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Trade-Offs for Climate-Smart Forestry in Europe Under Uncertain Future Climate

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Key Points:

- Strategies for climate-smart forestry under a range of climate scenarios always lead to trade-offs between different ecosystem services (ESs)
- Higher shares of unmanaged and broad-leaved forests are beneficial for numerous ESs, but lead to decreased timber provision
- The mitigation potential of forests strongly relies on substitution effects which depend on the carbon-intensity of the alternative products

Supporting Information:

Supporting Information may be found in the online version of this article.

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Abstract

Forests mitigate climate change by storing carbon and reducing emissions via substitution effects of wood products. Additionally, they provide many other important ecosystem services (ESs), but are vulnerable to climate change; therefore, adaptation is necessary. Climate-smart forestry combines mitigation with adaptation, whilst facilitating the provision of many ESs. This is particularly challenging due to large uncertainties about future climate. Here, we combined ecosystem modeling with robust multi-criteria optimization to assess how the provision of various ESs (climate change mitigation, timber provision, local cooling, water availability, and biodiversity habitat) can be guaranteed under a broad range of climate futures across Europe. Our optimized portfolios contain 29% unmanaged forests, and implicate a successive conversion of 34% of coniferous to broad-leaved forests (11% vice versa). Coppices practically vanish from Southern Europe, mainly due to their high water requirement. We find the high shares of unmanaged forests necessary to keep European forests a carbon sink while broad-leaved and unmanaged forests contribute to local cooling through biogeophysical effects. Unmanaged forests also pose the largest benefit for biodiversity habitat. However, the increased shares of unmanaged and broad-leaved forests lead to reductions in harvests. This raises the question of how to meet increasing wood demands without transferring ecological impacts elsewhere or enhancing the dependence on more carbon-intensive industries. Furthermore, the mitigation potential of forests depends on assumptions about the decarbonization of other industries and is consequently crucially dependent on the emission scenario. Our findings highlight that trade-offs must be assessed when developing concrete strategies for climate-smart forestry.

Plain Language Summary

Forests help mitigate climate change by storing carbon and via avoided emissions when wood products replace more carbon-intensive materials. At the same time, forests provide many other "ecosystem services (ESs)" to society. For example, they provide timber, habitat for various species, and they cool their surrounding regions. They are, however, also vulnerable to ongoing climate change. Forest management must consider all these aspects, which is particularly challenging considering the uncertainty about future climate. Here, we propose how this may be tackled by computing optimized forest management portfolios for Europe for a broad range of future climate pathways. Our results show that changes to forest composition are necessary. In particular, increased shares of unmanaged and broad-leaved forests are beneficial for numerous ESs. However, these increased shares also lead to decreases in harvest rates, posing a conflict between wood supply and demand. We further show that the mitigation potential of forests strongly depends on how carbon-intensive the replaced materials are. Consequently, should these materials become "greener" due to new technologies, the importance of wood products in terms of climate change mitigation decreases. Our study highlights that we cannot optimize every aspect, but that trade-offs between ESs need to be made.

1. Introduction

Forests contribute to climate change mitigation (Canadell & Raupach, 2008; Pan et al., 2011; Ramstein et al., 2019) through their significant role in the global carbon cycle. Annually, over 25% of global anthropogenic carbon emissions are absorbed by the terrestrial biosphere, primarily by forests (Friedlingstein et al., 2020; Harris et al., 2021; Pan et al., 2011). Consequently, Article 5 of the Paris Agreement states specifically that the world's existing forest carbon sinks should be conserved or enhanced (United Nations, 2015a). So far, this remains a challenging task since forest carbon sinks are vulnerable to climate change, mainly due to increased frequency and severity of disturbances (Dai, 2013; IPCC, 2014; Seidl et al., 2014; Spinoni et al., 2018).

European forests are estimated to take up 9% of Europe's emissions (Grassi et al., 2019). Since 90% of these forests are managed (IPCC, 2014), they additionally provide considerable amounts of wood products that store carbon and substitute carbon-intensive materials and fuels (Grassi et al., 2021; Harmon, 2019; Howard et al., 2021; van Kooten & Johnston, 2016). Furthermore, apart from climate change mitigation, forests provide numerous other ecosystem services (ESs) such as, for example, timber production, water regulation, local climate regulation, and recreation (Binder et al., 2017; Cordonnier et al., 2014; Mori et al., 2017). Therefore, future management strategies must also consider the continued provision of these manifold ESs (Brockerhoff et al., 2017; Díaz et al., 2006; Hua et al., 2022; Millenium Ecosystem Assessment, 2005), which is also reflected in the UN's sustainable development goals (United Nations, 2015b) and the declaration of sustainable forest management in Europe (Forest Europe, 2015a). This holistic approach of jointly considering adaptation, mitigation, and ESs in forest management has been coined "climate-smart forestry" (Kauppi et al., 2018; Nabuurs et al., 2017).

One approach to tackle the issue of assessing multiple goals in forest planning is using multi-criteria decision making (MCDM, Ishizaka and Nemery [2013], see also Uhde et al. [2015]). MCDM has been applied to find the best possible forest management regarding multiple ESs under different climate scenarios (e.g., in Spain, Diaz-Balteiro et al., 2017). Using models to develop and assess forest management portfolios—defined as the relative proportions of species and forest management alternatives—allows for a more nuanced assessment since these models integrate several ESs. In addition, portfolios can combine the benefits of multiple management options. This was recently demonstrated by Luyssaert et al. (2018) who created forest management portfolios from six simplified management options (e.g., species changes, conversion to coppice, or refraining from management) and applied these across Europe using the ORCHIDEE-CAN model. However, their optimization of three single-criterion objectives (maximize albedo, maximize carbon sink, minimize surface temperature) entailed trade-offs that must be made between the optimized objective and other ESs.

Management decisions in forestry are particularly complicated: The long life-spans of trees and changing societal demands require assumptions about the future resulting in large uncertainties. These uncertainties may be addressed with the concept of robust optimization (Ben-Tal & Nemirovski, 2002; Gorissen et al., 2015). One of the first studies that applied robust optimization to land-use portfolios was done by Knoke et al. (2015). Furthermore, several recent studies have shown how robust optimization and MCDM can be combined to optimize forest management portfolios for multiple ESs considering uncertainty (Knoke et al., 2016, 2020; Uhde et al., 2017).

Currently, one of the most uncertain aspects of forest planning is climate change and its local impacts (Lindner et al., 2014). For instance, it is challenging to find tree species that are well-suited for present and future conditions (Hickler et al., 2012). Such future uncertainty can be explored in modeling studies where various species can be assessed under different climate scenarios based on Representative Concentration Pathways (RCPs) covering several plausible trajectories of anthropogenic emissions and climates (Dufresne et al., 2013; IPCC, 2018; Meinshausen et al., 2011). However, recent studies on future forest management strategies do not embrace the full range of uncertainty; therefore ignoring the issue that a proposed forest management option deemed practical for one RCP might not be useful if another RCP materializes. Alternatively, it is crucial to broadly evaluate the full range of possible climates (Pedersen et al., 2020) and provide solutions that will be practical across all RCPs. This prevents costly adaptation if the world develops in unforeseen ways (Lawrence et al., 2020) and is also promoted by the IPCC which states that also low-likelihood outcomes need to be included in risk assessments (IPCC, 2021).

To address this, we considered RCP2.6, RCP4.5, RCP6.0, and RCP8.5, and determined how forest management portfolios may be constructed to provide all ESs across these RCPs. We combined the well-established dynamic vegetation model LPJ-GUESS (B. Smith et al., 2001; B. Smith et al., 2014; Lindeskog et al., 2021) with MCDM and robust optimization to consider the ESs climate change mitigation, timber provision, water availability, local climate regulation, and biodiversity habitat. Additionally, we covered climate change adaptation by evaluating which forest types can survive the different climate scenarios. Specifically, we focused on European forests, using six simplified forest management options similar to Luyssaert et al. (2018). Our approach allowed us to assess several ESs simultaneously under uncertain future climate development across Europe. We present a set of management portfolios for different European forest types that are viable across many future climate scenarios. We further investigated the potential trade-offs arising when implementing such portfolios, mainly by considering harvest rates and mitigation impacts. Lastly, we evaluated how different preferences and assumptions affected the portfolios.

2. Methods

We applied LPJ-GUESS for the European domain with a 0.5° × 0.5° spatial resolution and combined it with an optimization framework (Figure 1). We performed 24 simulations for $m = 6$ management options and $n = 4$ RCPs (RCP2.6, RCP4.5, RCP6.0, RCP8.5). We computed seven ES indicators (ESIs—representing the ESs climate change mitigation, local climate regulation, biodiversity habitat, timber provision, and water availability) to evaluate the different management options (Table 2). Finally, our optimization algorithm computed one management portfolio per grid cell which allowed an optimally balanced provision of all considered ESIs across all RCPs.

2.1. Dynamic Vegetation Model and Simulation Protocol

LPJ-GUESS (B. Smith et al., 2001; B. Smith et al., 2014) is a dynamic vegetation model that simulates establishment, growth, competition, management, and mortality of plant functional types (PFTs). Each PFT is represented by different parameters governing, for example, life-history strategy, phenology, growth form, drought tolerance, bioclimatic limits, and others (B. Smith et al., 2014). The modeled processes include photosynthesis and stomatal conductance based on BIOME3 (Haxeltine & Prentice, 1996), stochastic implementations of population dynamics (B. Smith et al., 2001), allocation of carbon to different compartments and allometric relationships for plant growth (Sitch et al., 2003). They also include the nitrogen cycle, soil, and litter processes (B. Smith et al., 2014). LPJ-GUESS has been designed to simulate the vegetation response under future climate change, including responses to climate extremes. As such it has been benchmarked to a wide variety of independent data-sets (e.g., Chang et al., 2017; Haverd et al., 2020; Ito et al., 2017; Lindeskog et al., 2021; B. Smith et al., 2014).

Mortality is implemented as a stochastic process considering a tree's growth efficiency and age. The simulations in this study were carried out in "cohort-mode," where one average individual represents a cohort of the same age. Each stand was modeled via 25 replicate patches representing random samples of the same stand.

