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Integrating remote sensing and site-specific data in forest planning: A multicriteria approach to prioritize forest restoration in Central Europe

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ABSTRACT

Recent severe droughts and storms has led to severe disturbances in Central European forests, highlighting the urgent need for adaptive forest management strategies and restoration. Using a multi-criteria decision analysis (MCDA) approach, this study prioritizes forest restoration areas across ownership boundaries by integrating remote sensing data, climate projections, and site-specific characteristics. Key indicators of forest resilience—-drought stress risk, stand-level tree species diversity, and landscape-level rarity—were assessed, revealing that approximately 18.5 % of the study area in Lüchow-Dannenberg, Germany, requires urgent restoration due to high drought stress risk, as well as low diversity and rarity. European beech and Norway spruce stands exhibit the highest projected drought stress risk, whereas Scots pine and oak stands show comparatively low drought stress risk. Pine-dominated stands exhibit the lowest diversity and landscape rarity, while beech and oak contribute significantly to biodiversity. Sensitivity analyses reveal that weighting different indicators affects restoration prioritization, emphasizing the need for context-specific adaptive management strategies that take into account site conditions (e.g., soil moisture, climate exposure) and forest type (e.g., converting pure pine stands into more diverse, mixed stands to increase resilience.

This work underscores the importance of forest inventory data in guiding a multidimensional approach to forest stewardship and highlights how big data analytics can enhance forest science and decision-making. The study provides a practical framework for policymakers and forest managers to allocate resources efficiently, enhancing the resilience and sustainability of Central European forests in a changing climate.

1. Introduction

In the ongoing battle against climate change, the imperative to adapt forests to drifting environmental conditions has become increasingly paramount. The cumulative effects of storms and drought in recent years have led to extensive disturbances in European forests and demonstrate the urgency of adaptation (Buras et al., 2020; Marini et al., 2017; Schuldt et al., 2020; Senf and Seidl, 2021). Large parts of the Central European forests are neither close to nature, which might grant resistance and resilience (BMEL, 2015; Hennenberg et al., 2017; PGL and LP, 2021), nor well mixed in order to spread risks (Bolte et al., 2009; Fuchs et al., 2024; Markowitz, 1991; Paul et al., 2020). The aims for the adaptation of Central European forests to climate change are to improve them structurally and functionally, i.e. enrich biodiversity by species composition and genetics, as well as by layering (Mansourian et al., 2020; Stanturf, 2016; WBW, 2022). Given the widespread forest damage and the resulting high demand for restoration under simultaneously limited capacities, further prioritisation of climate adaption measures is crucial (Anderegg et al., 2022; Pach et al., 2018).

Steps towards an order of preparing climate-smart forest ecosystems, have often been proposed but however in practice not implemented or realized yet (Knoke et al., 2020; Singh et al., 2015; Walentowski et al., 2017). Robust information on forest conditions and the potential for forest restoration is lacking so far, especially for informing policy

Abbreviations: CHM, Canopy Height Model; CMIP, Common Management Information Protocol; NFI, National Forest Inventory; ODS/ L, Other Deciduous with Short/ Long life expectancy; RCM, Regional Climate Model; RCP, Representative Concentration Pathway; SWB, Site water balance.

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makers. Up to now, in German forestry, federal state and ownership boundaries have been limiting for information availability and thus forest restoration, as forest management plans are not mandatory for small-scale private forests. In state-owned forests, a permanent ownership-wide inventory is conducted. However, the datasets from state forest services are different with the sampling methods varying greatly to cross-ownership national forest inventories (NFI; Böckmann et al., 1998; Gschwantner et al., 2022). This data gap makes it difficult to develop a comprehensive understanding of forest conditions and the necessary measures for climate adaptation as well as societal demands.

In the context of forest planning, decision support systems (DSS) have been used successfully for the long-term analysis of forest development in the past (Blattert et al., 2018; Linkevičius et al., 2019; Nordström et al., 2019; Thrippleton et al., 2023, 2021). For this purpose, forest inventories coupled with growth simulators and MCDA methods were utilized to assess different management scenarios in a changing climate (Blattert et al., 2017; Bugmann and Seidl, 2022; Maréchaux et al., 2021; Reyer et al., 2015; Wolfslehner and Seidl, 2010). In addition to the analysis of the optimal management, information on the prioritization of forest stands with regard to restoration efforts is crucial for decision makers. However, a practicable and robust DSS is largely lacking in this context.

The methodology outlined herein integrates various data sources, including climate data projections, such as site-specific characteristics and remote sensing data, to comprehensively assess each stand's **drought stress risk**, tree species diversity at the stand level (**stand-level diversity**), and **rarity** of each stand at the **landscape-level** (Grier and Running, 1977; Heinrichs et al., 2019; Möllmann and Möhring, 2017; Müller et al., 2022). While these indicators have individually been recognized for their relevance to ecosystem resilience, this study is, to our knowledge, the first to develop and apply a multi-criteria decision analysis (MCDA) that simultaneously incorporates all three criteria to prioritize forest restoration efforts across ownership boundaries at a landscape scale. Additionally, the temporal urgency of restoration is captured through a vegetation height model as a proxy for forest stand age (Senf et al., 2021; Socha et al., 2023), further refining the prioritization process. This novel approach aims to provide a robust, spatially explicit decision support tool that enhances the capacity of forest managers and policymakers to allocate restoration resources effectively and foster climate-resilient forest ecosystems.

2. Material and methods

2.1. Study region

The study area, encompassing about 47,000 ha of forest which cover 38 % of the area's land surface, lies within the county of Lüchow-Dannenberg in the North German Lowlands (Fig. 1a). This county in Lower Saxony also borders three other federal states (Fig. 1b). Characterized by its diverse topography and vegetative composition, the region offers a representative sample of the broader landscape dynamics typical of northern Germany temperate oceanic climate (Beck et al., 2018).

Comprising primarily coniferous and only few deciduous forests, the vegetation of the study area is notable for its dominance by Scots pine (*Pinus sylvestris* L.; hereinafter "pine"), constituting approximately 67 % of the forest cover (Fig. 1c). Pedunculate and Sessile oak (*Quercus robur* L., *Quercus petraea* (Matt.) Liebl., hereinafter "oak") represent the second most abundant species, accounting for 11 %, while Norway spruce (*Picea abies* L.; hereinafter "spruce") occupies a lesser proportion, amounting to 3 % of the forest. European beech (*Fagus sylvatica* L.; hereinafter "beech") only plays a minor role in the main stocking so far. The study area was further chosen because it is characterized by a wide range of soil moisture and soil nutrient availabilities. Elevation ranges from 6 to 145 m a.s.l., while in the period from 1991–2020 the climate was characterized by a 9.6 °C air temperature mean and a 629 mm m^2 a^{-1} annual precipitation sum (DWD, 2022). The future climate will change significantly in the study area and is characterised in Section 2.2.

The forest ownership in Lüchow-Dannenberg is dominated by private forests ranging from less than one hectare up to 5400 ha in size (Junack, 1989). Together they cover an area of 32,000 ha (69 %), while state forests comprise 13,000 ha (31 %) (vTI, 2015). Both state and private owned forests have an outstanding long tradition in forest restoration (Junack, 1924; von Unruh, 1936) –with only moderate achievement of the restoration goals. The famous big calamities in the study region have not ended until today (Habermann, 2017). All forest owners follow the integrative and multifunctional way of German

