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Shifts in forest management intensity mediate biodiversity outcomes under contrasting socio-economic and climate scenarios

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Abstract

Forest extent and forest management are both important in shaping biodiversity responses to climate and land use changes. Yet biodiversity impact models typically represent managed forests as an aggregated category despite advances in land use models that differentiate between forest management practices. This limits the ability to assess how shifts between management practices impact biodiversity under future scenarios. We addressed this gap by coupling the biodiversity model GLOBIO with the agent-based land use model CRAFTY-EU. GLOBIO simulates changes in mean species abundance (MSA) in response to drivers of biodiversity loss. CRAFTY-EU allocates land uses including six forest management practices with different timber harvesting intensities based on societal demands and socioeconomic and biophysical suitability. We integrated GLOBIO's climate change, land use, and updated forest management MSA estimates to evaluate biodiversity outcomes under three socioeconomic and climate scenarios (SSP1-RCP2.6, SSP3-RCP7.0, and SSP5-RCP8.5). Forest expansion partially buffered biodiversity loss only where agricultural abandonment was substantial and climate impacts were moderate. This effect was strongest under SSP1-RCP2.6, weaker under SSP5-RCP8.5 because climate losses dominated, and largely absent under SSP3-RCP7.0 due to limited abandonment and forest decline. Across all scenarios, shifts towards intensive forest management reduced MSA, showing that forest expansion alone does not guarantee biodiversity benefits, especially under management intensification and strong climate change. Land use and forestry policies that promote forest recovery while avoiding management intensification can moderate biodiversity losses, but are unlikely to compensate for strong climate-driven declines. Such policies align with the EU's Nature Restoration Regulation, which proposes planting 3 billion trees and restoring 20% of land by 2030. Successful implementation requires consideration of prior land use to avoid converting biodiversity-rich areas and planting and management strategies that ensure positive biodiversity outcomes.

1. Introduction

Forest management is important in conserving biodiversity under changing socio-economic and climate conditions (Luo *et al* 2024). While Europe's forests have remained relatively stable in terms of total forest cover and management intensity over recent decades (Sabatini *et al* 2018), rising demographic, economic, and climatic pressures threaten this stability, potentially altering both forest extent and management across Europe (Popp *et al* 2017, Luo *et al* 2024). Shifts in forest management practices, such as from selective cutting of naturally regenerating forests to monoculture plantations, or vice versa, can profoundly impact biodiversity (van 't Veen *et al* 2025). Furthermore, climate change increasingly affects forest biodiversity through rising temperatures (Schipper *et al* 2020) and altered disturbance regimes, such as prolonged droughts and fires (Nitschke and Innes 2007). Meanwhile, productivity gains in agricultural and forestry, arising from biotechnological developments and CO_2 fertilisation, may reduce land demand, enabling land abandonment and opportunities for reforestation and afforestation (Fayet *et al* 2022, Kok *et al* 2023). Understanding the capacity of European forests to contribute to biodiversity conservation requires unravelling how alternative socio-economic and climate futures shape forest extent and management.

Evaluating the role of forests in shaping biodiversity responses to socioeconomic and climate changes requires modelling frameworks that can simultaneously represent land use change, forest management decisions, and climate-driven biodiversity responses over space and time. Previous scenario studies primarily linked broad land cover categories and climate change to aggregate biodiversity indicators, such as species richness, habitat suitability, and intactness (Pereira *et al* 2024). For example, species distribution models have been used to estimate changes in species richness or range size under alternative climate and land use futures, with limited consideration of management impacts (e.g. Martin *et al* 2013, Visconti *et al* 2015, Sirami *et al* 2017, Chapman *et al* 2025). Additionally, models simulating biodiversity loss relative to reference ecosystems (e.g. undisturbed forests or grasslands), using indicators such as mean species abundance (MSA) and the Biodiversity Intactness Index, have been used to quantify overall biodiversity loss in response to land use change only (e.g. Hill *et al* 2018) or to land use and climate change (e.g. Schipper *et al* 2020, Kok *et al* 2023). Other studies have focused on forest biodiversity outcomes using proxies such as forest cover change, or naturalness indices, often assuming uniform biodiversity responses across forest types (e.g. Marzini *et al* 2026).

While prior modelling efforts have provided important insights into the magnitude and spatial distribution of biodiversity change under alternative socio-economic pathways (Schipper *et al* 2020), they typically treat forestry as a single land use category, distinguish only broad land use classes (van 't Veen *et al* 2025), or do not account for the direct effects of climate change combined with forest management on biodiversity intactness (Chapman *et al* 2025). While some studies test the effects of transitions in forest management on biodiversity, they only account for potential percentage shifts in forest management in response to scenarios (e.g. 10% forestry land shifts to reduced-impact logging) without allocating such land management changes in a spatially-explicit way (Kok *et al* 2023, 2024). As a result, they are limited in their ability to capture biodiversity impacts associated with shifts between different management strategies. This limitation persists, despite growing empirical evidence that forest management practices can strongly influence local (alpha) forest biodiversity outcomes (Chaudhary *et al* 2016, van 't Veen *et al* 2025). Consequently, the biodiversity implications of forest management under alternative socio-economic and climate futures remain poorly understood (Leclère *et al* 2020).

Recent advances in land-system and agent-based models that allow representation of alternative forest management practices over large spatial extents (e.g. CRAFTY; Murray-Rust *et al* 2014) now allow representation of diverse forest management practices. When coupled with biodiversity models that include forest management (e.g. GLOBIO; Schipper *et al* 2020, van 't Veen *et al* 2025), these frameworks enable assessment of biodiversity responses to both forest area changes and management shifts.

In this study, we ask the question: how could changes in forest extent and forest management in response to alternative socio-economic and climate change futures shape the contribution of European forests to biodiversity conservation? To address this question, we coupled the agent-based land use model CRAFTY-EU with the biodiversity model GLOBIO, enabling simulation of the impacts of different forest management practices on biodiversity under contrasting scenarios. Scenarios were defined using the shared socioeconomic pathways (SSPs) framework combined with the representative concentration pathways (RCPs) to represent contrasting socio-economic and climatic change futures (Riahi *et al* 2011, O'Neill *et al* 2014, Kok *et al* 2019).

2. Methodology

2.1. Modelling framework

We assessed biodiversity responses in European forests under alternative SSP–RCP scenarios by coupling the global biodiversity model GLOBIO with the agent-based land use model CRAFTY-EU. This integrated framework links land use, including forest management, to biodiversity responses, enabling consistent evaluation of changes in forest extent and management in the context of climate change.

