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Biodiversity implications of forest carbon payments

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Abstract

Carbon payments have been proposed as a means of incentivizing forest management that jointly optimizes timber production and carbon storage benefits. We analyze the impacts of such payment schemes on forest biodiversity using a detailed model describing Finland's forest resources, timber markets, and forest and energy industries. In the model, forest-owners' heterogenous valuations of forest amenities affect their harvesting behavior and willingness to join a carbon payment scheme. We measure forest biodiversity using an ecosystem condition index based on habitat-specific structural components. Our results indicate that forest carbon payments have a small to moderate positive effect on forest ecosystem condition. The improvements in ecosystem condition result from longer rotations and greater volumes of broadleaved trees and dead wood. The effects are weakest on low productivity sites. Under a partial-coverage carbon payment scheme, impacts on ecosystem condition vary according to forest-owner type, but are positive in aggregate. According to our sensitivity analysis, the ecosystem condition-improving effect of carbon payments is stronger under high demand for harvested wood products. Our results suggest that forest carbon payments are likely to have biodiversity co-benefits in boreal conditions, but halting biodiversity loss will require additional biodiversity-targeted policies.

Key words: biodiversity, carbon storage, forest ecosystems, sector model, timber harvesting

1. Introduction

Climate change and biodiversity loss are globally among the largest threats to ecosystem functioning and human welfare that depends on it. The deep interconnections between these twin crises are increasingly recognized, and call for simultaneous, coordinated action (Pörtner et al. 2023; IPBES 2024). Ideally, policies could be designed to harness synergies and control tradeoffs between biodiversity protection and climate change mitigation. The need for coordination is especially pressing in the forest sector, which can play a key role in climate stabilization, while also affecting large land areas with important implications for biodiversity. Carbon payments have been proposed as a means of incentivizing forest management that jointly optimizes timber production and carbon storage benefits (e.g., van Kooten et al. 1995; Sohngen and Mendelsohn 2003; Lintunen and Uusivuori 2016). The effects of carbon payment schemes on forests are mediated by forest-owners' behavior and market forces, both influenced by the socio-economic environment. In this paper, we analyze how forest carbon payment schemes with varying level of forest-owner participation (hereafter, coverage) impact forest biodiversity.

Biodiversity, or biological diversity, refers to the variability of life forms within a given ecosystem, region, or the entire planet. Biodiversity encompasses genetic variation within species, diversity of species, and variety of ecosystems (CBD 1992). Biodiversity is crucial for maintaining ecosystem stability, resilience, and providing humans with essential services. Yet forest biodiversity has been, and continues to be, heavily affected by land-use change and unsustainable forest management and harvesting practices (Haines-Young 2009; Green et al. 2020). This reflects the inadequate integration of the social value of biodiversity into forest management, decision support models, and policies (Thompson et al. 2011; Augustynczik et al. 2020).

While deforestation, forest degradation, and anthropogenic disturbance threaten forest biodiversity in the tropics (e.g., Barlow et al. 2016), the loss of forest biodiversity in temperate and boreal regions is currently mostly associated with intensive mechanized forest management and harvesting schedules that are poorly suited to recreate the natural disturbance dynamics and structural features forest-dependent species rely on (Kuuluvainen 2009; Kuuluvainen and Gauthier 2018). Key drivers of biodiversity loss in temperate and boreal forests include decreased tree species diversity and the low amount of living or decaying large-diameter trees (Siitonen 2001). The latter follows from short rotation periods (i.e., period from stand establishment to final felling) and low quantities of retention trees (Hagan and Grove 1999). Reconciling biodiversity and timber production objectives is possible (e.g., Mönkkönen et al. 2014; Augustynczik et al. 2020), but effective policies to drive changes in forest management are commonly lacking. In Finland, the number of retention trees or the minimum rotation length are currently not subject to, e.g., any legislative requirements, and this lack is only partly mitigated by voluntary forest certification schemes (Kuuluvainen et al. 2019).

The climate change mitigation potential of forests is widely recognized, and forest-based solutions are included in the national mitigation plans of many countries (Seddon et al. 2020). Forests generate a positive climate externality by storing carbon and keeping it away from the atmosphere. The level of the externality is influenced by forest management. The externality is undersupplied, as—in the absence of a price on forest carbon—private forest owners do not take its full value into account in their management decisions. Hence, more storage can be incentivized by subsidizing it (Tahvonen 1995; van Kooten et al. 1995), and socially optimal storage can be incentivized by subsidizing it according to its social value (Pigou 1920; Sohngen and Mendelsohn 2003). The analysis of the effects of forest carbon payments has advanced from stand-level models using a straightforward description of even-aged management (Plantinga and Birdsey 1994; van Kooten et al. 1995) to highly detailed stand-level models on the one hand (e.g., Tahvonen et al. 2024) and comprehensive forest-sector models (e.g., Sohngen and Sedjo 2006; Sjølie et al. 2013) on the other. At the market level, carbon payments tend to encourage afforestation and lighter harvesting (Sohngen and Mendelsohn 2003; Rautiainen et al. 2018), which leads to greater accumulation of carbon in forests.

Although increasing carbon storage in forests holds promise as an inexpensive means to reduce net emissions (e.g., Tahvonen et al. 2024), governments have been hesitant to implement policies that would incentivize private forest owners to do so. This is likely due to the multiple challenges involved, such as risks of harvest leakage and nonpermanence, unresolved additionality issues, transaction costs, and questions of socio-political acceptance (Sedjo and Sohngen 2012; Assmuth et al. 2024). To date, compliance-driven carbon payment schemes have been implemented only within the New Zealand national emissions trading scheme (NZ ETS) and the state of Californian carbon cap-and-trade scheme (Gren and Akilu 2016). However, forest-related projects (including reduced deforestation and reforestation but also improved forest management) produce a significant fraction of the units traded on the global voluntary carbon market (Forest Trends' Ecosystem Marketplace 2025).

In the boreal and temperate regions, a large fraction of forestland is commercially managed. The demand for forest products and forest bioenergy is a key driver of timber harvests. The demand for these final goods drives the demand for timber which is used as their raw material. The forest industry buys their timber from forest owners, who modify their harvesting activities according to the price development of different timber assortments. Harvest levels, in turn, along with the age-structure dynamics of forest growth, determine the net carbon sink of forest land (cf. Pilli et al. 2017; Hyyrynen et al. 2023). Hence, the future business-as-usual paths of timber harvest and forest sinks—and the need for, and the effects of, forests carbon payments schemes—depend on the demand for forest products, which is subject to considerable uncertainty. Furthermore, forest owners are heterogenous in their preferences, e.g., in terms of how much they value forest amenities, and this affects their harvesting behavior as well as their likely reactions to carbon payment schemes (e.g., Khanal et al. 2017; Husa and Kosenius 2021). Hence, understanding the likely effects of a carbon payment scheme requires the consideration of changes in both demand and forest-owner behavior.

The question of potential biodiversity side-effects of forest carbon policies is highly relevant for societies that aim to utilize forests for climate change mitigation while upholding their biodiversity commitments. A body of literature on the potential biodiversity co-benefits of climate change mitigation through forests exists, but most contributions are set in a tropical context and focus on the reduction of deforestation and forest degradation (e.g., Phelps et al. 2012), plantation activities (e.g., Bryan et al. 2016), or optimized forest management (Boscolo and Vincent 2003). Many contributions also study carbon storage and biodiversity co-benefits of forest conservation (e.g., Venter et al. 2009). Nevertheless, there is still a need for research on biodiversity side-effects in climate zones beyond the tropics, and on biodiversity side-effects resulting from changes in commercial forest management practices, as opposed to changes in land use or conservation status. López et al. (2024) model the effects of national-level biodiversity and carbon forest-sector policies in Norway, showing that the carbon pricing policy has a strong positive effect on old forest area and small positive effects on other examined biodiversity attributes (growing stocks of ungulate forage tree species and temperate tree species). Otherwise, the biodiversity effects of carbon policies targeting managed boreal forests have been little studied.

We contribute to this literature by focusing on the influence of policy coverage. We analyze the impacts of forest payment schemes on forest biodiversity using a detailed forest-sector model, FinFEP, which represents Finland's forest resources, timber markets, and forest and energy industries (Lintunen et al. 2015). As timber prices are endogenous in the model, it allows us to study the effects of carbon payment schemes on timber supply more accurately than stand-level models in which timber prices are exogeneous and the payments' effects on harvests may be overestimated. Moreover, forest-owners' heterogenous valuations of forest amenities affect their harvesting behavior and willingness to join a carbon payment scheme. We measure forest biodiversity using a well-established ecosystem condition index approach, describing ecosystem condition relative to a reference condition through a set of structural components based on ecological assessments (Kotiaho et al. 2016; Kangas et al. 2021). The objective is to establish the degree to which carbon payment policies generate co-benefits that are captured by the indicator used, and examine whether the results are sensitive to variation in forest-product demand which can affect annual harvests.

We find that forest carbon payments based on a €15 tCO_2^{-1} carbon price—sufficient to cause a 20–30 $MtCO_2$ $year^{-1}$ increase in net sink—have a small to moderate positive effect on forest ecosystem condition. The co-benefits are predominantly achieved by lengthening rotations, increasing the volume of broadleaved trees, and increasing the volume of deadwood. The clearest improvements in ecosystem condition are achieved on sites of medium-to-high fertility. Little to no improvement takes place on the least productive sites. Under a partial-coverage carbon payment scheme, that only attracts those forest owners who attach subjective value to forest amenities, ecosystem condition slightly declines in the forests of profit-maximizing forest owners due to domestic harvest leakage, but the aggregate effect on ecosystem condition (and net sink) is positive, albeit smaller than under a full-coverage scheme. Finally, we find that carbon payments have the strongest effect on ecosystem condition (relative to business-as-usual) if the demand for harvested wood products is high.

2. Models and methods

2.1. FinFEP model

The effects of the policies in different demand scenarios are examined using the Finnish Forest and Energy Policy model (FinFEP). This section provides a short summary of the model. A broader description is provided in Lintunen et al. (2015). The model has been updated since its initial release: the forest resource and production capacity data have been updated to reflect the current state, and the technological assumptions have been revised to reflect the current state and expected future development of key technologies.[1] The general structure of the model is nevertheless the same as in the original model description.

FinFEP is a dynamic partial equilibrium model of the Finnish forest and energy sectors. The regional supply and demand of wood (by species and timber grade) arise endogenously in the model. The supply of wood is based on detailed data from the national forest inventory and modelled forest owner behavior. The development of forest resources over time is determined by harvests and growth. The demand for wood arises from the production of final goods, such as sawn timber, pulp, paper, electricity, and heat. The combined demand for the final goods in the global and domestic market is depicted by exogenous demand functions. The demand for intermediate goods also has an exogenous component, but most of it arises endogenously from input demand within the Finnish forest and energy sectors. The technologies utilized in the production of the final goods are modelled in detail. The model's solution depicts a decentralized competitive market equilibrium within the limits of prevailing regulatory framework. The model is solved as a mixed complementarity problem (MCP) (Mathiesen 1985). The FinFEP runs are carried out for the period 2020–2065 with a time step of 5 years.

