than that in Lithuania. In the years 2014–2020, 2.2% of the total RDP budget was allocated for afforestation, which was the lowest support compared to that in the previous periods. Between 2007 and 2013, this expenditure accounted for 3.2% (representing 26% of the planned budget for afforestation), and under the first perspective, 2.7% of the total budget was spent. It is worth noting, however, that Poland and Lithuania joined the European Union on 1st May 2004, and, therefore, the period was shortened as compared to the old EU (EU-15).

It is also worth comparing the limits of support for afforestation in the analysed countries under the individual financial perspectives. Payment rates are provided in the Tables 2, 3 below.

The analysis of afforestation programmes in Poland and Lithuania under the Rural Development Programme in the years 2004–2020 confirms that each country has adapted the requirements to its own conditions and needs. The greatest effects were obtained during the first mentioned period, which was associated with the most favourable financial support. As Sioma (2019) emphasises, with the contribution of public funds, approx. 72.4 thousand ha of privately owned agricultural land were afforested between 2004 and 2013, resulting in a significant increase in the area of private forests, which are an important organisational and functional component of agricultural farms in Poland. Despite ambitious targets for subsequent years, interest in afforestation has declined, mainly due to overcomplicated procedures (Kaliszewski et al., 2016) and insufficient incentives for farmers. The analysis revealed numerous challenges in the implementation of afforestation programmes, including high competition between direct payments for agricultural production and afforestation premiums, which lowered interest in the latter. In addition, increasing the minimum area of plots eligible for afforestation and the exclusion of permanent grassland from the afforestation programme represented significant barriers. The effectiveness of the programmes was, therefore, variable, which suggests the need for further research and potential modifications to their structure and management.

Despite the implementation of measures to increase the country's forest cover, a reduction in the forest cover has been observed in many regions. Prusinkiewicz et al. (1983) predicted that forest areas would increasingly shrink as a result of growing pressure from industry, spatial development of cities and settlements, or investments related to the construction of technical infrastructure. Kurowska et al. (2014) demonstrated that 29 districts of the country have experienced a slight decrease in the forest cover. This phenomenon is most evident in Małopolskie Voivodeship, Łódzkie Voivodeship, and the southern part of Mazowieckie Voivodeship. After 2004, when Poland became a beneficiary of EU funds, a significant proportion of agricultural land was allocated for the construction of new road infrastructure and, starting in 2016, also the construction of railway infrastructure. As regards forest land, the actual change of their intended use for other purposes has been counterbalanced by the state policy of increasing the country's forest cover being implemented, especially on land that is least suitable for agricultural production (valuation class V and VI) (Kurowska et al., 2020).

As regards the EU policy, under the common agricultural policy (CAP), financial support for forests and forest management is provided through national rural development programmes, in particular those aimed at adapting to and increasing resilience to climate-related risks. Between 2014 and 2020, EUR 6.7 billion of the CAP forestry measures were allocated to support EU strategic objectives, in particular, afforestation (27%), forest fire and disaster prevention (24%), and investments in resilience building as well as ecological and social functions (19%). However, the level of implementation of forestry measures is low and has declined significantly over the programming period. This is due, for example, to the lack of knowledge needed to manage the administrative procedures associated with applying for access to funding, coupled with an insufficiently attractive premium and a lack of advisory services to support capacity building, as well as limited guidance on the implementation of forest resource-based climate change adaptation actions and measures aimed at preventing and mitigating hazards (e.g., environmental fires, soil erosion, diseases, floods) (New EU Forest Strategy 2030; NFS, 2021). It is also worth mentioning the EU Biodiversity Strategy for 2030, which is a key component of the European Green Deal (EU Biodiversity Strategy for 2030). The strategy is aimed at protecting and restoring biodiversity in Europe, which includes an ambitious plan to plant at least 3 billion additional trees by 2030, in full respect of ecological principles. The aim of the plan is not only to increase forest cover in the EU but also to improve the quality of forest ecosystems, protect biodiversity, and increase forests' resilience to climate change. Trees will be planted in a sustainable manner, taking into account species diversity and local environmental conditions, which is expected to provide long-term ecological benefits. The implementation of this plan is one of the key elements in combating climate change and environmental degradation while supporting the goals of climate neutrality and the restoration of natural ecosystems in Europe. The urgent need to protect biodiversity and improve environmental quality in the face of climate change necessitates the development and implementation of a new programme for increasing the forest cover and the provision of coherent tools to support the conversion of afforested agricultural land into forests, as confirmed by Kaliszewski and Jabłoński (2022) in their study.