Disturbances in LPJ-GUESS are modeled stochastically as patch-destroying disturbances. Our study followed Pugh et al. (2019) and used different average return times for different stand types. We used their global median values of 1000 and 500 years for broad-leaved deciduous and evergreen stands, respectively, and 300 years for needle-leaved and mixed stands. Note that we used these values even though they contain management disturbances, since return times in Europe are partly much lower than these global median values (Pugh et al., 2019). Additionally, we simulated fire using the GlobFIRM fire model (Thonicke et al., 2001) in the historical and spin-up phase to retrieve a reasonable representation of present-day European forests (see Section 3.1.1); however,

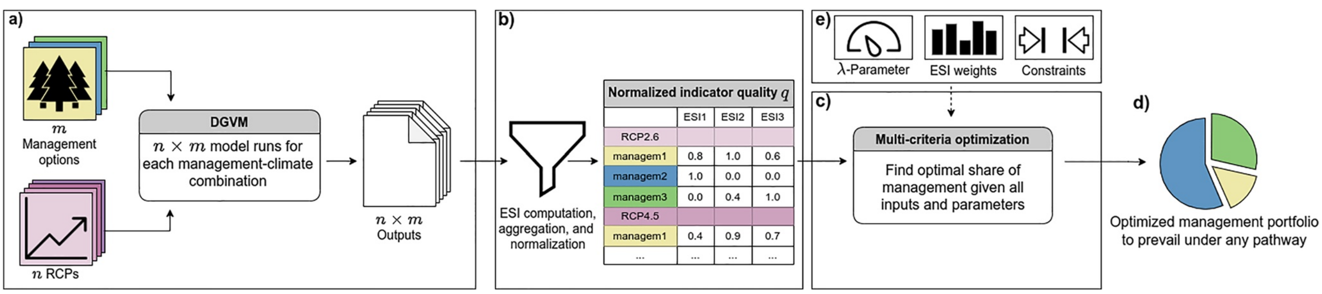

Figure 1. Visualization of the methodology, leading to one single portfolio per grid cell. Panel (a) $n \times m$ independent model simulations are run for the management options and Representative Concentration Pathways (RCPs). (b) Ecosystem service indicators (ESIs)—if not computed by LPJ-GUESS directly—are computed from the model outputs, then aggregated (e.g., 2100–2130 mean), and normalized to [0; 1], resulting in one table per grid cell containing the normalized values for all management types and RCPs. (c) For each grid cell, one optimization is run resulting in (d) one optimized portfolio per grid cell which for any RCP ensures a balanced provision of all ESIs. (e) Additional parameters may be passed into the process: A parameter $\lambda \in [0, 1]$ governs the focus on the balanced provision of the ESI. A low λ results in a more balanced provision of all ESIs whereas a high λ lays more focus on the average ESI values (and less balance) instead, see also Section 2.3. Weights can be added to put more focus on one or more ESI. Additional constraints can put bounds on the variables, for example, harvest needs to be at least at 2010 levels.

we excluded fire for the future period as it was included in the disturbance intervals. Furthermore, starting in 2010, we increased the disturbance rates by 1% each year to account for expected increasing disturbance frequencies in Europe (Senf & Seidl, 2021a).

We used specific parametrizations for 18 European tree species and four more generic PFTs for three shrubs and grass (Table S1 in Supporting Information S1). We used the parameters of LPJ-GUESS v4.0 (based on the values proposed by Hickler et al. [2012]) with adaptations as specified in Lindeskog et al. (2021). However, we removed the maximum temperature of the coldest month bioclimatic limit for establishment ($tcmax_{est}$) of certain species (as in Hickler et al. [2012]) after observing that otherwise modeled present-day occurrences were too far off observations for these species (Mauri et al., 2017).

2.1.1. Representation of Forest Management in LPJ-GUESS

Several forest management options are implemented in LPJ-GUESS (Lindeskog et al., 2021). Here we applied the thinning scheme following the self-thinning rule of Reineke (1933), which was implemented similarly as in Bellassen et al. (2010) as described in Lindeskog et al. (2021). Apart from thinning, forests were clear-cut when a stand fell below a corresponding target density predefined for broad-leaved summer green and needle-leaved forests (Bellassen et al., 2010; Lindeskog et al., 2021). We defined an additional target density for broad-leaved evergreen forests (75 ind/ha) to yield comparable rotation times between the forest types to facilitate comparison of harvests between forests in the limited time period of interest (2100–2130). In achieving the correct age distribution by 2010, clear-cuts were omitted between stand establishment and 2010 (see Section 2.1.3). For stands that attained the target density before 2010, clear-cuts were spread out through 2010–2020 to avoid a significant anomaly in 2010 (see Figure S20 in Supporting Information S1). During harvesting, 65% of the total carbon of the sapwood and heartwood was considered stem material of which 90% ("harvest efficiency") were taken out. Harvests were distributed to different product pools as described in Section 2.1.4. Small fractions of leaves and branches were assumed to be taken out and burned, and the rest was left to decay on-site together with the stem residues and coarse roots (see Lindeskog et al., 2021). We also implemented salvage logging, described in detail in Section S1.1 of Supporting Information S1.

2.1.2. Coppice Management Implementation in LPJ-GUESS

Coppice is a forest type that historically was very important in Europe (Albert & Ammer, 2012; Evans, 1984), still accounting for large forest shares in Southern European countries (Maganotti et al., 2018). Therefore, we implemented a simple form of coppice management into LPJ-GUESS similar to ORCHIDEE_CAN_r3069 (Luyssaert et al., 2018). If a broad-leaved tree is cut down, its coarse roots (assumed to be 22% of the total carbon mass) stay alive, and new shoots emerge from the stump (assumption: 10 per stump for all species). Choosing the initial number of shoots after a coppice event showed no significant difference after a few years as the mortality routines of LPJ-GUESS consequently resulted in strong self-thinning in cases with a higher number of shoots (see Figure S4 in Supporting Information S1). Besides this expected self-thinning (Johansson, 2008; Leonardsson & Götmark, 2015; Rydberg, 2000; Verlinden et al., 2015) we also applied anthropogenic thinning in coppices (Nicolescu et al., 2017). The shoots were harvested individually once they reached a diameter of 20 cm, our coppice forests were thus representative of traditional coppice forests, not short-rotation coppice plantations.

2.1.3. Simulation Protocol for LPJ-GUESS Simulations

For our simulations we followed the setup of Lindeskog et al. (2021) by using monthly temperature, radiation, and precipitation data (including the number of wet days) from CMIP5 (Taylor et al., 2012) simulations of the general circulation model (GCM) IPSL-CM5A-MR (Dufresne et al., 2013). We selected this GCM to compare our results with Luyssaert et al. (2018).

The climate data was interpolated bi-linearly from $2.5° \times 1.25°$ to $0.5° \times 0.5°$ spatial resolution and bias-corrected against 1961–1990 observations from CRU-NCEP (see Ahlström et al., 2012). For each climatic variable, this data covered a transient "historical" (1850–2005) and a future period (2006–2100) per RCP.

After a 1200-year spin-up period to bring soil carbon pools close to equilibrium, the simulations were run from 1800 until 2130. Before 1850, the 1850–1879 climate was detrended and recycled. It was similarly performed with each scenario's 2071–2100 data for the time after 2100. Our optimization considered the period 2100–2130. By then, most simulated forests had undergone a full rotation (Figure S22 in Supporting Information S1).

CO_2 concentrations of the RCPs were taken from Meinshausen et al. (2011), and a constant pre-industrial (year 1765) atmospheric CO_2 concentration of 278 ppmv was applied for the spin-up. Decadal values for Nitrogen deposition from 1850 to 2100 were used, depending on the grid cell and RCP (Lamarque et al., 2011).

To commence projections from a realistic representation of forests in Europe today, forest stands established during spin-up were successively clear-cut and replaced by stands matching the species distribution map from Brus et al. (2012). The timing of these clear-cuts was predefined to reproduce the observed 2010 age distribution from the global forest age data set (GFADv1.0, Poulter et al., 2018). This is why we turned off disturbances and clear-cuts in such stands between their establishment and 2010.

Old-growth forests (i.e., older than 140 years in 2010 according to GFAD) were not subject to any management before and after 2010. We kept the forest area constant at 2010 levels for future projections and focused on "forests remaining forests." Large-scale deforestation is unlikely due to the Paris Agreement (United Nations, 2015a) and national laws. Forest area increase however has recently been observed (Forest Europe, 2020). This trend might continue if land abandonment outweighs effects of increasing demand for agricultural products (Luyssaert et al., 2018; Perpiña Castillo et al., 2021; P. Smith et al., 2010). This competition is not part of our study.

We performed one simulation until 2010 for all 3124 grid cells in Europe to evaluate the present-day model outputs against observations and other models (Section 3.1.1). To create forest management portfolios, we implemented six simplified forest management options (Table 1) based on Luyssaert et al. (2018). We performed one future projection simulation (2010–2130) for each RCP and management option. For example, in the toBd ("to broad-leaved deciduous") simulation, all stands were converted to a broad-leaved deciduous forest after clear-cut. When changing a non-broad-leaved stand to coppice, the stand was initially converted to a broad-leaved deciduous forest and then managed as coppice.

We sampled grid cells every 2° latitude and longitude for future runs to reduce the computational load resulting in simulations for 193 grid cells across Europe. In the optimization, we combined the management options to create a portfolio for each grid cell since the stands within a grid cell were independent from one another. We excluded cells from the portfolio analysis where the average harvests between 1990 and 2010 were below 0.1 $gC/m^2/yr$ or less than 25% of the forest area was converted by 2100. The reason for this is that with such a low conversion, the forests looked very similar at the end of the simulation period and the difference in ESIs were only marginal, not allowing for a sensible comparison of management options. This resulted in a final set of 181 grid cells.

2.1.4. Contributions of Forests to Climate Change Mitigation

Apart from ecosystem carbon storage, the effect of forests in mitigating climate change depends on the life cycle of wood products, attributed to their ability to substitute carbon-intensive materials and fuels (Grassi et al., 2021; Harmon, 2019; Howard et al., 2021; van Kooten & Johnston, 2016). To measure this, we followed Krause et al. (2020) by using two wood product pools for medium and long-lived products, respectively, with corresponding decay rates described by two Gamma-functions (Klein et al., 2013, Figure S24 in Supporting Information S1).

Table 1
The Six Simplified Management Options

Name	Explanation
base	After clearcut, plant the same species as before, same management as before (thinning and clearcut)
toBd	After clearcut, convert to broad-leaved deciduous forest by planting the most common broad-leaved deciduous species in the stand. Continue with thinning and clearcut
toBe	Like toBd, but planting *Q. ilex*, the LPJ-GUESS broad-leaved evergreen species
toCoppice	Broad-leaved forests are clearcut and left to regrow from the stumps, needle-leaved forests are clearcut and replaced with broad-leaved species. The forest is managed as coppice from then on
toNe	Like toBd, but planting the most common needle-leaved evergreen species in the stand
unmanaged	Refrain from the clearcut and also refrain from any thinnings after this point in time, leaving the forests completely untouched

Note. A management decision was taken for each stand after 2010 as soon as it reached maturity (i.e., a target density). The conversion was implemented by planting the most common species of each forest type for that grid cell.

