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A procedure to rank pre-defined biodiversity hotspots in Finland

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ABSTRACT

The double crises of climate change and declining biodiversity are threatening boreal forests worldwide. We urgently need to effectively allocate resources to have more impactful conservation outcomes. Here, we examine the optimal targeting of carbon storage and deadwood production in forest landscapes. We specifically aim to discover (a) the regional differences in effectiveness and (b) the influence of the timing of targets. We conducted the study in two Finnish regions with pre-defined biodiversity hotspots. For each hotspot, the development of the forests over the next 30 and 50 years, including carbon storage and deadwood, was simulated. Then, mixed-integer linear programming was used to identify trade-offs between economic revenue, carbon storage of living trees, and deadwood volume. The cost-effectiveness varied both between and within regions, with the southern region achieving consistently larger carbon storage and more deadwood with the same loss of revenue. The cost-effectiveness was better in the 50-year time horizon than in 30 years, which was mainly due to the initial forest structure. The results indicate that to allocate conservation resources effectively, we need to consider the existing forest structure and its development, as well as regional differences and the time horizon of the targets.

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MILP; additionality; trade-offs; deadwood; climate change mitigation; forest management; Motti stand simulator

Introduction

The double crises of climate change and declining biodiversity are threatening the world's ecosystems. Climate change-induced extreme weather events and shifts in baseline conditions are pushing ecosystems towards tipping points (Brando et al. 2025; Lenton et al. 2025). To safeguard biodiversity, 44% of terrestrial areas require conservation action – ranging from protected areas to changes in land management (Allan et al. 2022). However, the capacity of forest ecosystems to mitigate climate change as carbon sinks has become uncertain (Pan et al. 2024), especially in Europe (Migliavacca et al. 2025). Furthermore, biodiverse natural and semi-natural forests have become rare both at the global level (Watson et al. 2018) and in Northern Europe (Östlund et al. 1997; Kuuluvainen et al. 2015). Forest management influences the levels of biodiversity by changing habitat structure, habitat availability, and abiotic circumstances (Peterken 1977; Kraus and Krumm 2013). Northern European boreal forests are particularly impacted by intensive commercial forestry (Kuuluvainen and Gauthier 2018), the pressure of which is predicted to grow further due to the transition from a fossil-fuel-based to a bio-based economy (Warman 2014; FOREST EUROPE, 2020). There is an urgent need to better reconcile the multiple demands and improve climate change mitigation and biodiversity conservation in forestry (Eyvindson et al. 2018).

Biodiversity encompasses genetic variation within species, diversity of species, and variety of ecosystems (Secretariat of the Convention on Biological Diversity, 2011). In Fennoscandia, the tradition of intensive silvicultural practices that favor conifers and rotation forestry has caused a decline in deadwood volume (Mönkkönen et al. 2022), broadleaf trees (Lindbladh et al. 2014) and old-growth forests (Kuuluvainen and Gauthier 2018), leading to diminishing species diversity – about 32% of forest-dwelling species in Finland are threatened (Hyvärinen et al. 2019). Habitat fragmentation and isolation caused by forest management further exacerbate the biodiversity loss (Gravel et al. 2016; Miller-Rushing et al. 2019). The need to both increase areas under strict conservation and to incorporate biodiversity into ecosystem management is well recognized (Convention

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on Biological Diversity, 2022). However, deciding which habitat is most valuable to be set aside based on its location, quality (Riva et al. 2024), and cost-effectiveness (Pienkowski et al. 2021) has proven to be difficult. The key issues include obtaining reliable data on biodiversity (Purvis and Hector 2000) and evaluating and interpreting the available data (Ranius et al. 2019). Obtaining comprehensive biodiversity data is time-consuming and thus expensive (Hekkala et al. 2023), which makes planning of the set-aside areas challenging at the regional levels. While there is an urgent need to develop methods that can provide reliable biodiversity data at a landscape scale, we also need methods to identify the locations where most additional climate and biodiversity benefits could be acquired with the current available data from e.g., forest models.

Forests are crucial for reaching national climate and biodiversity targets while simultaneously having high national and regional importance for the economy and employment (Lipiäinen et al. 2022; Räty et al. 2023). The conflicting targets lead to trade-offs between ecosystem services. Ecosystem services are the benefits that people obtain from nature, such as timber, nutrient cycling and clean water (Millennium Ecosystem Assessment 2005). Trade-offs between ecosystem services arise when improving one ecosystem service leads to a decline in another. To enable successful reconciliation of conflicting targets and resilient multipurpose land management, ecosystem service trade-offs and their temporal and spatial occurrence need to be recognized (Inostroza et al. 2017; Nikinmaa et al. 2023). Earlier studies have focused on assessing the trade-off between timber production and individual non-timber ecosystem services (Triviño et al. 2015; Luyssaert et al. 2018; Mazziotta et al. 2023), or on multi-objective maximization of ecosystem services (Eyvindson et al. 2018; Juutinen et al. 2019; Díaz-Yáñez et al. 2021; Peura et al. 2024; Rana et al. 2024) at one temporal scale. Therefore, despite a vast literature on trade-offs in forestry, there is a lack of studies tackling how trade-offs evolve in time (cf. Juutinen et al. (2020) for weighting time periods instead of setting different time spans). Understanding how time influences the impacts of target setting is important for long-term decision-making. Applying alternative time horizons and interest rates (López-Díaz et al. 2018) provides a fresh angle in assessing the trade-offs in forest management.

Here we examine the optimal targeting of forest management in landscapes inside the pre-selected biodiversity hotspots. We perform landscape-level optimization of forest management regimes, including a wide range of options from set-aside to intensive management in Southern and Northern Finland. The optimization method allows assessing the trade-offs between financial performance (expressed as net present value, NPV, € ha^{-1}), deadwood volume (a proxy for biodiversity, see e.g., Jonsson et al. 2010 and Hekkala et al. 2023, m^3 ha^{-1}) and average carbon storage of living trees (derived from living tree biomass, tCO_2 ha^{-1}). Our focus is on additionality: how much it costs to produce additional deadwood and carbon beyond the baseline economic maximum. Furthermore, the effects of time horizon on the trade-offs will be explored by running the computations over a 30- and 50-year horizon. To our knowledge, such analysis on the cost-effectiveness of producing additional deadwood volume and carbon storage that cover alternative time horizons has not been carried out before. We furthermore analyze the impact of interest rates on the trade-offs. The objective of this study is to identify the biodiversity hotspots where altering forest management and creating more set-aside areas will cost-effectively meet the target levels for deadwood volume and carbon storage in living trees.

In this study, we aim to answer the following questions:

- Do better growth conditions indicate better cost-effectiveness of producing additional deadwood and carbon?
- How do the cost-effective solutions to produce deadwood and increase carbon storage of living trees differ depending on the chosen time horizon?

Material and methods

This research consists of four phases: (i) producing the forest stand data for the biodiversity hotspots, (ii) forecasting the forest development for the next 30 and 50 years, (iii) optimizing forest management for each biodiversity hotspot, and (iv) ranking the cost-effectiveness of biodiversity hotspots to produce additional carbon storage and deadwood (Figure 1).