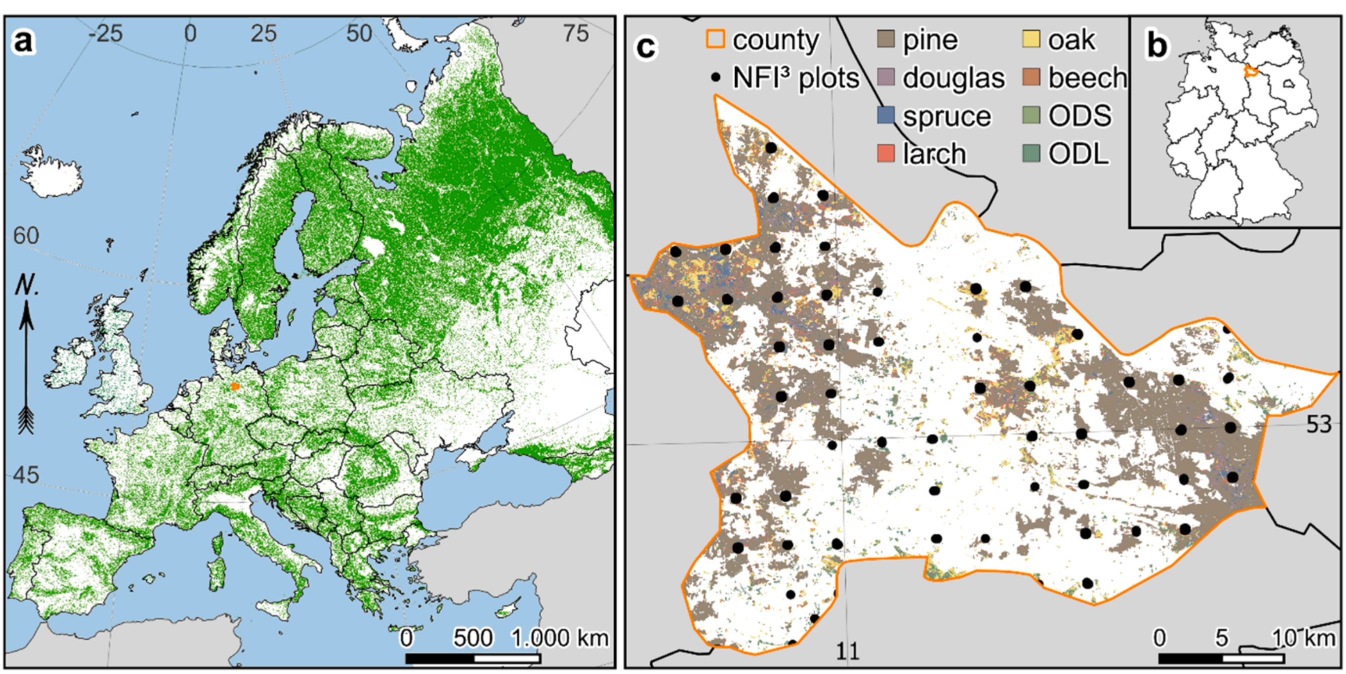

Fig. 1. a Forest cover in Europe (Burgoin et al., 2024) and b location of the Lüchow-Dannenberg county at 11° E lon. and 53° N lat. in the German federal state of Lower Saxony, c distribution of main tree spp. groups (ODS/ ODL: Other deciduous trees with short/ long life expectancy) according to Blickensdörfer et al. (2024, 2022).

forestry (Axer et al., 2023; Borrass et al., 2017; Larsen and Nielsen, 2007; Mölder et al., 2020; Simons et al., 2021). Alongside continuous cover forestry (Mason et al., 2022), operations such as small-scale clear-cuts are practised, which underlines a highly diverse management that could mitigate disturbances and facilitate restoration (Muys and Messier, 2023; Potterf et al., 2023). Furthermore, about 900 ha are strictly preserved forests where no management takes place at all (Bollmann and Braunisch, 2013; Engel, 2020; Steinacker et al., 2023).

According to the third German NFI, the forests in the study region have an mean age of 86 years in 2024 (vTI, 2015) which suggests a generational succession. Therefore, the biggest issue evolves around tree species selection under deep uncertainty (Albert et al., 2017; Knocke et al., 2024; Marchau et al., 2019) if priority restoration areas have been defined.

2.2. Remote sensing and site-specific data

A remote sensing-based dataset was used to determine the drought stress vulnerability of tree species, the tree species diversity within the stand and the rarity at landscape level (Fig. 1). Tree species detection was based on the spectral-temporal reflectance signature in Sentinel-2 satellite imagery, as assessed from Blickensdörfer et al. (2022). The resulting map provides information on the canopy-dominating tree species at a spatial resolution of 10 m for the stocked forest area, as defined by Langner et al. (2022). The mapped tree species distribution refers to the years 2017 and 2018, as satellite imagery of this time period was used for the analysis. The data set captures the condition of the forests before the extensive damage caused by the severe drought years of 2018–2020 in Germany. There were 11 tree species and tree species groups classified according to their spectral characteristics. NFI data was used to train the classification model and to estimate the classification accuracy at the national level. For a comprehensive description of the methods and data we refer to Blickensdörfer et al. (2024, 2022).

A Digital Surface Model (DSM) was utilized to delineate average forest stand heights (Fig. 2) (GeoBasis-DE/ LGLN 2024). Specifically, we employed the latest Canopy Height Model (CHM) with a resolution of 1 m, which is readily available by the land surveying office of the federal state Lower Saxony (GeoBasis-DE/ LGLN 2024). This height model was generated using airborne laser scanning data collected between 2013 and 2019. To get the average stand heights, we sampled the 1 m resolution CHM down to 50 m grid cells used in the other metrics. And to mitigate the effect of low canopy height values in forest gaps, which would lead to biased stand heights, the 95th percentile height within each grid cell was used.

In addition to the remote sensing data, climate and soil data was processed and used to derive the forest restoration area.

Climate projections for the period 2071 to 2100 were obtained from the ReKliEs-De project (Regional Climate Projection Ensemble for Germany; Hübener et al., 2017; Warrach-Sagi et al., 2018). Within the CMIP5 generation, the RCP8.5 scenario was chosen in order to investigate an extreme climate shift towards 4.8 °C temperature and 8.5 W m^{-2} radiation rise (van Vuuren et al., 2011). Thus, the General Circulation Model (GCM) 'Hadley Centre Global Environment Model' (HadGEM2) combined with the empirical-statistical regional climate model 'Wetterlagen-basierte Regionalisierungsmethode' (WettReg18) were chosen to represent the most extreme climate projection (Kreienkamp et al., 2013; Martin et al., 2011). By selecting these statistical climate projections, we stand in line with conservative estimates typical for forestry, which often consider unfavorable ('pessimum') conditions to evaluate the resilience of management options. The regional climate projections were regionalized to 50 × 50 m grid cells. For these projections, the climatic water balance (CWB) during the vegetation period (the difference between potential evapotranspiration and precipitation) was calculated for a 30-year climate period of 2071–2100 (Allen et al., 1998; Nuske, 2022).

In addition to the CWB, the soil's available water capacity (AWC) of a site was considered to determine the site water balance (SWB; Fig. 3; Grier and Running, 1977). Utilizing the pedotransfer function of Puhlmann and Wilpert (2011) and the input data from the forest site mapping, the soil water capacity was calculated where available. Otherwise, input variables from the 1 : 50,000 soil map were used (Gehrt et al., 2021).

2.3. Multi-criteria decision analysis

2.3.1. Forest restoration urgency

To identify and prioritize forest areas most in need of restoration, we apply a multi-criteria decision analysis (MCDA) approach. MCDA allows the integration of different ecological and management-related indicators into a single, comparable score.

Among the various MCDA methods, we use the multi-attribute value theory (MAVT), a well-established technique grounded in utility theory. MAVT is particularly suitable for our purpose because it can translate multiple, diverse indicators into a unified, dimensionless value between 0, representing the poorest condition, and 1, representing the best condition. This enables direct comparison across spatial units.

In our application, we do not use MAVT to select a single "best" alternative, as in classic decision problems, but rather to produce a spatially explicit ranking of forest stands according to restoration urgency. This ranking is based on three key indicators: drought stress risk, diversity at stand level, and rarity at landscape level.

For each grid cell i, the drought stress indicator D_i and the two rarity R_i and diversity E_i indicators are combined using equal weighting to produce the overall restoration value G_i:

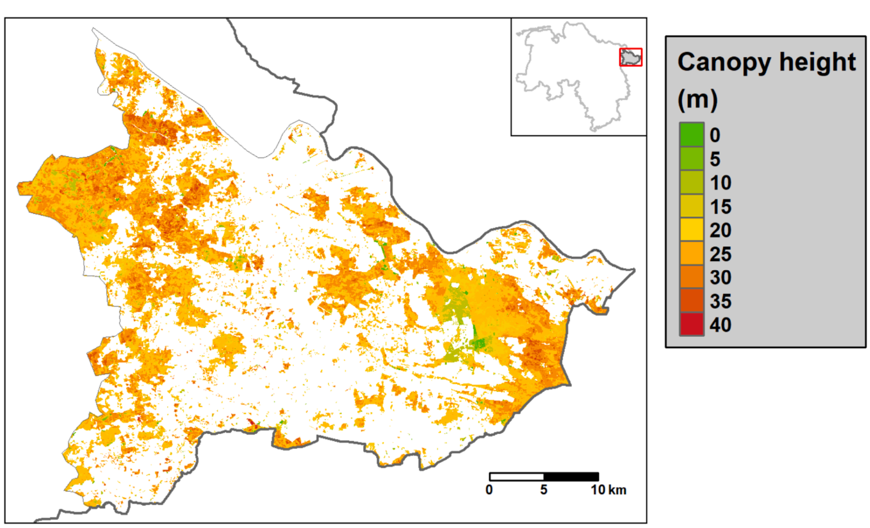

Fig. 2. Distribution of canopy heights in the study area. The map shows the canopy heights based on a Canopy Height Model (CHM) with 50 × 50 m resolution.