2.1.1. The GLOBIO model

The GLOBIO 4 model (hereafter referred to as GLOBIO) is a global open access model (https://github.com/GLOBIO4/GlobioModelPublic/wiki, last accessed 07.01.2026) operating at 10-arc second resolution (~300 m at the equator) that simulates changes in animal and plant biodiversity in response to drivers

Box 1
Mean species abundance (MSA) is a measure of biodiversity intactness. For forests, it is calculated by dividing the abundance of species in managed forest sites by the abundance of the same species in undisturbed forest sites. If the abundance response ratio is higher than 1, the value is set to 1 (maximum biodiversity intactness). If a species does not occur in the undisturbed forest sites, it is excluded from the calculation.

Table 1. GLOBIO drivers of biodiversity loss relevant to this study, the taxonomic groups they affect and the types of land use they are associated with. Natural land encompasses primary vegetation, secondary vegetation and forestry classes. For the land use and climate change drivers, the reported MSA values are the mean of the corresponding animal and plant MSA values.

Driver of loss	Taxonomic group	Land use
Land use/management	Animals & plants	All land use
Climate change	Animals & plants	All land use
Atmospheric nitrogen deposition	Plants	Only natural land
Habitat fragmentation	Animals	Only natural land
Road disturbance	Animals	All land use

of loss, including (i) land use change, (ii) habitat fragmentation, (iii) road disturbance, (iv) atmospheric nitrogen pollution and (v) climate change represented by global temperature increase (Schipper *et al* 2020). GLOBIO uses MSA, which is an indicator of biodiversity intactness and reflects the differences in species assemblages, including their abundances, between sites impacted by human disturbances and corresponding reference sites (e.g. undisturbed forests and grasslands) (Box 1) (see SI section 1 for model assumptions). We chose MSA as an indicator of biodiversity because it quantifies the difference in species composition between a disturbed (e.g. managed forests) and undisturbed state, allowing us to simulate the extent to which biodiversity recovers to its reference state. Unlike other biodiversity metrics (e.g. Biodiversity Intactness Index (Liu *et al* 2025)), effects of a wide range of biodiversity loss drivers on MSA have been established (Schipper *et al* 2020), allowing us to unravel the effects of changes in land use relative to the impact of other drivers, such as climate change (see SI section 1 for further rationale).

For each driver, a pressure-impact relationship has been introduced in GLOBIO, which is a beta-regression model where MSA is the response variable and the driver is the predictor. Not all drivers affect all taxonomic groups included in GLOBIO and some drivers only apply to natural land use, defined as primary vegetation, secondary vegetation and forestry classes (table 1).

The total terrestrial MSA per grid cell is derived by multiplying the MSA raster layers, using equation (1):

$$\mathrm{MSA}_{g,i} = \prod_{x} \mathrm{MSA}_{x.g,i} \qquad (1)$$

where $\mathrm{MSA}_{g,i}$ is the MSA of species group g in grid cell i, and $\mathrm{MSA}_{x,g,i}$ is the MSA of species group g in response to biodiversity loss related to driver x (e.g. land use) in grid cell i.

Once MSA is estimated for each driver and effects of all drivers are combined, the proportional effect of each driver can be determined using equation (2):

$$P_{x,g,i} = \frac{1 - \mathrm{MSA}_{x,g,i}}{\sum_{x=1}^{x=m}\left(1 - \mathrm{MSA}_{x,g,i}\right)} * \left(1 - \mathrm{MSA}_i\right) \qquad (2)$$

where $P_{x,g,i}$ is the proportional contribution of driver x to the MSA loss of species group g in grid cell i. Calculating proportional contributions ensures that the sum of the MSA loss by each driver of loss equals total MSA loss (Schipper *et al* 2020). This allows for an assessment of changes in the relative contributions of different drivers to MSA loss across different scenario assumptions.

2.1.2. The CRAFTY-EU model

CRAFTY is an agent-based modelling framework that simulates land use and management dynamics (Murray-Rust *et al* 2014). In this study, we used CRAFTY-EU, representing the European Union (EU), Norway, Switzerland and the United Kingdom land areas as a grid of 1 × 1 km cells (EPSG:3035), each characterised by levels of natural, human, social, financial, and manufactured resources, or capitals, that determine local ecosystem service production potentials (e.g. for timber, crops or non-material services). Agents are land managers that are categorised into agent functional types (AFTs) (Arneth *et al* 2014) with distinct behavioural traits, levels of ecosystem service provisioning, and sensitivities to resources (table 2; also see SI table A1 for a detailed description of AFTs). The baseline distribution of AFTs is defined by empirical land use and management data from HILDA+ (Winkler *et al* 2025), the EU land systems map (Dou *et al* 2021) and a tree species map (De Keersmaecker *et al* 2024), with the latter two sources providing intensity and species choice within the broad HILDA+ classes (see Brown *et al* 2026, for more information). AFTs utilise resources to produce ecosystem services, in response to societal demands for these services, while seeking to maximise their utility under prevailing socio-economic and environmental conditions (see SI section 2 for more details).

CRAFTY-EU sits within the LandSyMM modelling framework (Rabin *et al* 2020, Henry *et al* 2022, Arendarczyk *et al* 2025), operating in interaction with a dynamic vegetation model (i.e. LPJ-GUESS) (Smith *et al* 2001, Smith *et al* 2014, Martín Belda *et al* 2022) and a global societal demand and trade model (PLUM) (Alexander *et al* 2018, Arendarczyk

Table 2. Mapping of CRAFTY-EU AFTs and GLOBIO forest management (LUM) classes to derive the CRAFTY-GLOBIO LUM classes adhered to in this study. See table 1 in van 't Veen *et al* (2025) for more detailed definitions of the GLOBIO forest management classes distinguished. See SI table A2 for associated MSA values. See SI section 1 for detailed hypotheses on how MSA was expected to respond to forest management-related alterations of forest structure and diversity.