Study area

We utilized data from two administrative regions in Finland (Figure 2). The chosen regions represent (i) southern boreal forests with relatively high productivity in Southern Finland (Pirkanmaa region) with a mean growing stock volume of 163 m^3 ha^{-1} on forest land and a mean annual increment of 7.0 m^3 ha^{-1} a^{-1}, and (ii) central and northern boreal forests with lower-productivity forest in Northern Finland (Northern Ostrobothnia region) with a mean growing stock volume of

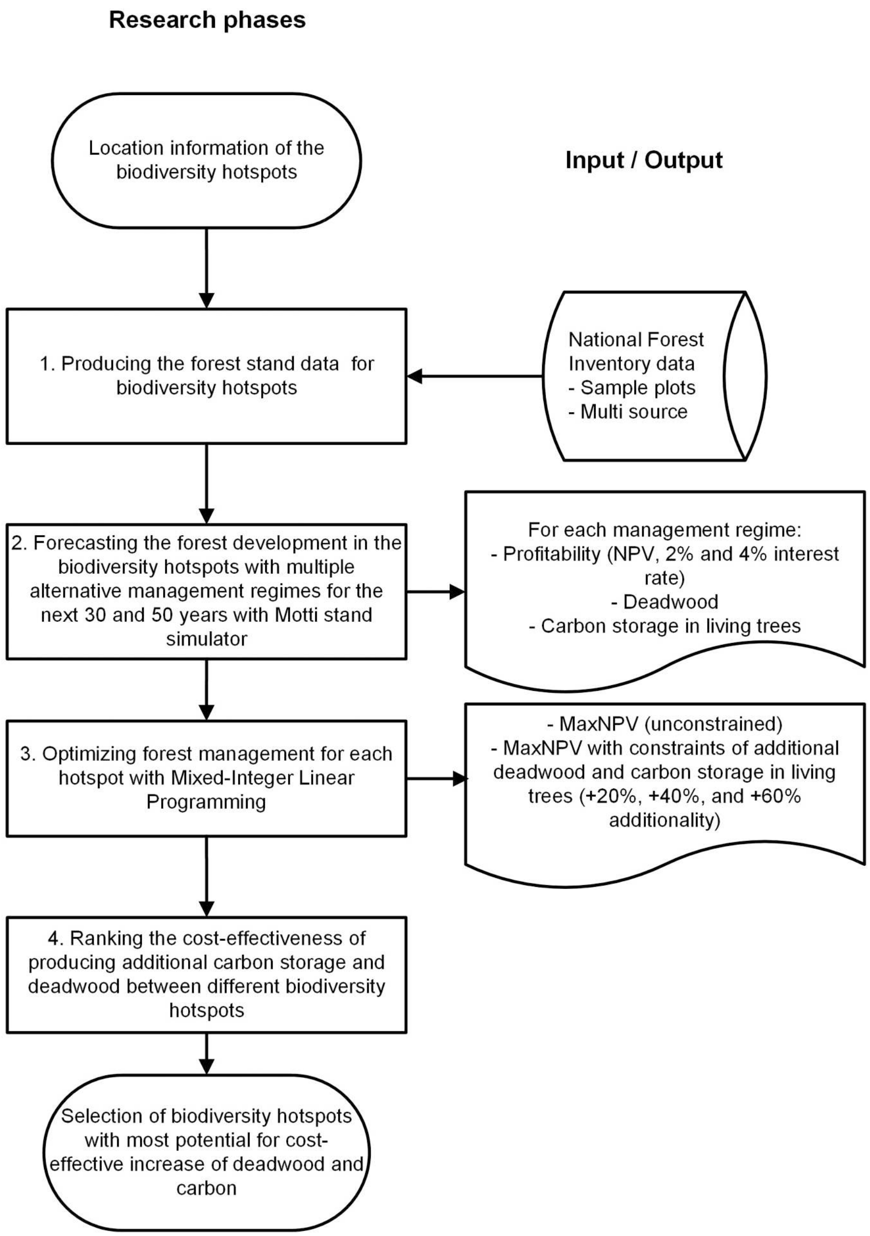

Figure 1. The flowchart of the different analysis phases in the methods. The terms are explained in the text in more detail.

105 m^3 ha^{-1} and a mean annual increment of 4.1 m^3 ha^{-1} a^{-1} (Forest Resources – statistics discontinued [web publication]).

Previous to this study, the biodiversity hotspots outside established nature protection areas in both regions were selected by the Economic Development Centres (previously ELY Centres) (Economic Development Centre, 2026) in collaboration with the local stakeholders and the Finnish Environment Institute (Finnish Environmental Institute (Syke), 2026). The selection of the hotspots was based on the existing information about the regional biodiversity (e.g. Kontula et al. 2021; The Council of Tampere Region 2023), which was updated with the most recent data on species and habitats. The individual hotspots in these regions cover large areas, with the smallest being 15 176 ha and the largest 81 316 ha. A total of 10 (5 from each region) were selected for this analysis. The initial state of forests in these 10 biodiversity hotspots is shown in Table 1. The distribution of different site types and initial volume of forest for the two regions are shown in Table 2.

Producing forest data for biodiversity hotspots

The forest parameters in the biodiversity hotspots were assessed based on two main forest resource datasets: the National Forest Inventory (NFI) (Korhonen et al.

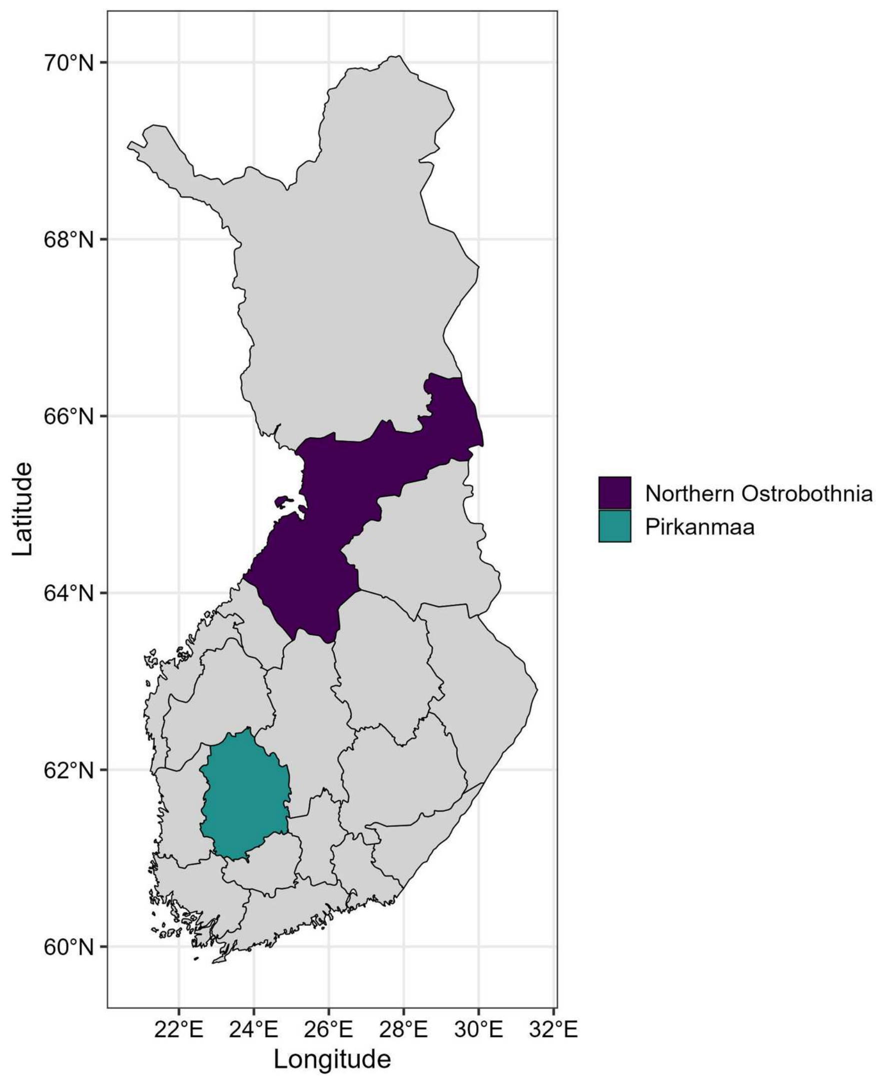

Figure 2. Study regions Pirkanmaa (green) and Northern Ostrobothnia (purple) in Finland.