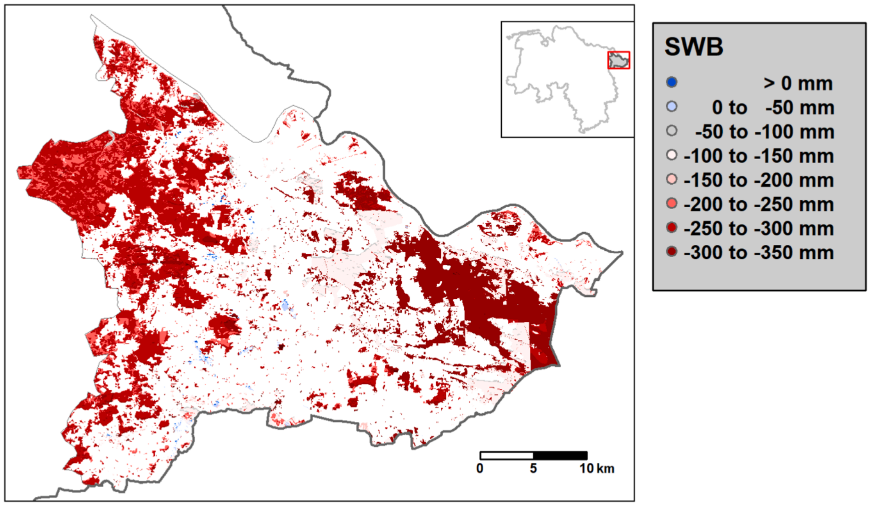

Fig. 3. Spatial visualisation of the site water balance (SWB) as the sum of the available water capacity and the climatic water balance for the period 2071–2100 for the HadGEM2/WETTREG climate model.

$$G_i = 0.5 \times D_i + 0.25 \times R_i + 0.25 \times E_i \quad (1)$$

The result is a value G_i that describes the urgency of the restoration. The higher the value, the lower the urgency of restoration. The lower the restoration value, the greater the urgency of restoration.

A value function can be applied either by linear transformation or by direct evaluation (Demetriou, 2014; Eisenführ et al., 2010). While the results of a linear transformation are dependent on the data set and change with it, direct evaluation is independent and globally applicable.

Both the choice of value functions and their weights have a very strong impact on the results of the respective decision problem and thus on the decision as such. Depending on the objective, however, a weighting of the indicators could be useful in order to further specify the forest restoration activity. A sensitivity analysis was therefore carried out with all possible combinations. The sensitivity analysis was conducted to explore the impact of different weightings on the forest restoration value G_i.

The temporal component of urgency is represented by the vegetation height model (cf. 2.3. Remote Sensing and Site-specific Data). Forestry management practices often define certain height or age thresholds at which a stand should be regenerated (Albert et al., 2015; Knocke et al., 2023, p. 98; Pach et al., 2018). If the height data indicates that the stand has reached or is approaching these thresholds, this suggests an increasing urgency to initiate regeneration measures—such as underplanting.

2.3.2. *Drought stress risk assessment*

Drought stress risk is a crucial criterion as it directly impacts the resilience and health of forest ecosystems, particularly in the face of climate change. By assessing drought stress, the methodology helps to identify areas and vegetation types most vulnerable to water scarcity, which can lead to increased susceptibility to diseases, reduced growth rates (Albert et al., 2018; Schmidt, 2020), and ultimately higher mortality rates among trees (Bigler et al., 2006). The inclusion of drought stress as a factor allows for targeted adaptation measures, such as the selection of drought-tolerant species (Albert et al., 2017) or the implementation of water management practices to ensure the sustainability and functionality of ecosystems.

Based on the SWB for the selected climate scenario and model (Fig. 3), the drought stress vulnerability of the current stand is estimated for each 50 × 50 m grid cell. This indicator integrates the AWC and the CWB for a given period (Grier and Running, 1977). The value function of drought stress risk is based on tree species-specific SWB threshold values derived from Albert et al. (2017). A monotonically increasing sigmoid curve was selected as the shape of the function (see Fig. 4). The threshold values of the tree species from Albert et al. (2017) were set as intersections of the curve with the values 0.25 (high risk threshold) and 0.75 (low risk threshold). The values used were selected to allow more differentiation in the dry and moist areas and thus keep the loss of information to a minimum.

Let $(x1_j,\ 0.25)$ and $(x2_j, 0.75)$ be the two points that define the monotonic sigmoid function for a given tree species j representing the drought stress threshold values. The general formula is:

$$d_{ij} = \frac{1}{1 + \left(\frac{1}{0.75} - 1\right) * e^{\left(-r \times \left(x_i - \left(\frac{x1_j - x2_j}{2}\right)\right)\right)}} \quad (2)$$

The function interpolates sigmoidal relationships between these two points for a given tree species j. $x_i$is the SWB value for the i-th grid cell and r is the slope parameter of the sigmoid function. It is determined using the *nleqslv* function to solve the equation associated with the sigmoid curve, given the input coordinates (Hasselman, 2023).

Let d_{ij} be the standardized drought stress score specifically associated with the j-th tree species for the i-th grid cell. p_{ij} is the proportion of the j-th tree species in the i-th grid cell, where j ranges from 1 to the total number of tree species n. Then the weighted standardized drought stress score D_i for the i-th grid cell can be calculated as:

$$D_i = \sum_{j=1}^{n} p_{ij} \times d_{ij} \quad (3)$$

The weighted standardized drought stress score can range from 0 to

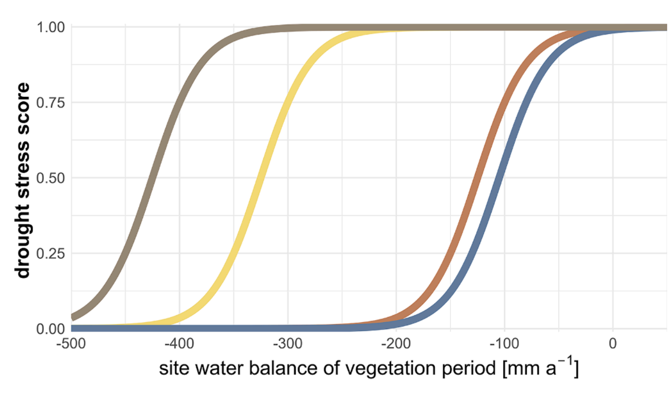

Fig. 4. Drought stress score depending on the site water balance for the different tree species groups.

1, with lower values indicating higher drought stress risk.

2.3.3. Diversity at stand level

Stand-level diversity is selected to evaluate the variety of tree species within a stand, which is essential for maintaining ecosystem resilience and ecological balance. Diverse stands are generally more resilient to pests, diseases, and environmental stressors, as different species and genetic variability can buffer the system against disturbances (Pretzsch and Grote, 2023; Yachi and Loreau, 1999). By assessing diversity at this level, the methodology aims to support biodiversity conservation and to foster stands that can adapt better to changing environmental conditions. High biodiversity also contributes to enhanced ecosystem services, such as carbon sequestration, soil fertility, and overall ecosystem productivity (del Río et al., 2022; Pretzsch et al., 2015; Steckel et al., 2020).

Information on the species composition of dominant trees at the stand level (0.25 ha) was derived from remote sensing data and used to calculate diversity indices for identifying stands with low species diversity. Firstly, the species richness (S) as the total number of tree species within the i-th 50 × 50 m grid cell was determined. For this purpose, the 10 × 10 m cells of the remote sensing data set (Fig. 1c) were aggregated.

$$H' = -\sum_{i=1}^{S} p_i \times \ln(p_i) \quad (4)$$

Subsequently, the Shannon index (H') at the 50 × 50 m was derived from the species richness (S) and the proportions (p_i) of 10 × 10 m pixel cells belonging to the i-th species using the following formula (Shannon and Weaver, 1949):

$$E_H = \frac{H'}{lnS} \quad (5)$$

Evenness (E_H), on the other hand, is a measure of how evenly the tree species are distributed among the different species within the 50 × 50 m grid. It is often expressed as:

H' is the Shannon index and S is the total number of species in the community. Evenness ranges from 0 (completely unequal distribution of pixel cells among species) to 1 (total even distribution).

By computing evenness, an evaluation of each stand based on tree species composition can be conducted. This allows the identification of stands with particularly low species diversity, which can then be diversified for risk prevention purposes.

2.3.4. Rarity at landscape level

Landscape-level rarity is selected to assess the contribution of rare tree species to the gamma (γ) diversity, or overall biodiversity, across a landscape. Rare elements can significantly enhance γ-diversity by adding unique species compositions and ecological niches that would not otherwise be represented (Qiao et al., 2023; Schall et al., 2020). This criterion allows for the identification of areas that, despite covering smaller spatial extents, contribute disproportionately to the richness and diversity of the entire landscape (Sebald et al., 2021). By focusing on rarity, the methodology supports conservation strategies that aim to protect these uncommon elements, thus maximizing γ-diversity and fostering ecosystem resilience at a larger scale.