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These functions account for the fact that the entire amount of wood entering a product pool will typically remain there for some time before it starts to be returned to the atmosphere.

We based product flows on Klein et al. (2013) and Eurostat (2021a), where for conifers (non-conifers), 23% (2.5%) of stem mass entered the long-lived pool, and 9.4% (11.9%) entered the medium pool. Meanwhile, the rest (plus some harvest residues) was converted to fuel wood or short-lived products and transferred to the atmosphere within one year (Figure S19 in Supporting Information S1). For coppice, we assumed that 2% entered the medium product pool. The rest was turned over immediately, because most wood from coppice forests is used for energy, heating, and paper, and only very small portions for, for example, furniture (Maganotti et al., 2018). For the medium and long-lived pools the median residence times were 18 and 93 years, respectively, that is, 50% of products remained in that pool after this time (Figure S24 in Supporting Information S1).

To assess the emission reductions, we used displacement factors (defined as avoided emissions in relation to the mass of carbon in the wood product, Arehart et al., 2021) for fuels and materials, respectively. We used 0.67 tC/tC as the displacement factor for fuels (Knauf et al., 2015). It is based on a simple assessment of the possible energy provision of wood compared to light oil (Rüter, 2011). It is comparable to those of studies of other developed nations (Myllyviita et al., 2021) and well in the range of "less than 0.5 up to about 1.0" as suggested by Sathre and O'Connor (2010). The displacement factor for material substitution is based on an analysis of 16 key products and their alternatives, making up 90% of the wood usage spectrum (Knauf et al., 2015). Its value of 1.5 tC/tC was originally estimated for Germany and is lower than the commonly used value of 2.1 (Sathre & O'Connor, 2010) which is a mean value over studies considering different applications and countries across the world. It does not contain end-of-life assessment. We used this displacement factor for 77% of wood products and assumed energy recovery at the end of their lifetime. In Europe, 23% of all waste ends up in landfills (Eurostat, 2021b). Consequently, for the remaining 23% of products we used a lower displacement factor of 1.1 tC/tC to also account for emissions from landfills (Sathre & O'Connor, 2010).

Wood fuels emit more CO_2 per unit of energy than fossil fuels, and it requires a long time until a forest has regrown and absorbed the initially emitted carbon ("carbon debt," Cherubini et al., 2011). Consequently, wood-based fuels are not carbon-neutral, at least on short and medium time-scales (Booth et al., 2020; Cherubini et al., 2011; Holtsmark, 2012; Leturcq, 2020). It is crucial to take this into account, which is done here by considering the total carbon effect including stocks in forests and products, similarly as performed by Knauf et al. (2016). This approach quantifies the actual emissions from burning the wood.

The actual number of avoided emissions heavily depends on the energy mix at a given time or the type of materials that are replaced and is expected to decrease in the future (Harmon, 2019). We accounted for this by gradually discounting the displacement factors based on the RCPs, similar to Brunet-Navarro et al. (2021, Figure S23 in Supporting Information S1). Additionally, we evaluated our approach without discounting (see Section 4.3).

Finally, it is crucial to take into account additionality. Any measurement of mitigation from forests needs to be compared to a reasonable baseline, because only an additional measure would contribute to climate change mitigation. In our case this is done implicitly by including a baseline simulation ("base"). Any change in forestry can thus be compared to this baseline.

2.2. Ecosystem Services and Their Indicators

Our study investigated the potential changes in seven ESIs (Table 2 and Section S1.2 in Supporting Information S1). As indicators for the provision of timber we used both the simulated total harvest values ($t\,ha^{-1}\,yr^{-1}$ dry mass) and harvests used only for long-lived wood products (HLP).

We used one combined indicator for a forest's (global) climate change mitigation potential by combining the total simulated carbon pool (vegetation + soil + litter + products) with cumulative avoided emissions through material and energy substitution. For local climate regulation we included evapotranspiration (ET) and surface roughness. Since the *local* effects of albedo have previously been found to be small compared to ET and surface roughness (Winckler et al., 2019), we included albedo only in an additional experiment. Although we acknowledge that biogeophysical variables also possess non-local and global effects (Pongratz et al., 2010; Winckler et al., 2017), these might be small compared to the biogeochemical effects (Pongratz et al., 2010). Therefore, we solely evaluated increases in these variables as an indication for local cooling, omitting possible feedback on global

Table 2
The Ecosystem Service Indicators Used in the Main Part of This Study

Variable Name	Ecosystem service indicator (ESI)	Explanation
Harvests	Total harvests	Total wood provision (including products like firewood, pulp etc.)
HLP	Harvests for long-lived wood products	Timber provision for long-lived products (e.g. construction)
Mitigation	Carbon storage plus material and energy substitution effects	Total carbon in vegetation, soil, litter, and products, plus avoided emissions from substitution with wood products
z_0	Surface roughness	Indicator for atmospheric conductance, influencing heat fluxes. Higher roughness results in higher fluxes
ET	Total evapotranspiration	Indicator for latent heat fluxes. More ET means more local cooling
Ψ_{soil}	Soil water potential	Yearly minimum of monthly values, indicator of water availability and drought stress
Bio	Combined indicator (RCP-normalized mean) of amount of coarse woody debris, Shannon entropy of 5 cm diameter-at-breast-height (DBH) classes and number of trees with DBH >50 cm	Coarse woody debris, large trees, and an abundance of various tree sizes provide high numbers of habitats and resources (Cordonnier et al., 2014)

Note. Albedo was included in an additional experiment. See Section S1.2 in Supporting Information S1 for technical details.

climate. The yearly minimum of monthly soil water potential Ψ_{soil} was used as an indicator for water availability (Rajasekaran et al., 2018), where a decrease in Ψ_{soil} indicates less water available for plants. We implemented a combined indicator for biodiversity based on tree sizes and dead wood, adapted from Cordonnier et al. (2014). Due to the model setup (i.e., planting the most common species of the grid cell), it was not sensible to include an indicator for the diversity of tree species. Finally, to assess the changes in ESI performances, we compared their values at the beginning of the next century (mean of 2100–2130) to present-day values (mean of 2000–2010).

2.3. Robust Multi-Criteria Optimization

To compute portfolios that provide various ESs of different units and magnitudes, we used MCDM (e.g., Ishizaka & Nemery, 2013; Marler & Arora, 2004). To account for uncertainty in the inputs stemming from the RCPs, we combined this with robust optimization (Ben-Tal & Nemirovski, 2002). The solution of such an optimization guarantees feasibility for all realizations of the uncertainty set and can be written as a maxi-min linear program (Gorissen et al., 2015). To enable comparison, the computed ESIs from the simulations were first normalized to [0, 1]. This was done for each ESI and RCP separately, since ESI values of different RCPs do not belong to the same environmental conditions and we argue that they cannot be compared to an ESI from an entirely different climatic future. Our normalization thus yields a better interpretability of the worst case results: When the worst case ESI is, say, 0.4, this means that in every RCP we will achieve 40% of the achievable outcome in this emission pathway. For each grid cell this resulted in one linear program creating a single portfolio for all RCPs:

$$\max_{\omega}\left((1-\lambda)\min_{esi,rcp}\sum_{s}\omega_s W_{esi}q(esi,s,rcp)+\lambda\sum_{esi,rcp}\sum_{s}\omega_s W_{esi}q(esi,s,rcp)\right) \tag{1}$$

$$\text{subject to}\quad \sum_{s\in S}\omega_s = 1 \tag{2}$$

$$\omega_s \geq 0 \qquad \forall\ s\in S \tag{3}$$

$$\sum_{esi} W_{esi} = 1 \tag{4}$$

$$W_{esi} \geq 0 \qquad \forall \ esi \tag{5}$$

$$\text{fpc}(2100, s, rcp) \geq \min(0.1, \text{fpc}(2010)) \tag{6}$$

where $S = \{\text{base, toBd, toBe, toCoppice, toNe, unmanaged}\}$

ω_s : Share of management type s in the optimized portfolio

W_{esi} : Preference / weight for esi

$\text{fpc}(year, s, rcp)$: Foliar Projective Cover of the grid cell under management option s in RCP rcp in year $year$

$q(esi, s, rcp)$: Normalized quality of esi for management option s in rcp

$\sum_{s \in S} \omega_s q(esi, s, rcp)$: Quality of esi for the whole grid cell for a portfolio ω (without ESI weights)

The objective function, Equation 1, expresses that we wanted to find the portfolio ω that results in optimally balanced ESIs over all RCPs. It is a weighted combination of the worst case ESI and the average ESI. Optimizing the worst case would lead to the most balanced solution but is quite pessimistic, since a bad performance of one ESI cannot be compensated with the excellent performance of another ESI. Meanwhile, optimizing the average ESI, results in a much less balanced solution but could improve some ESIs. In the theory of sustainability, the former approach termed "strong sustainability" (Ruiz et al., 2011) is preferred (Diaz-Balteiro et al., 2018). We combined the two approaches (Diaz-Balteiro et al., 2018) by adding a trade-off parameter $\lambda = 0.2$ that resulted in balanced ESI performances and robustness to small changes in ESIs, thus obtaining more similar results in adjacent grid cells. We argue that a higher λ-value is unreasonable since it would invalidate the approach of treating all RCPs and ESIs equally. Also, mathematically, a higher λ would decrease the relevance of the worst cases (left summand) due to their naturally lower value than the averages (right summand).

Equations 2 and 3 ensure that our portfolio contained non-negative fractions for all strategies and allotted the whole managed forest area of the grid cell. Additionally, weights can be given to each ESI (Equations 4 and 5). Throughout this study we applied equal preferences W_{esi} unless otherwise noted. Finally, Equation 6 ensures the maintenance of forest cover at the beginning of the next century, using a value of foliar projective cover of 10% based on FAO's definition of forests (FAO, 2020), or the 2010 foliar projective cover if lower than 10%. Notably, violation of this equation led to a management option not to be considered for a grid cell, since such a low foliar projective cover indicates that this type of forest does not grow well under the given conditions. This also led to the exclusion of this management option from the normalization process and the q-values for the other management options. The normalization was excluded from the equations above for brevity, and the optimization was implemented using SciPy (Virtanen et al., 2020).