2024) sample-plot data and the Multi-source National Forest Inventory (MSNFI) (Mäkisara et al. 2022) thematic forest maps in raster format with a pixel size of 16 m. To obtain a representative description of the current status of the forests in the hotspots, we developed a novel method to produce forest characteristics for each hotspot (Figure 3). The purpose of the method was to enable NFI sample plots to be used as calculation units instead of MSNFI map pixels. The small size and high number of pixels would have made them unfeasible as forest development simulation units. Therefore, we utilized information from the MSNFI thematic maps for calculating the area weights represented by each NFI plot within the hotspot area data sources (Tuominen et al. Forthcoming) . The method was based on the k-Nearest Neighbor (k-NN) estimation method

Table 1. The characteristics of the 10 biodiversity hotspots.

Region	Biodiversity hotspot	Area (ha)	Share of low productive forest (%)	Share of peatland (%)	Mean stand volume (m^3 ha^{-1})	Mean stand age (yr)	Mean basal area (m^2 ha^{-1})
Pirkanmaa	0	15 996	0.1	4.3	168	47	19
	1	15 176	0.8	14.5	138	50	17
	2	36 318	0.2	6.4	160	44	18
	3	33 066	0.4	9.3	165	51	19
	4	24 532	0.9	15.3	141	54	17
Northern Ostrobothnia	0	18 038	12.9	56.8	70	55	11
	1	10 696	11.7	51.7	67	53	11
	2	26 472	10.4	23.5	79	65	13
	3	81 316	8.0	41.2	85	54	13
	4	16 115	8.0	41.8	74	53	12

Table 2. The share of site types of mineral soil (Cajander 1949) and peatland (Laine et al. 2012) and initial total volume of growing stock in the selected hotspots in Pirkanmaa and Northern Ostrobothnia.

		Pirkanmaa		Northern Ostrobothnia	
Main soil type	Site type	Area (%)	Volume (%)	Area (%)	Volume (%)
Mineral soil	Herb-rich forest	3.8	3.6	0.1	0.1
Mineral soil	Herb-rich and mesic heath forest	69.4	75.0	28.9	41.5
Mineral soil	Sub-xeric and xeric heath forest	17.0	13.0	30.1	26.7
Mineral soil	Barren heath forest	0.0	0.0	0.0	0.0
Mineral soil	Rocky terrain	0.1	0.1	0.1	0.0
Peatland	Herb-rich type of drained peatland	0.1	0.1	0.1	0.0
Peatland	Herb-rich and *Vaccinium myrtillus* type of drained peatland	4.5	4.8	9.1	6.5
Peatland	*Vaccinium vitis-idaea* and dwarf shrub type of drained peatland	4.9	3.4	30.4	25.0
Peatland	*Cladonia* type of drained peatland	0.2	0.0	1.1	0.2

(Cover and Hart 1967; Kilkki and Päivinen 1987) and on a combination of MSNFI thematic forest map variables and measured forest variables of the NFI sample plots.

The calculation process was carried out as follows:

(1) MSNFI data in the form of a 16 × 16 m grid were obtained from each biodiversity hotspot. Each pixel represented an area of 256 m^2.
(2) For each pixel, the most similar NFI sample plot within a geographic range of 200 km was identified using the k-NN method. The similarity, i.e. nearness, was based on the volumes of the main tree species, the mean diameter of the stand, and the forest site type.
(3) The area represented by any NFI sample plot in the hotspot was determined by the number of pixels for which the plot was the nearest neighbor.
(4) Pixels with the same NFI plot as their nearest neighbor constituted a virtual stand, the area and location of which were defined by pixels. The forest attributes for the virtual stand were defined according to the measured variables of the NFI plot.
(5) The stand development simulations (see section 2.3) were carried out for the NFI sample plots, and the results were used for the corresponding pixels forming the virtual stands.

Forecasting the development of the biodiversity hotspot forests

We simulated alternative forest management regimes for the NFI sample-plot stands representing forests in the biodiversity hotspot with the Motti stand simulator. Motti is a stand-level decision-support tool for assessing the effects of forest management on stand dynamics (Salminen et al. 2005; Hynynen et al. 2015; Lee et al. 2026) and, e.g. carbon sequestration, harvest revenues and costs. The simulator covers the development of growing stock and the accumulation and decay of deadwood. Deadwood originates from natural mortality, harvest residues, and high stumps left in cuttings. Predictions of natural mortality are based on mean probabilities of trees surviving, excluding events regarded as stochastic by nature, such as storms and insect outbreaks. The decay of deadwood decreases the total accumulated deadwood volume. The models by Mäkinen et al. (2006) predict the remaining relative mass of dead trees and the probability of snags falling. Decaying functions depend on factors such as tree species, stem diameter and type (snag or fallen), and the time elapsed since tree death. A detailed description of the framework and growth models incorporated into Motti are presented in supplementary data in Lee et al. (2026). Here we used Motti to simulate the future stand attributes from present to 30 and 50 years. A typical strategic planning horizon is approximately 30 years (Mazziotta et al. 2025), and the 50-year time horizon was chosen to correspond to a recent study tackling wood production, carbon sequestration, and the produced deadwood volume under alternative management regimes in Fennoscandia (Hynynen et al. 2025).

The alternative management regimes were based on predefined chains of silvicultural treatments and harvesting for different stand-characteristics groups (i.e. dominant tree species, site type, and geographic location). To cover a wide range of management options for landscape-level optimization, there were up to 99 and at least four alternative forest management regimes for each stand, each of which resulted in alternative harvest removals. The simulated regimes were mainly based on rotation forestry, RF (i.e. even-aged management) and thinnings were conducted from below, but some options also represent continuous cover forestry (CCF), depending on the stand-characteristics group. A set-aside regime without any treatment was also included. We altered timings of the thinnings (expressed as a dominant height), intensity of the thinnings (expressed as a decrease in basal area), and timing for clear cutting (expressed by stand mean

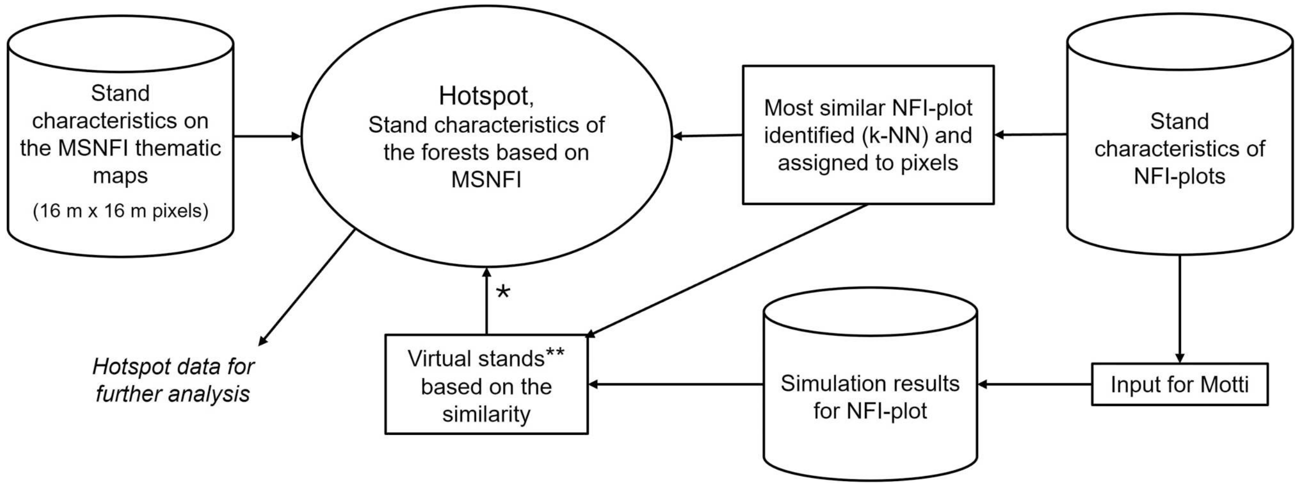

Figure 3. Overview of the method used to produce forest characteristics and forecasts of their further development for the biodiversity hotspot. * Simulation results transferred to all pixels of the hotspot based on k-NN. ** Virtual stand = MSNFI-pixels representing the same NFI-plot.

diameter) for each stand: each one was simulated according to altered values for the timings of thinnings, timing of clear cutting, and the choice of regeneration method (artificial or natural), resulting in numerous management regimes for that particular stand (for a detailed description of identical procedure, see Haikarainen et al. (2021)). For a similar approach, see Ahtikoski et al. (2024).