The model consists of five modules that depict the behavior of the agents at different phases of the production chain. The modules (namely: forests, energy processing, pulp and paper processing, wood-product processing, and final consumption) are linked by material flows, as shown in Fig. 1.

The three processing modules (i.e., wood-product, pulp and paper, and energy processing) depict the behavior of profit-maximizing, price-taking firms. The firms optimize input use to maximize profits from the production of final goods. Demand for the final goods is determined partly endogenously (i.e., when the firms' outputs are utilized as intermediate goods by other firms), and partly exogenously (i.e., by exogenous demand functions). Likewise, the supply of inputs is partly endogenous (when firms use intermediate goods produced by other firms in the model) and partly exogenous (when firms use inputs, such as fossil fuels, that are imported or produced outside the sector). Production processes are represented at an aggregated regional level.

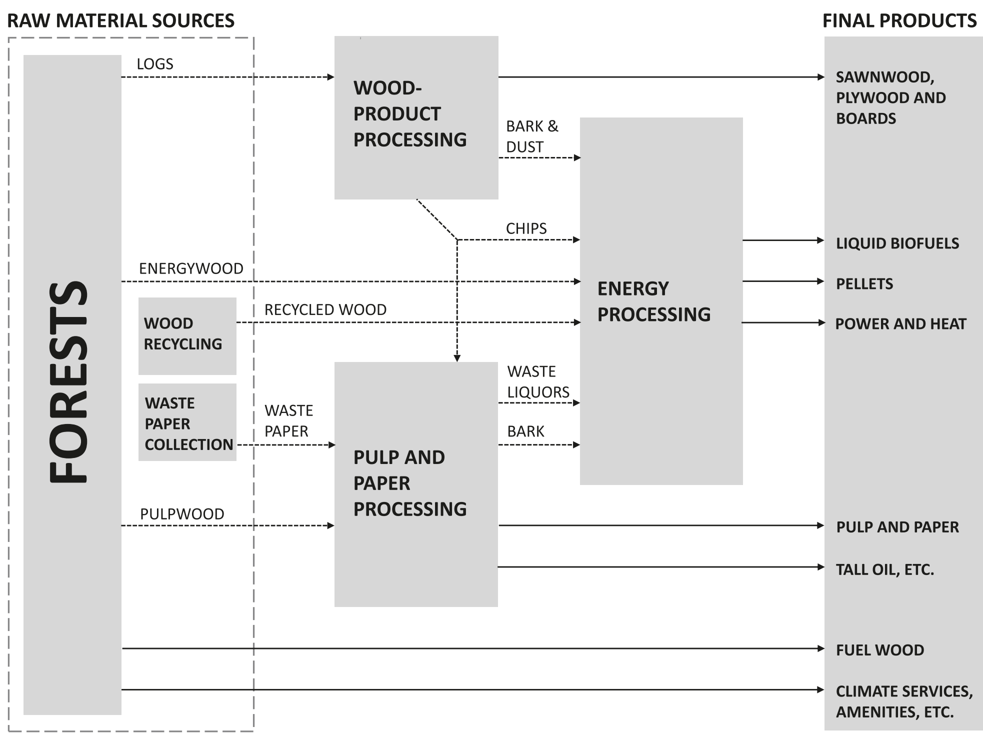
The supply of wood is founded on national forest inventory data and modelled forest owner behavior. Forest inventory data are used to initialize regional timber stocks by productivity class, species, and age class. Wood is harvested by thinning and final felling. Harvests are determined by stationary harvesting rules that depict the landowners' optimized thinning and clear-cutting behavior at the prevailing timber prices in each period (in aggregate). The rules are founded on the modelled behavior of individual landowners that optimize the management of individual forest stands under uncertainty of future prices, i.e., assuming that the development of prices over time follows an exogenous autoregressive random process. These results are then aggregated into rules that can be applied to forest resources depicted at the market level.

At the stand level, landowners seek to maximize the present value of their profits and amenities provided by their forests (Hartman 1976; Amacher et al. 2003). Profits comprise forestry income and carbon rent—in the cases where landowners participate in a forest carbon payment scheme. Landowners in the model are heterogenous in terms of their valuation of amenities. Three landowner types are included, and each type assigns a different weight to amenities. The first type is the *Faustmannian landowner*, who only values profit and gives no weight to amenities, i.e., maximizes the net present value (NPV) of forest income. The share of the forest stock owned by the *Faustmannians* is 32%. The two other types are *mildly and strongly Hartmanian landowners*, 42% and 26% share of forest stock, respectively, who also value amenities in addition to profit. The allocation of forest resources to the forest-owner types is based on calibration, in which there is no apparent increase or decrease in the harvest levels with the current price levels in the first period of the model run. Hence, the calibration rationalizes the observed age–class structure of the forests.

The supply of wood, which is driven by forest owners' harvest decisions, and the demand for wood by the rest of the forest sector are linked through the markets. In each period, the markets for all timber grades clear: supply meets demand at

[1] The model's general structure and data sources are described in Lintunen et al. (2015). However, many aspects of the model have been updated recently. The forest data has been updated to correspond to data from the national forest inventory from the years 2018–2022. The description of forest and energy sector installations is now based on data from the years 2023–2024. New energy technologies have been added to the model to account for technological development that has occurred during the past 15 years. The model has been calibrated to reflect the state of the sectors in the years 2019–2023. However, given the rapid change in the energy sector in recent years, the last years in the interval are emphasized in the calibration. Wood imports from Russia have been reduced permanently, which is in line with the current trade ban between Finland and Russia.

Fig. 1. Material flows between modules in the FinFEP model.



equilibrium prices. From an individual forest-owner's point of view the prevailing market prices provide information that can be used when deciding whether, and to what extent, to thin or clearcut stands. In each period, these management decisions are made based on the current state of the stands, personal amenity valuations, current prices, and future price expectations. High current prices incentivize the immediate felling and thinning of suitable stands, whereas low prices encourage postponing the harvests to a later date when prices are expected to be higher. At the market level, the supply of timber is determined by the aggregated optimal harvesting rules of individual forest owners of each type that are applied to the combined pool of forest resources controlled by the owner type. Hence, each individual forest owner does not need to be separately depicted in the market-level model (see details in Lintunen et al. 2015, Pohjola et al. 2018, or Rautiainen et al. 2025).

2.2. Measuring forest biodiversity

Biodiversity can be measured in several ways, each focusing on different aspects of the complex phenomenon (Purvis and Hector 2000). Classic approaches include indices that describe species richness, species evenness, and their combination (Shannon–Weaver Index) (cf. DeJong 1975). Some biodiversity measurement approaches focus on evaluating the condition of an ecosystem relative to its natural state, undisturbed by humans, as this is the state that best promotes the survival and thriving of the species community adapted to it. In this study, we utilize the latter approach by adapting a well-established ecosystem condition index for boreal forest habitats, developed by ecologists to evaluate the degradation of ecosystems and to aid in designing and prioritizing conservation measures (Matveinen et al. 2015; Kotiaho et al. 2016). A somewhat similar approach is utilized in a tropical context by Boscolo and Vincent (2003), where biodiversity is measured as proximity to a climax state.

The ecosystem condition index describes the condition of ecosystems relative to a reference (natural or target) condition via a set of structural components, and expert panels are used to obtain specific parameter values for each ecosystem type (Kotiaho et al. 2016). A key strength of the ecosystem condition approach in the context of forestry is that its structural components have a clear link to forest management and protection activities and that data on their state can be obtained from the Finnish National Forest Inventory (NFI) (cf. Ćosović et al. 2020). The ecosystem condition index in its original form has been used by the Finnish restoration prioritization working group for regular reporting on the state of ecosystems in Finland (e.g., Kotiaho et al. 2016; Kangas et al. 2023). It has previously been applied in economic studies (e.g., Kangas et al. 2021; Kangas and Ollikainen 2023) in the context of conservation allocation and ecological compensation, but not for studying the effects of commercial forest management in extensive simulated policy scenarios.

The ecosystem condition index is a multiplicative composite index, which in its normalized form (Kangas et al. 2021) obtains values between 0 and 1. The interval's endpoints indicate a *completely degraded condition* and a *natural condition*, respectively. The index, I_e, is calculated separately for each ecosystem type, $e \in E$. In this study, the ecosystem types are the different forest site types included in FinFEP. In FinFEP, forests are distinguished by region, productivity class, and main species. Here, the notation is simplified so that the subindex e summarizes variation in all three.[2]

Following Kotiaho et al. (2016), the index score is determined by the current state of structural ecosystem components (distinguished by the subindex $c \in C$). The current state, S_{ec}, is compared to its reference state, R_{ec}. Different weighting factors, w_{ec}, are assigned to distinct structural components in the calculation of the index. Altogether four ecosystem components are kept track of in the original index version for forest ecosystems. They are (1) the number of large trees, (2) the volume of deadwood, (3) the volume of broadleaved trees, and (4) the annual burnt area. However, only three components are assigned nonzero weighting factors in the calculation of the index for each ecosystem type. The volume of broadleaves is included at mesic sites ($w_{\mathrm{e3}} > 0$), while burnt area is not ($w_{\mathrm{e4}} = 0$). The opposite holds for the least productive xeric and barren heath forest sites ($w_{\mathrm{e3}} = 0$ and $w_{\mathrm{e4}} > 0$). The normalized formula (Kangas et al. 2021) for calculating the index is

$$(1) \qquad I_{\mathrm{e}} = \frac{\prod_{c\in C}\left[1 - w_{\mathrm{ec}}\left(1 - \min\left\{1, \frac{S_{\mathrm{ec}}}{R_{\mathrm{ec}}}\right\}\right)\right] - \prod_{c\in C}(1 - w_{\mathrm{ec}})}{1 - \prod_{c\in C}(1 - w_{\mathrm{ec}})}$$

In eq. 1, the index is calculated for a single ecosystem type. Calculating an aggregate index, $\tilde{I}$, requires aggregation over multiple ecosystem types. Following Kotiaho et al. (2016), we calculate aggregate indices as a weighted average in which index values for individual ecosystem types are weighted relative to their areal extent, i.e.,

$$(2) \qquad \tilde{I} = \frac{\sum_{e\in E} A_{\mathrm{e}} I_{\mathrm{e}}}{\sum_{e\in E} A_{\mathrm{e}}}$$

where A_{e} is the total area (ha) of ecosystem type $e \in E$ in a respective aggregate.

In this paper, the boreal ecosystem condition index is adapted to be compatible with the FinFEP model, which means that certain modifications are made to some of its components. First, as in Kangas and Ollikainen (2023), we replace the structural component "number of large trees" by the age of the stand, because stand age is standard output of the model, whereas the number of large trees is not, but the two variables are strongly correlated. Moreover, many forest-dwelling species rely on trees that are not only large but also old as their habitats, but current production forests often lack such trees (Nirhamo et al. 2023). Second, as the volume of broadleaves is not a decision variable in FinFEP, nor targeted by carbon payment schemes, we use age-dependent fitted functions for volume shares of broadleaves based on NFI data. The volume share functions are estimated for forest stand types specified by site productivity class and main species using regional disaggregation to North and South Finland. Deadwood volume, consisting of decaying and other dead tree stems created through natural mortality and large enough pieces of harvest residues, is measured as in the original version of the index (Kotiaho et al. 2016). The amount of deadwood is kept track of in FinFEP, by allocating deadwood in the classes of different decay rates (Rautiainen et al. 2025) derived from the Yasso07 soil model (Tuomi et al. 2011*a*, 2011*b*). The initial stocks are known only as regional averages, but their allocation into decay classes is not known. Hence, the level of the deadwood structural component might be biased—however, the potential bias does not affect the changes in the level due to carbon policies. Finally, as the annual burnt area is not described by FinFEP, on the least productive sites we assume based on Matveinen et al. (2015) that the current annual burned area is constant and equals only 1/100 of the reference state. In modern Finland, the successful prevention and control of forest fires has led to the burnt area being very low compared to natural forest fire dynamics (Metsähallitus 2018).