At the landscape level, the regional importance of each tree species is assessed concerning its suitability for restoration. To achieve this, a rarity value was calculated for each stand based on the proportions of tree species, which examines the relative rarity of each tree species within the landscape. The rarity value function was adjusted so that rare species have a high rarity weight, while common tree species have a very low rarity weight (Fig. 3). To determine the rarity of a tree species, the proportions of tree species within a radius of 1700 m were extracted from remote sensing data (Fig. 1c). This represents a landscape area of 907 ha. Let A_{ij} represent the proportion of the j-th tree species within the i-th buffer. The rarity value function for each tree species within the buffer can be formulated as follows:

$$R_{ij} = 1 - A_{ij}^{\ k} \quad (6)$$

The exponent k is calibrated based on a reference proportion pc_x and a target rarity score pc_y, and is calculated as follows:

$$k = \frac{\log\ (1 - pc_y)}{\log\ (pc_x)} \quad (7)$$

For the rarity value, it was assumed that a species has a rarity value of 0.75 if it represents 50 % of the species composition (Fig. 3). Then, to calculate the rarity-weighted values for the raster-cell RW_i, we used the proportions of tree species within the raster cell p_{ij} to weight the rarity values R_{ij} accordingly.

$$RW_i = \sum_{j=1}^{n} p_{ij} * R_{ij} (8)$$

3. Results

3.1. Drought stress risk

The analysis of the drought stress scores for different tree species shows a remarkable differentiation with regard to their susceptibility to drought stress at their sites in the study area (Table 1): Oak shows high values with an average drought stress score of 0.86 and a median of

Table 1
Summary statistics of drought stress scores for different tree species. The table presents the mean, median, standard deviation (SD), minimum, and maximum values of drought stress for each tree species. It is important to note that these drought stress scores are derived from the combination of observed site water balance (SWB) and species composition.

Tree species	mean	median	SD	min	max
Oak	0.86	0.95	0.20	0.05	1.00
Beech	0.12	0.01	0.25	0.00	1.00
Other deciduous trees with short life expectancy	0.95	0.99	0.10	0.32	1.00
Other deciduous trees with long life expectancy	1.00	1.00	0.00	0.95	1.00
Spruce	0.06	0.00	0.15	0.00	0.99
Douglas fir	0.86	0.92	0.18	0.15	1.00
Pine	0.99	1.00	0.01	0.87	1.00
Larch	0.97	0.99	0.04	0.53	1.00

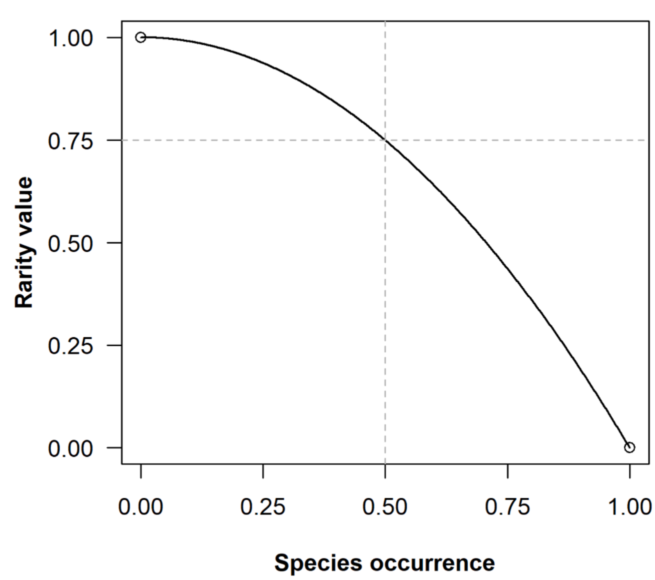

Fig. 5. Rarity value function depending on the proportion of tree species. A rarity score of 0.75 is assigned for a tree species proportion of 0.5.

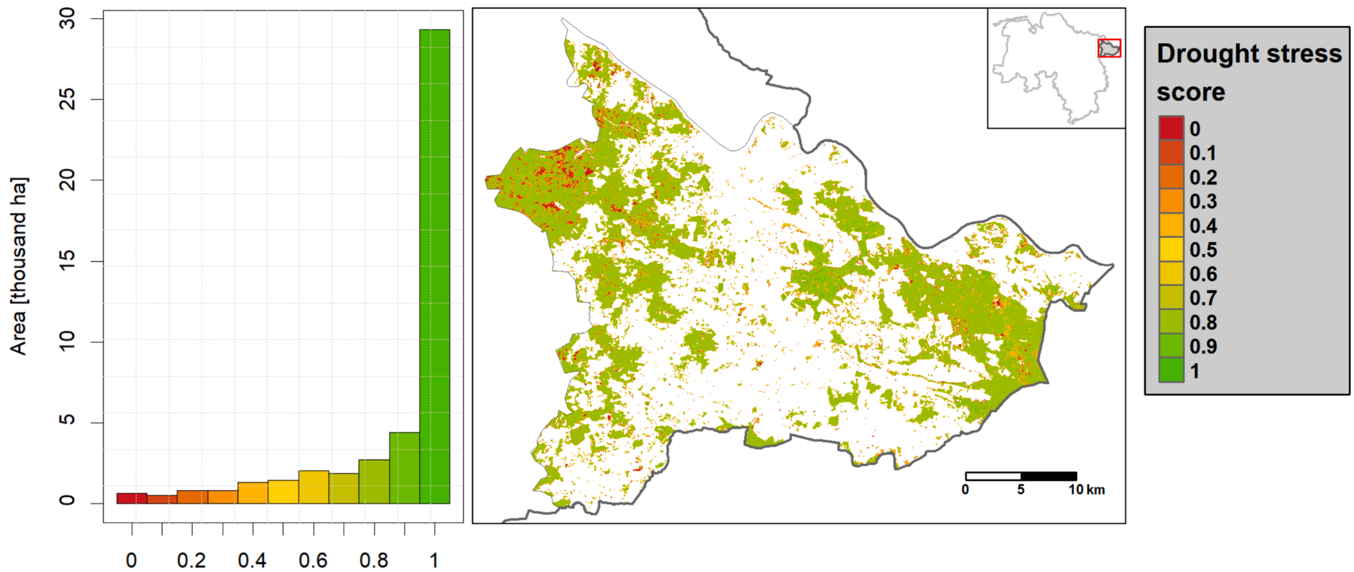

Fig. 6. Spatial representation of drought stress score within the study area. The map illustrates the distribution of drought stress score across different regions, with colour intensities indicating varying levels of risk. The histogram displays the frequency distribution of dry stress risk values across the entire area, with the x-axis representing the risk levels and the y-axis representing the cumulative hectare values.

0.95, which indicates a relatively low susceptibility to drought stress. Pine with a mean value of 0.99 or ODS and ODL appear to be similarly unsusceptible. This means that these tree species are less affected by drought stress under the projected SWB (Fig. 3). In comparison, beech and spruce show significant differences in their drought stress scores. A mean drought stress score of 0.12 is projected for the existing beech stands and a mean drought stress score of 0.06 is projected for the spruce stands. However, there are also a few sites with a lower drought stress score for both tree species (Table 1 and Fig. 5).

Based on the drought stress analysis, the area required for forest restoration can initially be quantified on a site-specific and climatic basis. Approximately 32,000 ha have a drought stress score above 0.9 and thus a low risk of drought stress (Fig. 6). The high percentage of pine trees is responsible for this (Fig. 1), as they receive a very low drought stress score (Table 1 & Fig. 4). Nevertheless, approx. 5,000 ha of the study area have a standardized drought stress score < 0.5 (Fig. 6). This means that this combination of tree species and sites is probably at risk in the face of climate change. Such areas are mainly stocked with spruce or beech.

3.2. Diversity at stand level

From the distribution of the number of species and their proportions (Fig. 1), the evenness for the study area also shows that many stands are very homogeneous in terms of species composition (Fig. 7). An area of 35,134 ha of forest has an evenness of <0.5. Only 5,363 ha have a high evenness score above 0.9. In the north-west of the study area in particular, there are more mixed stands, resulting in higher evenness scores. Especially in the centre-east of the study area, there are large stands of pine trees without a mixture.