3. Results

3.1. Simulation Results

3.1.1. Model Evaluation

We found that LPJ-GUESS reproduced present-day data from observations and models (Table 3, Figures S6–S8 in Supporting Information S1). For example, modeled vegetation measures such as tree cover, biomass, net and gross primary productivity (NPP and GPP), ET, and runoff were close to estimates from the literature, on average over the continent and more regionally. Total carbon was overestimated. Additionally, simulated fellings were above values from country reporting. We found a historical increase in GPP of 30% in response to changing environmental conditions such as increased CO_2-concentrations from 1900 to 2010 (Figure S1 in Supporting Information S1).

3.1.2. Projected Changes in ESIs Until 2100–2130

In our simulations, a European forest landscape with solely unmanaged forest by 2100–2130 resulted in the highest carbon stocks (including products) with 12.1–12.4 kgC/m^2 (Tables S3–S6 in Supporting Information S1) over the 181 analyzed grid cells which is on average 19% above present-day values and 24% above the base scenario. Other management options generally resulted in carbon stocks below present-day values in all RCPs.

Table 3
Comparison of Our Present-Day (2000–2010 Average) Simulation Outputs to Results From Other Studies and Publicly Available Datasets

	This study	Literature	Data source[a]	Reference
Vegetation C (PgC)	13.7	13	I	Pan et al. (2011)
		11.7[b]	C	Forest Europe (2015b)
		11.6[c]	S	Liu et al. (2015)
Total C (PgC)	59.8	40.9[d]	I	Pan et al. (2011)
Forest-GPP (gC/yr/m^2)	1175	1107–1199	E, F	Luyssaert et al. (2010)
Forest-NPP (gC/yr/m^2)	426	447	I	Luyssaert et al. (2010)
Fellings (10^6 m^3 yr^{-1})	663	562	C	Forest Europe (2015b)
Total Tree Cover (%)	24.9	26.8[e]	S	GFC v1.7 Map (Hansen et al., 2013)
ET (mm/yr)	459[f]	490	S, M	GLEAM (Martens et al., 2017)
Runoff (mm/yr)	286[f]	297	G, M	UNH-GRDC (Fekete et al., 1999)

Note. The considered region is the European continent excluding Russia, Iceland, Cyprus, and Malta, but including Kaliningrad region and European part of Turkey, resulting in 3124 grid cells for our simulation. Values are representative only for the forest area of this region, except for tree cover, ET, and runoff, which are averaged over the entire region.

[a]Data source meanings: S, satellite-derived; I, inventory data; C, country reporting; M, modeled; G, Gauge-observations; F, flux measurements from ecological site; E, ecological site studies. [b]Using aboveground biomass = 79% of vegetation biomass. [c]Includes Turkey. [d]Only upper 100 cm of soil considered in their study. Our value is for the upper 150cm. [e]Tree cover here refers to canopy closure for all vegetation taller than 5 m (GFC) and LPJ-GUESS simulated crown cover of all simulated forests. GFC contains tree cover also for non-forest areas. [f]Assuming C3 grass growing everywhere outside of forest areas.

The values for simulated total harvest increased with higher-emission RCPs. The toNe management option showed higher harvests (2.9–3.5 m^{3}ha^{-1} yr^{-1}) than toBd (2.1–3.3 m^{3}ha^{-1} yr^{-1}, Figure S20 in Supporting Information S1), whereas converting to coppice caused strong reductions in harvests. Harvests for long-lived products (HLP) were much higher in the toNe management option than in toBd and lowest in coppice forests (Figure S21 in Supporting Information S1). For the unmanaged option harvests converged to zero by the end of the simulation period, as stands were successively left untouched.

Climate mitigation strongly depended on the RCP. For RCP2.6, the unmanaged forest had a higher cumulative mitigation potential than the base management (1.6 vs. 0.8 kgC/m^2), the situation was opposite for RCP8.5 (1.9 vs. 2.2 kgC/m^2).

The broad-leaved deciduous forests showed the highest roughness values, among other reasons due to the low plant area index in winter after senescence and a higher forest density than the needle-leaved evergreen forests. Also, the unmanaged forests showed higher surface roughness, majorly attributed to their tall trees.

Evapotranspiration was projected to be highest in coppices in our simulations (505 mm yr^{-1} on average) but showed higher variation between the RCPs (Figure S5 in Supporting Information S1). Additionally, the management options revealed small differences in ET when averaging over the continent, with much larger differences locally (not shown).

The simulations also showed that soil water potential varied between RCPs due to different projected patterns in rainfall and CO_2 concentrations (Figure S10, Tables S3–S6 in Supporting Information S1). Soil water potentials of the base management option were projected on average 10% lower than present-day values. Different vegetation and management affected the soil water potential, with the lowest values in coppice (around −2.1 MPa).

Higher biodiversity habitat provision was observed in the unmanaged stands compared to other forest types. It was also higher in the toBd option compared to toNe, which had higher values than coppice.

3.2. Optimized Management Portfolios

3.2.1. Optimized Portfolios and Species Shares in European Regions

The portfolios were optimized for the 2100–2130 period but it is important to keep in mind that the different management transformations occurred gradually (Figure S22 in Supporting Information S1). Across Europe we

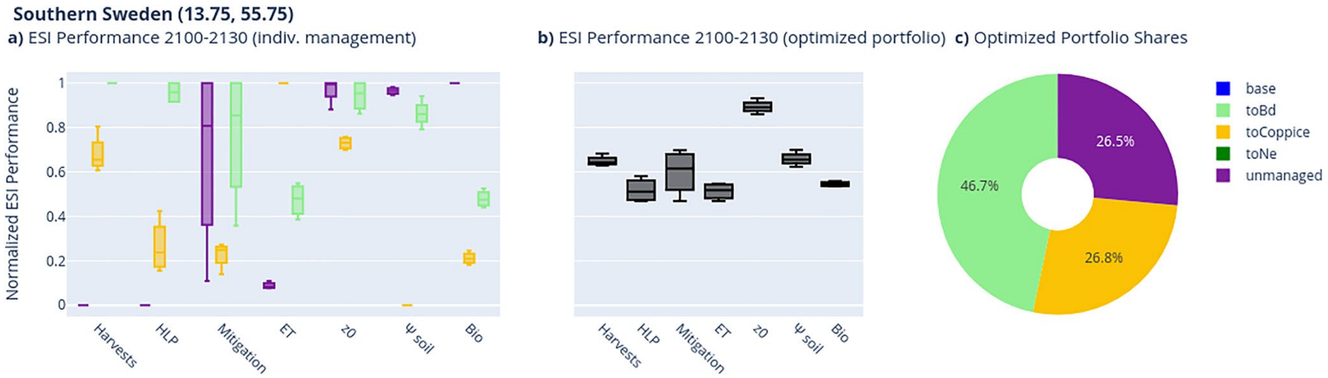

Figure 2. Example of the multi-criteria robust optimization approach for one grid cell. (a) Each box plot shows the spread of the normalized performance of the ecosystem service indicators (ESIs) over the four Representative Concentration Pathways (RCPs) when converting the forest of the entire grid cell to broad-leaved deciduous (toBd, green), to coppice (toCoppice, yellow), or to unmanaged forest (purple). ESIs for the other management options are omitted for clarity. Panel (b) is similar to (a), but for the optimized portfolio. Panel (c) shows the distribution of management options in the optimized portfolio for this grid cell. The figure illustrates how balanced the optimized portfolio is (b): regardless of the RCP, every ESI will be provided at a reasonable level (i.e., no ES performance will deteriorate in the future), whereas any other single management option would lead to very different performance of different ESIs. A value of 1 here means that this is the best attainable performance of an indicator given all management options. For example, in (a), in this grid cell, ET is highest for coppice for all RCPs, which is why the box plot is reduced to a single point at 1. This is not the case for, for example, unmanaged forest regarding mitigation: The box plots spans from 0.1 to 1 meaning that for some RCPs, mitigation is best for unmanaged, but for other RCPs it is quite bad (though never the worst, since the box does not go down to 0).

generally found that the optimized portfolios were quite diversified, comprised a large area of newly unmanaged forests, net increases in broad-leaved species shares, relevance of broad-leaved evergreen taxa in the South, and a divided importance of coppice (Figure 3).

Also, continually leaving a high share of the currently managed forest unattended played an important role in the optimized portfolios to maintain balanced ESIs. This share, 29% (Figure 3a), was similarly high across the different regions in Europe.

According to our optimization, the optimized species distribution generally shifted toward more broad-leaved species, especially in Northern Europe (Figure 3d) where the current high share of conifers decreased from 86% to 51%. Transitions in species proportion were visible in all regions with a net shift to more broad-leaved taxa from 38% to 55% by 2100–2130 across Europe (Figure 3a). However, for some grid cells the optimization caused a shift from broad-leaved species to needle-leaved species corresponding to 11% of current broad-leaved forests (mainly in Southern Europe). In comparison, roughly 34% of current needle-leaved forests were converted to broad-leaved ones by the next century (Figure 4). This transition included an increased extent of broad-leaved evergreen species (represented in this simulation by *Quercus ilex*) in Southern Europe with 22% of forests in that region consisting of broad-leaved evergreens (in the toBe, base, and unmanaged fractions of the portfolios).

According to our optimization, large regional differences were observed for coppice in the future forest management with no forests in the Southern and Alpine regions managed as coppice. At the same time, it had higher importance in Atlantic, Northern, and Continental grid cells (7%, 14%, and 7%, respectively, Figures 3c–3e).

3.2.2. Optimized Portfolio Compared to Present-Day Values and Base Management

According to our optimized portfolios (row "optimized" in Table 4 and Tables S3–S6 in Supporting Information S1), total carbon stocks were 1%–5% above present-day levels by the beginning of the next century. On the other hand, maintaining the base management (present-day species composition) in all regions across Europe decreased stocks by 3%–7%. The base management resulted in many regions becoming carbon sources, particularly in Central Europe and Southern Scandinavia (see Figure 5b, for visualization of RCP 4.5). Instead, the optimized portfolio provided more carbon storage in almost all grid cells than the base scenario (Figure 5d).

The mitigation ESI was higher in the optimized portfolio than base management in RCPs 2.6 and 4.5 whereas it was about equal in RCPs 6.0 and 8.5. When considering additionality, this means that in the lower RCPs, the optimized portfolio offered a positive net impact on climate change mitigation compared to base management, whereas in higher RCPs, there was no net effect on global climate.