For each stand and each alternative management regime, the Net Present Value, NPV, was assessed according to the following equation:

$$NPV_s = \sum_{i=0}^{T} b^{t_i}\left[\sum_{k=1}^{K} p_k h_{ki}\right] - \sum_{l=1}^{L} w_l b^{t_l},$$

where NPV_s = net present value for stand s, $i = 0, \ldots, T$ (with t_0 and t_T denoting the beginning and the end of the time horizon at 5-year time steps, respectively); k denotes timber assortments ($k = 1, \ldots, K$); and p_k represents the stumpage price (€ m^{-3}) of each timber assortment k (note that there are different tree species). Let b be the discount factor, defined as $b = 1/(1+r)$, where r is the annual interest rate in real terms (here 2%). The cost of a silvicultural measure l is w_l, € ha^{-1} (note that L refers to the last silvicultural measure within a time horizon). The removal of each timber assortment k at year t_i is denoted by h_{ki}, expressed in m^3. The NPV of the hotspot was then transformed to per-hectare values by dividing the total NPV of the hotspot by the area of the hotspot. For choosing the time horizon and NPV, see Supplementary Material 1. For NPV calculations, we applied a 2% interest rate. In addition, a 4% interest rate was applied in the sensitivity analysis. These interest rates can be interpreted to represent a lower and upper bound for long-term investments in forestry, respectively (see, e.g. Weitzman (2010) for theoretical background and Price (2018) for empirical evidence). The stumpage prices and silvicultural costs underlying the NPVs were based on a time series of ten years (2014–2023) (Official Statistics Finland (OSF): Volumes and prices in industrial roundwood trade [web publication]). The nominal prices and costs were deflated according to the cost-of-living index (Official Statistics of Finland (OSF): Consumer price index [web publication]) to convert them into real terms. The stumpage prices and silvicultural costs in real terms are presented in Supplementary Material 2.

Optimizing forest management for each biodiversity hotspot

To analyze the trade-offs between timber production, carbon storage and deadwood, we focused on the cost of producing additional carbon storage in living trees and deadwood in comparison to the economic maximum. We conducted a mixed-integer linear programming task for each biodiversity hotspot as follows. First, NPV was maximized without any constraints on deadwood and carbon storage (unconstrained maxNPV) to have the economic maximum as a baseline (Luptácik 2010). Second, we optimized the additional carbon storage and deadwood volume simultaneously in comparison to the amounts of the two in the economically optimal solution by minimizing the loss of NPV with the carbon storage and deadwood volume set as constraints. Since the initial stand structures (and therefore the average carbon storage and deadwood volume in absolute terms) fluctuated between biodiversity hotspots, we applied specific additionality levels instead of absolute values to allow comparison between the hotspots. The additionality levels applied were +20%, +40%, and +60% for both carbon storage in living trees and the deadwood volume in comparison to the baseline solution (output of Figure 1, phase 3).

The mathematical formulation of the optimization problem can here be described as follows:

$$Maximize_A(\, f_i(x))$$

subject to:

$$x \in X$$

where A represents a hotspot area, $f_i(x)$ is the objective function, and X is the set of alternative management regimes generated by the Motti stand simulator. Here, the objective function is NPV (€) for all stands within a biodiversity hotspot. The value of the objective function depends on x, i.e. the management regime applied. Then, let $s = 1, 2, \ldots, m$ denote the index of forest stands and $r = 1, 2, \ldots, n$ the index of forest management regimes. The decision variables x_{sr} are binary variables belonging to {0,1}. It should be noted that the above-mentioned equation is presented as an unbounded notation, i.e. without any constraints. This approach differs from, e.g. the ε-constraint method, which is usually applied to identify Pareto optimal solutions (see, e.g. Miettinen 1998; Mazziotta et al. 2025). In this study, only one objective function was solved at a time in an unconstrained form. After that solution (baseline), MILPs with constraints according to the additionality levels of deadwood and carbon storage were executed. The set of feasible solutions to maximize the objective function was defined by:

$$X = \left\{ x = (x_{s,r})_{m*n} \in \{0, 1\}^{m*n};\ \sum_{r=1}^{n} x_{sr} = 1, \quad \textit{for each } s = 1, \ldots, m \right\}$$

In other words, each feasible solution is a management plan where one particular management regime is chosen for each forest stand (see, e.g. Eyvindson et al. (2024) for technical implementation). Within this optimization framework, trade-offs between (i) NPV and deadwood, and (ii) NPV and carbon storage could be quantified so that, in principle, the biodiversity hotspots could be ranked according to the cost-effectiveness of producing deadwood and increasing the carbon storage.

Technically, optimizations were conducted with Python 3.13. We applied mixed-integer linear programming with PuLP (Mitchell et al. 2011) and Coin-or Branch and Cut, CBC (Forrest et al. 2024). The code is presented in Supplementary Material 3.

Ranking the regional biodiversity hotspots according to their cost-effectiveness

We constructed a cost-effectiveness ranking system between biodiversity hotspots in Pirkanmaa and in Northern Ostrobothnia as follows. First, we applied min–max normalization separately for additional carbon storage and deadwood according to

$$x'_{ip} = a + \frac{(x_{ip} - \min(X_{ip}))(b - a)}{\max(X_{ip}) - min(X_{ip})},$$

where x'_{ip} is the normalized value within the range [a, b], x_{ip} is the original value for i (either deadwood or carbon storage) according to the p optimal solution (+20% optimal solution, +40% optimal solution or +60% optimal solution), min (X_{ip}) is the smallest value for X_{ip} in the dataset, max (X_{ip}) is the largest value for x_{ip} in the dataset. To avoid 0 coefficients (a case where an optimal solution presents the smallest value for both deadwood and carbon storage), a and b were set to 1 and 2, respectively.

The next step was to use the normalized values to modify the original NPV losses associated with the optimal solutions. Since the normalized values were calculated separately for carbon storage and deadwood, calculating an average of the normalized values would reflect both variables. The cost-effectiveness index was calculated as follows:

$$CE_v = \frac{\Delta NPV_{vp}}{x'_{vp}},$$

where $CE_v =$ cost-effectiveness metric for biodiversity hotspot v, $\Delta NPV_{vp} =$ NPV loss associated with p optimum solution in biodiversity hotspot v, x'_{vp} is the average of normalized values (X'_{ip}) of carbon storage and deadwood according to the optimal solution p in biodiversity hotspot v. The best performer, i.e. the most cost-effective biodiversity hotspot, has the lowest cost-effectiveness value.

Results

The analysis shows that the chosen time horizon, region and additionality level affect the ranking of the cost-effectiveness of biodiversity hotspots. In both regions, the variation in the cost-effectiveness order was sensitive to additionality level (+20%, +40%, and +60%) only in one of the chosen time horizons. For example, in Pirkanmaa, the rank order did not change with increasing additionality in the 30-year time horizon, but in the 50-year time horizon at least two hotspots changed ranks with the increasing additionality levels (Table 3). In Northern Ostrobothnia, the situation is

Table 3. Cost-effectiveness rankings between biodiversity hotspots (hotspot0 – hotspot4) in Pirkanmaa. The cells with normal font show the results for the 30-year time horizon, and the cells with italics show the results for the 50-year time horizon.