In the rest of the paper, this adapted version of the index is simply referred to as the ecosystem condition index. The weighting factors and reference values of each structural component are shown in Table 1, based on Matveinen et al. (2015).

2.3. Climate policies

Based on the current policy environment and likely near-term future policy developments, we assume differentiated climate policies for fossil and biogenic emissions. Fossil CO_2 emissions from large industrial and energy production facilities are regulated through the EU Emissions Trading Scheme (EU ETS) for which we assume an emission allowance price of €90 $tCO_2{}^{-1}$ in the base year (2020) and an increasing trend with a rate of 2% per year. Fossil CO_2 emissions from smaller facilities are not regulated by the EU ETS, but are subject to national carbon taxation, which is assumed to be at the same level as the EU ETS allowance price.

For forestry, climate policy is either nonexistent (business-as-usual (BAU)) or consists of carbon payments based on a constant carbon price of €15 $CO_2{}^{-1}$.[3] The forest carbon payments are paid as carbon rent (Sohngen and Mendelsohn 2003), i.e., the forest owner receives periodic subsidies based on the amount of carbon stored in their forest at the time.

[2] The productivity class is an indicator of the growth conditions, as it depends on the fertility of the site. Productivity classes apply to both organic and mineral soils. On drained peatlands the productivity classes correspond to the following ecological site types: (0) herb-rich type drained peatland forest, (1) *Vaccinium myrtillus*-type drained peatland forest, (2) *Vaccinium vitis-idaea*-type drained peatland forest, (3 and 4) dwarf shrub-type drained peatland forest. On mineral soils the productivity classes correspond to the following Cajander site types (Cajander, 1926): (0) OMT (herb-rich heath forests), (1) MT, (2) VT, (3) CT, (4) CIT.

[3] Based on test runs of the model, a carbon price of €15 $CO_2{}^{-1}$ within a full-coverage carbon payment scheme is sufficient to cause large increases in the forest carbon sink, roughly aligned with Finland's target of climate neutrality by 2035 (Ministry of Economic Affairs and Employment of Finland 2022). On the other hand, based on test runs, such a carbon price implies significant decreases in timber supply especially in the short run, and a higher carbon price would amplify this shock.

Table 1. The components of the ecosystem condition index, their weighting factors, and reference values.

	Structural component	Weighting factor	Reference value
Mesic sites	Stand age, years	0.4	120
	Deadwood volume (m^3 ha^{-1})	0.6	80
	Broadleaved tree volume (m^3 ha^{-1})	0.4	50
Xeric and barren heath forest sites	Stand age (years)	0.4	140
	Deadwood volume (m^3 ha^{-1})	0.4	40
	Annual burnt area (ha)	0.5	5000

Note: The reference for annual burnt area is an aggregate over Finland.

The periodic carbon rent represents the benefit of postponing the release of carbon for one period, and it is calculated by multiplying the carbon stock by the carbon price, the annual discount rate (here, 3%), and the period length in years (here, 5 years). Under standard assumptions, the carbon rent policy can be applied to incentivize identical forest management decisions as the flux-pricing policy (Lintunen et al. 2016), in which emissions are taxed and removals are subsidized (van Kooten et al. 1995). However, the carbon rent approach may have administrative advantages relative to the carbon subsidy and tax approach as payments flow in one direction only: from the regulator to the forest owners.

We examine four cases, in which the policy is implemented at varying degrees of coverage (or not at all):

1. *Business-as-usual*. No forest carbon payment scheme is implemented.
2. *Full coverage*. All forests in Finland are included in the carbon payment scheme. In practice, this means that participation in the scheme is mandatory.
3. *Partial coverage*. Participation in the scheme is voluntary, and only those Finnish forest owners with mild or strong Hartmanian preferences opt in. Participation rate, measured as the share of the forest stock included in the scheme, is 68%.
4. *Low coverage*. Participation in the scheme is voluntary, and only those Finnish forest owners with strong Hartmanian preferences opt in. Participation rate is 26%.

The notion that not all forest owners would be willing to join a voluntary carbon payment scheme may seem counterintuitive given that the scheme represents a potential source of additional income. However, empirical research on forest owners indicates that a considerable share of forest owners may be unwilling to join a carbon payment scheme (e.g., Markowski-Lindsay et al. 2011; Tzemi et al. 2025). The reasons for this may stem from, e.g., unwillingness to enter contracts that may be difficult to withdraw from, attitudes towards climate change and climate policy, distrust of the authorities managing the payment, or simply resistance to changing practices that have become habitual (Markowski-Lindsay et al. 2011; Khanal et al. 2017; Karppinen et al. 2018). Existing research suggests that knowledge of and willingness to adopt forest management practices promoting climate change mitigation is likely to be associated with multiobjective or amenity goals in forest ownership (Khanal et al. 2017; Husa and Kosenius 2021).

Based on an earlier application of the FinFEP model where the effects of a carbon rent system were studied (Pohjola et al. 2018), we understand the basic mechanisms through which carbon payments affect timber markets. Carbon payments give value to the maintenance of the carbon stock in living trees. Hence, for those forest owners who participate in the carbon payment scheme, the payments generate an incentive to reduce the intensity of thinning and postpone final felling. For those forest owners who do not participate, market signals (higher timber prices) incentivize more intensive harvests. The total effects depend on both the extent of the scheme's coverage and the carbon price.

2.3. Demand scenarios

A large part of the demand for Finnish forest industry products comes from global markets, which could develop along several alternative trajectories depending on economic, political, and social factors (cf. Daigneault et al. 2019; Perrels et al. 2022). We examine three demand scenarios: baseline demand, low demand, and high demand. The role of the demand scenarios is to serve as a backdrop for studying how the examined carbon payment schemes affect forest biodiversity in alternative market conditions. In FinFEP, the demand for final products is depicted by exogenous demand functions of the form $q_t = d_{0t}(p_t/p_0)^{-\varepsilon}$, where q_t is the quantity demanded in period t, d_{0t} is the scale parameter, p_t is the price in period t, p_0 is the calibration price, and ε is the price elasticity of demand. The scale parameter, d_{0t}, can be adjusted over time to reflect temporal shifts in demand for products belonging to different product groups (sawn timber, paper, paper board, tissue, softwood cellulose, and hardwood cellulose) in alternative scenarios.

In the baseline scenario, the calibration of the scale parameters of the demand function for different products was done approximately following assumptions made in a baseline scenario for Finland's greenhouse gas emissions and energy balances up to 2055 with existing energy and climate measures formulated by Koljonen et al. (2024). The low and high demand scenarios were defined by scaling the baseline growth rates so that the demand projections are roughly consistent with forest product demand paths proposed by

Table 2. Annual growth rates used in scaling the demand functions.

Forest product	Baseline demand	Low demand	High demand
Sawn timber	2.00%	– 1.90%	3.3%
Paper	– 1.50%	– 1.50%	– 1.50%
Paper board	1.00%	– 0.95%	1.6%
Tissue	0.00%	0.00%	0.00%
Softwood cellulose	1.00%	– 0.95%	1.6%
Hardwood cellulose	0.00%	0.00%	0.00%

Daigneault and Favero (2021) for relevant Shared Socioeconomic Pathways (SSPs).[4] The demand function for paper was treated as an exception, as we assumed that the decreasing consumption trend caused by digitalization will dominate in all three scenarios. The assumed annual percentage change in the scale parameters of different product groups' demand functions are shown in Table 2.

3. Results

3.1. Effects of carbon payment schemes on harvests and net sink

In the BAU case, i.e., without forest carbon payments, the total harvested volume is 75 Mm^3 in 2020 and stays approximately on the same level throughout the examined time horizon 2020–2065 (Fig. 2*c*). A little less than half of this volume consists of thinned volume and a little more than half of clearcut volume (Figs. 2*a* and 2*b*). Under a full-coverage forest carbon payment scheme, thinned volume is only moderately reduced compared to BAU, but clearcut volume is strongly reduced for the first 30 years as the rotations become longer. This results in a notable decrease of total harvested volume relative to BAU during the first decades, but the effect mostly tapers off by 2065.

Under the partial-coverage carbon payment scheme (that attracts only forest owners who hold mild or strong Hartmanian preferences, i.e., NPV maximizers are assumed not to participate in the scheme), the negative effect on total harvests is initially weaker but stable. Clearcut volume remains at a lower level throughout the time horizon. Thinned volume increases after the first decade. However, the low-coverage carbon payment scheme (where exclusively strong Hartmanians participate) decreases harvests only little relative to BAU as the number of such forest owners is small. Total growing stock increases from approximately 2200 to 2800 Mm^3 during 2020–2065 under BAU (Fig. 2*d*). The full-coverage carbon payment scheme leads to a notably stronger increase in growing stock, while the partial-coverage scheme falls in between the BAU and the full-coverage scheme. Under the low-coverage scheme, the growing stock is close to the BAU path.

Compared to BAU, the full-coverage carbon payment scheme strengthens the net sink in forests and wood products by 20–30 MtCO$_2$ year^{-1} (Fig. 3*a*). The net sink includes CO$_2$ fluxes to and from standing trees, dead organic matter, and harvested wood products (HWP). Postponing clearcuts increases the sink in standing trees greatly (i.e., up to 40 MtCO$_2$ year^{-1}) relative to BAU. On the other hand, the sink in dead organic matter decreases strongly initially, as less harvest residue feeds into the pool due to decreased harvests. The policy's effects on the HWP stock and fossil emissions are much smaller. The effects of the partial-coverage scheme are qualitatively similar but smaller (Fig. 3*b*). The low-coverage scheme causes little change in CO$_2$ fluxes compared to BAU (Fig. 3*c*), implying that it will not fulfill any ambitious climate targets for the forest sector.