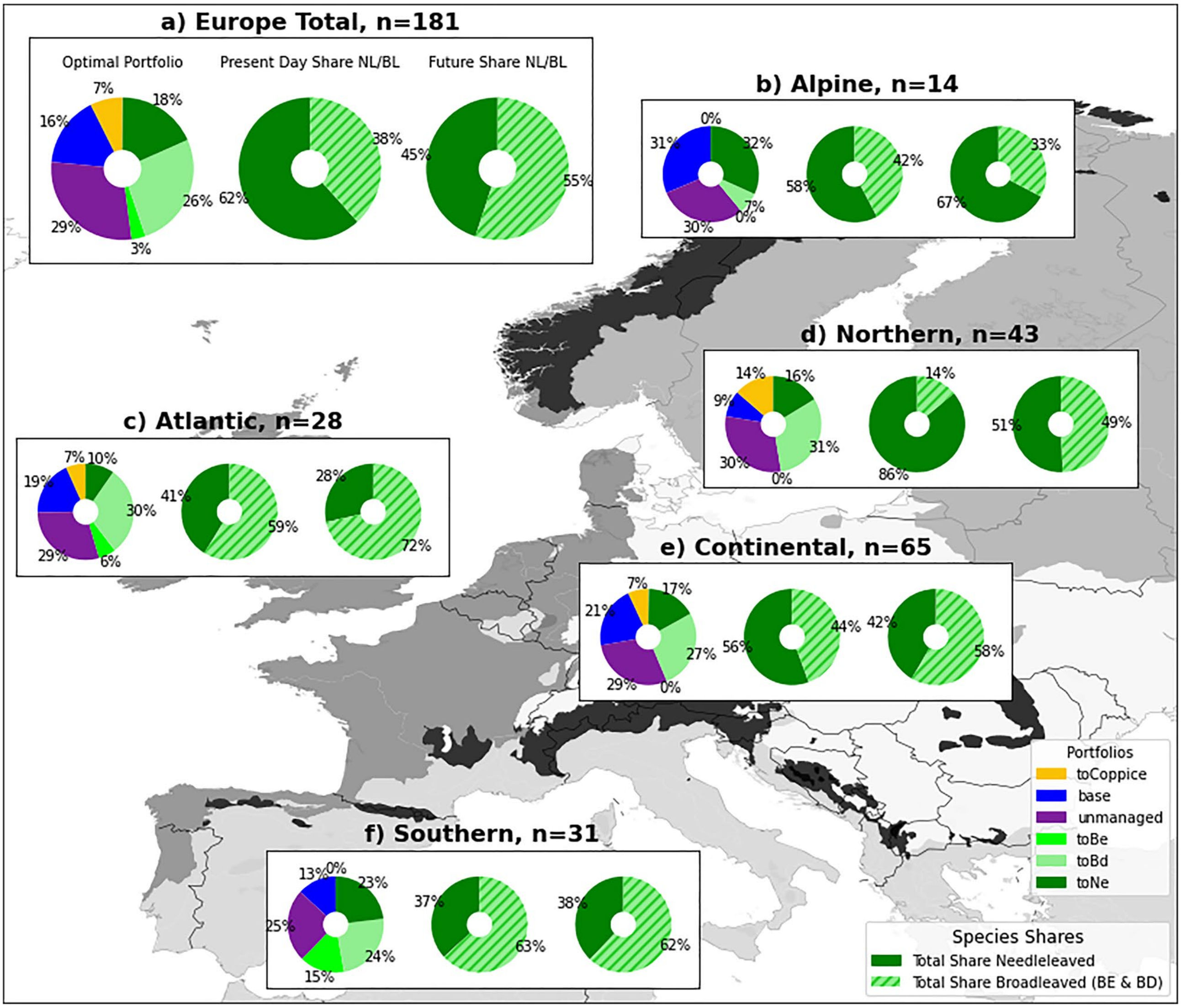

Figure 3. Proposed management portfolios in Europe, aggregated over all simulated grid cells (a) and for different regions (b–f). The five regions (indicated by different shades of gray) are based on the climatic zones of Metzger et al. (2005) but aggregated as in the IPCC AR5 (Kovats et al., 2015). The leftmost chart in each box always shows the optimized share of management options in the entire region. The middle and right charts show the current and future shares of needle-leaved and broad-leaved tree species, respectively. The depiction of broad-leaved shares is shaded as it may contain both deciduous and evergreen species. Note that portfolios and future species shares differ between cells of a region. The computation is done for each grid cell independently (Figure S3 in Supporting Information S1). Here we show the forest-area-weighted aggregated shares.

The increase in carbon storage came at the cost of lower harvests compared to the base management option. Harvests for the optimized portfolio were simulated to be 4%–29% lower than present-day values, whereas for base they increased by 2%–33% depending on the RCP (Tables S3–S6 in Supporting Information S1). Additionally, Figure 6 shows the spatial patterns of these changes in the harvest volumes. Harvests were highest in Scandinavia, both today and in the future; however, in the optimized portfolio they decreased in various parts of Europe (Figure 6c).

Regarding the biogeophysical effects, roughness length was 17%–26% higher for the optimized portfolio than in present-day and 18%–21% higher than the future base values. These increases occurred in most regions except for a few cells (Figure S17c in Supporting Information S1). The optimized portfolio showed spatially different changes in ET with increases in Central, Eastern, and Northern Europe; however, they decreased in Atlantic and Southern regions (Figure S16c in Supporting Information S1). The effect of optimization regarding ET was

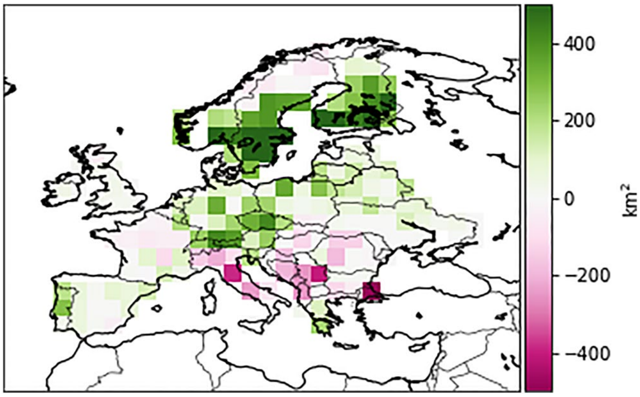

Figure 4. Net forest conversion from needle-leaved to broad-leaved forests in km². Negative values indicate a conversion from broad-leaved to needle-leaved forests.

small compared with the base management according to our simulations (Figure S16d in Supporting Information S1).

3.2.3. Optimized Portfolios With Alternative Assumptions

The previous results were based on the standard settings equally valuing all ESIs. However, decision-makers will likely encounter specific constraints or have regional preferences for some ESIs. Therefore, we briefly express our results when adding weights or constraints to the optimization.

First we found that counteracting the decrease in harvests by adding a constraint to keep total harvest at present-day levels in every grid cell is an unsuitable strategy: In about a fourth of the analyzed grid cells, such harvest levels could not be maintained until 2100–2130 for all RCPs since many stands did not reach the density and size conditions for a clear-cut. This made the optimization infeasible for many grid cells as the harvest constraint could not be met.

However, when valuing harvests twice as important as all other ESIs ("double harv"), the unmanaged fraction of the optimized portfolio was reduced to 18% (Figure S11 in Supporting Information S1). Harvests were higher than in the default optimization, ranging around the present-day levels when aggregating over the continent, nonetheless, they depended highly on the RCP (−16% to +12%, Table 4 and Tables S3–S6 in Supporting Information S1), with lower values for HLP.

Doubling the importance of climate mitigation ("double mit") resulted in a higher share of unmanaged forest such that 41% of currently managed European forests were proposed to be successively left untouched (Figure S12 in Supporting Information S1). This resulted in carbon stocks 5%–9% higher than present-day values, with a 4% higher mitigation potential than the standard optimization. There were also benefits for other ESIs compared with the standard optimization, but with even lower harvests than in the standard optimization portfolio. Additionally, coppice virtually vanished from the continent. Adding summer and winter albedo as ESIs did not change the overall portfolios much (Figure S14 in Supporting Information S1).

Furthermore, optimizing for RCP2.6 only led to slightly larger fractions of unmanaged forest (31% vs. 29%) and lower shares of coppice forests (3% vs. 7%). Also the total share of broad-leaved forests decreased from 55% to 49%

Table 4
Ecosystem Service Indicator Mean Performances if One Management Were Applied to the 181 Grid Cells

	C-Pool	Mitigation	Harvests	HLP	ET	Ψ*soil*	*z*0	Bio
Unit	kgC/m²	kgC/m²	m³/ha/yr	m³/ha/yr	mm/yr	MPa	m	unitless
base	9.8(±0.2)	1.7(±0.6)	2.99(±0.36)	0.77(±0.09)	495(±22)	−1.55(±0.10)	0.53(±0.01)	0.47(±0.01)
toBd	10.1(±0.1)	1.7(±0.7)	2.77(±0.52)	0.43(±0.07)	498(±17)	−1.66(±0.12)	0.75(±0.03)	0.55(±0.02)
toBe[a]	3.0(±0.1)	0.5(±0.2)	0.78(±0.08)	0.12(±0.01)	457(±47)	−2.76(±0.08)	0.41(±0.02)	0.08(±0.00)
toCoppice	9.9(±0.1)	1.3(±0.5)	1.77(±0.35)	0.11(±0.01)	505(±22)	−2.12(±0.10)	0.60(±0.03)	0.21(±0.01)
toNe	9.7(±0.2)	1.7(±0.6)	3.20(±0.29)	0.98(±0.10)	490(±25)	−1.44(±0.07)	0.40(±0.01)	0.42(±0.02)
unmanaged	12.2(±0.2)	1.8(±0.2)	0.08(±0.01)	0.02(±0.00)	492(±23)	−1.53(±0.07)	0.66(±0.04)	0.77(±0.04)
Present	10.3	0.0	2.48	0.65	501	−1.41	0.52	–[b]
Optimized	10.7(±0.1)	1.7(±0.5)	2.10(±0.29)	0.44(±0.05)	498(±22)	−1.64(±0.09)	0.63(±0.02)	0.57(±0.01)
Double harv	10.4(±0.2)	1.7(±0.6)	2.47(±0.32)	0.55(±0.06)	498(±22)	−1.62(±0.09)	0.62(±0.02)	0.54(±0.01)
Double mit	11.0(±0.1)	1.8(±0.5)	1.75(±0.26)	0.34(±0.04)	495(±21)	−1.58(±0.09)	0.67(±0.02)	0.64(±0.00)

Note. The values are the area-weighted means averaged over the 4 RCPs, values in brackets are standard deviation and indicate the spread over the RCPs. The lower rows of the table correspond to the optimized portfolio in the standard setting ("optimized") as well as for the experiments with different weights and constraints as described in the text.