Additionality*	Best	Second	Third	Fourth	Last
+20%	hotspot2	hotspot0	hotspot3	hotspot1	hotspot4
	hotspot0	*hotspot2*	*hotspot4*	*hotspot1*	*hotspot3*
+40%	hotspot2	hotspot0	hotspot3	hotspot1	hotspot4
	hotspot0	*hotspot2*	*hotspot1*	*hotspot4*	*hotspot3*
+60%	hotspot2	hotspot0	hotspot3	hotspot1	hotspot4
	hotspot2	*hotspot3*	*hotspot1*	*hotspot4*	**

*additional proportion, % (simultaneously, carbon storage and amount of deadwood) compared to the values corresponding to MaxNPV. ** With a 50-yr time horizon and 2% interest rate in hotspot0 +60% additionality resulted in an infeasible optimal solution, indicating that carbon storage and the amount of deadwood cannot be simultaneously increased by 60% (compared to the MaxNPV solution).

reversed (Table 4). In both regions, some hotspots have constantly better cost-effectiveness: e.g. in Pirkanmaa hotspot0 and hotspot2 alternate between the first and second rank, and in Northern Ostrobothnia the situation is similar with hotspot3 and hotspot4.

The chosen time horizon influenced the absolute values of the cost-effectiveness index greatly by decreasing (i.e. becoming cheaper) for the same level of additionality when moving from the 30-year to 50-year time horizon (Figure 4). The cost-effectiveness difference between the hotspots tended to increase with increasing additionality and was, in general, higher in Northern Ostrobothnia than in Pirkanmaa for the 30-year time horizon (Figure 4). In Northern Ostrobothnia, the hotspots in which the highest relative harvest, i.e. the harvested amount of merchantable wood (total m^3 during the time horizon) in the additionality scenario divided by the harvested amount of merchantable wood in the economic optimum, had the best cost-effectiveness. This relationship was less clear in Pirkanmaa, where the relationship between the amount of harvested merchantable wood for each additionality level and the economic optimum was more similar between the hotspots.

Notably, there is a great difference in the possibility to produce additional carbon storage and deadwood between the regions. The optimal solution is achieved when the additionality can be (i) economically (i.e., no negative NPV values) or ecologically (i.e., the additional amount of carbon storage or deadwood is within the simulation results) achieved. In Pirkanmaa, in 30 years, it was possible to achieve +60% additionality with less than 60% reduction in the amount of harvested timber, whereas in Northern Ostrobothnia, there was no optimal solution for three hotspots and for two it was achieved with more than 80% reduction in the amount of harvested timber. The results are very different for the 50-year time horizon: in both regions, the difference in cost-effectiveness between the additionality levels decreases notably. Furthermore, the two regions have similar cost-effectiveness for the same level of additionality. The corresponding development of mean stand volume and harvesting amounts are shown in Supplementary Material 4.

The costs (i.e., NPV losses compared to economic optimum) associated with alternative additionality levels (+20%, +40%, and +60%) are presented in Figure 5. Due to the differences in the initial stand structures between biodiversity hotspots, the same additionality level resulted in slightly different absolute values of additional carbon storage (tCO_2 ha^{-1}) and additional deadwood volume (m^3 ha^{-1}) in different biodiversity hotspots (Figure 5). Nevertheless, the differences between the costs of producing additional carbon storage and additional deadwood volume when comparing the two regions are clearly visible (Figure 5(a-b vs. c-d)). For instance, for the 30-year time horizon and interest rate of 2%, the costs of producing +20%

Table 4. Cost-effectiveness rankings between biodiversity hotspots (hotspot0 – hotspot4) in Northern Ostrobothnia. The cells with normal font show the results for a 30-year time horizon, and the cells with italics show the results for a 50-year time horizon.

Additionality*	Best	Second	Third	Fourth	Last
+20%	hotspot3***	hotspot4	hotspot0	hotspot1	hotspot2
	hotspot4	*hotspot0*	*hotspot3*	*hotspot1*	*hotspot2*
+40%	hotspot3	hotspot4	hotspot0	hotspot2	hotspot1
	hotspot4	*hotspot0*	*hotspot3*	*hotspot1*	*hotspot2*
+60%	hotspot3	hotspot2	**	**	**
	hotspot4	*hotspot0*	*hotspot3*	*hotspot1*	*hotspot2*

*additional proportion, % (simultaneously, carbon storage and amount of deadwood) compared to the values corresponding to MaxNPV, ** With a 30-yr time horizon and a 2% interest rate in hotspot0 +60% additionality resulted in an infeasible optimal solutions, indicating that carbon storage and the amount of deadwood cannot be simultaneously increased by 60% (compared to the MaxNPV solution).

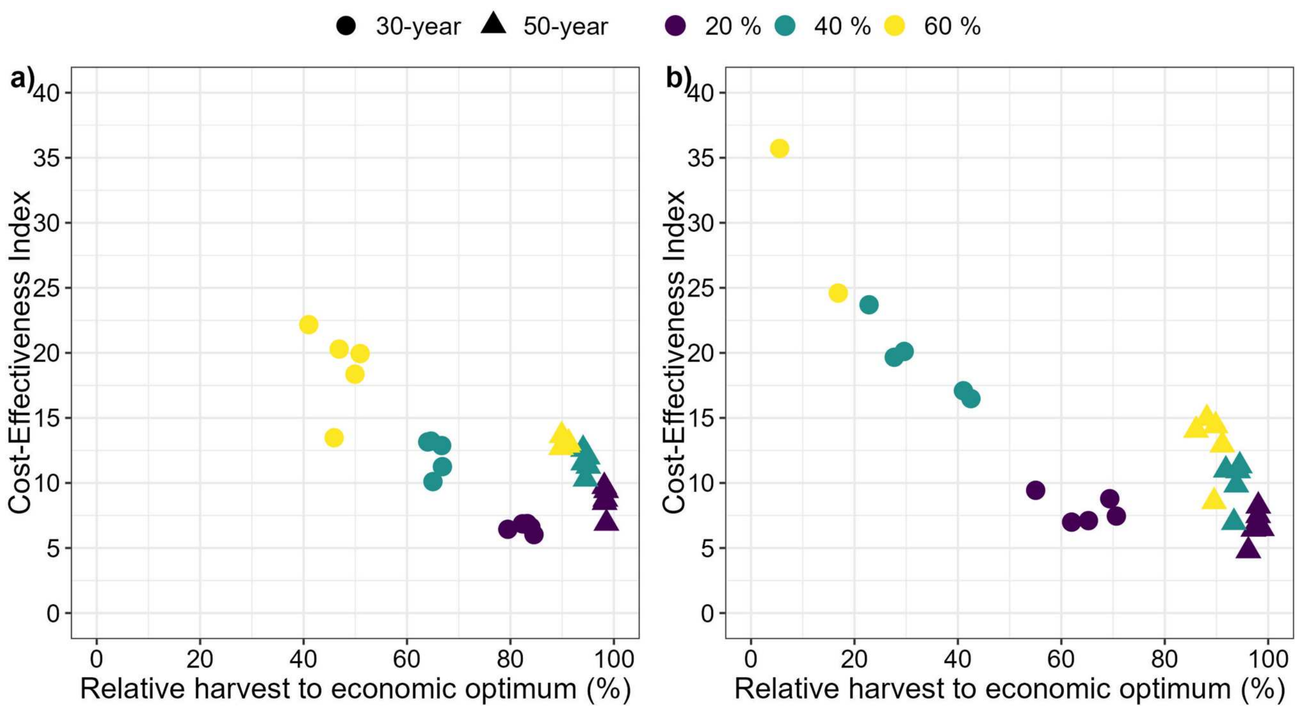

Figure 4. The effect of additionality on relative harvest and cost-effectiveness in (a) Pirkanmaa and (b) Northern Ostrobothnia for the 30-year (circle) and 50-year (triangle) time horizons at a 2% interest rate. The x-axis shows the relative harvest of the additionality scenario to the economic optimum. The y-axis shows the cost-effectiveness index. The colors show the levels of additionality.