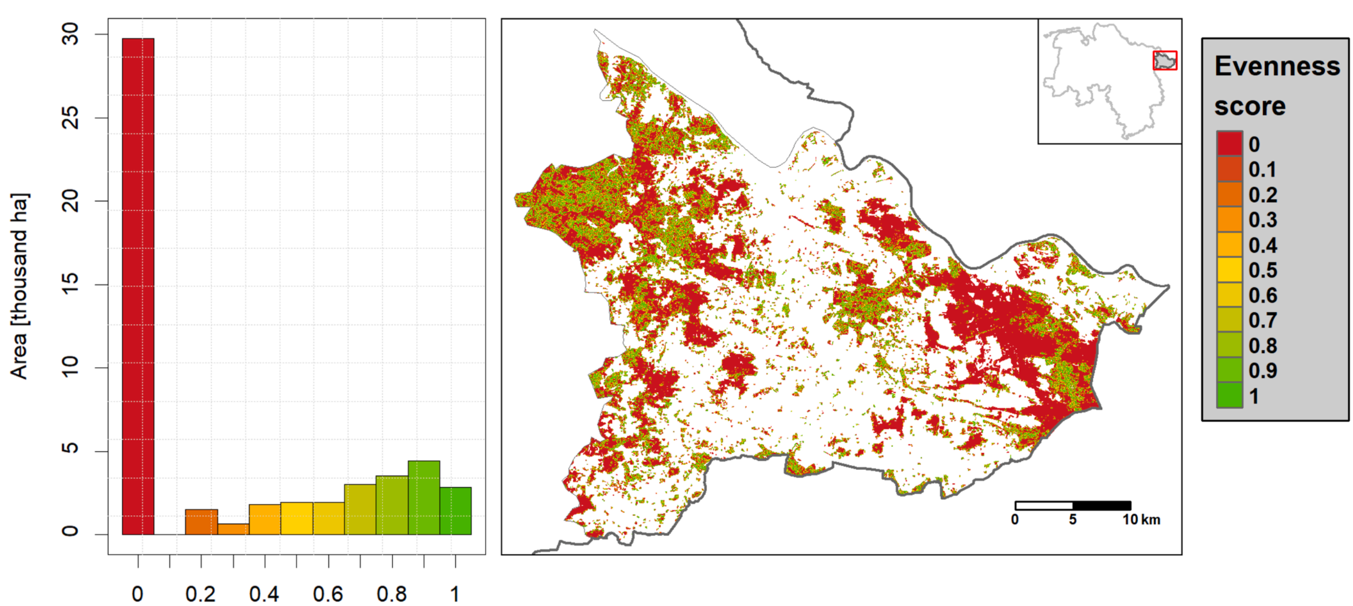

Fig. 7. Spatial representation of evenness within the study area. The map displays the distribution of evenness across different regions, with colour intensities reflecting gradients of evenness. The histogram depicts the frequency distribution of evenness values across the entire area, with the x-axis representing evenness and the y-axis representing the cumulative hectare values.

Table 2
Summary statistics of Evenness scores for different tree species. The table presents the mean, median, standard deviation (SD), minimum, and maximum values of Evenness scores for each tree species.

Tree species	mean	median	SD	min	max
Oak	0.58	0.72	0.35	0.00	1.00
Beech	0.65	0.75	0.30	0.00	1.00
Other deciduous trees with short life expectancy	0.72	0.78	0.23	0.00	1.00
Other deciduous trees with long life expectancy	0.65	0.74	0.30	0.00	1.00
Spruce	0.66	0.73	0.27	0.00	1.00
Douglas fir	0.67	0.73	0.26	0.00	1.00
Pine	0.25	0.00	0.36	0.00	1.00
Larch	0.71	0.76	0.23	0.00	1.00

Table 3
Summary statistics of rarity scores for different tree species. The table presents the mean, median, standard deviation (SD), minimum, and maximum values of rarity scores for each tree species.

Tree species	mean	median	SD	min	max
Oak	0,67	0,79	0,31	0,00	1,00
Beech	0,82	0,95	0,24	0,04	1,00
Other deciduous trees with short life expectancy	0,58	0,61	0,29	0,04	1,00
Other deciduous trees with long life expectancy	0,61	0,67	0,31	0,02	1,00
Spruce	0,78	0,84	0,20	0,04	1,00
Douglas fir	0,77	0,83	0,21	0,04	1,00
Pine	0,43	0,40	0,28	0,00	1,00
Larch	0,72	0,79	0,25	0,04	1,00

Looking at individual tree species, pine stands show the greatest homogeneity with an average evenness of 0.25 (Table 2). Spruce (mean = 0.66) and Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco; mean = 0.67) show moderate evenness values, reflecting more balanced species distributions, though they still experience some dominance. Oak (mean = 0.58) and beech (mean = 0.65) also exhibit moderate evenness, with oak and beech stands maintaining relatively higher diversity compared to pine-dominated areas (Table 2).

3.3. Rarity at landscape level

Fig. 8 shows the rarity of the tree species at landscape level, i.e. γ-diversity. It can be seen that, above all, mixed oak or beech stands in pine-dominated areas, for example in the north-west of the study area, have a high rarity and receive correspondingly high rarity weights. The analysis of tree species-specific rarity values provides further insights into these patterns. Oak (mean = 0.67) and beech (mean = 0.82) exhibit relatively high rarity values (Table 3). This indicates their contribution to γ- diversity, as they are less common in the broader context.

In areas of neighboring pine stands, there is a particular lack of species at the landscape level and the pine stands show low rarity values (mean = 0.43; Table 3). 29,135 ha have a rarity of <0.5. Of this area, 14,000 ha have a rarity <0.2, which indicates very low mixing at the landscape level (Fig. 8). These monocultures exhibit low diversity at the landscape level.

3.4. Combining the indicators

Based on the derived drought stress risk assessment, the species diversity at stand level and the rarity at landscape level, the final step is to prioritize the forest restoration areas using a multi-criteria evaluation.

Fig. 9 shows the aggregated restoration values with equal weighting of the indicators for the study area. It can be seen that the eastern part of the study area, in particular, is highly worthy of restoration due to an increased future risk of drought stress and low diversity at stand and landscape level. At the same time, there are forest stands, for example, in the north-western part of the study area, which show a low risk of drought stress and a high species diversity at stand and landscape level.

Assuming a threshold value of 0.5 for a high forest restoration priority, approximately 18.5 % (8,689 ha) of forest stands have a high restoration priority, while 81.5 % (38,214 ha) have a low restoration priority.

Fig. 10 shows the histogram of canopy height distribution for the 8,689 ha identified with high restoration priority (i.e., restoration values

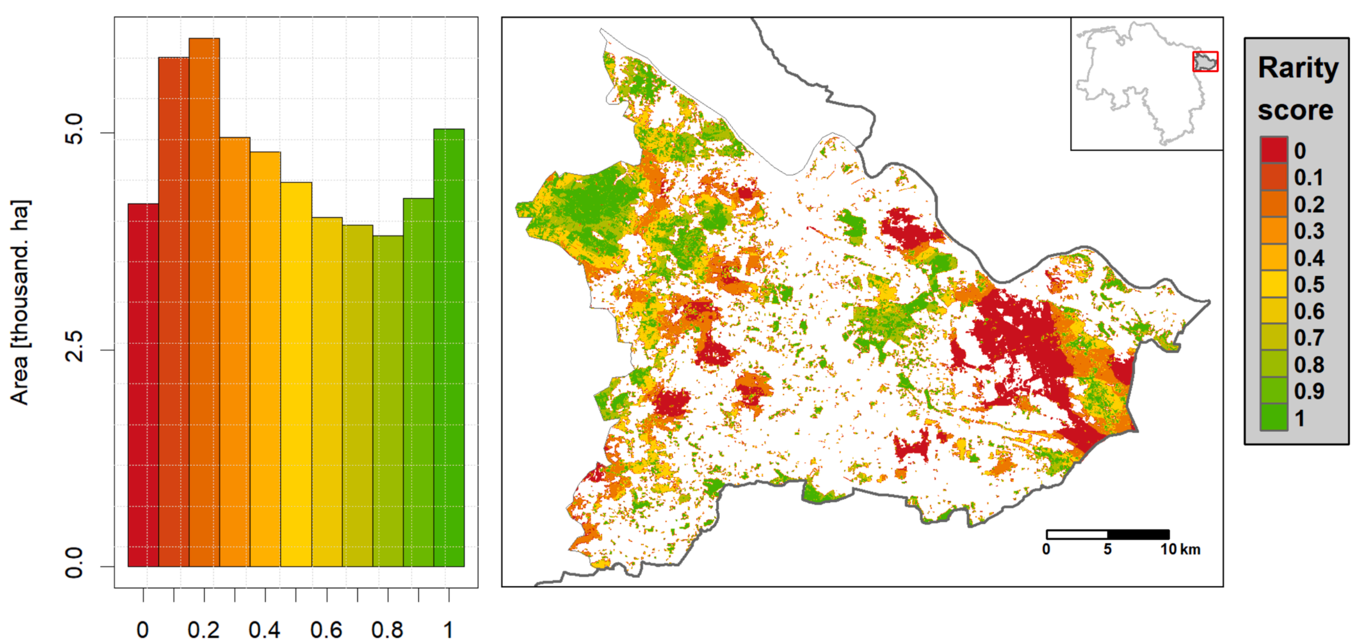

Fig. 8. Spatial representation of landscape-level species rarity within the study area. The map illustrates the species rarity of each cell, with colour intensities indicating varying degrees of rarity. The rarity of each grid cell was calculated in relation to the proportion of tree species within a radius of 1700 m The histogram displays the frequency distribution of rarity scores across the entire area, with the x-axis representing rarity levels and the y-axis representing the cumulative hectare values.