[a]toBe management was only feasible in few regions in Europe, the reported values for this case are only for those regions ($n = 38$). In the optimized portfolio of a grid cell, only management was considered that ensured forest cover by the end of the century. [b]Computation of the Bio indicator is not sensible for present day, as it is solely a relative value between management options.

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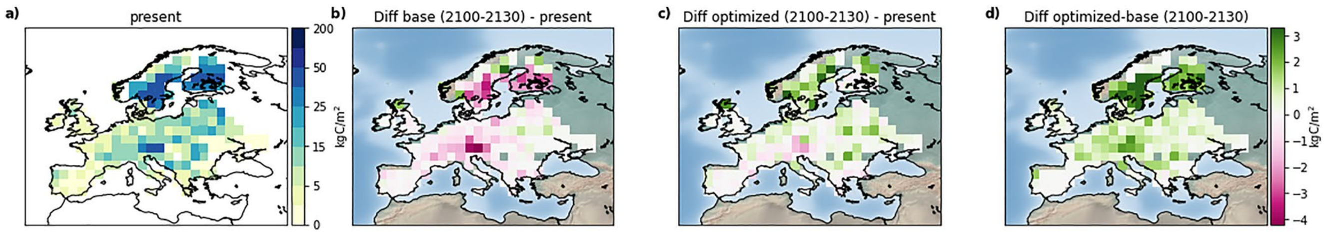

Figure 5. (a) Simulated present-day and (b–d) future carbon storage (above and below ground in kgC/m^2) in European forests for RCP4.5. Applying the base management (keeping species shares at present-day proportions, see Table 1) mainly led to decreases in carbon stock across Europe (b), the optimized management portfolio led to stable or increased carbon stocks in most parts of Europe (c), particularly in the East and North. By 2100–2130, the optimized portfolio had higher carbon stocks throughout Europe compared to the base scenario (d).

(Figure S15 in Supporting Information S1). Keeping displacement factors constant resulted in lower fractions of unmanaged forests (17%; in the Northern and Atlantic region around 10%), but a stronger focus on broad-leaved trees and coppice, compared to the standard optimization (Figure S13 in Supporting Information S1).

4. Discussion

4.1. Evaluation of LPJ-GUESS Simulation Results

LPJ-GUESS was able to reproduce present-day forest vegetation characteristics (Table 3 and Figures S6–S8 in Supporting Information S1) on which we based our management scenarios. However, soil carbon (and consequently total carbon) in our simulations was larger than estimates. This could result from the fact that we did not account for forest management prior to 1871. Since wood was never removed from the forest, trees that died simply decomposed on-site (Lindeskog et al., 2021). However, the estimate of Pan et al. (2011) also only considered the first 100 cm of the soil layer. A crucial point in projections via DGVMs is the response to changing environmental conditions, including elevated CO_2-concentrations. In our runs, the increase in GPP between 1900 and 2010 was around 30% as observations also suggest (Figure S1 in Supporting Information S1, see Campbell et al., 2017). Regarding historical changes in the land sink, LPJ-GUESS shows an intermediate response compared to other models used in the Global Carbon Budget (Figure S2 in Supporting Information S1, see Friedlingstein et al., 2020).

4.2. Species Composition of Future Forests

According to our simulations and optimizations, implementing climate-smart forestry and adapting European forests to be productive under uncertain future climate conditions may imply strong changes in their composition, as well as trade-offs that need to be addressed.

First of all, throughout the continent a shift toward more broad-leaved species is proposed by our optimization. This result aligns with the reduce-air-temperature-portfolio of Luyssaert et al. (2018). However, it is in stark contrast to their maximize-carbon-sink-portfolio, proposing higher shares of conifers in the future. Since they optimized single objectives, their maximization of carbon sink focused mostly on coniferous forests due to their overall higher volumes and avoided emissions from substitution effects. On the other hand, our optimization

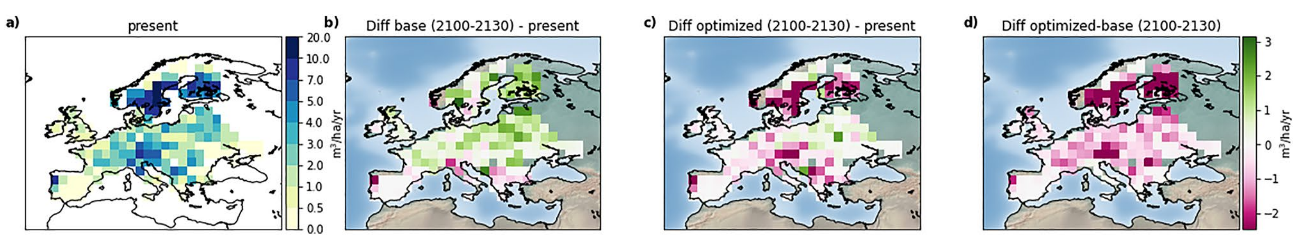

Figure 6. (a) Simulated present-day and ([b–d] future harvest; in $m^3\ ha^{-1}\ yr^{-1}$ dry biomass) in European forests for RCP4.5. Harvests decreased in most regions in Europe in the optimized portfolios (c) but mostly increased if present-day management was extended into the future (b), leading to lower harvests for the optimized scenario compared to base (d). The main reason for this is the high share of unmanaged forests throughout Europe in the optimized portfolios.

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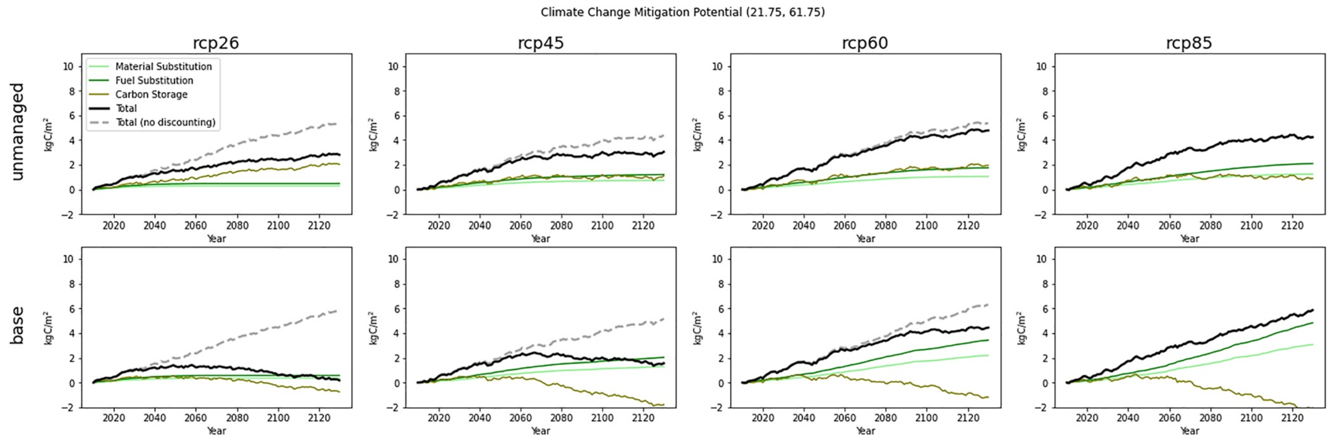

Figure 7. Climate change mitigation of forests highly depends on the Representative Concentration Pathway (RCP): In this grid cell in Finland (21.75, 61.75), continuing the base management led to higher mitigation in RCP 8.5 compared to successively leaving the forest unmanaged (same species). This was different in low-RCP scenarios where the substitution of carbon-intensive fuels and materials through the usage of wood products was assumed to abate rather quickly, making the carbon storage in forests and products the main driver for mitigation potential. The reason for there still being harvests in the unmanaged option is that the change to unmanaged forest only happened gradually, since only once each stand reached maturity a decision about future management was made. See also Figures S20 and S22 in Supporting Information S1. The dashed lines show the mitigation if constant discounting factors are assumed, where in all RCPs, wood products keep avoiding the same amount of emissions today and in the future.

considered multiple ESIs, dampening the importance of conifers due to their negative performance in terms of other ESIs. Additionally, we assumed a higher risk of disturbance for needle-leaved compared to broad-leaved species (Figure S25 in Supporting Information S1, Pugh et al., 2019; Seidl et al., 2017), another reason for the lower needle-leaved shares in our portfolios. Since multiple factors govern air temperature, the reduce-air-temperature portfolio of Luyssaert et al. (2018) implicitly optimized for more than one variable. Therefore, its similarity to ours, in terms of species composition, is not surprising.

Propositions for more broad-leaved forests in Europe have previously been made regarding lower summer temperatures (Schwaab et al., 2020) and lower fire risk (Astrup et al., 2018). Felton et al. (2010) indicated similar potential benefits for biodiversity as our approach, particularly for replacing of coniferous monocultures in Scandinavia. Nevertheless, it must be noted that in our study, those broad-leaved forests were modeled using stands of one or two species. An actual implementation should naturally include more diverse forests.

Second, our simulations suggested a large (but gradual) shift toward more unmanaged forest attributed to its benefits for biodiversity, water availability, and surface roughness (see, e.g., Figure 2). These benefits of unmanaged forests regarding multiple ESs were also found in an MCDM case-study by Diaz-Balteiro et al. (2017). Also, we found that the role of unmanaged forests was divided regarding global climate change mitigation, strongly dependent on the assumptions about the decarbonization of the construction sector (Section 4.3, Figure 7). Luyssaert et al. (2018) similarly showed a 30% fraction of unmanaged forest in their maximize-carbon-sink-portfolio (note that their values reflect the entire European forest whereas we considered only the currently managed part). On the other hand, their reduce-air-temperature-portfolio only contained 19% of unmanaged forest. This matches our optimization results when we kept displacement factors constant as they did (see Section 3.2.3 and Figure S13 in Supporting Information S1). This emphasizes the importance of a thorough analysis of wood substitution effects and the carbon-intensity of other sectors (more detail in Section 4.3).

Third, coppice, currently an abundant forest type in Southern Europe with areas well above one third of total forest area in some countries (Maganotti et al., 2018), is proposed to vanish from Southern European forests by our optimization. The optimization penalized its disadvantages such as the missing provision of long-lived wood products, low surface roughness and water availability (Figure 8), the latter agreeing with findings by Drake et al. (2009), see also Hartwich et al. (2014). This computed decrease in coppice forests due to ES provision complements an observed and proposed transition of coppice forests toward other management forms in some regions of Southern Europe, in part due to economical aspects (e.g., Stajic et al., 2009; Vacchiano et al., 2017). In other areas where coppice is currently less relevant (but potentially used to be), its role could increase as the high LAI of dense coppice forests leads to high transpiration rates and consequently local cooling.