The new CAP (for the years 2023–2027) provides more flexibility in designing forest-related interventions according to national needs and specificities and reduces bureaucracy while linking and ensuring a synergistic approach between the European Green Deal, national forest policies and the EU's environmental and climate acquis. The Commission will seek to achieve a greater utilisation of funds allocated for rural development and available for the objectives of this strategy (New EU Forest Strategy 2030; NFS, 2021).

4.2 A comparative analysis of the expenditure on afforestation under the RDP in Poland and Lithuania

Based on data from EU reports, simulated trends in expenditure on afforestation are presented for Poland, Lithuania, and the EU level for the period of 2004–2024. Subsequent graphs will illustrate the expenditure *per capita* and per area and the correlations with other environmental and economic variables.

It can be seen from the graph below (Figure 2) that there has been a significant increase in expenditure on afforestation in each of the two countries and in the EU as a whole, with the largest increase in expenditure at the EU level.

Another study analysed the assumption that afforestation is related to the national income (GDP) for both countries, and it carried out a simulation of the income and afforestation data (Figure 3).

The graph shows a linear regression model illustrating the relationship between afforestation and the national income of Poland and Lithuania. The regression line for each country shows a trend indicating that an increase in the national income is linked to an increase in the afforestation area. The higher coefficient for Poland suggests that an increase in income has a greater impact on afforestation in Poland than in Lithuania. The negative value for Poland indicates a model that best fits the data at higher income values.

In the next stage of the research, analyses were carried out on the expenditure on afforestation in Poland and Lithuania in relation to the area of each country and their populations, with agricultural income also taken into account. The analysis examined the dependence of expenditure on afforestation on agricultural income, the population, and the area of the country. The following multiple linear regression model formula was adopted:

$$Y = \beta0 + \beta1X1 + \beta2X2 + \beta3X3 + \epsilon$$
$$Y = \beta0 + \beta1X1 + \beta2X2 + \beta3X3 + \epsilon$$

where:

YY - expenditure on afforestation *per capita*,X1X1 - agricultural income,X2X2 - population,X3X3 - country's area.

The model for Poland shows that the afforestation expenditure *per capita* is relatively stable and insensitive to changes in agricultural income. In contrast, the model for Lithuania suggests that agricultural income has a significant effect on afforestation expenditure *per capita*, with a more pronounced increase occurring as this income increases (Figure 4).