3.2. Effects of carbon payment schemes on biodiversity

Carbon payment schemes change the timing and intensity of harvests, which, in turn, results in changes in various stand structural components that are important for ecosystem condition and, hence, biodiversity. Specifically, postponing clearcuts increases the prevalence of older stands, in which natural mortality is higher, thus contributing to the volume of deadwood. Longer rotations and less intensive thinning increase the average volume of broadleaved trees. Figure 4 shows the changes (vs. BAU) in the components of the ecosystem condition index caused by carbon payment schemes of varying coverage (absolute values are shown in Fig. S1 in the Supplementary Material). Average stand age (Fig. 4*a*) increases monotonously, by approximately 11 (8) years by 2065, as a result of the full-coverage (partial-coverage) scheme. The low-coverage scheme decreases average stand age very slightly compared to BAU, likely due to leakage of harvests to forests that are not enrolled in the scheme. Average deadwood volume increases by approximately 1 m^3 ha^{-1} as a result of the full-coverage scheme; impacts of the partial- and low-coverage schemes are smaller (Fig. 4*b*). Average broadleaf tree volume increases as a

[4] In the context of climate policy, SSPs represent an established approach for tackling uncertainties related to global socio-economic development (Riahi et al. 2016). SSPs are scenarios that describe different ways the world might develop in terms of demographics, economics, and energy use up to the year 2100, helping policymakers and researchers understand how various interlinked factors might influence greenhouse gas emissions and climate change. The low demand scenario is roughly in line with SSP3 (Regional Rivalry) with decreased global trade and slower economic development, whereas the high demand scenario roughly corresponds to SSP5 (Fossil-fueled Development) with high economic growth. SSPs are often used in conjunction with representative concentration pathways, climate change scenarios to project future greenhouse gas concentrations.

Fig. 2. Annual harvested volumes and growing stock under business-as-usual (BAU) and various coverages of the carbon payment policy. Note: Baseline demand scenario, carbon payment level €15 tCO_2^{-1}.

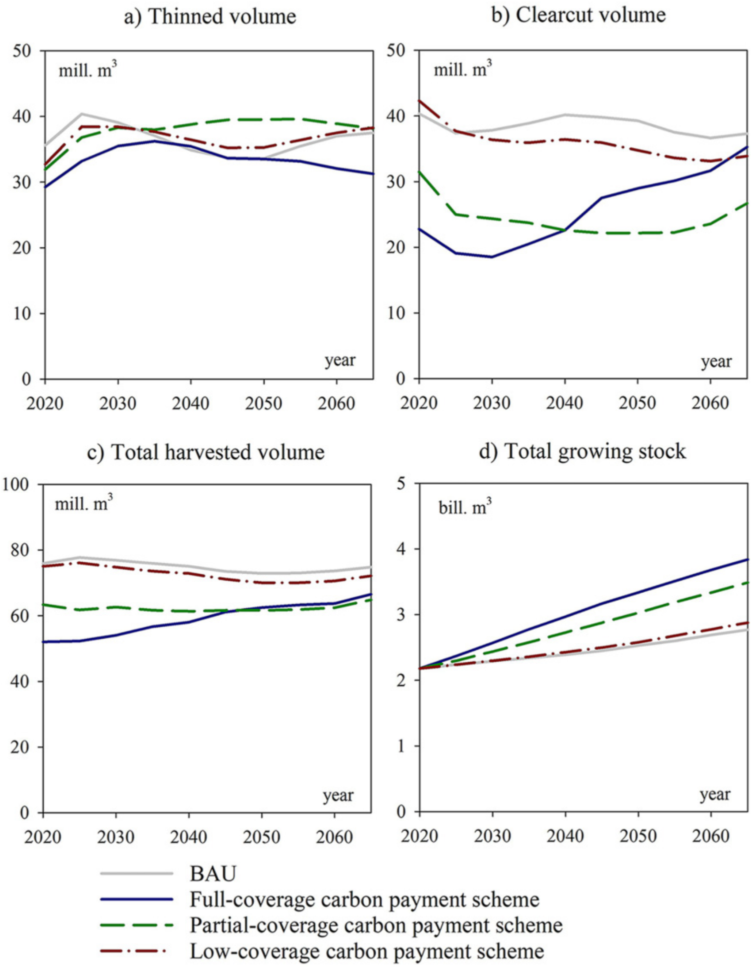


Fig. 3. Change in CO_2 flux (vs. business-as-usual (BAU)) under (*a*) full-coverage, (*b*) partial-coverage, and (*c*) low-coverage carbon payments. Note: Baseline demand scenario, carbon payment level €15 tCO_2^{-1}.

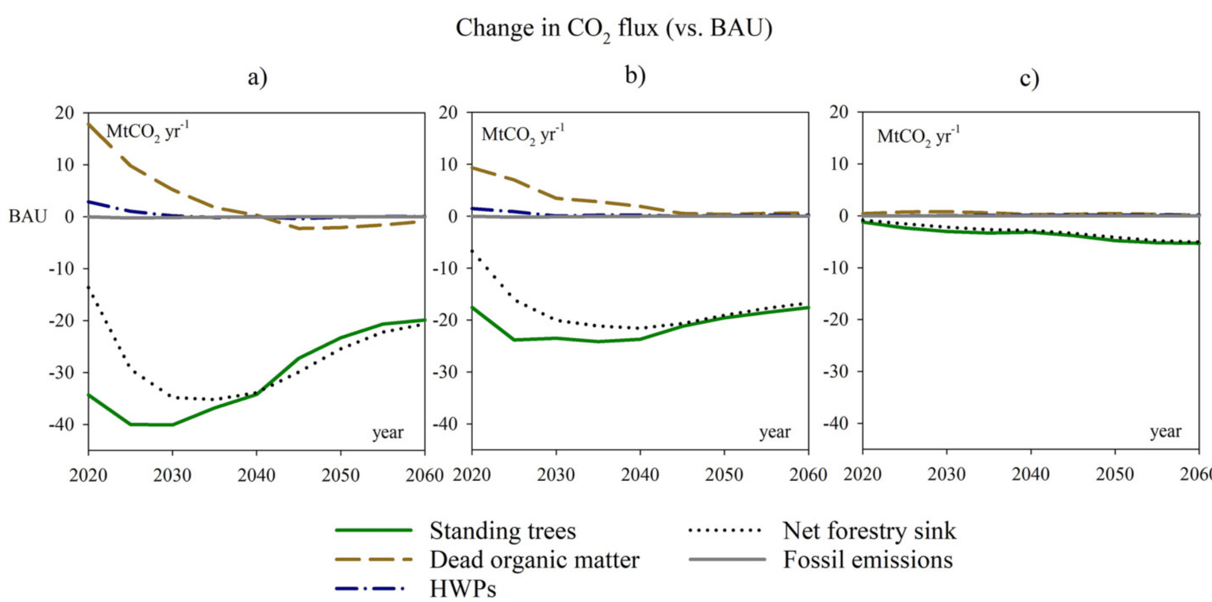
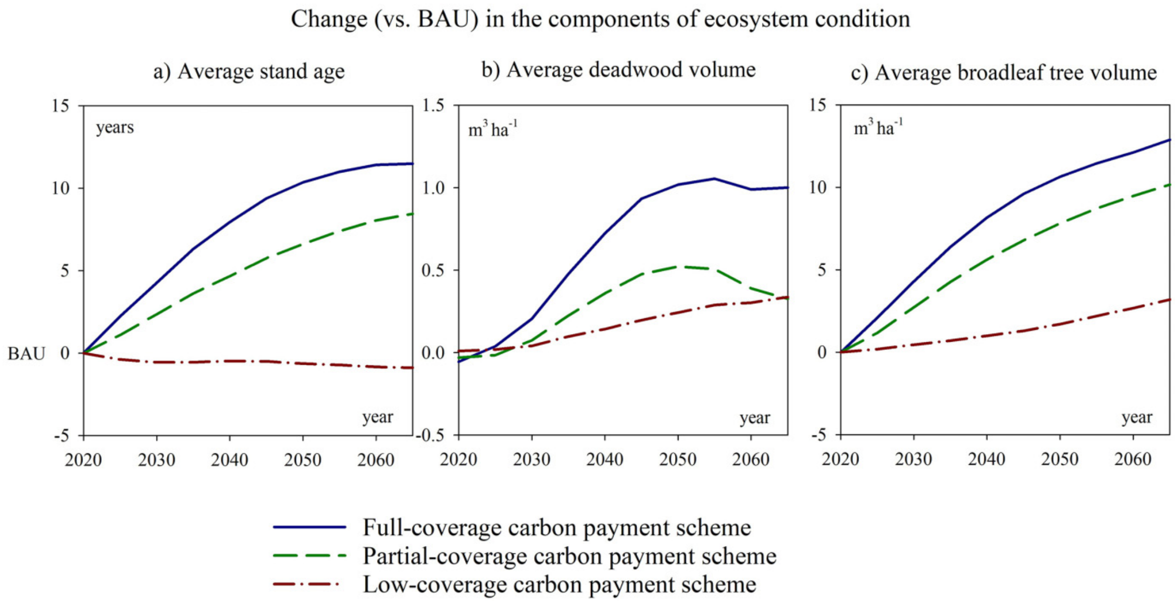


Fig. 4. Change (vs. business-as-usual (BAU)) in the components of the biodiversity index by policy coverage: (*a*) average stand age, (*b*) average deadwood volume, and (*c*) average broadleaf volume. Note: Baseline demand scenario, carbon payment level €15 $tCO_2{}^{-1}$.



result of carbon pricing under all scheme coverages; the positive effect of the full-coverage scheme equals approximately 13 m^3 ha^{-1}. Burnt area stays constant (and very low) as—in our scenarios—it is not affected by harvesting decisions but determined exogenously.

Assuming BAU forest management, the state of forest biodiversity—approximated by the ecosystem condition index—is rather weak throughout the examined time horizon. The ecosystem condition score increases from 0.14 to 0.17 during 2020–2065 (solid black line in Fig. S2a in the Supplementary Material). (Recall that index scores 0 and 1 represent a completely degraded condition and a natural condition, respectively). Figures 5 and 6 show the changes in ecosystem condition score relative to the BAU path, caused by the carbon payment schemes of varying coverage.

In the full-coverage carbon payment scheme's case (Fig. 5*a*), the effect of carbon payments on ecosystem condition is positive on sites of all productivity classes, denoted in an order of decreasing fertility (see Section 2.2). The positive effect is the strongest on the fertile sites (productivity classes 0–2). On these sites, all the structural components of the ecosystem condition index are favorably affected by carbon payments (Fig. 4). On the least fertile sites (productivity classes 3 and 4), the positive effect on ecosystem condition is smaller and tapers down towards the end of time horizon. The total change in ecosystem condition score (solid black line) is the weighted average of the respective changes in all productivity classes. After 2055, the total ecosystem condition score is approximately 0.04 points higher than without carbon payments, i.e., the total ecosystem condition score increases from 0.16 to 0.20. The improvement in ecosystem condition takes place gradually as it takes time for the changes in management to influence the structural attributes of the stands.

Under the partial-coverage carbon payment scheme, the effect of carbon payments on biodiversity is positive on all site types except the least productive one, but the effect is weaker than under the full-coverage scheme (Fig. 5*b*). The total ecosystem condition score is around 0.02 points higher than in BAU. Under the low-coverage carbon payment scheme, the total effect of carbon payments on ecosystem condition is negligible. Only very small positive effects on high-productivity sites and very small negative effects on low-productivity sites are discernible (Fig. 5*c*).