CRAFTY-GLOBIO	CRAFTY		GLOBIO	
LUM class	AFT	Definition	LUM class	Definition
Intensive forestry	Broadleaf forest (intensive) Conifer forest (intensive)	Broadleaf forest with thinning of 40% of trees every 30 years Conifer forest with thinning of 40% of trees every 30 years	Forest plantations	Trees are planted to produce a plantation. Replanting occurs after plantations are harvested.
Extensive forestry	Broadleaf forest (extensive) Conifer forest (extensive) Multifunctional mixed woodland (extensive)	Broadleaf forest with thinning of 10% of trees every 30 years Conifer forest with thinning of 10% of trees every 30 years Mixed broadleaf and conifer forest with thinning of 10% of trees every 30 years	Clear cutting Selective cutting	Removal of all trees in a forest area. Natural forest regeneration through succession. Selective harvesting of trees, resulting in an uneven forest structure.
Conservation forest	Conserved Woodland	Protected forests with very little or no management	Primary vegetation	Undisturbed ecosystems (e.g. forests and grasslands), showing no evident signs of human activity for at least 80 years.
Secondary vegetation	Abandoned land	Land where no agent actively manages the land. A cell is classified as abandoned when no agent selects an active management option under the scenario-specific social, economic, natural and institutional constraints.	Secondary vegetation	Reforestation or afforestation through natural regeneration or tree planting, ultimately transitioning from the prior land use status to 'Conservation Forest' over a period of 100 years.

et al 2025). This setting allows scenario-driven changes in, among other things, climate, land productivity, and trade conditions to influence agent behaviour. Crop, pasture and forest harvest potentials are derived from LPJ-GUESS and production demands for timber and crops are provided from PLUM (e.g. Brown et al 2022).

2.2. Coupling GLOBIO and CRAFTY

We coupled GLOBIO and CRAFTY so that the spatial distribution of CRAFTY AFTs was used to derive MSA estimates for all drivers of biodiversity loss (table 1).

2.2.1. Land use/management

We estimated the average impact of each AFT on MSA by linking GLOBIO non-forest land use classes (i.e. cropland, pastures, and urban areas) described under Schipper et al (2020) and forest management classes described under van 't Veen et al (2025) to the corresponding CRAFTY AFT classes (SI tables A1 and A2) based on assumptions about the relative contribution of each AFT to MSA loss. In the process, we produced distinct CRAFTY-GLOBIO land use and management classes. Differences in the level of detail regarding land use and management classes mean that the classifications do not align perfectly.

Forest management-related MSA values for GLOBIO have been recently updated, providing more detailed and robust estimates (van 't Veen et al 2025). GLOBIO distinguishes between forest plantations (lowest MSA values), clear cutting and regrowth (medium MSA values) and selective cutting (highest MSA values) (van 't Veen et al 2025) (see table 2 for definitions; and SI table A2 for MSA values). As CRAFTY did not explicitly differentiate these forest practices, we matched CRAFTY AFTs to the GLOBIO forest-management MSA values based on their expected relative biodiversity impact rather than exact equivalence between management definitions. Thus, the GLOBIO labels should be interpreted as impact classes used to assign MSA values, not as literal descriptions of CRAFTY silvicultural practices. Specifically, intensive broadleaf and conifer forestry were assigned the lowest forest-management MSA values because they are the most production-oriented AFTs and have the highest assumed harvest intensity in CRAFTY. Extensive broadleaf and conifer forestry were assigned intermediate MSA values because they involve lower-intensity but recurrent biomass removal. As GLOBIO represents forestry impacts through disturbance-based classes rather than thinning intensity, these AFTs were assigned the 'clear cutting and regrowth' MSA values as an intermediate disturbance-impact proxy, not as a literal clear-cutting classification. Multifunctional mixed woodland was assigned higher MSA values due to its mixed composition and lower-intensity, multifunctional management, while conserved woodland was assigned the highest MSA values. The resulting CRAFTY-GLOBIO classes were: (i) intensive forestry, (ii) extensive forestry, (iii) conservation forest, and (iv) secondary vegetation. Definitions of conservation forest and secondary vegetation followed the respective CRAFTY AFT descriptions.

In CRAFTY-EU, abandoned land represents cells where no agent actively selects a management option. We represented these cells as secondary vegetation in GLOBIO to approximate passive recovery following cessation of active management. This should be interpreted as a simplified recovery pathway, not as an assumption that all abandoned land necessarily develops into closed-canopy forest. Secondary vegetation ultimately transitions to conservation forest through a process of either reforestation (i.e. forest growth on land that was forested prior to land use through natural regeneration) or afforestation (i.e. forest planting on land that was not forested prior for conservation purposes), ultimately assumed to transition to the MSA value 1. In line with assumptions and findings of other studies on biodiversity recovery periods (Akçakaya et al 2018, 2025, Rydgren et al 2020), we assumed that biodiversity takes 100 years to recover in all land use types, regardless of land use history. We implemented the biodiversity recovery in secondary vegetation as follows. For cell i at time t after the transition time t_0 from class a to class b, we compute a lagged $\mathrm{MSA}_{\mathrm{LUT},t}$, using equation (3):

$$\mathrm{MSA}_{\mathrm{LUT},t} = \begin{cases} \min\left(\mathrm{MSA}_a + \dfrac{\mathrm{MSA}_b - \mathrm{MSA}_a}{y_b}\,(t - t_0)\,, \mathrm{MSA}_b\right), & \text{if } \mathrm{MSA}_b > \mathrm{MSA}_a, \\ \mathrm{MSA}_b, & \text{otherwise.} \end{cases} \quad (3)$$

where $\mathrm{MSA}_{\mathrm{LUT},t}$ indicates a recovery time, such that MSA does not increase instantly to MSA_b but takes y_b years. If the new AFT has a lower MSA, such as in the conversion of forest to cropland, we assume MSA reduces to the value of the new AFT in one year.

2.2.2. Other drivers of loss

GLOBIO includes the effects of nitrogen deposition, fragmentation and roads on biodiversity (table 1) by applying pressure-impact relationships to pressure data for the respective driver of loss (Benítez-López et al 2010, Midolo et al 2019, Nunez et al 2019, Schipper et al 2020). We integrated these pressure impact relationships in CRAFTY, using CRAFTY AFT maps as input, as well as additional data needed to predict MSA losses, including annual global mean temperature data from the MPI-ESM1-2-HR climate model (Müller et al 2018), global road inventory project (GRIP) data (Meijer et al 2018), and atmospheric nitrogen deposition from IMAGE (Stehfest et al 2014). This specific climate model was used to be consistent with the inputs of the vegetation model LPJ-GUESS, which provides CRAFTY-EU with potential productivity information. A detailed description of the implementation of each pressure-impact relationship can be found in SI section 3, and SI table A3 provides coefficients of the pressure-impact relationships.