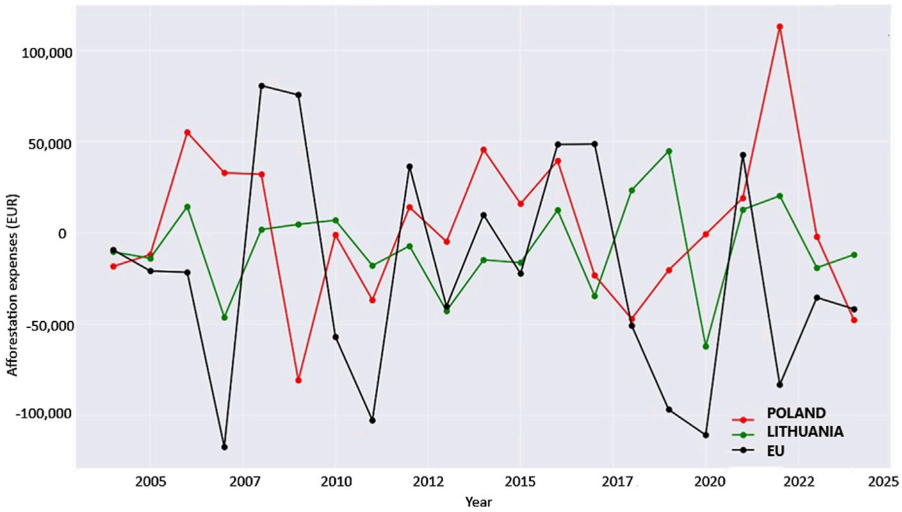

FIGURE 2
The trend in expenditure on afforestation in Poland and Lithuania compared to the EU.

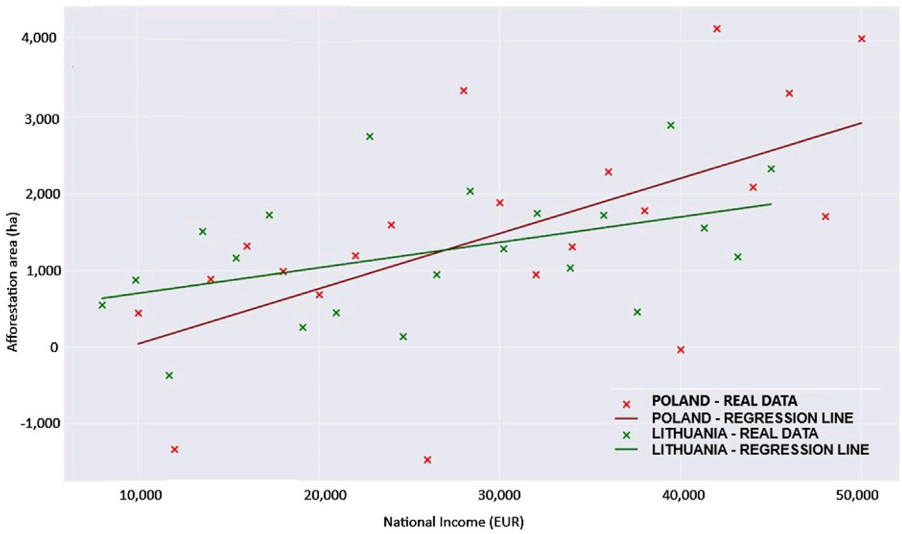

FIGURE 3
The relationships between afforestation and GDP in Poland and Lithuania.

In the final stage of the analyses, a Q-Q (quantile-quantile) plot was created as a graphical tool to assess whether the dataset follows a specific theoretical distribution, such as a normal distribution (Figure 5).

The graphs above (Figure 5) show a clear dependence of afforestation expenditure *per capita* on agricultural income. The regression lines for the two countries indicate a positive correlation, where higher agricultural income is linked to higher expenditure on afforestation.

In conclusion, the regression models showed a positive correlation between agricultural income and afforestation expenditure *per capita*, indicating the economic determinants of afforestation activities in the two countries.

An earlier study by Źróbek-Różańska et al. (2014), which assessed financial feasibility using the net present value (NPV) criterion commonly applied to assess the effectiveness of investments in the property market, confirms the low profitability of these activities. Based on afforestation statistics and considering the 5% discount rate in the Polish forestry market, that study showed the highest increase in cumulative net cash flows over the first 5 years, with a gradual decline in subsequent years. The longer the investment period, the lower the return is, even after excluding the discount rate. Investments of this type are difficult to terminate, as forests younger than 20 years are difficult to sell at a price that covers increasing outflows. Afforestation is a long-term investment that benefits future generations. These benefits need to be considered more from a social and environmental point of view. Private owners assess afforestation from the perspective of their own benefits, mainly