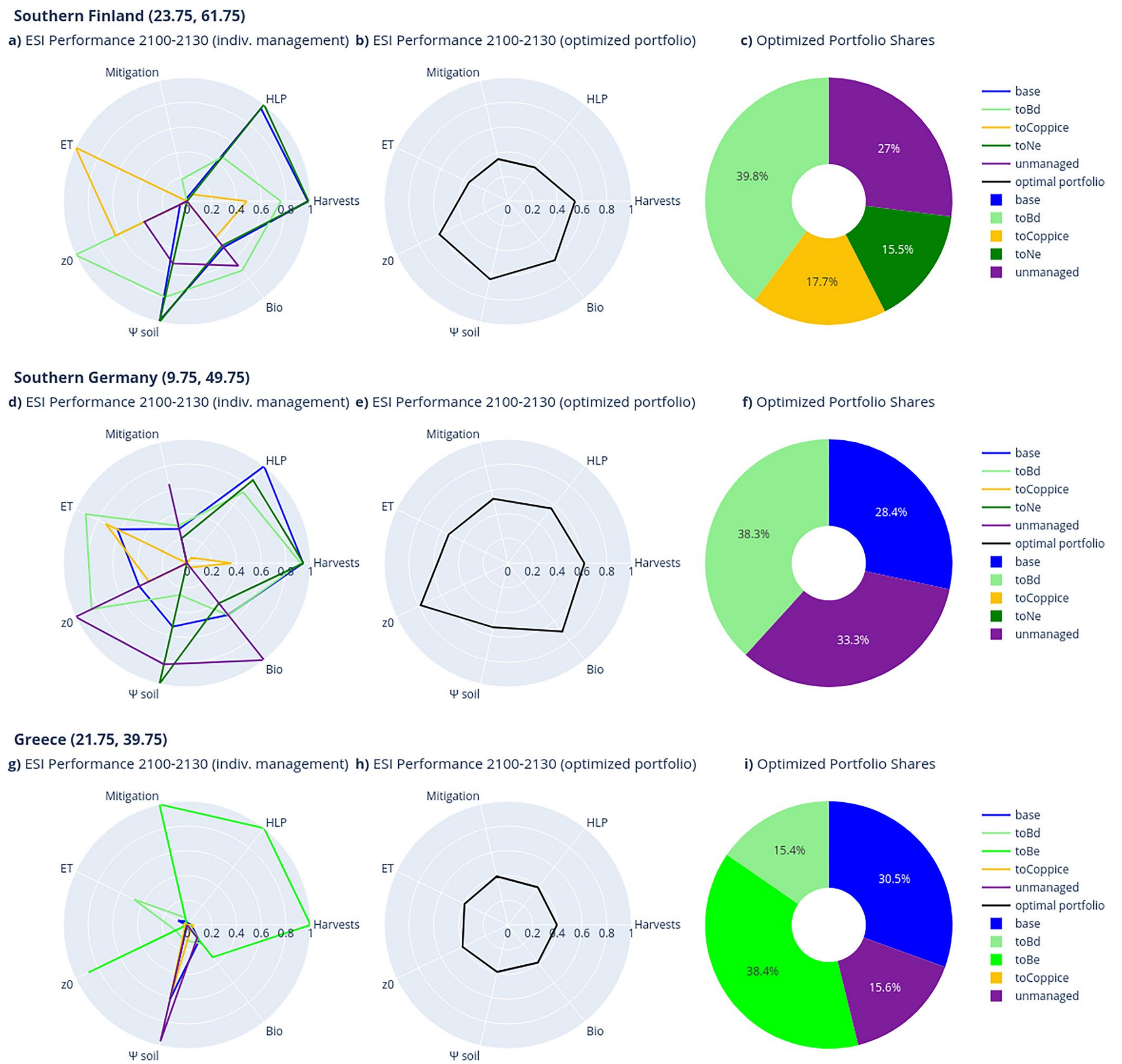

Figure 8. Examples of the optimization across a North-South gradient. Plots (a, d, and g) show the *worst cases* over the Representative Concentration Pathways (RCPs): a value of 1 in these radar charts thus indicates that this management had the best performance for this Ecosystem service indicator (ESI) in any given RCP. A value of 0 consequently does not necessarily mean that this management was always the worst for this ESI, it only means that this management had the worst performance *in at least one RCP*. The optimized portfolio of the grid cells contained multiple forest types (c, f, and i) because each type had its own advantages and disadvantages (a, d, and g). Coppice was mixed into the portfolio for the Finnish cell (c) but not into that of the cell in Germany (f) because in the latter, the merits of coppice (i.e., ET) were not high enough to outweigh its disadvantages ([d] compare to [a]). The grid cell in Greece was dominated by broad-leaved evergreen forests (i) because it had many advantages. It contained a much smaller fraction of unmanaged forests because other than Ψ_{soil}, they performed poorly compared to the other options (g). Panels (b, e, and h) also show how the proposed portfolio provided the ESIs in a very balanced fashion: In the worst cases, all ESIs were provided at a balanced level (compare Figure 2). Note that we used $\lambda = 0.2$, thus not only optimizing for the worst case, but partly also for the average ESI performance which explains some peculiarities, for example, the high share of base management in (i).

Fourth, our simulations indicated a trend to more drought-tolerant species especially in Southern Europe attributed to the low availability of water (Figure S10 in Supporting Information S1). Also, in our simulations, drought-induced higher mortality was intensified by lower regeneration due to the additional warmer winter temperatures that led to decreased natural reproduction. Consequently, species that are more drought-tolerant and do not rely on winter chilling such as *Q. ilex* and *P. halepensis* were indicated to dominate in those regions, a result that is supported by the literature (López-Tirado & Hidalgo, 2018).

4.3. Forests and Climate Change Mitigation

The European forest carbon sink has been stable in recent decades (Grassi et al., 2019; Pan et al., 2011) but changes to forest management strategies already have been proposed necessary to sustain it (Nabuurs et al., 2013). One means of sustaining or increasing the sink could be to increase the forest area which happened in the past decades (Forest Europe, 2020) and could continue to do so (Perpiña Castillo et al., 2021). This however is not part of our study as we focused solely on "forests remaining forests." According to our simulations, keeping the area of managed forest constant will transform the sink into a source (Table 4, Tables S3–S6 in Supporting Information S1); however, leaving some forests unmanaged could sustain a small sink until the beginning of the next century (1%–5% increase in carbon stocks including products).

Nevertheless, climate change mitigation is not only about carbon storage. Substitution of fossil fuels and carbon-intensive products plays another significant role. Although many studies previously found that substitution effects are crucial for mitigation highlighting the importance of managed forests, recent research has indicated that the benefits of substitutions will likely decrease, depending on the RCP (Brunet-Navarro et al., 2021; Harmon, 2019). This trend is caused by an increased share of renewables in the energy mix and technological improvements in the construction sector (see, e.g., IEA, 2020; Lehne & Preston, 2018). Therefore, the importance of harvests for our mitigation ESI decreased since we explicitly considered such decreasing substitution effects (Figure S23 in Supporting Information S1) and optimized for multiple RCPs. Hence the high proposed share of unmanaged forests. This share even increased when only looking at RCP2.6 where substitution effects diminished very quickly (Figure S15 in Supporting Information S1). The opposite was true when keeping substitution effects constant for all RCPs (Figure S13 in Supporting Information S1).

It is important to note that our proposed shift to more unmanaged forests was gradual, that is, forests were successively taken out of management. Consequently, our approach still strongly utilized the substitution effects in the shorter term when they are still important in low emission scenarios. The total benefit of management or non-management in terms of mitigation depended on the time frame and RCP; hence our portfolios offer a diversified mitigation strategy.

Additionally, our assessment is somewhat unbalanced: The production of steel and concrete combined account for 14% of global carbon emissions, and their improvements in carbon-intensity partly depend on breakthrough technologies (IEA, 2020; Lehne & Preston, 2018). Although we accounted for such improvements, we ignored potential innovations in wood usage. Many new technologies will likely facilitate higher amounts of wood usage in the construction sector, including strategies to use wood from broad-leaved trees and increase the building materials' carbon content (Churkina et al., 2020). Considering this, the increased usage of wood as construction material (as opposed to burning) and longer product lifetimes could drastically affect the mitigation potential. Future studies should focus on this change which is currently rarely explicitly assessed (e.g., Dugan et al., 2018; Smyth et al., 2014, 2018). Another aspect that is important but out of scope is the impact of different materials and energy sources on other environmental and social dimensions (see, e.g., Blankendaal et al., 2014; Hertwich et al., 2016; Santangeli et al., 2016).

4.4. Management and Disturbances

Future management must consider increases in disturbances (Seidl et al., 2014, 2017; Senf & Seidl, 2021a, 2021b). A common strategy is to diversify risks against these disturbances (Jandl et al., 2019; Mitchell, 2013), which to some extent is implicitly included in our propositions: Since we assessed the provision of multiple ESIs under increasing disturbances, our optimization automatically resulted in diverse forest compositions. Additionally, our portfolios contained an increased share of broad-leaved forests which are less prone to disturbances (Figure S25 in Supporting Information S1, Astrup et al., 2018; Pugh et al., 2019). Due to the uncertainties and complexity related to disturbances (e.g., Ahlström et al., 2015; Reyer et al., 2017; Seidl et al., 2011), we chose a rather simple representation of the change of disturbance regimes. While we used species-specific disturbance intervals, we refrained from making their increases species-specific, too, because research has shown that also species that were previously thought to be rather resilient have experienced heavy impacts of disturbances in recent years, for example, European beech in Central Europe (Buras et al., 2020; Schuldt et al., 2020).

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The direct effect of management regarding disturbances was only partly contained in our study, namely through salvage logging in managed stands and because in managed stands, some trees are already harvested and thus the impact of a disturbance is lower (Figure S25 in Supporting Information S1). The impact of salvage logging on mitigation was rather small since disturbed wood also remains in the forest for many years before it decays back to the atmosphere (Suzuki et al., 2019), and because we assumed low fractions of salvaged wood to be used in products. However, an increased material usage of salvaged wood and measures to dampen disturbance impacts will foster the relevance of forest management for mitigation. Examples are harvesting trees that are more likely to be affected by disturbance, adapting the age structure (O'Hara & Ramage, 2013), or active fire suppression (Agee et al., 2000; Fernandes et al., 2013). This should be addressed in more detail in future studies.