additional carbon storage and deadwood volume ranged from 272.3 to −375.5 € ha^{-1} in Pirkanmaa (Figure 5(a)). The additional carbon storage ranged from 36.0 to −49.2 tCO_2 ha^{-1} and the additional deadwood volume from 3.6 to −5.6 m^3 ha^{-1}. The corresponding costs in Northern Ostrobothnia were between 224.6 and 315.4 € ha^{-1}, and the range of additional carbon storage was from 29.7 to −31.0 tCO_2 ha^{-1} and the range of additional deadwood was from 2.0 to −2.4 m^3 ha^{-1} (Figure 5(c)). Further, with a 50-year time horizon and 2% interest rate, the costs of +40% additional carbon storage and deadwood volume fluctuated between 315.4 and 516.6 € ha^{-1} in Pirkanmaa, with the additional carbon storage ranging from 18.5 to −36.8 tCO_2 ha^{-1} and the additional deadwood volume from 8.8 to −13.3 m^3 ha^{-1}. (Figure 5(b)), whereas in Northern Ostrobothnia, the corresponding costs were from 140.1 to −184.7 € ha^{-1} and the additional carbon from 8.2 to −9.5 tCO_2 ha^{-1} and the additional deadwood from 6.1 to −6.8 m^3 ha^{-1} (Figure 5(d)). The general trend of Pirkanmaa having higher absolute values of additional carbon and deadwood is due to the differences in stand structures between the regions, namely, the Pirkanmaa region possessing higher stand volumes at the onset of the simulations.

In optimal solutions, the shares (%) of the three forest management regimes (RF, CCF and set-aside) varied considerably across regions, additionality levels, and for Northern Ostrobothnia also between 30-year and 50-year time horizons. In principle, the biggest change was between economic optimums and +60% additionality levels. For Pirkanmaa, on the 30-year time horizon, the share of RF ranged from 95.2% to 62.9%, the share of CCF from 19.1% to 2.3% and the share of set-aside from 34.1% to 0.1%. The range was smaller on the 50-year time horizon: the share of RF ranged from 95.3% to 74.9%, the share of CCF from 19.2% to 1.0% and the share of set-aside from 13.2% to 0.1%. For Northern Ostrobothnia, on the 30-year time horizon, the share of RF ranged from 86.2% to 34.2%, the share of CCF from 14.3% to 4.2%, and the share of set-aside from 61.6% to 2.8%. In Northern Ostrobothnia, the 50-year time horizon decreased the ranges more than in Pirkanmaa: the share of RF ranged from 86.5% to 77.8%, the share of CCF from 14.6% to 2.7%, and the share of set-aside from 16.1% to 2.7%. A comparison of how forest management changes between the unconstrained solution and the +40% additionality optimum at 30-year and 50-year time horizons is shown in Figure 6. In Pirkanmaa, the share of set-aside areas increased substantially under the +40% additionality level compared to the unconstrained solution both at 30-year and 50-year time horizons, but the increase was smaller for the 50-year time horizon, and both were clearly lower than in Northern Ostrobothnia (Figure 6(a,b)). In Northern Ostrobothnia,

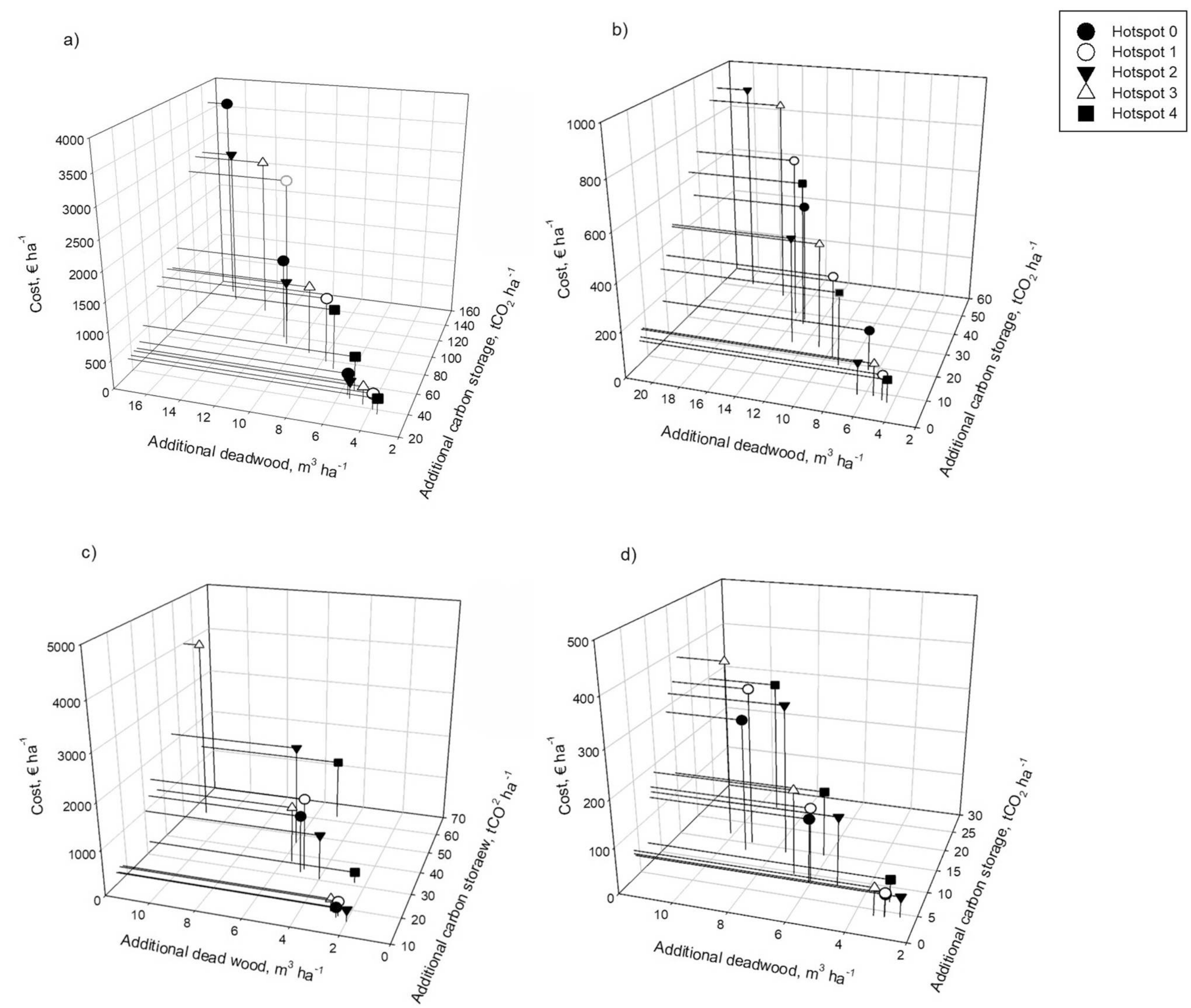

Figure 5. A 3D graph on the costs associated with simultaneous additional deadwood and carbon storage compared to the MaxNPV, € ha^{-1} for (a) Pirkanmaa region, time horizon 30 yrs, interest rate 2%, (b) Pirkanmaa region, time horizon 50 yrs, interest rate 2%, (c) Northern Ostrobothnia region, time horizon 30 yrs, interest rate 2%, and (d) Northern Ostrobothnia region, time horizon 50 yrs, interest rate 2%. Additionality levels were +20%, +40%, and +60%. Note the different scales of the axes.

the share of set-aside areas rose to over +30% at +40% additionality at the 30-year time horizon and showed a greater variability among hotspots than in Pirkanmaa (Figure 6(c)). The increase in set-aside areas was much smaller at the 50-year time horizon (Figure 6(d)). The additional carbon storage and deadwood were primarily achieved by increasing the set-aside area, typically at the expense of both RF and CCF.