2.3. SSP-RCP scenario building and simulation

We simulated SSP1-RCP2.6 (sustainability-orientated development), SSP3-RCP7.0 (regional rivalry), and SSP5-RCP8.5 (fossil-fuel-driven growth) (Riahi et al 2011, O'Neill et al 2014, Kok et al 2019) from the base year 2020 up to 2100 to assess the effects of alternative socio-economic and climatic futures on forest biodiversity in Europe. SSPs describe contrasting socioeconomic trajectories, including population, economic growth, and land use demand (O'Neill et al 2017), while RCPs represent different levels of radiative forcing resulting from greenhouse gas concentrations (Riahi et al 2011). We focused on these three scenarios as they depict three distinct global futures in terms of climate change mitigation and socio-economic development, shaping future wood consumption and land use demand, thereby influencing forest management for conservation, recreation and wood production (Popp et al 2017, Leclère et al 2020). They span contrasting land-use-climate combinations rather than a single gradient. SSP5-RCP8.5 provides a high-climate/low-land-conversion contrast to SSP3-RCP7.0, helping assess whether biodiversity outcomes are dominated by climate change even where land abandonment and forest expansion occur. Table A5 describes the SSP assumptions of CRAFTY-EU for different socio-economic dimensions.

2.4. Forest biodiversity responses to SSP-RCPs

We quantified biodiversity dynamics in response to changes in forest extent and forest management over time under the different SSP-RCPs. We first assessed SSP-RCP induced transitions in land use (table 2) in relation to other drivers of loss. Thereafter, we assessed the mean and total proportional changes in MSA related to the different drivers of loss, splitting the contribution of land use into the CRAFTY-GLOBIO forest management classes and non-forested land (table 2), using equations (4) and (5):

$$G_{x,r} = \frac{\sum P_{x,g,i}}{A_r} \tag{4}$$

where A_r is the area in km^2 of region r and $G_{x,r}$ is the mean proportional contribution of driver x to MSA loss in MSA*km^{-2} across region r,

$$P_{x,r} = \sum P_{x,g,i} \tag{5}$$

where $P_{x,r}$ is the total proportional contribution of driver x to MSA loss across region r in MSA*km^2. We then assessed the temporal and spatial changes in overall mean and total MSA loss in forests. Finally, we assessed MSA loss over time since the base year in response to secondary vegetation recovery (3) to quantify the magnitude and temporal dynamics of biodiversity gains associated with reforestation (and afforestation).

3. Results

3.1. Forest management and land use dynamics

From 2020 to 2100, the scenarios produced distinct trajectories in land use change and forest management, reflecting differences in their underlying assumptions (table A5) and management pathways (figures 1 and 2). Under SSP1-RCP2.6, total forest area including secondary vegetation increased by 30%, while managed forest area declined by 8%. Extensive forestry decreased by 17%, mainly through conversion to non-forest land and secondary vegetation (figure 1), whereas intensive forestry remained stable, raising its share of managed forest from 49% to 53%. Pastureland and cropland area decreased by 37% and 15%, respectively, by 2100 compared to the 2020 areas (figure A5). Land abandonment affected 707 000 km^2 (15% of land area), 61% originating from non-forest land, with widespread abandonment (figures 1 and A3) driven by projected pasture, fodder and overall food demands in this scenario (figure A2), alongside marginal increases in crop, pasture and timber productivities (figure A1).

Under SSP3-RCP7.0, total forest area declined by 18% due to the expansion of non-forest land, largely at the expense of extensive forestry (–35%) (figure 1). Managed forest area decreased by 20%, while intensive forestry expanded slightly (+1.5%), increasing its share from 49% to 60%. Compared to 2020, cropland area increased by 32% and pastureland area decreased by 9.5% in 2100 (figure A5). Land abandonment was limited (~0.8% of land area) due to the scenario's high land demand and declining socio-economic resources limiting supply of hardwood and softwood (figures A1–A4).

Under SSP5-RCP8.5, total forest area (including secondary vegetation) increased by 7%, while managed forest area declined by 20%, with substantial

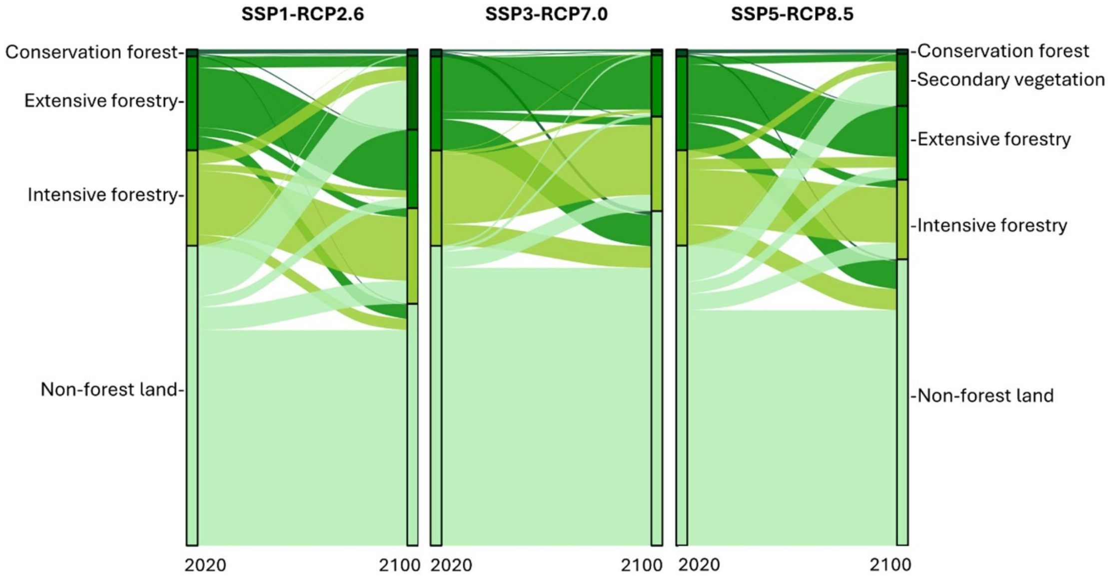


Figure 1. Changes in forest and non-forest land over time. Table 2 provides definitions of the CRAFTY-GLOBIO forest management classes distinguished.

losses in both extensive (−22%) and intensive (−16%) forestry (figure 1). This occurred despite the scenario's increase in timber demands (figure A2) due to increases in potential timber productivity (figure A1). Consequently, the share of managed forest under intensive harvest regimes rose slightly from 49% to 51%. Compared to 2020, pasture area decreased by 43% whilst cropland area increased by 13% and most notably, urban area doubled by 2100 (figure A5). Land abandonment across Europe (figure 2) made 500 000 km^2 (11% of the land area) available for secondary vegetation by 2100, with 72% of abandoned land originating from non-forest areas.