The extent to which forest owners change their harvesting behavior—and, hence, the ecosystem condition of their stands—in response to carbon payments depends both on their amenity preferences and the assumed coverage of the carbon payment scheme. Under the full-coverage scheme, the strongest positive effects on ecosystem condition are seen in the stands of mild Hartmanian forest owners (Fig. 6*a* presenting change vs. BAU in ecosystem condition score by forest-owner type; absolute scores are shown in Fig. S3 in the Supplementary Material). As most of the forest owners are NPV maximizers, the change (vs. BAU) in ecosystem condition score observed in their stands is close to the change in ecosystem condition score of all forest owners combined (black line). The improvement in ecosystem condition is the smallest in stands of strong Hartmanians, because they apply long rotation ages even without carbon payments. The effects under a partial-coverage scheme (Fig. 6*b*) mostly resemble those observed under the full-coverage scheme, but the positive total effect on ecosystem condition is smaller. The only noteworthy difference is that, under partial coverage, ecosystem condition is weakened in the forests of NPV-maximizing owners who opt out from the scheme (red dashed line).

Fig. 5. Change (vs. business-as-usual (BAU)) in ecosystem condition score by site productivity class under (*a*) full-coverage, (*b*) partial-coverage, and (*c*) low-coverage carbon payments. Note: Baseline demand scenario, carbon payment level €15 $tCO_2{}^{-1}$.

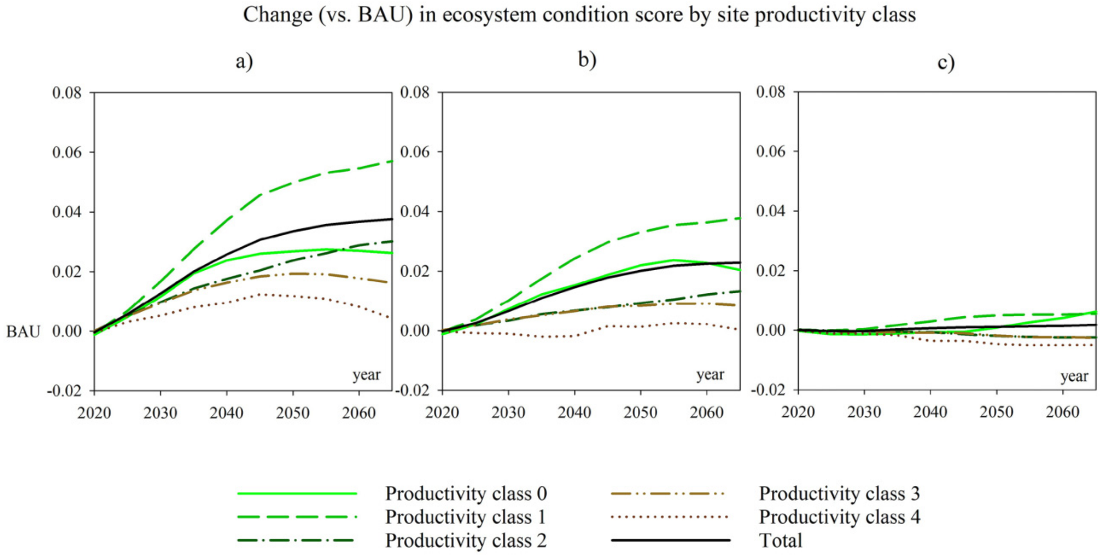
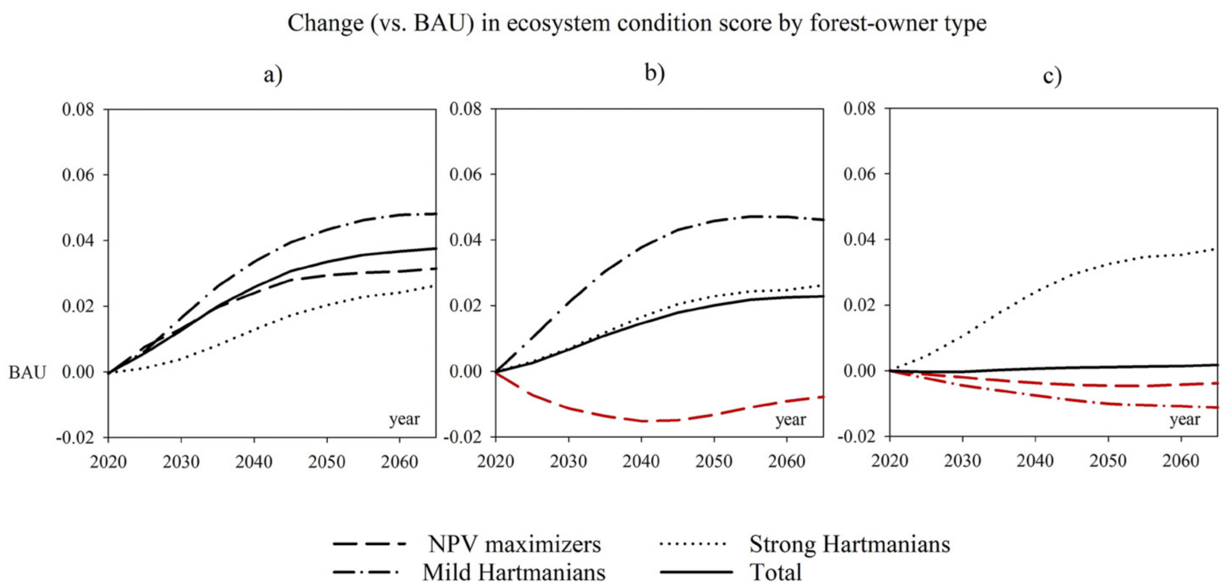


Fig. 6. Change (vs. business-as-usual (BAU)) in ecosystem condition score by forest-owner type under (*a*) full-coverage, (*b*) partial-coverage, and (*c*) low-coverage carbon payments. Red lines indicate forest-owner types not participating in the scheme in question. Note: Baseline demand scenario, carbon payment level €15 $tCO_2{}^{-1}$.

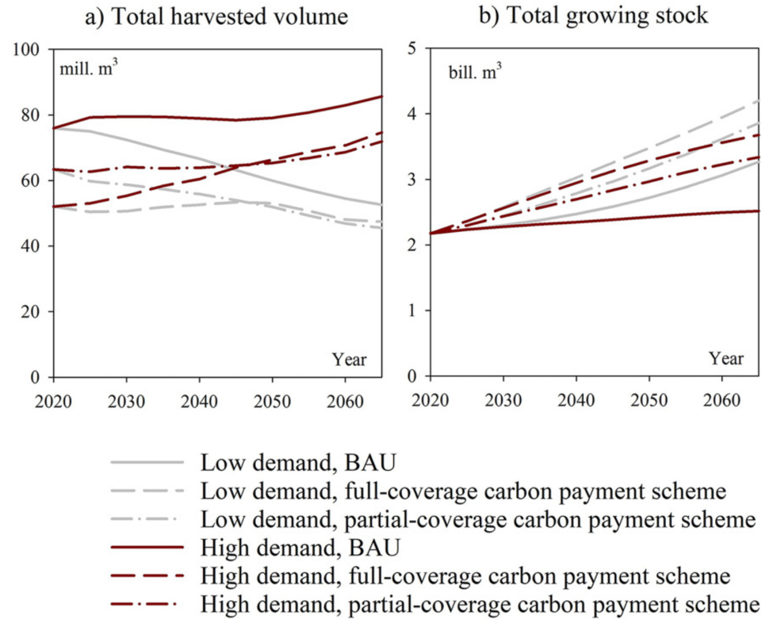


This effect is caused by the following mechanism. When Hartmanian forest owners decrease harvests in response to the carbon scheme, timber prices increase (see Fig. S4 in the Supplementary Material), prompting the NPV maximizers to harvest more–i.e., harvests *leak* to forests that do not participate in the scheme. In the low-coverage case (Fig. 6*c*), the leakage is amplified, and combined with low participation the scheme has negligible total effect on the ecosystem condition. Improvements in ecosystem condition in the forests covered by the scheme (owned by strong Hartmanians) are roughly offset by small declines in the extensive forests not covered by scheme (owned by mild Hartmanians and NPV maximizers).

3.3. Sensitivity to demand scenarios

To estimate how conceivable changes in demand for forest-sector products might affect our results, we performed a sensitivity analysis utilizing the low and high demand scenarios described in Section 2.4. We examine solutions assuming business-as-usual and solutions under full-coverage and partial-coverage policy, but for brevity we exclude the low-coverage policy as its effects are negligible (see results in Sections 3.1 and 3.2). We find that without carbon payments, i.e., in the BAU scenarios indicated by solid lines in Fig. 7*a*, harvest volumes increase slightly under high demand and decrease gradually but strongly under low demand. In the high-demand case, carbon payments with full

Fig. 7. Total harvested volume and growing stock under low and high demand scenario and business-as-usual (BAU) and various coverages of the carbon payment policy. Note: Carbon payment level €15 tCO_2^{-1}.



coverage (red dashed line) initially decrease harvests by over 20 million m^3 relative to BAU, but then harvests gradually increase to 75 million m^3 over time. In the low-demand case with full coverage carbon payments (light grey dashed line), harvested volume remains low throughout the time horizon. Payments with partial coverage (dash-dotted lines) reduce harvest more mildly than those with full coverage, in both high and low demand cases. Under high demand, harvests increase slightly over time, whereas under low demand, they decrease. In short, the policy choice (i.e., full vs. partial coverage) affects harvests immediately, whereas the differences between the demand scenarios arise gradually as defined in Section 2.4.

Differences in the harvest volumes under low and high demand and alternative policy coverages are reflected in the development of total growing stock (Fig. 7b). Total growing stock stays approximately constant under high demand and BAU and increases throughout the time horizon in all other demand and policy scenario combinations. Unsurprisingly, the largest growing stock is reached when low demand and full-coverage carbon payments are assumed.

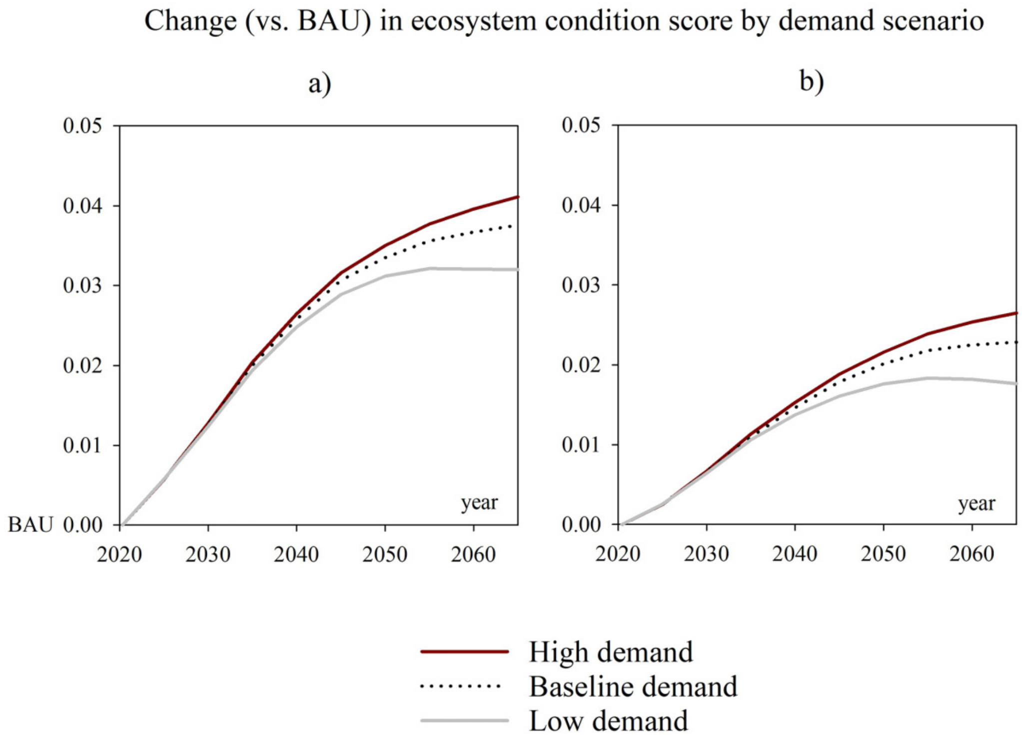
Assumptions on product demand and policy coverage also have an impact on ecosystem condition. Figure 8 presents the change (vs. BAU) in ecosystem condition score by demand scenario under full-coverage (Fig. 8a) and partial-coverage (Fig. 8b) carbon payments (absolute values are shown in Fig. S5 in the Supplementary Material). The positive effect of the full-coverage carbon payment scheme on ecosystem condition is stronger under the high demand scenario than under baseline or low demand (Fig. 8a). This is because higher demand implies higher harvest pressure, and carbon payments work against increasing harvests. The findings are qualitatively similar for the partial-coverage scheme (Fig. 8b) but with smaller improvement in ecosystem condition due to the carbon payments.