Our analysis of using 2% and 4% interest rates in the optimization problem resulted in a somewhat conflicting outcome: it was more cost-effective to increase the carbon storage and deadwood volume with a 4% interest rate rather than with 2% (Table 5).

This contradictory result can be explained as follows. Optimizing with 4% results in higher harvesting removals than with 2% during the first half of the time horizon, i.e., within 15 to 25 years depending on the time horizon applied (30 or 50 years). For example, in the Pirkanmaa region (hotspot 1 and hotspot 4), average annual harvesting removal according to optimization with a 4% interest rate was over 30 m^3 ha^{-1} higher than the harvesting removal with a 2% interest rate within the first two decades (results not shown). Furthermore, the value (+30 m^3 ha^{-1}) was mainly due to increased final harvesting associated with 4%. This, in turn, creates conditions where clearcut stands have more time to grow towards the end of the time horizon. Therefore, with a 4% interest rate, there is timewise more potential to produce additional deadwood and carbon storage, compared to optimization with

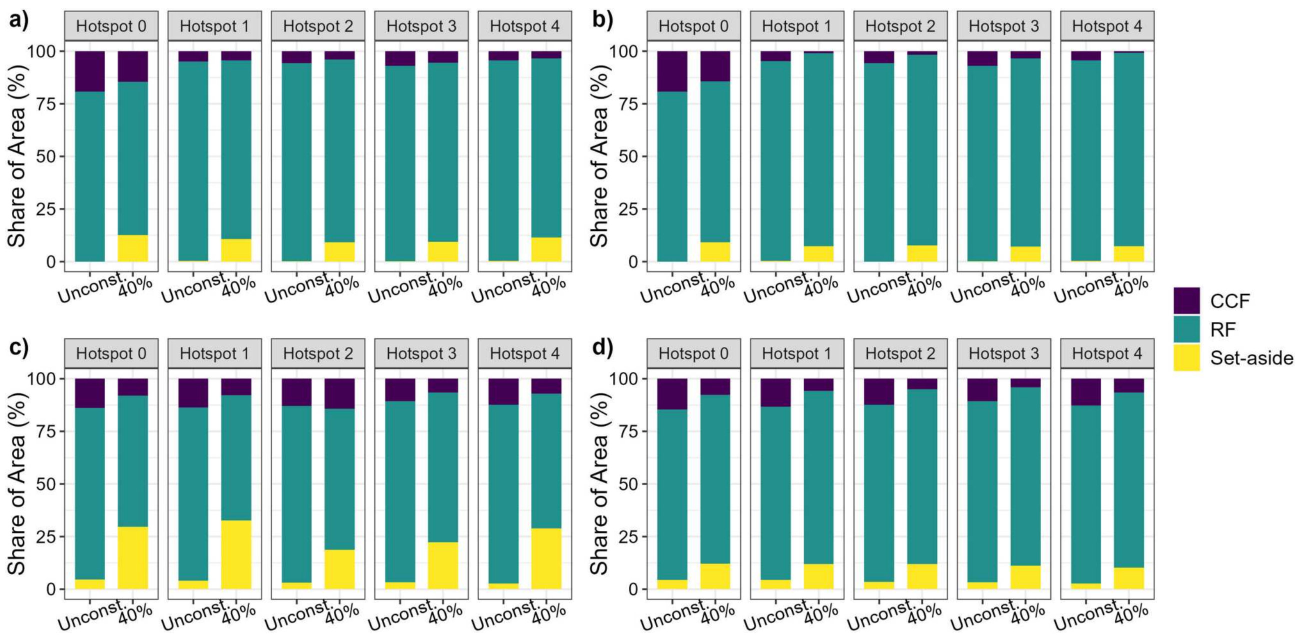

Figure 6. The share of area under a specific forest management regime (RF, CCF and set aside) in Pirkanmaa at (a) 30-year and (b) 50-year time horizon, and in Northern Ostrobothnia at (c) 30-year and (d) 50-year time horizon describing the unconstrained and +40% additionality solutions. The x-axis shows the results for the unconstrained and +40% additionality of carbon storage and deadwood in each biodiversity hotspot, and the y-axis shows the percentage of the area under continuous cover forestry, rotation forestry and set aside.

2%. At the same time, the absolute economic values (i.e. NPV expressed in euros) representing the optimal solution with 4% are considerably lower than the corresponding values with 2%. These two effects (longer time for established stands to develop after clear-cut and drastically lower net present values in absolute terms) associated with a 4% interest rate bring about outcomes where it tends to be more cost-effective (i.e., less costs for identical additional environmental benefits) to apply 4% than 2% interest rate.

Discussion

The need to sustainably provide several ecosystem services in an increasingly uncertain future requires new perspectives and better capacity to use existing information in decision-making (Polasky et al. 2011). Here, we provided a novel approach for forest management optimization at different time horizons and locations to illustrate the trade-offs between NPV, carbon storage in living trees, and deadwood volume.

Our results show that the considerable differences in general growing conditions and productivity between the forests of Pirkanmaa and Northern Ostrobothnia influence the capacity of the selected biodiversity hotspots to produce additional carbon storage and deadwood cost-effectively. In the short term, the capacity is clearly better in Pirkanmaa than in Northern Ostrobothnia. The growing conditions affect the options to alter forest management: higher growth rates in better site types in Pirkanmaa provide a broader space of options to achieve the different additionality levels, whereas the less fertile site types and shorter growing season in Northern Ostrobothnia limit the options of achieving the higher additionality levels. Both carbon stocks and deadwood are produced the most in the short term when forests are unmanaged (Alrahahleh et al. 2017; Petty et al. 2024). Therefore, in Northern Ostrobothnia, more set-aside areas are needed to achieve the same level of additionality than in Pirkanmaa, which makes it less cost-effective.

The time horizon set to reach the targets of additional carbon storage and deadwood matters. The better capacity to produce additionality can be seen especially at the 30-year time horizon, whereas it decreases when examined at the 50-year time horizon (Figure 4). The significant decrease in costs of producing additional carbon storage and deadwood when moving from the 30-year to 50-year time horizon is explained twofold. First, the space of options is broader: more time, in general, allows for more combinations of different management interventions, which provide more opportunities to produce carbon storage or deadwood. Second, the initial stand structures influenced the timing of interventions. In both regions, the final-cutting maturity of most hotspots' stands was not reached yet at 30 years but was

Table 5. The results of a sensitivity analysis of cost-effectiveness using 2% and 4% interest rates for different additionality levels of biodiversity hotspots in Pirkanmaa.

Additionality level	Interest rate	Mean cost-effectiveness index	Standard deviation	Min	Max
+20%	2%	213.71	39.77	169.71	272.24
	4%	73.03	14.70	54.40	91.60
+40%	2%	835.78	173.31	633.47	1067.19
	4%	267.14	53.59	200.02	332.17
+60%	2%	1972.45	336.75	1550.67	2425.22
	4%	577.7	114.17	425.65	719.02

at 50 years (Supplementary Material 4). Due to the initial stand structure, the harvesting possibilities (cuttings simulated according to the management guidelines) were relatively low for 30 years, but clearly higher in the late part of the 50-year time horizon (Supplementary Material 4). This, combined with the NPV-maximizing optimization that tends to lead to harvesting and clear-cutting earlier (Amacher et al. 2009), results in a lower level of stand stocking at the end of the period. This indicates that any additionality at the 30-year time horizon would require setting aside forests close to final-cutting maturity, which would result in a high loss of NPV. At the 50-year time horizon, however, there will be a high share of young forests that have inherently smaller carbon storage than older forests (Harmon et al. 1990; Luyssaert et al. 2008) and have less deadwood (Siitonen 2001). The additional carbon storage and deadwood would then require setting aside younger forests, which is less expensive as the trees are not yet economically so valuable, and less in absolute terms as the additionality is always counted in relation to the economic maximum of that time point. This highlights the importance of time in cost-effectiveness analysis, as longer time horizons show a more complete picture of the long-term consequences of decisions (O'Mahony 2021). An earlier study in boreal forests (Mazziotta et al. 2016) demonstrated that a cost-effective strategy for forest conservation varies with the budget limit, discount rate (under economic and ecological time) and the time horizon applied (Mazziotta et al. 2016). Further, time horizon plays a crucial role in adaptive forest management against natural disturbances (Mazziotta et al. 2026).