3.2. General biodiversity responses to SSP-RCPs

Figure 3 shows relative changes in mean (MSA·km^{-2}) (equation (4)) and total MSA (MSA km^2) (equation (5)) over time compared to base year 2020 values for the five drivers of biodiversity loss. The figure indicates how driver pressures change over time and how the contribution of a driver to MSA loss increases or decreases relative to other drivers. MSA declined on average in all scenarios, with the smallest average declines under SSP1-RCP2.6 (2.9% by 2100), followed by SSP3-RCP7.0 (5.1%) and SSP5-RCP8.5 (14.0%) (figure 3). Under SSP1-RCP2.6, average and total MSA losses relative to the base year 2020 were driven by nitrogen deposition, fragmentation, climate change, and increasing land use intensity in non-forested areas (figures 3(d)–(i)). Under SSP3-RCP7.0 and SSP5-RCP8.5, average MSA declines were mainly driven by climate change (figures 3(d)–(i)). Under SSP3-RCP7.0, expansion of non-forested land in Western and Central Europe further reduced MSA, although this effect decreased proportionally over time as the impacts of climate change increased.

3.3. Role of forests in biodiversity conservation

Figure 4 shows relative land use and management changes in mean (MSA·km^{-2}) (equation (4)) and total MSA (MSA km^2) (equation (5)) over time compared to the base year 2020. The figure highlights the relative contribution of forested land to reductions and/or increases in MSA loss over time. Total MSA in managed forests increased by 24.7% under SSP1-RCP2.6 and 11.1% under SSP5-RCP8.5 between 2020 and 2100, primarily due to an expansion of secondary vegetation areas and concurrent increases in MSA within these secondary vegetation areas over time (figure 4). In contrast, cumulative MSA in managed forest land declined by 16.7% by 2100 under SSP3-RCP7.0, in response to conversion of forests into non-forested land. The proportional MSA loss caused by forest management decreased as climate change impacts increased under both SSP3-RCP7.0 and SSP5-RCP8.5.

We observed slight average reductions in MSA loss over time compared to 2020 related to forest management in West-Central Europe (e.g. United Kingdom, The Netherlands, Germany, Poland) under both SSP1-RCP2.6 and SSP5-RCP8.5, primarily in response to an increase in secondary vegetation (figure 4). Under SSP3-RCP7.0, MSA loss in response to forest management changes in the Benelux area also declined due to marginal increases in land abandonment (figure 4), but less so than under SSP1-RCP2.6 and SSP5-RCP8.5. MSA losses related to forest management rose on average in Scandinavia and the Mediterranean areas in response to shifts from extensive to intensive forestry classes, particularly in SSP1-RCP2.6 and SSP3-RCP7.0 (figure 4).

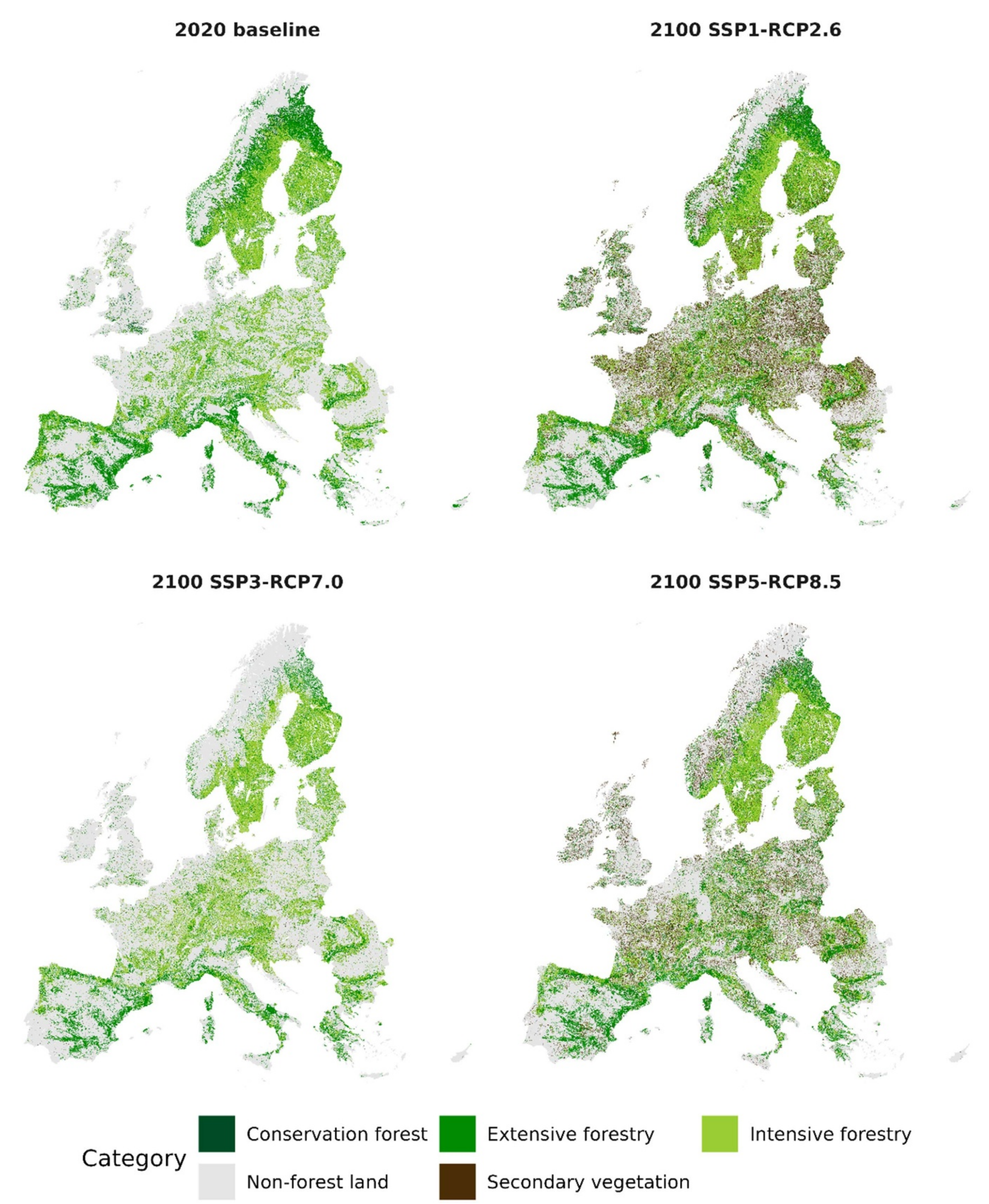


Figure 2. Distribution of forest area and management practices in 2020, and in 2100 for SSP1-RCP2.6, SSP3-RCP7.0 and SSP5-RCP8.5.