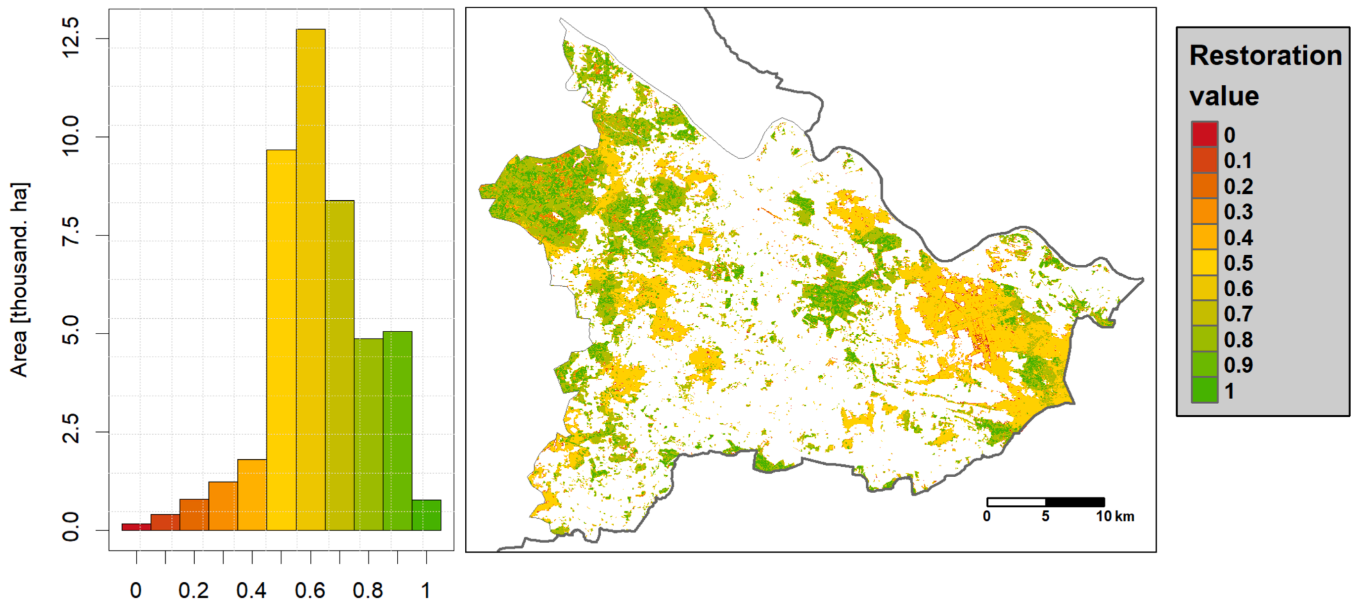

Fig. 9. Aggregated restoration values with equal weighting of the indicators for the study area. This map illustrates the prioritization of forest restoration areas based on a multi-criteria evaluation, incorporating drought stress, stand-level species diversity, and landscape-level rarity.

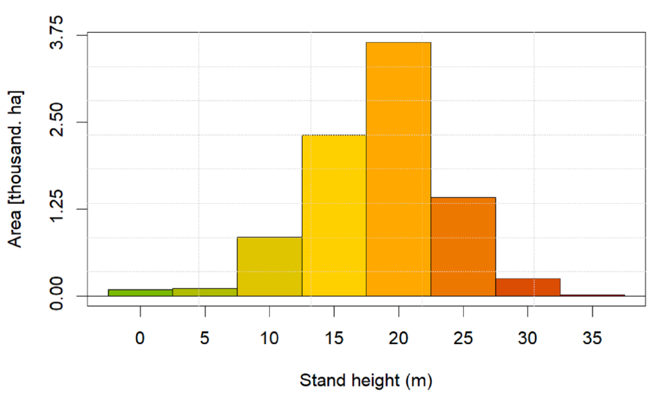

Fig. 10. Distribution of the canopy height of stands with increased restoration priority.

< 0.5) in the study area. The distribution highlights the varying stages of forest development within these areas. Most of the high-priority restoration areas exhibit average canopy heights indicative of mid-successional stages, suggesting that these stands may benefit from interventions aimed at promoting structural diversity and resilience. A minor proportion of forest stands have average canopy heights towards the lower or upper end of the range. These forest stands in the early or late successional stage may require different management approaches. The varying average stand heights, as a proxy for successional development stages, suggest that targeted forestry measures, such as thinning or species diversification, could be strategically applied depending on the specific successional stage and restoration goals. This can guide the temporal prioritization of restoration efforts.

3.5. Sensitivity

The sensitivity analysis reveals the impact of different weighting schemes on the restoration value. Overall, the sensitivity analysis highlights the importance of carefully selecting appropriate weightings based on the objectives of the study. This tailored approach enables a more nuanced understanding of forest restoration dynamics and enhances the specificity of the results.

The largest area of restoration is observed with a weighting of 0.5 each for rarity and diversity, and 0 for drought stress (Fig. 11). With this combination, 33,600 ha are analyzed that have a restoration value lower than the threshold value of 0.5. Conversely, if only the drought stress risk is considered and the other two variables are weighted with 0, the lowest forest restoration area is obtained. Around 5,000 ha with a restoration value lower than 0.5 will then be derived, demonstrating that drought stress alone identifies a much smaller area needing restoration.

4. Discussion

Drought stress risk, diversity at the stand level, rarity at the landscape level and stand age are the key criteria underscoring the multifaceted nature of forest resilience. By applying our indicator-based approach, we can quantify these aspects across the study area, enabling the identification of priority areas for management and restoration. The following discussion evaluates the strengths and limitations of our approach while interpreting the findings in the context of existing research.

4.1. Drought stress risk

In the 21st century, a warmer and, in many regions, drier climate is expected (IPCC, 2014). For Central Europe, mean precipitations may not change substantially, but seasonal shifts –particular drier summers –are projected (Hübener et al., 2017). Consequently, an increase in the frequency and intensity of forest disturbances such as drought, storms, and bark beetle outbreaks is anticipated (Seidl et al., 2017; Senf and Seidl, 2021; Anderegg et al., 2022; Patacca et al., 2023). These changes imply significant impacts on forest functioning and resilience (Allen et al., 2010; Senf et al., 2020). These impacts can be stress, disease and tree mortality (Rykiel, 1985). Causes of mortality (i.e., factors, agents, pathogens) are complex and are often classified as predisposing, inciting (primary) and contributing (secondary) factors, which can form a decline spiral and possibly lead to death (Manion, 1991, p. 333). In this context, climate change-induced droughts become the most prominent factor (Buras et al., 2020; Senf et al., 2020). Secular drought has to be seen as a predisposing factor and heavy drought events depict inciting causes for lethal stress (Bigler et al., 2006; Rykiel, 1985). Generally, it is

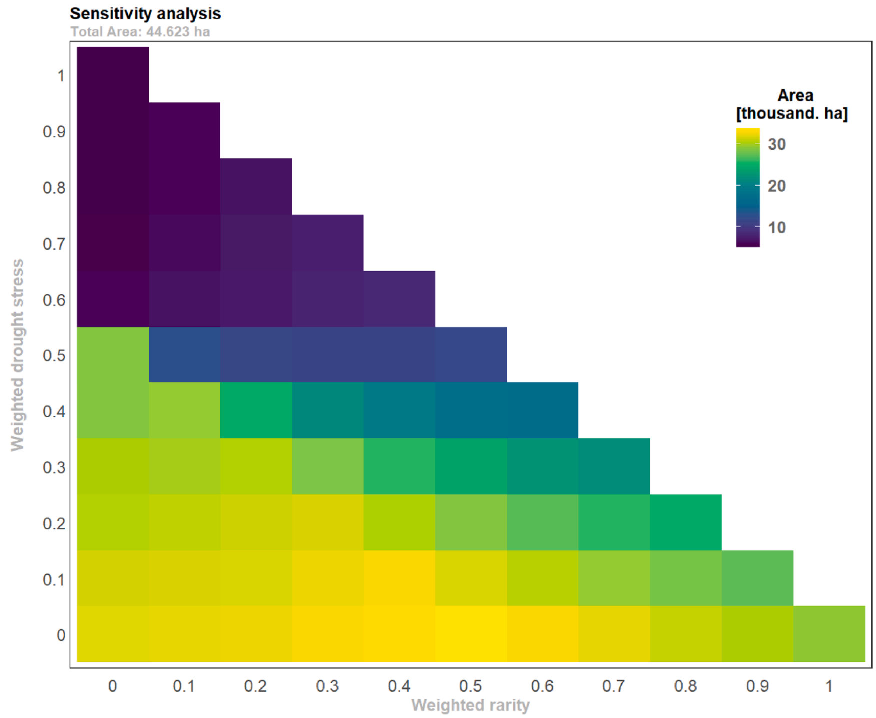

Fig. 11. Illustration of the sensitivity analysis, in which the variables drought stress, rarity at landscape level and diversity in the stand were weighted variably. The weighting of evenness is derived from the difference between the drought stress weight and the rarity weight. The forest restoration area with a multi-attribute value < 0.5 is shown as a coloured gradient.

difficult to distinguish if tree mortality is caused by a pathogen itself, or if a tree has died with this pathogen present.