4.5. Harvest Reductions

The large fractions of successively unmanaged forest in the portfolios naturally resulted in a gradual reduction in harvest volumes in favor of other ESIs. Future harvests of the optimized portfolios in low-emission RCPs were simulated to be 20%–29% lower than present-day values, but closer to present-day values in the high-emission RCPs, likely due to simulated CO_2-fertilization (Tables S3–S6 in Supporting Information S1). Furthermore, long-lived wood products provision would be further reduced in the optimized portfolio as it includes broad-leaved forests and coppice, which provide lower shares of wood being useable for those products (especially coppice, almost completely used for firewood).

From a purely financial perspective, a decrease in harvests for mitigating climate change is unremarkable. Previous reports indicated that financial incentives are probably necessary to implement forest management measures for mitigation (e.g., Khanal et al., 2017). However, a decrease in harvests in Europe will lead to increased wood imports, possibly stemming from unsustainable sources leading to dislocation of emissions and increased ecological pressure in other regions (Berlik et al., 2002; Mayer et al., 2005).

The increasing demand for wood in various industries (FAO, 2022; Nabuurs et al., 2007) and a proposed timber usage rise for construction (Churkina et al., 2020) combined with increasing floor areas per capita (e.g., Bierwirth & Thomas, 2015) will cause great pressure on forests in terms of wood production. Therefore, this stands in direct conflict with the various other ESs offered by forests. Countermeasures could be to redirect harvests for fuel toward long-lived products (Churkina et al., 2020), and societal changes in the amount of wood required per capita. Other means could be forest expansion, changes in harvest intensity, and inclusion of fast-growing species such as Douglas fir (Thomas et al., 2022). Sensible intensities of such measures and their impact on ESs could also be assessed by our framework.

4.6. Provision of Ecosystem Services and Trade-Offs

Our study was aimed to assess the possibility of achieving climate-smart forestry by combining mitigation, adaptation, and the provision of other ESs, under the uncertainty of future climate pathways. Above we mainly focused on forest adaptation (e.g., some species are no longer suitable in certain regions, see Section 4.2) and the issues arising for mitigation and harvests. But our portfolios can help enabling the continued provision of other ESs.

Roughness length was estimated to be about 22% higher than present-day values. It is mostly governed by plant area index and canopy height (Moene & van Dam, 2014; Raupach, 1994). Higher roughness decreases aerodynamic resistance and consequently increases heat fluxes. The effect of canopy height explains the high simulated roughness values of unmanaged forests. Also, the change to more deciduous species increased the average roughness, since due to senescence their winter plant area index was quite low leading to lower resistance (Figure S9 in Supporting Information S1).

Future ET values heavily depended on the RCP, especially in the high-emission scenario, where we found strong ET reductions in all management options (Table S6, Figure S5 in Supporting Information S1). With elevated atmospheric CO_2 concentrations, plants do not need to open their stomata as much, hence losing less water (Keenan et al., 2013). Consequently, trees are more drought-resistant but the local cooling reduces, an implicit trade-off between adaptation and local climate regulation. However, in low-emission RCPs, this effect was not large enough to compensate for increased precipitation and temperature, causing a net increase in ET. Note that vegetation-climate feedbacks will likely amplify temperature changes through ET, increasing warming in Southern Europe, and reducing warming in Central Europe (Wramneby et al., 2010).

Although our simulations indicated positive effects of elevated CO_2 levels on productivity through higher water use efficiency, soil water potentials decreased until the beginning of the next century for all RCPs except RCP2.6 (Figure S10 in Supporting Information S1). Furthermore, our water availability indicator depicting the minimal monthly soil water potential decreased for almost all management options and RCPs compared with the present-day values, indicating higher drought risk in the future. Needle-leaved forest showed the highest water availability—for RCP 2.6 even slightly higher than today. In contrast, coppice showed the lowest, due to its high LAI and water usage.

In terms of biodiversity, our approach favors unmanaged forests over managed ones due to the higher abundance of different age classes, dead biomass, and large trees. This result is confirmed by Vuidot et al. (2011) who argue that the number of different habitats makes unmanaged forests preferable in terms of biodiversity. However, more important than the "management versus conservation antagonism" is the diversity of landscapes (e.g., Schall et al., 2021) which our portfolio-based approach implicitly offers. Additionally, making management mimic features of unmanaged forests and especially avoiding clear-cuts with species changes could further improve the suitability of managed forests in terms of biodiversity (Paillet et al., 2010; Vuidot et al., 2011). Although diversity is often evaluated at a within-stand level, concentrating on landscape-scales similar to this study can be a reasonable approach (Schall et al., 2018). Importantly, using biodiversity indicators such as those used in this study is encouraged, though still speculative (Davies et al., 2008) and insufficient biodiversity assessment. Also, simulations with more heterogeneous species are necessary, for a larger variety of species and microhabitats (Vuidot et al., 2011).

4.7. Different Experiments and Applicability of the Methodology

In a few experiments (Section 3.2.3) we showed the implications of changing the optimization focus. Interestingly, a stronger focus on mitigation resulted in a near-complete disappearance of coppice management from the continent (Figure S12 in Supporting Information S1) due to low material substitution and low carbon stocks. Its apparent benefit for fuel substitution is also decreased since other management options obtain similar fuel substitutions because large fractions of wood products can displace fossil fuels at the end of their lifetime (Knauf et al., 2015). In the considered time frame, the fuel substitution of coppice thus did not have a significant impact. We consequently suggest that more research is necessary to assess the currently rising interest in coppice due to its apparent potential for bioenergy (Maganotti et al., 2018). Note that we mean traditional coppice here, not short-rotation plantations.

Inclusion of albedo only changed portfolios marginally (Figure S14 in Supporting Information S1) because of the strong focus on worst cases. Since we already assessed seven ESIs, chances were high that the "crucial" ESI was already included. Nevertheless, in some cells, including albedo led to changes, often toward more unmanaged forest, partly because these were simulated to be less dense, that is, some highly reflective grass shone through (not shown).

Increasing the importance of harvests in the optimization disclosed that harvest levels can be increased compared with the standard optimization while still adequately representing other ESs (Section 4.5), with harvests spread around present-day values, depending on the RCP. Adding constraints to the optimization to keep harvests at present-day levels in every grid cell could not provide a mathematically feasible solution for many grid cells. This was because harvest rates in these grid cells inevitably decreased until the end of the century, demonstrating that collaboration between countries will be necessary to meet future wood demands and climate objectives.

Our weighting was done equally for the entire continent, but the importance of ESs actually depends on the geographic location and local socioeconomic circumstances. Furthermore, our weights were independent of the RCP although different RCPs could yield different ESs to be most important. While technically possible, including ES-weights that depend on the RCP is a major task of its own and was excluded from this study. We would like to note however, that adaptation was accounted for as RCP-dependent in our study, since Equation 6 guarantees a healthy forest cover under all RCPs.

Considering all RCPs simultaneously is a very risk-averse strategy, especially considering the decreasing likelihood of RCP8.5 (Hausfather & Peters, 2020). However, including the full range of uncertainty could avoid costly adaptation (Lawrence et al., 2020) and is also suggested by the IPCC (IPCC, 2021). We however also applied our methodology to RCP2.6 only where at least the shift to broad-leaved forests is not as pronounced as when considering all RCPs (Figure S15 in Supporting Information S1).

Finally, our results showed management options and potential impacts on a very coarse 0.5° × 0.5° resolution. In practice, an application of our methodology must be made on a fine scale taking into account site-specific characteristics, ideally with more detailed regional assessments of the inputs. Our general results can serve as a guideline for such an application, in which also ES preferences should be included, obtained for example by conducting regional stakeholder surveys (see, e.g., Knoke et al., 2020). Some ESs could also be ignored by setting weights to zero. This could for example, be the case when the mitigation potential of a particular forest is rather small and other ESs are more important. Additionally, harvest adjacency and avoidance of fragmentation (Baskent & Keles, 2005; Millar et al., 2007) could be directly included as constraints in our optimization, but lead to increased (computational) complexity.

4.8. Conclusion

The goal of our study was to obtain insights on possible climate-smart forest management portfolios across Europe allowing for a balanced provision of many ESs. The resulting portfolios with high shares of broad-leaved forests and unmanaged forests revealed a consequent trade-off between better performance of a few ESs against a balanced provision of all ESs with potentially lower performances.

The main trade-off shown in this study lies in harvest reductions in favor of other ESs, such as increased biodiversity, mitigation, and local cooling. Such harvest reductions and potentially increasing wood demand will put additional pressure on forest ecosystems, especially when forest areas remain constant as assumed here. This might also entail further ecological issues when wood is imported from unsustainable sources. Possible countermeasures might be increasing the fraction of wood harvests used for long-lived products (Churkina et al., 2020) and changing societal behavior. We also indicated how adding a preference to wood harvests in the optimization could alleviate the harvesting decreases to some extent; however, at the cost of other ESs.

Furthermore, our approach revealed another trade-off related to climate uncertainty. Portfolios optimized for a various climatic futures show weaker performance in some ESs than optimizing for a certain future, a cost of deriving robust solutions. However, concurrently, this is an important benefit of our approach: the possibility to make valid propositions for a wide range of climatic futures. We found this uncertainty-related trade-off to be particularly pronounced when assessing climate change mitigation. Mitigation depends on the forest carbon sink as well as wood products that substitute carbon-intensive materials and fuels in various economic sectors. However, in low-emission scenarios, the potential future decarbonization of these sectors may cause a decreased importance of substitution effects compared to the carbon sink, whereas the opposite is true in high-emission scenarios. Our portfolios offer an intermediate mitigation strategy, based on both, enhancing the carbon sink and continued material and fuel substitution. In any case, a crucial means for enhancing mitigation is increasing the fraction of harvested wood used for long-lived products. Further work should focus on this aspect and more detailed assessments of management options and ESIs. Also, the uncertainty space should be further explored by including forcing data from multiple GCMs and multiple process formulations in other DGVMs.

Conclusively, we applied a combination of forest management simulations with a multi-criteria robust optimization framework. Our study gives recommendations on climate-smart forest management options capable of providing many ESs in the future under a broad range of future climate scenarios. Our results provide insights on a general direction of European climate-smart forestry that may be used as a baseline when developing regional forest management strategies in practice. However, we also revealed that such a holistic approach does not eliminate all trade-offs. Although we acknowledge that further considerations are required, this study lays the groundwork for future research that examines trade-offs of forest management strategies in terms of ES provisioning and especially climate change mitigation under a highly uncertain future.

Data Availability Statement

The code to compute the ESIs, analyze them, and run the optimization is publicly available at https://doi.org/10.5281/zenodo.6667489 together with data for the four grid cells used for the illustrations. The entire data set to reproduce all optimizations for the European continent is available at https://doi.org/10.5281/zenodo.6612953.

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