4. Discussion

Our results demonstrate that future biodiversity intactness outcomes in European forests are shaped by the combined effects of climate change, forest extent, and forest management, with their relative importance varying across SSP–RCP combinations. We show that, even though land abandonment and forest expansion can partially mitigate against biodiversity losses, biodiversity outcomes are increasingly dominated by climate change under high-emission pathways. Furthermore, our results show that shifts towards widespread intensive forest management can negatively affect biodiversity.

4.1. Climate change mitigation is a prerequisite for biodiversity conservation

Consistent with previous GLOBIO-based assessments, the climate-related component of projected biodiversity loss increases over time under SSP3-RCP7.0 and SSP5-RCP8.5 (Schipper et al 2020), progressively outweighing biodiversity loss in response to land use and management change. This highlights that climate change mitigation, and

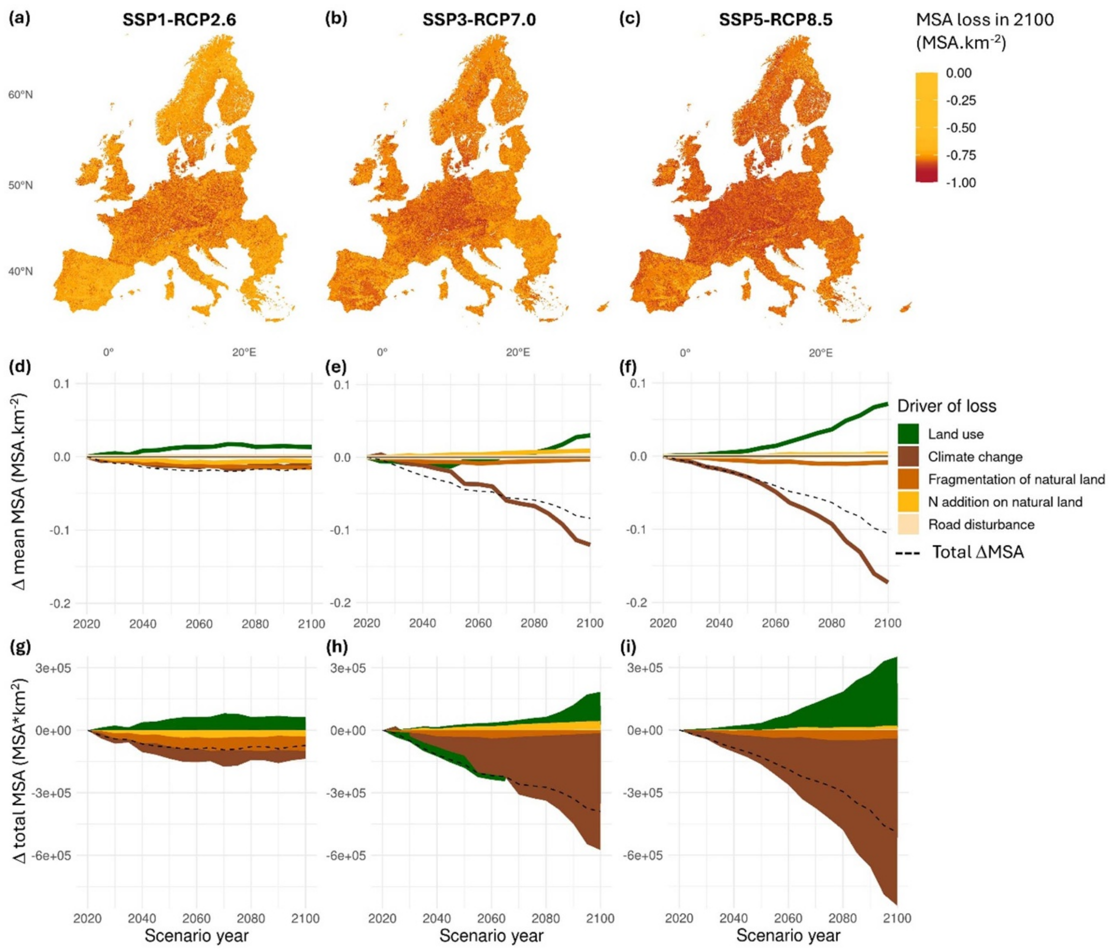
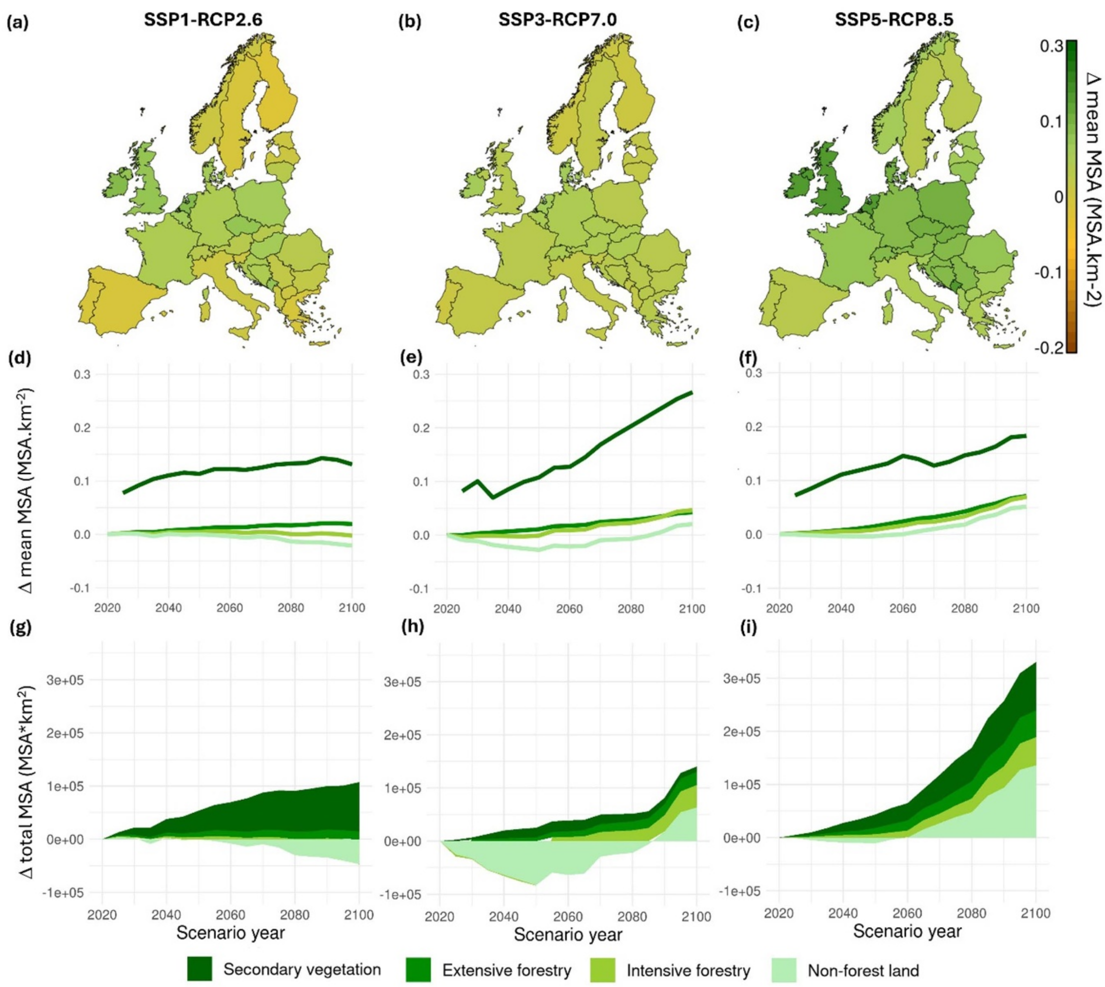