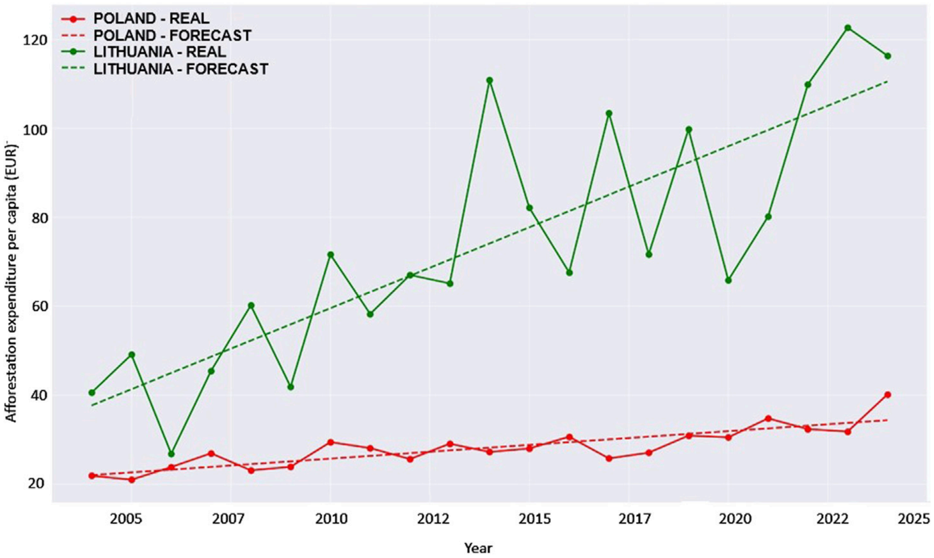

FIGURE 4
Expenditures on afforestation (forecast vs. reality).

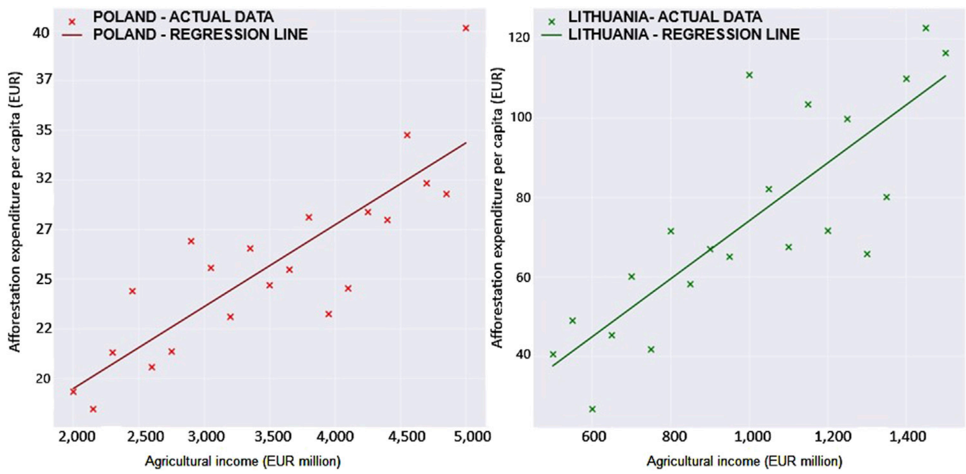

FIGURE 5
The relationships between afforestation expenditure and agricultural income.

cost-effectiveness. The merits of forestry investment in different countries depend not just on the local silvicultural forestry credentials but also on local costs of capital or discount rate, inflation, risk, and land acquisition costs (Kurowska and Kryszk, 2017; Chappell, 2019).