Actual vulnerability to drought can also be clouded by background mortality (i.e., influenced by competition), especially in dense forests (Archambeau et al., 2020; Kulha et al., 2023; Sánchez-Salguero et al., 2015) or at edges (Biber and Pretzsch, 2022; Buras et al., 2018). Therefore, our drought stress risk assessment was designed to identify site-specific areas where mortality and vitality reduction are most likely. This encapsulates both growth reactions and the risk of exposure to biotic factors using the SWB (Jaime et al., 2019; Martinez del Castillo et al., 2024).

Norway spruce stands in the study area grow on a SWB of $\leq$ 0 mm within the vegetation period and thus possess the highest drought stress risk according to Albert et al. (2017; Table 1). Our findings are supported by the significant reduction in radial growth in drought years (Lévesque et al., 2014; Vitali et al., 2017) and increased susceptibility to biotic pests. Biotic pests, such as the spruce bark beetle (Jönsson et al., 2012; Marini et al., 2017; Stadelmann et al., 2013), contribute secondarily to tree death (Manion, 1991, p. 333). Therefore, the drought stress assessment we have applied (Fig. 4) is aligned with present studies that identify spruce stands as highly endangered (Aldea et al., 2024; Brandl et al., 2020; Fuchs et al., 2024; Honkaniemi et al., 2020; Lévesque et al., 2014).

Within the study region, Scots pine is the most abundant species (Fig. 1). However, it is also seen as having high drought stress, i.e. mortality risk (Bigler et al., 2006; Haberstroh et al., 2022; Hartmann et al., 2022; Lemaire et al., 2022; Patacca et al., 2023; Rehschuh and Rühr, 2021), while its hardiness and (epigenetic) adaptation potential is simultaneously praised (Bose et al., 2020, 2024; Brichta et al., 2023). Predisposing factors for tree death of pine are often associated with drought, as soil and climatic stressors affect health in the long term. Stony, shallow soils with low water capacity (Bose et al., 2024) and insufficient nutrient supply (Bose et al., 2020) primarily challenge pine in continental regions (Diers et al., 2024; Enderle et al., 2024). High stand densities (del Río et al., 2017; Jaime et al., 2019; Sohn et al., 2016), senescing old trees and missing (Steckel et al., 2020) or inappropriate admixtures (Špulák, 2023) also predispose pines.

The growth trends of pine in Europe vary strongly (Pretzsch et al., 2023). Central European pine stands, which are found in our study area, appear to compensate for the growth impairments caused by warmer and drier summers by stimulating growth through warmer late winter periods (Diers et al., 2024, 2023). Furthermore, there are indications that larger pines are more affected by drought than smaller trees (Merlin et al., 2015; Mueller-Dombois, 1987). This size–vulnerability relationship is directly represented in our CHM (Fig. 2) and taken into account in further prioritization (Fig. 10).

Appropriate forest management of pine stands can mitigate drought stress. Thinning reduces competition for water, thereby lowering drought stress (Sohn et al., 2016, p. 20). The risk of pest outbreaks (i.e., bark beetle, butterfly or jewel beetle gradations), such as those linked to drought and higher temperatures (Netherer and Schopf, 2010; Ray et al., 2016; Skrzecz et al., 2020), further increases stress and vulnerability of pine. Even if biotic pathogens increase in intensity and frequency in the future (Jabłoński et al., 2019), they will continue to be linked to a drought-related weakening of the trees. This favors plant and fungal parasites as dieback due to mistletoe or *Diplodia sapinea* (Fr.) Fuckel (Brodde et al., 2023; Dobbertin and Rigling, 2006).

Recent studies show that Sessile and Pedunculate oak have a high resistance to drought and show less damage and lower mortality compared to beech and spruce (Brandl et al., 2020). Our assessment confirms these findings (Fig. 4): Stands with a high proportion of oak showed the lowest drought stress values for the projections within the study area (Table 1). While other tree species in the northern German lowlands show a decline in growth with higher temperatures, oaks show a positive basal area increment trend (Enderle et al., 2024). This capacity to adapt to the warmer conditions and the increasing likelihood of summer droughts projected for Central Europe under future climate scenarios emphasizes their importance as a key species for the design of climate-resilient forests (Gribbe et al., 2024; Rubio-Cuadrado et al., 2018; Vrška et al., 2017).

From the preceding discussion it is apparent that assessing drought risk is complex (Pretzsch et al., 2013). Our methodology addresses this by weighting tree species according to their proportional representation in a stand (Section 2.3.2), providing a practice-oriented framework for classifying drought stress risk. As a result, less drought-tolerant species such as beech and spruce benefit from being mixed with drought-tolerant species such as oak or pine (Fig. 4). While intuitive, this approach only partially accounts for species interactions, such as facilitation or competition (Bauhus et al., 2017; Pretzsch et al., 2013). In mixed forests, synergistic effects –such as complementary water and nutrient use –can reduce overall drought stress, even when individual species exhibit high stress levels. For pine stands, the inclusion of drought-tolerant species like *oaks* enhances resilience through resource complementarity (Giberti et al., 2023; Steckel et al., 2020). Conversely, competition from spruce, even when limited to the understory, can intensify drought stress for co-occurring species (Špulák, 2023). A higher survival rate was observed for spruce in mixed stands, also in a drier and warmer climates (Brandl et al., 2020; Neuner et al., 2015). As noted by Grossiord et al. (2014) and Forrester et al. (2016), drought stress can sometimes be lower in mixed forests. However, this is not a general pattern they found, as there were many two-species mixtures where both species had higher water stress compared to their monocultures.

Despite the complexity of mixed forest effects, the presented drought stress risk classification to prioritize forests for restoration (Fig. 6) reflects increasing mortality projections for lowland conditions (Allen et al., 2010; Nothdurft, 2013). The chosen climate model (Fig. 4) can be updated and adapted at any time to reflect new data or advances in climate science (Eyring et al., 2016; Pirani et al., 2024), ensuring that the drought risk assessment remains robust and relevant under changing conditions.

4.2. Diversity at stand level

The evaluation of biodiversity at the stand level provides valuable insights into its resilience (Yachi and Loreau, 1999). The incorporation of remote sensing data allowed for wall-to-wall mapping of tree species diversity in the study area and across ownership boundaries. Earlier approaches mainly served nature conservation purposes (Foody and Cutler, 2006; Madonsela et al., 2017; Redowan, 2015; Starčević et al., 2020). By calculating diversity indices such as species richness, Shannon index, and evenness, we identified critical regions for forest restoration requiring diversification.

The species distribution data we applied to model diversity indices is derived from optical satellite imagery with a spatial resolution of 10 m (Sentinel-2 data; Blickensdörfer et al., 2024). Consequently, there are certain limitations of this data in terms of species mapping and thus biodiversity indices. As wavelengths of optical satellite imagery are not able to penetrate the upper canopy layer, we only get information about the topmost trees. Potential species only present in the understory are not assessed. Because the reflectance of 10 m pixels consists of mixed signals, the processed information is only valid for the dominant species and might miss smaller trees; it is therefore less accurate for mixed forest stands. Additionally, the model was trained to accurately map the main tree species and is less accurate for rarer tree species. These limitations will most likely lead to underestimated biodiversity indices and should not be compared to indices based on field data. All these factors have to be considered when interpreting the results of the present tree species distribution and biodiversity indices. Nonetheless, the applicability remains, especially in the case of structurally poor stands (Fig. 1c). The uncertainties of the species map have been quantified in Blickensdörfer et al. (2024), which should guide the interpretation of results.

Our findings demonstrate that many stands within the study area exhibit low canopy tree species diversity, with a significant proportion showing evenness values below 0.5 (Fig. 7). As stands with low species diversity are generally more susceptible to pests, diseases, and environmental stressors (Messier et al., 2022), this highlights a notable vulnerability. A review by Jactel et al. (2005) confirmed the 'diversity-stability theory', stating that mixed stands are generally less affected by pests than pure stands.

The comparison of tree species within our study shows that pine stands are particularly species-poor (Table 2). Numerous findings indicate that these species-poor pine stands are more severely affected by pests: The susceptibility of pure pine stands to needle-feeding insects was shown early on to decrease when deciduous trees are admixed (Lüdge, 1971). Higher densities of parasitoid wasps as antagonists of pest insects have been detected when deciduous trees, especially oak, were admixed (Jäkel and Roth, 2004). Furthermore, the probability of bark beetle infestation was higher for stands with a high pine basal area (Jaime et al., 2019). However, under the influence of recent megadroughts in Central Europe (Bose et al., 2022; Büntgen et al., 2021), evidence has emerged suggesting that very high species diversity can increase the likelihood of functionally redundant neighbors. This, in turn, reduces niche complementarity and intensifies interspecific competition, ultimately leading to higher mortality (Searle et al., 2022; Shovon et al., 2024).