Figure 3. (a)–(c) Total MSA loss in response to all drivers in Europe for three SSP-RCP combinations in the year 2100 compared to the base year 2020 in MSA·km^{-2}. (d)–(f) Proportional changes in mean MSA (average for Europe) in MSA·km^{-2} compared to the base year 2020 for different drivers of biodiversity loss for three SSP-RCP combinations. The lower the mean proportional MSA of a driver, the more this driver on average contributes to MSA loss. Average MSA values are always between 0 and 1, hence, average MSA changes can range between −1 and 1. (g)–(i) Proportional changes in total MSA in MSA*km^2 compared to the base year 2020 for different drivers of biodiversity loss for three SSP-RCP combinations. Negative Δ total MSA values indicate that proportionally more MSA loss occurs as a result of certain drivers of loss (e.g. climate change) than in the base year 2020 across when the MSA values of these drivers are summed across each grid cell. Positive Δ total MSA values indicate that proportionally less MSA loss occurs as a result of certain drivers of loss (e.g. land use) when the MSA values of these drivers are summed across each grid cell. For example, increases in climate change over time in all grid cells causes a total decline in MSA. Altogether, this means that transitions in, for example, land use can positively affect MSA, but this positive effect may be outweighed by the negative effects of climate change, ultimately causing a loss in total MSA over time. Δ Total MSA = the combined change in MSA across all drivers of change.

the associated required behavioural changes, are a prerequisite for long-term biodiversity conservation and calls for stringent climate change mitigation measures (Warren *et al* 2013). Under higher-emission pathways, we also observed that gains in biodiversity associated with forest expansion are modest relative to the magnitude of climate-driven losses. Our results underline the importance of integrating climate change mitigation and biodiversity conservation strategies, where reducing climate change pressures reduces biodiversity loss and shifts in land management have been previously shown to mitigate climate change and its consequent biodiversity losses (IPBES 2019, Hagenström *et al* 2025).

Overall, our finding that the proportional contribution of climate change to biodiversity loss increases over time relative to the contributions associated with land use change and transitions among conservation, extensive, and intensive forest management corroborates findings of prior studies (e.g. Newbold 2018, Pereira *et al* 2024). Previous studies find that, while land use change has historically been the most important driver of biodiversity loss, climate change impacts are expected to overtake the effects of land use change over time (Newbold 2018, Pereira *et al* 2024). These findings contrast, however, with studies that find a similar impact of land use change and climate change on biodiversity over time under varying SSP-RCP combinations (Vermaat *et al* 2017, Mouratidis *et al* 2025). These differences partly reflect the distinct ecological processes and biodiversity dimensions represented by alternative modelling approaches.



Figures 4. (a)–(c) Proportional changes in MSA in MSA·km^{-2} in managed forest land (see table 1) in 2100 compared to the base year 2020 averaged per European country simulated in CRAFTY for three SSP-RCP combinations. (d)–(f) Changes in average proportional MSA in MSA·km^{-2} for four different types of managed forests and non-forested land from the base year 2020–2100 for three SSP-RCP combinations. The lower the mean proportional MSA loss of a class of land use/management, the more this driver on average contributes to MSA loss. Average MSA values are always between 0 and 1, hence, average MSA changes can range between −1 and 1. (g)–(i) Changes in total proportional MSA in MSA*km^2 over time in four different types of managed forests and non-forested land between 2020 and 2100 for three SSP-RCP combinations. The area size of a land use/management in the graphs indicates the MSA values times the total area extent the land use/management covers. Changes in total proportional MSA could, therefore, result from both changes in the area (e.g. a conversion from non-forested land to secondary vegetation) and changes in the types of non-forested land (e.g. transitions from pasture to cropland). Transitions from other land uses/management to secondary vegetation positively affect MSA, yet it remains lower than primary vegetation for the assumed period of 100 years. As a result, secondary vegetation contributes to the net loss in MSA relative to the primary state of ecosystems that would have occurred without human influence.

GLOBIO is well suited to the present analysis because it provides a spatially consistent, pressure-based framework that integrates land use change, forest-management intensity, fragmentation, infrastructure, and climate-related pressures across large regions and multiple scenarios. This enables direct comparison of the relative contributions of these pressures to changes in biodiversity intactness and provides a close conceptual link to the land use and management variables represented in our scenario framework.