The identified determinants of forest cover growth in Poland and Lithuania are similar. These obstacles include the low supply of land for afforestation, the competitiveness of direct subsidies for agricultural production and the relatively low attractiveness of support for afforestation, limitation of the minimum area of an afforested plot, complicated procedures for applying for afforestation subsidies, insufficient education and promotion of afforestation among farmers, limitations of afforestation in Natura 2000 sites, and the exclusion of permanent grassland from afforestation. These factors are also confirmed by other studies conducted in Poland (Kurowska and Kryszk, 2017; Gołos et al., 2021; Kaliszewski and Jabłoński, 2022) and in Lithuania (Šepetienė et al., 2014; Veteikis and Piškinaitė, 2019; Mozgeris et al., 2021; Tiškutė-Memgaudienė and Tiškutė-Memgaudienė, 2021). In addition, Sulewski (2018) emphasises that in Poland, however, achieving positive effects in terms of increasing farm income is determined by the possibility of carrying out afforestation and maintenance works with the involvement of one's own labour force only.

As emphasised by Gołos et al. (2021), in addition to voluntary programmes based on owners' intrinsic motivation, which include afforestation under the RDP, programmes based on extrinsic motivation should be available, in line with the self-

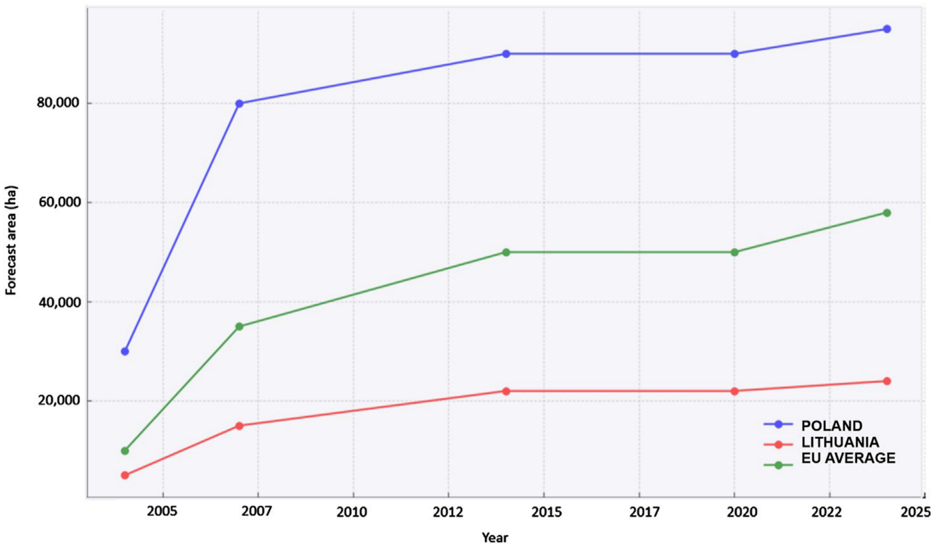

FIGURE 6
The balance of afforestation in Poland and Lithuania, compared to the European Union, in the years 2004–2020.

determination theory. This theory envisages, *inter alia*, a financial compensation scheme that could support the generation of public value (Mikša et al., 2020). Such a solution appears to be particularly desirable under conditions of the ineffectiveness of the existing policy and regulatory solutions when there is a growing concern that public benefits from private forest ownership will not be sufficiently ensured under the existing forest management schemes (Lindhjem and Mitani, 2012; Juutinen et al., 2021). Wysocka-Fijolek et al. (2020a) also emphasised that in order to increase interest in afforestation, there should be more support for young farmers who could be offered additional incentives to afforest land that is less useful to them as part of farm specialisation.

Poland saw the highest increase in the forested area in the years 2007–2013, which was due to intensive programmes supporting the afforestation of marginal agricultural land. Lithuania, while on a smaller scale, has also implemented afforestation measures, but their pace has slowed down since 2020. The average value for EU countries reflects the differences in afforestation policies of the individual Member States, taking into account their geographical and economic conditions. This is shown in the graph below (Figure 6).