4. Discussion

In this study, we examine whether forest carbon payments are likely to have positive or negative side-effects on biodiversity in boreal commercially managed forests. We find that Finland's aggregate state of forest biodiversity, measured through an ecosystem condition index, is improved by carbon payments. This result supports the findings of co-benefits in earlier research on different climate zones and forestry contexts (Bryan et al. 2016; Dybala et al. 2019; Venter et al. 2009), and are in line with results from sector-level policy modelling of Norwegian forestry (López et al. 2024). Our result suggests that, at national scale, forest carbon payments may be well aligned with the "Do No Significant Harm" principle recently emphasized by the EU (Beltran Miralles et al. 2023), stating that the implementation of climate policies should not hamper biodiversity protection objectives. The biodiversity co-benefits suggested by our results are also an encouraging sign that carbon payments targeted to existing forest land seem to have better implications for biodiversity compared to carbon payments focusing on afforestation, which may cause unfavorable biodiversity effects (e.g., Matthews et al. 2002).

Our results suggest that the carbon sink can be notably strengthened through carbon payments, even when the carbon price is relatively low. With a €15 tCO_2^{-1} carbon price, a 20–30 $MtCO_2$ $year^{-1}$ increase (relative to BAU) in the net sink in forests and HWP is achieved under the full-coverage carbon payment scheme. For reference, Finland's net sink in forest land and HWP has ranged from −33.7 to −0.4 during 2010–2023. While the effect of increasing carbon sink is qual-

Fig. 8. Change (vs. business-as-usual (BAU)) in ecosystem condition score by demand scenario under (*a*) full-coverage and (*b*) partial-coverage carbon payments. Note: carbon payment level €15 $tCO_2{}^{-1}$.



itatively in line with the results of Swedish (Guo and Gong 2017) and Norwegian (Lopez et al. 2024) forest-sector studies, the observed reaction to carbon pricing is clearly stronger in our results. The possible reasons for this difference have been discussed at length in Pohjola et al. (2018), López et al. (2024), and Rautiainen et al. (2025). In short, both supply and demand are more elastic in FinFEP than in the other Nordic forest-sector models due to more flexibility in harvest decisions and a stronger link to global markets for forest industry products. There may also be other, more subtle differences between the models that are harder to pinpoint.

While we find that carbon payments affect ecosystem condition positively, the positive effect is relatively small. Hence, even an ambitious carbon payment scheme cannot alone solve the problem of biodiversity loss in boreal forests. Halting biodiversity loss will require separate, biodiversity-targeted policies. The finding is consistent with the Tinbergen Rule stating that for each distinct policy target there must be at least one separate policy instrument (Tinbergen 1952). A similar conclusion was also reached by Bryan et al. (2016) in the context of reforestation of agricultural land in Australia, where biodiversity co-benefits were only negligible with uniform carbon payments, but more significant when these payments were supported by other policy measures. López et al. (2024) found that carbon pricing alone was able to strongly increase old forest area, but combining carbon pricing with specific biodiversity-related constraints was helpful for other forest features important for biodiversity.

Our modelling approach accounts for the fact that even if carbon payments increase forest biodiversity on an aggregate level, the effects on some site types may be negative. Moreover, if the payment scheme is not mandatory and only some of the forest owners choose to participate, biodiversity effects will be heterogeneous across different owner types' forests. We find that the clearest biodiversity co-benefits are achieved on sites of medium-to-high fertility, and little to no improvement takes place on the least productive (i.e., ecologically xeric) sites. Carbon payments improve all the structural components that are important for ecosystem condition on high- and medium-fertility sites, but the effect on burnt area–that supports biodiversity on boreal xeric sites alongside stand age and deadwood volume–was beyond the scope of our model. Our results suggest, however, that if a carbon payment scheme is implemented, additional biodiversity-targeted policies may be especially important for low-productivity sites. Such policies could involve, e.g., promoting prescribed burning, which would have negative climate effects—representing a biodiversity–climate tradeoff worth further study. On the other hand, low-productivity sites are more likely to acquire protection status than high-productivity sites due to smaller economic opportunity cost; in this sense, the finding that carbon payments may improve ecosystem condition on high-productivity sites under commercial management is promising.

Our examination of various policy coverages was motivated by the notion that the implementation of a full-coverage, mandatory carbon payment scheme may entail considerable administrative challenges and face strong political opposition, at least in the short term. It might be more feasible to launch a small-scale pilot scheme and then advance gradually towards higher coverage. We find that under a partial-coverage scheme, that is based on voluntary participation and attracts forest owners with mild or strong Hartmanian preferences, ecosystem condition slightly declines in the forests of NPV-maximizing forest owners due to domestic harvest leakage, but the aggregate effect on ecosystem condition is positive albeit smaller than under a full-coverage scheme. This means that the carbon payment scheme's positive aggregate effects on the net sink and ecosystem condition are only partly impeded by harvest leakage, and the role of leakage decreases as the fraction of forest owners joining the scheme increases. In the case of the low-coverage scheme, where only those forest owners with strong Hartmanian preferences participate, the net sink is increased very little and effects on ecosystem condition are negligible. Hence, our results emphasize the value of promoting wide forest-owner enrollment through, e.g., a participatory scheme design process and information campaigns (cf. Assmuth et al. 2024).

Our results should be interpreted with some caution as our modelling approach entails limitations that may affect validity. One question relates to the ability of the chosen biodiversity metric, an ecosystem condition index based on certain structural components, to measure the key aspects of such a complex and dynamic phenomenon as biodiversity. It can be rightly argued that aggregating several ecological aspects into one composite index oversimplifies the effects of forest management on ecological condition and biodiversity and reduces transparency. An alternative approach, applied, e.g., in López et al. (2024), would be to look only at a set of individual structural components of ecosystem condition (see Fig. 4) instead of combining them to form a composite index. However, we argue that the index approach is useful, as it distills established ecological understanding of the relative importance of various structural components for maintaining biodiversity on different ecosystem types (Kotiaho et al. 2016) and facilitates the communication of results to decision-makers. We see that the positives exceed the negatives, especially when the limitations in modelling of individual components are clearly stated. Ecological research suggests that the quality of habitats and the survival probability of forest-dwelling species in Finland is correlated with the presence of similar structural components, as those applied in our model (Koivula et al. 2022). However, it is uncertain how the diversity of species (or, even more so, genetic diversity) develops over time in response to changes in stand structure. Habitat connectivity is an additional aspect that is highly important for biodiversity but not addressed in the ecosystem condition index.

Moreover, the structure of the FinFEP forest-sector model limits how precisely the effects of carbon payments on ecosystem condition can be modelled. One limitation relates to the ecosystem condition index's structural component "volume of broadleaf trees". In general, there are two ways forest owners could alter the broadleaved volume: species selection at regeneration and in thinning operations. In our model, when predicting the future development, we assume that the same species is optimal from one rotation to the next. It is also assumed that in thinning operations the volume share of broadleaved trees develops as in recent history, i.e., the thinning decisions do not include species-specific choices of harvesting or leaving trees. Hence, the volume of broadleaves is determined by total stand volume and age-dependent, stand-type- and region-specific volume share functions for broadleaves based on NFI data. The central question is whether the carbon policy would create incentives to change this behavior. Stand-level research suggests that carbon payments may (by increasing optimal stocking levels on fertile sites) motivate increasing the share of shade-tolerant spruce at the expense of less-shade-tolerant birch (Assmuth et al. 2021). Nevertheless, they may also incentivize maintaining trees of commercially nonvaluable broadleaf species in the stand (Assmuth et al. 2021). Together these results demonstrate that carbon payments can affect the share of broadleaves through separate mechanisms, which work in opposite directions. Such fine-detail effects cannot be captured by our forest-sector model's current version. Hence, it is possible that our modelling over- or underestimates the carbon payments' positive impact on this structural component, depending on site type and management practices regarding commercially nonvaluable broadleaf species. It seems, however, that the main driver of the increased stocking volume of broadleaved trees is the shift to longer rotations—which increases stocking volume in general and, therefore, also the volume of broadleaved trees. This main mechanism is present in the model and represented in the results.

Another modelling limitation relates to the structural component "burnt area" as our modelling does not include any mechanism through which the carbon payments could affect fire probability or burnt area. In theory, carbon payments could have an indirect effect on fire probability, if, e.g., older forests were more or less likely to catch fire, but since there is no such endogenous mechanism in the model, we cannot really make any inferences about these changes. However, as described in Section 2.2, forest fires are rare in Finland due to effective fire prevention and control, making it unlikely that carbon payments by themselves would cause significant changes in forest fire regimes. Overall, we argue that our approach to modelling ecosystem structural components is suitable for addressing the research question at hand, but leaves room for refinement in future research.

Our modelling approach incorporates the potential for harvest leakage within Finland if a carbon payment policy covers only some part of Finland's forests. However, we do not study harvest leakage to forests beyond Finland nor its effects on the biodiversity of those forests. Such *transborder carbon leakage* could result from the unilateral implementation of a carbon payment scheme through two mechanisms. The first mechanism is increased imports, as Finnish forest industries might seek to make up for reduced domestic harvests by importing wood from other countries. The second mechanism is migration. That is, forest industries might migrate from Finland to other countries where more resources are available. In practice, the migration could occur gradually, through altered investment patterns, i.e., decreased investments to the maintenance of domestic production facilities, and increased investments to the expansion of new facilities abroad owned by Finnish or foreign companies. A meta-analysis by Pan et al. (2020) indicates that the carbon leakage rate in the forestry sector is higher than in the sector of energy-intensive industries, approximately 40% on average, and even higher leakage rates have been presented for EU countries (e.g., Kallio and Solberg 2018). However, evaluating the biodiversity impacts of transnational harvest leakage, i.e., "biodiversity leakage", is challenging because these impacts depend both on the functioning of global markets for forest industry products and on the characteristics, age structure, and utilization practices of forests in competitor countries (c.f. García et al. 2018). Further research on the subject is needed.