Species distribution models (SDMs), in contrast, provide greater detail on species-specific climatic and habitat suitability, range shifts, and compositional responses (e.g. Baisero et al 2020, Jung 2023). However, SDMs do not inform on intactness, as they are based on recent observations. GLOBIO therefore offers breadth and cross-pressure comparability, while species distribution models offer greater taxonomic detail. Their projections should be regarded as complementary, with differences arising from the biodiversity components and ecological conditions that each approach emphasises or omits. The contrasting results between indicators emphasises the importance of comparing biodiversity indicators and models to derive the full picture of possible biodiversity responses to alternative socio-economic and climatic futures (Zurell et al 2026).

4.2. Forest extent and management jointly shape biodiversity intactness outcomes

Within the broader climate constraint, changes in forest extent and forest management conditionally shape biodiversity outcomes. Land abandonment under SSP1-RCP2.6 and SSP5-RCP8.5 enabled emergence and then recovery of secondary vegetation, increasing forest-related biodiversity. We interpret this as an emergent land-sparing mechanism (Kok *et al* 2018): reduced land demand, largely driven by productivity gains (SSP5) or dietary shifts toward plant-based consumption (SSP1), allowed natural regeneration and afforestation without explicit conservation policies (Brown *et al* 2018). These findings align with prior studies showing that abandoned land can contribute to policy goals (Ustaoglu *et al* 2018, Fayet *et al* 2022). Under SSP3-RCP7.0, our results highlight the potential to mitigate biodiversity loss by reducing land requirements through policies such as dietary shifts (Alexander *et al* 2015) or agricultural intensification to spare land for conservation forests (Kok *et al* 2018).

We found that biodiversity gains from land abandonment are partially moderated by increases in forest management intensity across all SSP-RCP combinations. Shifts from extensive or conservation-oriented forestry to intensive practices reduced biodiversity, in some cases counteracting the benefits of forest expansion. This underscores that the biodiversity value of land abandonment depends critically on how regenerating forests are managed. Because the SSP-RCP combinations represent exploratory rather than predictive futures, these results should be interpreted as identifying conditions under which forestry intensification could erode biodiversity gains, rather than as outcomes that are expected to occur inevitably. Policies that discourage such intensification could therefore help steer future land use trajectories away from these outcomes.

Such policies align with the EU Forest Strategy for 2030, which promotes 'closer-to-nature' practices such as selective cutting and extended rotations (European Commission 2021) and would facilitate a land-sharing approach, where wood production occurs over larger areas with lower local biodiversity impacts (Kok *et al* 2018, 2023). However, this may require more land to meet timber demand, potentially displacing biodiversity pressures to regions outside Europe (Balmford *et al* 2025, Di Fulvio *et al* 2025). Since our analysis does not account for displacement effects of the international trade of timber outside Europe, apparent biodiversity gains in Europe should be interpreted cautiously. Future work with scenario modelling frameworks, such as those used here, could produce more detailed insights in the trade-offs between land sparing and sharing policies.

4.3. Limitations

Several limitations should be acknowledged. First, the differences between CRAFTY AFTs and GLOBIO land use/management classes means biodiversity intactness outcomes are approximate, based on assumed impacts of each CRAFTY AFT relative to GLOBIO classes. Future work could improve this by aligning biodiversity estimates with CRAFTY AFTs.

Second, our treatment of abandoned land as secondary vegetation is a simplification and our assumption that biodiversity intactness increases over 100 years following reforestation/afforestation depends on natural regeneration or active forest establishment and management. Recovery rates vary with prior land use (e.g. croplands or mines) (Meli *et al* 2017, Ekka *et al* 2023) and regeneration methods (e.g. planting vs natural regeneration) (Crouzeilles *et al* 2016, 2017). In addition, the model applies biodiversity losses immediately following conversion to a more intensive land use, while gains occur gradually. This asymmetric treatment may overestimate the short-term rate of biodiversity decline following land use change; however, it errs on the side of caution by avoiding an overly optimistic representation of biodiversity impacts.

Abandoned land may follow multiple trajectories, including forest regeneration, shrub encroachment, wetland development or persistence of open semi-natural grasslands. Our results should therefore be interpreted as outcomes under an assumed secondary-vegetation recovery pathway. Additionally, while we assume afforestation benefits biodiversity, afforestation can also have negative effects, such as trade-offs with species-rich grasslands (Ustaoglu and Collier 2018, Prangel *et al* 2023). Advances in modelling prior land use impacts and regeneration approaches are needed, requiring improved parameterisation in GLOBIO (van't Veen *et al* 2025).

Third, our results are also shaped by the biodiversity metric used. GLOBIO's MSA indicator measures biodiversity intactness relative to a historical reference state, meaning that climate-driven changes in species composition are represented as reductions in intactness. Alternative biodiversity metrics, including those emphasising species richness or compositional change differently, could yield different outcomes, as these not only account for losses of reference species but also for gains in species that migrate in response to climate change (Bellard *et al* 2012).

Fourth, our results are dependent on the SSP-RCP scenario framework, which has been shown to have largely negative outcomes for nature (e.g. Alexander *et al* 2023). Frameworks that more explicitly embed biodiversity objectives, such as the Nature Futures Framework (Pereira *et al* 2020), may yield different, potentially more positive, outcomes (Alexander *et al* 2023).

Inherent in the scenario approach are limits on the quantifiable uncertainty in results. The simulations conducted here were designed to compare forest management under consistent assumptions, across scenarios. They do not constitute a probabilistic sample from the complete space of plausible model structures and parameter values. Consequently, the plotted trajectories should be interpreted in terms of their comparative patterns, directions of change and relative differences rather than as precise predictions with an associated statistical confidence level. A comprehensive uncertainty analysis would require probability distributions to be defined for uncertain inputs and parameters, followed by a substantially larger simulation ensemble and formal variance-based sensitivity analysis. Such an analysis is identified as a valuable topic for future research.

5. Conclusion

Our results demonstrate that forest management critically shapes biodiversity outcomes under future socio-economic and climate scenarios. While land abandonment and conservation-oriented practices can partially offset biodiversity loss, their effectiveness is limited under high-emission pathways without robust climate mitigation. Explicitly accounting for forest management in spatial biodiversity assessments is therefore essential for evaluating land use and climate policies. Future work could leverage coupled land use and biodiversity models to assess the impact of targeted policy interventions on reducing biodiversity loss across space and time.

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Data availability statement

The data that support the findings of this study is openly available at the following DOI: https://doi.org/10.5281/zenodo.22048932 (Raymond *et al* 2026).

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Conflict of interest

The authors have no conflicts of interest to declare.

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