The balance of afforestation in the years 2004–2024 indicates significant differences between Poland, Lithuania, and the European Union average. In terms of forested areas, Poland was the region's leader during the first two financial perspectives, but this activity has been slowing down since 2014. As for Lithuania, the afforestation rate was more stable but on a smaller scale. The EU average reflected the diversity of national policies, which gradually shifted priorities from simply increasing the forest area to improving the quality of forest ecosystems, in line with long-term climate and biodiversity protection goals.

Figure 6 shows the assessment of afforestation in Poland and Lithuania and the European Union average in the years 2004–2024, taking into account the milestones in the development of afforestation policies under the Rural Development Programme (RDP). Analysis of the budget periods reveals the varying afforestation rate, resulting from political, economic and geographical factors that affected the intensity of the measures being implemented.

The first financial perspective under RDP 2004–2006 was a crucial moment for the implementation of afforestation policy on a large scale, especially in Poland. The programme supported farmers in converting marginal agricultural land into forest land, which contributed to the afforestation of approx. 30,000 ha in Poland, and approx. 5,000 ha in Lithuania. The high afforestation rate during this period resulted from intensive efforts to promote an increase in forest cover in regions with a high proportion of low-quality land, which was in line with the EU environmental objectives, such as the protection of soils and biodiversity.

The second financial perspective, RDP 2007–2013, saw even more intensive afforestation measures. Poland afforested approx. 50,000 ha, which represented a significant increase compared to the previous period. This was linked to higher amounts of support and simplified administrative procedures, which encouraged more farmers to get involved in afforestation projects. Lithuania, while on a smaller scale, also increased its operations and afforested a further 10,000 ha. The EU average during this period also increased, thanks to extensive afforestation programmes in central and eastern European countries.

In the 2014–2020 perspective, the afforestation rate in Poland clearly declined, with only 10,000 ha afforested. This was due to a shift in EU policy priorities towards other environmental objectives, such as biodiversity conservation and organic farming. Many support programmes were focused on protecting the existing forests and converting them into multi-functional forests, which contributed to a reduction in new afforestation. During this period, Lithuania afforested a further 7,000 ha, and the EU average remained stable despite the fact that regional variations in the intensity of afforestation activities were observed.

For the period 2020–2024, a clear slowdown in afforestation has been observed in Poland and Lithuania, which is in line with general

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trends in the EU, where afforestation policy focuses more on the quality of forests than on increasing the forest area. In Poland, approx. 5,000 ha were afforested, compared to approx. 2,000 ha afforested in Lithuania. In the European Union, in line with the Biodiversity Strategy for 2030, new goals have emerged, such as planting 3 billion trees by 2030, in full respect of ecological principles, which could bring new impetus to further afforestation activities in the coming years.

5 Conclusion

Afforestation of agricultural land is a key element of the two countries' environmental policies that contribute to improving biodiversity, carbon storage, and soil and water quality. Afforestation programmes in both countries have significantly contributed to an increase in forest cover, especially considering the reasonable use of space and afforestation of land that is least suitable for agricultural production.

Poland, thanks to its active policy, financial support, and the availability of agricultural land, has implemented extensive afforestation programmes. Lithuania has been afforesting land on a smaller scale due to limited land resources and different priorities. Interest in afforestation grants under the CAP and the average area of afforestation in the EU is due to the diversity of climatic, economic and political conditions in the individual Member States.

Poland and Lithuania have different approaches to afforestation. Although Lithuania has achieved a higher level of forest cover due to more decisive and consistent afforestation activities carried out even before joining the European Union, the study showed that, as time progressed, there was less interest in afforestation due to less favourable financial conditions. In the case of Poland, there were restrictions on qualifying land for afforestation. It is worth noting that the complex application procedures may discourage potential beneficiaries from joining the programme. This indicates the need to simplify processes and provide better information and advice to farmers, not only in the afforestation process but also in subsequent forest management. Currently, financial support is possible for up to 5 years after afforestation. Once the afforested land has been converted into a forest, the forest is supervised by the competent authority without additional financial support. Obviously, an afforestation premium is paid for up to 12 years in order to compensate for the permanent exclusion of agricultural land from agricultural production. This is a very short period of time, which is why this form of support for afforestation is hardly competitive with other programmes dedicated to agricultural land implemented under the RDP.