Finally, the study at hand does not consider the effects of climate change on forest growth and natural disturbance risk. Increasing rotation ages may increase the risk of drought stress-triggered pest outbreaks, especially in spruce-dominated stands (cf. Seidl et al. 2017). On the other hand, more decayed dead wood may promote species that are antagonist to some important forest pests (Johansson et al. 2007). Increasing tree species diversity in boreal forests would—in addition to being beneficial for biodiversity—likely improve forest resilience (Jactel et al. 2017). Improved understanding of the interactions between forest structure, biodiversity loss, and disturbance risk is needed to support the development of policies that incentivize both climate change mitigation and adaptation in forestry.

5. Conclusions

We analyzed the effects of forest carbon payments on forest biodiversity utilizing a detailed Finnish forest-sector model and a biodiversity metric based on ecosystem condition. We found that carbon payment schemes of full or partial coverage have biodiversity co-benefits in boreal commercially managed forests, especially on medium-to-high-productivity sites. However, the extent of these co-benefits is unlikely to be sufficient to halt biodiversity loss in forest ecosystems, emphasizing the need for separate, biodiversity-targeted policies.

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Data generated or analyzed during this study are available from the corresponding author upon reasonable request.

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Supplementary material

Supplementary data are available with the article at https://doi.org/10.1139/cjfr-2025-0184.

References

Amacher, G., Conway, M.C., and Sullivan, J. 2003. Econometric analyses of nonindustrial forest landowners: is there anything left to study? J. For. Econ. **9**(2): 137–164. doi:10.1078/1104-6899-00028.

Assmuth, A., Autto, H., Halonen, K.M., Haltia, E., Huttunen, S., Lintunen, J., and Usivuori, J. 2024. Forest carbon payments: a multidisciplinary review of policy options for promoting carbon storage in EU member states. Land Use Policy **147**: 107341. doi:10.1016/j.landusepol.2024.107341.

Assmuth, A., Rämö, J., and Tahvonen, O. 2021. Optimal carbon storage in mixed-species size-structured forests. Environ. Resour. Econ. **79**: 249–275. doi:10.1007/s10640-021-00559-9.

Augustynczik, A.L.D., Gutsch, M., Basile, M., Suckow, F., Lasch, P., Yousefpour, R., and Hanewinkel, M. 2020. Socially optimal forest management and biodiversity conservation in temperate forests under climate change. Ecol. Econ. **169**: 106504. doi:10.1016/j.ecolecon.2019.106504.

Barlow, J., Lennox, G.D., Ferreira, J., Berenguer, E., Lees, A.C., Nally, R.M., and Gardner, T.A. 2016. Anthropogenic disturbance in tropical forests can double biodiversity loss from deforestation. Nature, **535**(7610): 144–147. doi:10.1038/nature18326. PMID: 27362236.

Beltran Miralles, M., Gourdon, T., Seigneur, I., Arranz Padilla, M., and Pickard Garcia, N. 2023. The implementation of the ‘Do No Significant Harm’ principle in selected EU instruments. Publications Office of the European Union, Luxembourg. doi:10.2760/18850, JRC135691.

Boscolo, M., and Vincent, J.R. 2003. Nonconvexities in the production of timber, biodiversity, and carbon sequestration. J. Environ. Econ. Manage. **46**(2): 251–268. doi:10.1016/S0095-0696(02)00034-7.

Bryan, B.A., Runting, R.K., Capon, T., Perring, M.P., Cunningham, S.C., Kragt, M.E., and Wilson, K.A. 2016. Designer policy for carbon and biodiversity co-benefits under global change. Nat. Clim. Change, **6**(3): 301–305. doi:10.1038/nclimate2874.

Cajander, A.K. 1926. The theory of forests types. Acta For. Fenn. **29**: 1–108. doi:10.14214/aff.7193.

Convention on Biological Diversity (CBD). 1992. Article 2: Use of Terms. Available from https://www.cbd.int/convention/text/ [accessed December 2025].

Ćosović, M., Bugalho, M.N., Thom, D., and Borges, J.G. 2020. Stand structural characteristics are the most practical biodiversity indicators for forest management planning in Europe. Forests, **11**(3): 343. doi:10.3390/f11030343.

Daigneault, A., and Favero, A. 2021. Global forest management, carbon sequestration and bioenergy supply under alternative shared socioeconomic pathways. Land Use Policy, **103**: 105302. doi:10.1016/j.landusepol.2021.105302.

Daigneault, A., Johnston, C., Korosuo, A., Baker, J.S., Forsell, N., Prestemon, J.P., and Abt, R.C. 2019. Developing detailed shared socioeconomic pathway (SSP) narratives for the global forest sector. J. For. Econ. **34**(1–2): 7–45. doi:10.1561/112.00000441.

DeJong, T.M. 1975. A comparison of three diversity indices based on their components of richness and evenness. Oikos, 222–227. doi:10.2307/3543712.

Dybala, K.E., Matzek, V., Gardali, T., and Seavy, N.E. 2019. Carbon sequestration in riparian forests: a global synthesis and meta-analysis. Global Change Biol. **25**: 57–67. doi:10.1111/gcb.14475.

Forest Trends’ Ecosystem Marketplace. 2025. State of the Voluntary Carbon Market 2025. Forest Trends Association, Washington, DC.

García, J.H., Orlov, A., and Aaheim, A. 2018. Negative leakage: the key role of forest management regimes. J. For. Econ. **33**: 8–13. doi:10.1016/j.jfe.2018.09.001.

Green, E.J., McRae, L., Freeman, R., Harfoot, M.B., Hill, S.L., Baldwin-Cantello, W., and Simonson, W.D. 2020. Below the canopy: global trends in forest vertebrate populations and their drivers. Proc. R. Soc. B Biol. Sci. **287**(1928): 20200533. doi:10.1098/rspb.2020.0533.

Gren, M., and Aklilu, A.Z., 2016. Policy design for forest carbon sequestration: a review of the literature. Forest Policy Econ. **70**: 128–136. doi:10.1016/j.forpol.2016.06.008.

Guo, J., and Gong, P. 2017. The potential and cost of increasing forest carbon sequestration in Sweden. J. For. Econ. **29**: 78–86. doi:10.1016/j.jfe.2017.09.001.

Hagan, J.M., and Grove, S.L. 1999. Coarse woody debris: humans and nature competing for trees. J. For. **97**(1): 6–11. doi:10.1093/jof/97.1.6.

Haines-Young, R. 2009. Land use and biodiversity relationships. Land Use Policy, **26**: S178–S186. doi:10.1016/j.landusepol.2009.08.009.

Hartman, R. 1976. The harvesting decision when a standing forest has value. Econ. Inq. **14**(1): 52–58. doi:10.1111/j.1465-7295.1976.tb00377.x.

Husa, M., and Kosenius, A.K. 2021. Non-industrial private forest owners’ willingness to manage for climate change and biodiversity. Scand. J. For. Res. **36**(7–8): 614–625. doi:10.1080/02827581.2021.1981433.

Hyyrynen, M., Ollikainen, M., and Seppälä, J. 2023. European forest sinks and climate targets: past trends, main drivers, and future forecasts. Eur. J. For. Res. **142**(5): 1207–1224. doi:10.1007/s10342-023-01587-4.

IPBES. 2024. Summary for policymakers of the thematic assessment of the interlinkages among biodiversity, water, food and health (nexus assessment) of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. *Edited by* P.D. McElwee.

Jactel, H., Bauhus, J., Boberg, J., Bonal, D., Castagneyrol, B., Gardiner, B., et al. 2017. Tree diversity drives forest stand resistance to natural disturbances. Current For. Rep. **3**(3): 223–243. doi:10.1007/s40725-017-0064-1.

Johansson, T., Gibb, H., Hjältén, J., Pettersson, R.B., Hilszczański, J., Alinvi, O., and Danell, K. 2007. The effects of substrate manipulations and forest management on predators of saproxylic beetles. For. Ecol. Manage. **242**(2–3): 518–529. doi:10.1016/j.foreco.2007.01.064.

Kallio, A.M., and Solberg, B. 2018. Leakage of forest harvest changes in a small open economy: case Norway. Scand. J. For. Res. **33**(5): 502–510. doi:10.1080/02827581.2018.1427787.

Kangas, J., and Ollikainen, M. 2023. Reforming a pre-existing biodiversity conservation scheme: promoting climate co-benefits by a carbon payment. Ambio, 1–14. doi:10.1007/s13280-023-01833-4.

Kangas, J., Kullberg, P., Pekkonen, M., Kotiaho, J.S., and Ollikainen, M. 2021. Precision, applicability, and economic implications: a comparison of alternative biodiversity offset indexes. Environ. Manage. **68**(2): 170–183. doi:10.1007/s00267-021-01488-5. PMID: 34100133.

Kangas, J., Majasalmi, T., Juva, K., Kotiaho, J.S., and Ahlvik, L. 2023. Suomen luonnon tila ja tulevaisuus—skenaariotarkastelu luontokadon pysäyttämiseksi vaadittavista toimista. Suomen Luontopaneelin Julkaisuja 4B/2023. Available from http://hdl.handle.net/10138/611393 [accessed December 2025].

Karppinen, H., Hänninen, M., and Valsta, L. 2018. Forest owners’ views on storing carbon in their forests. Scand. J. For. Res. **33**(7): 708–715. doi:10.1080/02827581.2018.1480800.

Khanal, P.N., Grebner, D.L., Munn, I.A., Grado, S.C., Grala, R.K., and Henderson, J.E. 2017. Typology of nonindustrial private forest landowners and forestry behavior: implications for forest carbon sequestration in the southern US. Small-Scale For. **16**: 419–434. doi:10.1007/s11842-017-9363-4.

Koivula, M., Louhi, P., Miettinen, J., Nieminen, M., Piirainen, S., Punttila, P., and Siitonen, J. 2022. Talousmetsien luonnonhoidon ekologisten vaikutusten synteesi: Synteesiraportti. Luonnonvara—ja biotalouden tutkimus 60/2022. Available from http://urn.fi/URN:ISBN:978-952-380-472-2 [accessed December 2025].

Koljonen, T., Silfver, T., Soimakallio, S., Koreneff, G., Lehtilä, A., Markkanen, J., and Koskivaara, O. 2024. Perusskenaariot energia-ja ilmastotoimien kokonaisuudelle kohti päästöttömyyttä (PEIKKO) [In Finnish with English abstract. English title: Baseline scenarios for energy and climate policy package towards zero emissions.]. Available from http://urn.fi/URN:ISBN:978-952-383-219-0 [accessed December 2025].

Kotiaho, J.S., Kuusela, S., Nieminen, E., Päivinen, J., and Moilanen, A. 2016. Framework for assessing and reversing ecosystem degradation—Report of the Finnish restoration prioritization working group on the options and costs of meeting the Aichi biodiversity target of restoring at least 15 percent of degraded ecosystems in Finland. Available from https://urn.fi/URN:ISBN:978-952-11-4603-9 [accessed December 2025].