In addition to biotic risks (Guyot et al., 2016), the diversity of forest stands also influences their resistance to abiotic factors, such as forest fires. Forest fires are expected to increase as a result of climate change due to lower precipitation and higher temperatures (Fig. 3). Evidence from fire-prone regions indicates that mixed forests containing both coniferous and deciduous tree species are less fire-prone than pure coniferous forests (Hély et al., 2001, 2000). In addition, fires in mixed forests tend to be spatially less extensive and their damages on the tree level less intense (González et al., 2006; Silva et al., 2009; Wang, 2002).

The preceding discussion shows that evenness emerges as a suitable indicator for assessing the resilience of forest stands, as low values indicate heightened susceptibility to biotic and abiotic stress factors. The predominance of pure pine stands in the center-east of the study area, underscores the need for targeted interventions to improve yield, ecological stability and economic viability (Knoke et al., 2008; Silva Pedro et al., 2015; Spathelf and Ammer, 2015). Pure stands could be diversified through management strategies such as mixed-species planting, seeding or natural regeneration facilitation to enhance structural and functional diversity (Axer et al., 2022; Fischer and Fischer, 2012; Fischer et al., 2016; Huth et al., 2017; Spathelf et al., 2018; Stanturf, 2016; von Lüpke and Sennhenn-Reulen, 2023).

Conversely, the higher evenness scores in the north-west region of the study area reflect a greater proportion of mixed stands, suggesting these areas may serve as benchmarks or reservoirs of resilience. This contrast between regions emphasizes the importance of spatially explicit strategies in forest management. By prioritizing the diversification of homogenous stands, especially in regions with low evenness, forest managers can mitigate risks and enhance the adaptability of forest ecosystems to changing environmental conditions (Schoene and Bernier, 2012).

4.3. Rarity at landscape scale

Forest management thrives to diversify stands regarding species, structures (layering) and genetics, i.e. α-diversity (Kimmins, 2004; see 4.2. Diversity at stand level). However, this process inevitably leads to an alignment of forest types across the landscape (Fuchs et al., 2024; Pretzsch, 2019). As different forest sites promote β-diversity, so that swamplands or weak nutrient levels differentiate stocking (Stein et al., 2014), there are calls to focus more on β- and γ-diversity across the landscapes. β- and γ-diversity have been identified as important elements for the functioning of ecosystems (Mori et al., 2018; Schuler et al., 2017). Sebald et al. (2021) showed that mixing tree species between stands is at least as effective as mixing tree species within stands to mitigate disturbances. This includes structures, species and management concepts (Müller et al., 2022).

We developed an approach for how to combine both viewpoints by enhancing diversity within stands through the evenness index (Larsen and Nielsen, 2007; Markowitz, 1991; Pretzsch et al., 2013) and by keeping rare elements at a landscape scale through the rarity index, even if they are monospecific or maladapted (Heinrichs et al., 2019). The approach of lowering the weight of common species supports diversity maximisation, while taking into account the ecological importance of rare species (Fig. 3). As numerous studies have shown, individual trees or groups of old oaks or beeches in homogeneous coniferous forests, in particular, create diverse ecological niches for different species groups (Koch Widerberg et al., 2012; Pilskog et al., 2016; Wehnert et al., 2020). Assessing the ecological value of single tree admixtures is even more important in landscapes with predominance of a single tree species over large areas (Fig. 1).

A further example is rare indicator species (Dufrêne and Legendre, 1997), such as the forest owl *Glaucidium passerinum* L.. They are especially present in lowland Spruce forests, whose silvicultural aspects must be restored (cf. drought assessment in Fig. 6), and could function as a tradeoff and be preserved from the viewpoint of ecology (Rothgänger, 2023). Light-transmissive coniferous species overall host many specialized taxa, which promote biodiversity (Brändle and Brandl, 2001; Heinrichs et al., 2019) and the multifunctionality of forests with their ecosystem services (Schuler et al., 2017).

In addition to nature conservation aspects, the rarity indicators also appear to be suitable for forest management. Rare tree species are of particular value at the landscape scale as seed trees, as they contribute significantly to natural regeneration and thus serve as cornerstones for forest restoration (Axer et al., 2022; Dobrovolny and Tesař, 2010; Kunstler et al., 2004; Zerbe, 2002). Rare elements are weighted particularly highly (Fig. 3) and can thus be considered as natural regeneration potential after disturbances (Schüle et al., 2023; Tiebel et al., 2020). This consideration becomes even more crucial when we account for the increasing risk of bark beetle infestations, which rises sharply with spruce volume at the landscape level (Stadelmann et al., 2013). Simulation studies confirm that reducing the proportion of spruce in the landscape increases the resilience of the remaining spruce trees (Honkaniemi et al., 2020). A study by Kautz et al. (2011) showed that the dispersal of the European spruce bark beetle *Ips typographus* L. is strongly dependent on distance and that 95 % of new infestations occurred within a radius of 500 m of the previous year's infestation sites.

From the preceding discussion, the assumption that rare tree species make a higher contribution to biodiversity at the landscape scale provides a basis for decision making in forest restoration. The approach provides a robust methodology for calculating rarity scores that can be linked to site-specific data to support forest restoration prioritisation (Fig. 8). These findings underscore the utility of remote sensing in biodiversity assessment and decision-making, enabling the identification of priority areas for intervention.

4.4. Standardization and value functions: global applicability and impact of weighting

As already mentioned in chapter 2.3. Multi-Criteria Decision Analysis, the specific definition of the value functions is of crucial importance for the results. In order to be able to apply the value functions globally, it is equally important not to carry out a purely linear scaled transformation. This would make comparability between different scenarios impossible. Since the decision model should also be able to be adapted to changes in the input data with as little effort as possible, the global methods for defining value functions are also suitable here. This also allows changes between different data statuses to be quantified.

Furthermore, the results of the decision model are heavily dependent on the weighting of the individual indicators. The sensitivity analysis shows that the area increases significantly with a higher weighting of the diversity indicators (Fig. 11). However, it is up to the decision-maker to determine the weighting. There are various approaches for methodically determining the weighting (e.g. Analytical Hierarchy Process; Saaty et al., 2012).

5. Conclusions

Our study highlights key factors influencing forest resilience, including drought stress risk, stand-level diversity, rarity at the landscape scale, and stand age. While previous approaches in Germany end at the ownership boundary (Grüll et al., 2020), we demonstrate how to combine site- and remote sensing information at the landscape level. The integration of multi-source spatial datasets enables a landscape-level assessment, providing robust guidance for forest restoration planning.

Key contributions and implications:

- **For forest owners:** Prioritize restoration of highly vulnerable or species-poor stands, balancing timber production with biodiversity enhancement.
- **For planners and authorities:** Use our multidimensional indicators to allocate resources efficiently, identify priority restoration areas, and adapt funding mechanisms and silvicultural guidelines.
- **For restoration practitioners:** The approach supports planning of planting, thinning, and species-mixing strategies, incorporating rare species and enhancing ecosystem resilience.

Limitations and future steps:

- While our indicators provide a comprehensive framework, further refinement could integrate additional local site conditions or climate projections.
- Long-term monitoring is needed to validate model predictions and to adapt management strategies over time.
- Expanding the approach to include socio-economic constraints and stakeholder preferences could further improve decision-making.

Overall, the analysis emphasizes the necessity of a comprehensive and flexible framework for forest restoration prioritization (Cannon et al., 2020; Cavalcante et al., 2022; Rayden et al., 2023; Vettorazzi and Valente, 2016). Our study demonstrates that a flexible, indicator-based framework can accelerate forest adaptation to climate change, foster species-rich and resilient ecosystems, and support coordinated action among forest owners, policymakers, and practitioners. Given limited resources such as seedlings, labor, and subsidies, strategic prioritization based on these indicators is crucial for effective forest management (Charron and Hermanutz, 2016; Silva et al., 2023).

CRediT authorship contribution statement

Maximilian Axer: Writing – original draft, Visualization, Software, Methodology, Formal analysis, Data curation, Conceptualization. **Hans Hamkens:** Writing – review & editing, Visualization, Software, Methodology, Formal analysis, Data curation, Conceptualization. **Hergen Knocke:** Writing – original draft, Visualization, Formal analysis. **Jens Wiesehahn:** Writing – review & editing, Software, Formal analysis, Data curation.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Maximilian Axer reports article publishing charges was provided by Northwest German Forest Research Institute. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Data availability

The authors do not have permission to share data.

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