Since 2014, there has been a noticeable decrease in the rate of new afforestation, both in Poland and Lithuania, which is linked to a change in European Union policy priorities. The EU has begun to place greater emphasis on protecting the existing forests, enhancing their ecological quality, and tackling climate change, which has reduced the emphasis on converting new areas to forests.

There is also a need to adapt national afforestation strategies to changing climatic and economic conditions, taking into account long-term sustainability goals. Not only does afforestation contribute to environmental improvement, but it also offers new economic opportunities for rural areas through the creation of forestry- and tourism-related jobs.

In recent years, afforestation activities in the EU countries, including Poland and Lithuania, have increasingly become part of long-term environmental goals. The implementation of the Biodiversity Strategy for 2030, including the plan to plant 3 billion trees, shows that future measures will focus on ensuring a balance between the quantity and quality of forests, with full respect for ecological principles. Countries with higher levels of natural forest cover, e.g., the Scandinavian countries, focus on managing and protecting the existing forests, while Central and Eastern European countries, such as Poland and Lithuania, have reforested new areas to a greater extent, especially at the beginning of the period under analysis.

Efficient management and financing of afforestation activities are crucial to achieving climate, environmental and economic objectives both at the national level and in the context of the European Union's policies. European Union support for afforestation should continue to the extent that it ensures improved land use and prospects for long-term economic activity in rural areas.

Data availability statement

The datasets presented in this study can be found in online repositories. The names of the repository/repositories and accession number(s) can be found in the article/supplementary material.

Author contributions

HK: Conceptualization, Data curation, Formal Analysis, Methodology, Project administration, Resources, Software, Supervision, Visualization, Writing–original draft, Writing–review and editing. JV: Formal Analysis, Software, Supervision, Validation, Visualization, Writing–original draft, Writing–review and editing. DJ: Conceptualization, Data curation, Formal Analysis, Writing–original draft. AM: Formal Analysis, Supervision, Validation, Visualization, Writing–original draft. KK: Formal Analysis, Methodology, Supervision, Writing–review and editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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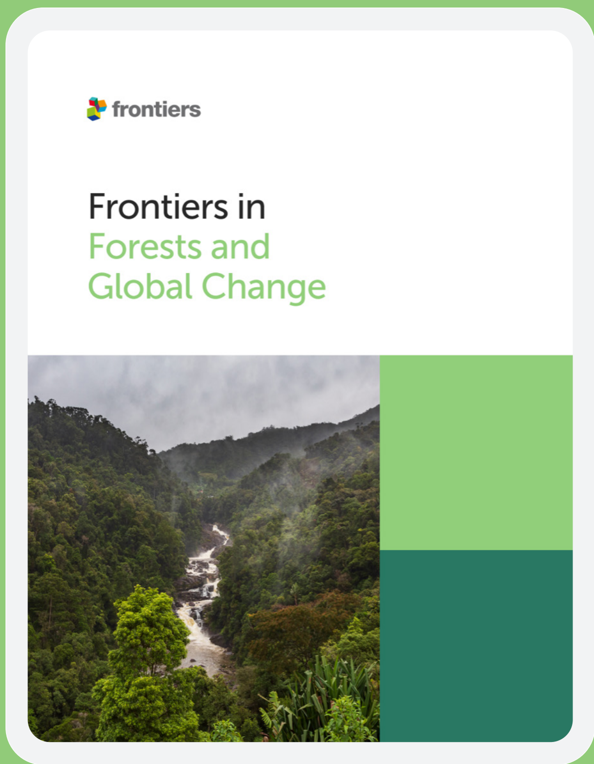

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