When comparing the hotspots within a region, the sensitivity of cost-effectiveness ranking to the level of additionality shows that the level of additionality can influence the cost-effectiveness ranking. These findings are supported by Juutinen et al. (2008), who found that the level of biodiversity indicator targets matters in reaching the conservation goals under budget constraints, as the availability of suitable sites might be insufficient for achieving the targets. Therefore, an examination of the ranges of target levels and their influence on the cost-effectiveness at different time horizons is an important consideration in nature conservation target setting, e.g. at national scale, when budget allocations for regions are decided.

Recently, many studies have analyzed the trade-offs between different objectives in forest management (Triviño et al. 2015; Díaz-Yáñez et al. 2021; Rana et al. 2024). The overarching consensus is that diversifying forest management from the current RF regime is the optimal way to reach the objectives with limited economic losses. However, these optimizations were done mostly for a 100-year period (except Triviño et al. 2015, who used a 50-year period), which indicates that the time to reach the objective levels of desired ecosystem services is considerable. Our results indicate that, in the relatively short term, the increase of both carbon storage in living trees and the deadwood volume is mainly achieved by increasing the share of set-aside areas in biodiversity hotspots (Figure 6), which, especially in Northern Ostrobothnia, is a substantial share of the area of biodiversity hotspots. This insight is important for resource allocation decisions in nature conservation, as providing the decision-makers with a diverse and sufficiently comprehensive set of solutions to choose from facilitates better decisions (Hwang and Masud 1979; Navarro-Cerrillo et al. 2025). The results presented here provide decision-makers with not only more information on the level of carbon storage and deadwood reachable with different temporal scales but also on the economic implications of each choice. Furthermore, while not shown here, the results can be transferred to maps to illustrate how the additionality influences forest management spatially. Therefore, the method could help to identify forest areas important for connectivity also outside the biodiversity hotspots (Wang et al. 2025). The new ability of projecting the optimal forest management solutions into maps is an important development that has practical benefit for decision makers who plan restoration and conservation policies. Such "theme maps" could concretely demonstrate different areas to be allocated to, e.g. set aside, intensive and extensive management, as well as their cost-effectiveness.

As most of the forest considered in this study is privately owned, the increase in the additionality levels would result in a loss of income for forest owners. While the forest owners could technically be obliged to set aside forest through legislation changes, this would likely not be socially accepted. Therefore, a compensation scheme would be needed. The scheme could, e.g. take the form of a Payments for Ecosystem Services (PES) scheme in which the selection criteria would consider both biodiversity values and carbon storage (Kangas and Ollikainen 2022). The fact that the compared regions have different outcomes in cost-effectiveness indicates that homogeneous compensation for increasing an identical deadwood volume or carbon storage is neither cost–nor budget-effective and that we should rather consider output-based or differentiated compensation schemes for forest owners (Harzer and Quaas 2026). Differentiated compensation schemes could also address the issue of local benefits of conservation being inferior to national benefits (Kniivilä et al. 2002; Naidoo and Ricketts 2006): the compensation schemes could be designed to mitigate the effect of increased set-aside areas on local economies. However, the actual design and implementation of a compensation scheme would need to tackle several issues that remain beyond the scope of this article, such as a broader spectrum of biodiversity indicators, ownership structure, landowner behavior, transaction costs, compatibility with legal frameworks, and leakage potential (cf. Assmuth et al. (2024) on implementation challenges of forest carbon payment schemes). Hence, the prioritization method presented in this article cannot be the sole basis of conservation decision-making but can support it considerably.

An argument could be made that our optimization approach is a rather simple one. The rationale for the chosen approach was to create a single optimization problem with constraints representing environmental and climate benefits (carbon storage and deadwood volume). Had we chosen, e.g. assessing Pareto frontiers would have required a multi-objective optimization approach (see, e.g. Miettinen 1998; Eyvindson et al. 2023; Mazziotta et al. 2025). For the scope of this study, a single optimization problem is adequate since the focus here was to assess trade-offs between financial performance and the deadwood volume and carbon storage in living trees in different time horizons. Furthermore, simultaneous increments for deadwood and carbon storage were assumed to emphasize synergy between biodiversity and climate change mitigation (Smith et al. 2022; Assmuth et al. 2026). Had we applied different additionality levels for deadwood and carbon storage, it would have resulted in more combinations for deadwood and carbon storage values. This would have distinctly hampered the interpretations of the optimal solutions.

We do recognize that the results presented here are subject to the assumptions that (i) carbon storage is only increased in living trees and not, e.g. in soils, and (ii) the deadwood volume alone is a comprehensive indicator of biodiversity. Previous studies on especially biodiversity have shown that the cost-effectiveness of conservation depends on the chosen indicators (e.g. Juutinen and Mönkkönen 2007; Wiersma and Sleep 2018). It is well known that in boreal forests, soils can be a large carbon storage and sink, which tends to increase if the forest is not harvested (Lindroos et al. 2022). Therefore, it could be that the area needed to be set aside is an overestimation. For deadwood, we only considered the modeled deadwood volume, not e.g. the measured deadwood in a landscape, deadwood diversity or the different decay stages. Estimates of deadwood volume derived from the simulations are subject to uncertainty, which, in turn, may have influenced the cost-effectiveness index. However, as the same model was applied to all hotspots, the comparison of the hotspots is unlikely to change. Furthermore, as deadwood volume correlates with its diversity (Peura et al. 2024), on a coarse scale it can be said that higher volumes of deadwood imply a more diverse spectrum of, e.g., decay stages. However, even the most accurate estimations of deadwood volumes and properties alone are not adequate to comprehensively assess biodiversity. To have a fully comprehensive strategy for increasing carbon storage and biodiversity, this analysis could be conducted with several alternative indicators to analyze the cost-effectiveness of the strategy combined with other biodiversity mapping strategies.

It is important to note that as we focused on identifying where it would be the most cost-effective to create additional carbon storage and deadwood in comparison to the economic maximum, we did not place any value on existing carbon storage or deadwood and therefore did not create constraints that would have preserved existing valuable sites. With these results, we do not mean to indicate that current old-growth-type forests should be harvested, but rather we provide a method to identify areas where creating additional carbon storage and deadwood, when the known ecologically important sites have been protected, is cost-effective. To halt biodiversity loss, we need to increase conservation efforts for remaining forest habitats cost-effectively to reduce the potential negative impact on the economy. The results of our analysis clearly indicate that the different capacities of areas to produce different biodiversity or climate benefits should be explicitly considered when targets for these are set.

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Author contributions

CRediT: **Laura Nikinmaa:** Conceptualization, Formal analysis, Investigation, Visualization, Writing – original draft, Writing – review & editing; **Soili Haikarainen:** Investigation, Methodology, Resources, Validation, Writing – review & editing; **Sakari Tuominen:** Methodology, Resources, Software; **Mika Lehtonen:** Resources, Software; **Aino Assmuth:** Formal analysis, Writing – review & editing; **Hannu Salminen:** Resources, Software; **Anssi Ahtikoski:** Conceptualization, Formal analysis, Funding acquisition, Methodology, Supervision, Visualization, Writing – review & editing.

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