Kuuluvainen, T. 2009. Forest management and biodiversity conservation based on natural ecosystem dynamics in northern Europe: the complexity challenge. Ambio, **38**(6): 309–315. doi:10.1579/08-A-490.1.

Kuuluvainen, T., and Gauthier, S. 2018. Young and old forest in the boreal: critical stages of ecosystem dynamics and management under global change. For. Ecosyst. **5**: 26. doi:10.1186/s40663-018-0142-2.

Kuuluvainen, T., Lindberg, H., Vanha-Majamaa, I., Keto-Tokoi, P., and Punttila, P. 2019. Low-level retention forestry, certification, and biodiversity: case Finland. Ecol. Processes, **8**(1): 1–13. doi:10.1186/s13717-019-0198-0.

Lintunen, J., and Uusivuori, J. 2016. On the economics of forests and climate change: deriving optimal policies. J. For. Econ. **24**: 130–156. doi:10.1016/j.jfe.2016.05.001.

Lintunen, J., Laturi, J., and Uusivuori, J. 2015. Finnish Forest and Energy Policy model (FinFEP)—a model description. Natural Resources and Bioeconomy Studies 29/2015. Available from https://urn.fi/URN:ISBN:978-952-326-119-8 [accessed December 2025].

Lintunen, J., Laturi, J., and Uusivuori, J. 2016. How should a forest carbon rent policy be implemented? For. Policy Econ. **69**: 31–39. doi:10.1016/j.forpol.2016.04.005.

López, L.N., Sjolie, H.K., Nabhani, A., and Aguilar, F.X. 2024. Impacts of biodiversity and carbon policies on the management of Norwegian forest and its ecosystem services. Silva Fenn. **58**(4):. doi:10.14214/sf.23067.

Markowski-Lindsay, M., Stevens, T., Kittredge, D.B., Butler, B.J., Catanzaro, P., and Dickinson, B.J. 2011. Barriers to Massachusetts forest landowner participation in carbon markets. Ecol. Econ. **71**: 180–190. doi:10.1016/j.ecolecon.2011.08.027.

Mathiesen, L. 1985. Computational experience in solving equilibrium models by a sequence of linear complementarity problems. Oper. Res. **33**(6): 1225–1250. doi:10.1287/opre.33.6.1225.

Matthews, S., O'Connor, R., and Plantinga, A.J. 2002. Quantifying the impacts on biodiversity of policies for carbon sequestration in forests. Ecol. Econ. **40**: 71–87. doi:10.1016/S0921-8009(01)00269-5.

Matveinen, K., Lilja-Rothsten, S., Junninen, K., Bäckman, M., Eteläaho, E., Kajander, L., et al. 2015. Metsäelinympäristöt. *In* Elinympäristöjen tilan edistäminen Suomessa. ELITE-työryhmän mietintö elinympäristöjen tilan edistämisen priorisointisuunnitelmaksi ja arvio suunnitelman kokonaiskustannuksista. *Edited by* J.S. Kotiaho, S. Kuusela, E. Nieminen, and J.J. Päivinen. Suomen ympäristö 8/2015. Available from http://urn.fi/URN:ISBN:978-952-11-4462-2 [accessed December 2025].

Metsähallitus. 2018. Metsien ennallistamisen ja luonnonhoidon asiantuntijaryhmä Metsä-ELO: Tuli Suomen metsissä -teemakokous [Expert group on forest restoration and nature management: thematic meeting "Fire in Finnish forests"]. Available from https://tapio.fi/wp-content/uploads/2019/10/MetsaELO_Tuli-Suomen-metsissa_muistio.pdf [accessed December 2025].

Ministry of Economic Affairs and Employment of Finland. 2022. Carbon neutral Finland 2035—national climate and energy strategy. Available from http://urn.fi/URN:ISBN:978-952-327-843-1 [accessed December 2025].

Mönkkönen, M., Juutinen, A., Mazziotta, A., Miettinen, K., Podkopaev, D., Reunanen, P., and Tikkanen, O.P. 2014. Spatially dynamic forest management to sustain biodiversity and economic returns. J. Environ. Manage. **134**: 80–89. doi:10.1016/j.jenvman.2013.12.021.

Nirhamo, A., Pykälä, J., Jääskeläinen, K., and Kouki, J. 2023. Habitat associations of red-listed epiphytic lichens in Finland. Silva Fenn. **57**(1): article id 22019. doi:10.14214/sf.22019.

Pan, W., Kim, M.K., Ning, Z., and Yang, H. 2020. Carbon leakage in energy/forest sectors and climate policy implications using meta-analysis. For. Policy Econ. **115**: 102161. doi:10.1016/j.forpol.2020.102161.

Perrels, A., Haakana, J., Hakala, O., Kujala, S., Láng-Ritter, I., Lehtonen, H., et al. 2022. Assessment of the cost of inaction regarding climate change (KUITTI). Publications of the Government's analysis, assessment and research activities 2022:37. Prime Minister's Office, Helsinki.

Phelps, J., Webb, E.L., and Adams, W.M. 2012. Biodiversity co-benefits of policies to reduce forest-carbon emissions. Nat. Clim. Change **2**(7): 497–503. doi:10.1038/nclimate1462.

Pigou, A. 1920. The economics of welfare. Macmillan and Company, London.

Pilli, R., Grassi, G., Kurz, W.A., Fiorese, G., and Cescatti, A. 2017. The European forest sector: past and future carbon budget and fluxes under different management scenarios. Biogeosciences, **14**(9): 2387–2405. doi:10.5194/bg-14-2387-2017.

Plantinga, A.J., and Birdsey, R.A. 1994. Optimal forest stand management when benefits are derived from carbon. Nat. Resour. Model. **8**(4): 373–387. doi:10.1111/j.1939-7445.1994.tb00190.x.

Pohjola, J., Laturi, J., Lintunen, J., and Uusivuori, J. 2018. Immediate and long-run impacts of a forest carbon policy—a market-level assessment with heterogeneous forest owners. J. For. Econ. **32**: 94–105. doi:10.1016/j.jfe.2018.03.001.

Pörtner, H.O., Scholes, R.J., Arneth, A., Barnes, D.K.A., Burrows, M.T., Diamond, S.E., and Val, A.L. 2023. Overcoming the coupled climate and biodiversity crises and their societal impacts. Science, **380**(6642): eabl4881. doi:10.1126/science.abl4881.

Purvis, A., and Hector, A. 2000. Getting the measure of biodiversity. Nature, **405**(6783): 212–219. doi:10.1038/35012221. PMID: 10821281.

Rautiainen, A., Lintunen, J., and Uusivuori, J. 2018. Market-level implications of regulating forest carbon storage and albedo for climate change mitigation. Agric. Resour. Econ. Rev. **47**(2): 239–271. doi:10.1017/age.2018.8.

Rautiainen, A., Pohjola, J., Assmuth, A., and Lintunen, J. 2025. Side effects of a forest carbon rent policy on soil GHG emissions and nutrient export from forests. Forestry.doi:10.1093/forestry/cpaf082.

Riahi, K., Van Vuuren, D.P., Kriegler, E., O'Neill, B., and Rogelj, J. 2016. The shared socio-economic pathways (SSPs): an overview. Available from https://unfccc.int/sites/default/files/part1_iiasa_rogelj_ssp_poster.pdf [accessed December 2025].

Seddon, N., Daniels, E., Davis, R., Chausson, A., Harris, R., Hou-Jones, X., et al. 2020. Global recognition of the importance of nature-based solutions to the impacts of climate change. Global Sustainability, **3**: e15. doi:10.1017/sus.2020.8.

Sedjo, R., and Sohngen, B. 2012. Carbon sequestration in forests and soils. Ann. Rev. Resour. Econ. **4**(1): 127–144. doi:10.1146/annurev-resource-083110-115941.

Seidl, R., Thom, D., Kautz, M., Martin-Benito, D., Peltoniemi, M., Vacchiano, G., et al. 2017. Forest disturbances under climate change. Nat. Clim. Change, **7**(6): 395–402. doi:10.1038/nclimate3303.

Siitonen, J. 2001. Forest management, coarse woody debris and saproxylic organisms: Fennoscandian boreal forests as an example. Ecol. Bull. 11–41. Available from https://www.jstor.org/stable/20113262 [accessed December 2025].

Sjølie, H.K., Latta, G.S., and Solberg, B. 2013. Potentials and costs of climate change mitigation in the Norwegian forest sector—does choice of policy matter? Can. J. For. Res. **43**(6): 589–598. doi:10.1139/cjfr-2012-0457.

Sohngen, B., and Mendelsohn, R. 2003. An optimal control model of forest carbon sequestration. Am. J. Agric. Econ. **85**(2): 448–457. doi:10.1111/1467-8276.00133.

Sohngen, B., and Sedjo, R. 2006. Carbon sequestration in global forests under different carbon price regimes. Energy J. **27**(3_suppl): 109–126. doi:10.5547/ISSN0195-6574-EJ-VolSI2006-NoSI3-6.

Tahvonen, O. 1995. Net national emissions, CO2 taxation and the role of forestry. Resour. Energy Econ. **17**(4): 307–315. doi:10.1016/0928-7655(95)00002-X.

Tahvonen, O., Suominen, A., Parkatti, V.P., and Malo, P. 2024. Optimizing high-dimensional forestry for wood production and carbon sinks. Can. J. For. Res. **54**: 877–894, doi:10.1139/cjfr-2023-0267.

Thompson, I.D., Okabe, K., Tylianakis, J.M., Kumar, P., Brockerhoff, E.G., Schellhorn, N.A., and Nasi, R. 2011. Forest biodiversity and the delivery of ecosystem goods and services: translating science into policy. Bioscience, **61**(12): 972–981. doi:10.1525/bio.2011.61.12.7.

Tinbergen, J. 1952. On the theory of economic policy. 2nd ed. Amsterdam, North Holland.

Tuomi, M., Laiho, R., Repo, A., and Liski, J. 2011a. Wood decomposition model for boreal forests. Ecol. Modell. **222**(3): 709–718. doi:10.1016/j.ecolmodel.2010.10.025.

Tuomi, M., Rasinmäki, J., Repo, A., Vanhala, P., and Liski, J. 2011b. Soil carbon model Yasso07 graphical user interface. Environ. Modell. Software, **26**(11): 1358–1362. doi:10.1016/j.envsoft.2011.05.009

Tzemi, D., Leppänen, J., Autto, H., and Haltia, E. 2025. Finnish forest owners' willingness to participate in result-based carbon offset schemes. For. Policy Econ. **177**: 103531. doi:10.1016/j.forpol.2025.103531.

van Kooten, G.C., Binkley, C.S., and Delcourt, G. 1995. Effect of carbon taxes and subsidies on optimal forest rotation age and supply of carbon services. Am. J. Agric. Econ. **77**(2): 365–374. doi:10.2307/1243546.

Venter, O., Laurance, W.F., Iwamura, T., Wilson, K.A., Fuller, R.A., and Possingham, H.P. 2009. Harnessing carbon payments to protect biodiversity. Science, **326**(5958): 1368–1368. doi:10.1126/science.1180289. PMID: 19965752.