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Original Article



# Side effects of a forest carbon rent policy on soil GHG emissions and nutrient export from forests

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## Abstract

Member states of the European Union (EU) have targets to increase net $CO_2$ removals in the Land Use, Land Use Change and Forestry sector. As Finland threatens to fall short of its target, measures to strengthen its forest $CO_2$ sink have been called for. One option would be to implement a carbon rent policy to incentivize increased carbon storage in standing timber and wood products. However, such a policy might have unintended side effects on nutrient loads in runoff from forests, and soil GHG emissions from forests on peatlands. Here, we assess these side effects using FinFEP, a dynamic partial-equilibrium model of the Finnish forest and energy sectors. Our results suggest that a carbon rent policy could be expected to decrease nitrogen (N) and phosphorus (P) loads in the short run, but not in the long run. Net GHG emissions from the decomposition of peat could be expected to increase, but the increase would be small compared to the policy's overall effect on the forest $CO_2$ sink. Hence, neither of these examined side effects appears to provide a reason to reject the policy.

*Keywords:* nutrient export; carbon rent; climate policy; LULUCF; boreal forest; organic soil; GHG emissions

## Introduction

Climate policy challenges the continuation of the prevailing practices of forest management and utilization in Finland. As a member of the European Union (EU), Finland has a binding target to increase net $CO_2$ removals in the Land Use, Land Use Change and Forestry (LULUCF) sector beyond the previously observed level (European Parliament and Council of the European Union 2023). The country, however, threatens to fall short of this target (Haakana et al. 2023, Hyyrynen et al. 2023) and, hence, measures to strengthen the forest sink have been called for (Finnish Climate Change Panel 2023). One option would be to subsidize the maintenance of forest biomass carbon stocks through a carbon rent policy (Sohngen and Mendelsohn 2003, Lintunen et al. 2016). Paying carbon rent would internalize the value of forest carbon storage into private forest management decisions. It would make managing forests jointly for timber and carbon more profitable than managing them for timber only and, hence, give landowners an incentive to reoptimize their forest management. At the market level, the policy would decrease annual harvest volumes and the share of timber obtained from final-felled sites (Pohjola et al. 2018). These changes would strengthen the forest $CO_2$ sink, but they could also have unintended side effects. That is, changes in forest management activities, such as final felling and thinning, could alter nutrient loads in runoff (Laurén et al. 2021, Nieminen et al. 2023) and greenhouse gas (GHG) emissions (Ojanen et al. 2010, Ojanen and Minkkinen 2019, Minkkinen et al. 2020, Korkiakoski et al. 2023) from boreal peatland forest soils. The nutrients (nitrogen, N, and phosphorus, P) accumulate in waterbodies and cause eutrophication. The GHGs (carbon dioxide, $CO_2$, methane, $CH_4$, and nitrous oxide, N2O) accumulate in the atmosphere and contribute to climate change. In this article, we assess the magnitude of these side effects and discuss their implications regarding the implementation of a carbon rent policy.

A carbon rent policy's *net* effects on nutrient export and soil GHG emissions are not evident *a priori*. Subsidizing carbon storage tends to decrease annual harvests compared to a business-as-usual baseline, at least in the short-run (Lintunen and Uusivuori 2016, Pohjola et al. 2018, Rautiainen et al. 2018). When harvests are decreased, forests grow older and denser and accumulate more timber and carbon. Nevertheless, *decreased harvests* and *the increased accumulation of standing timber volume* tend to force nutrient loads in runoff in opposite directions—and, therefore, establishing the policy's net effect requires determining which effect dominates. Logging operations temporarily increase N and P loads in runoff both on mineral and organic soils (Finér et al. 2010, Nieminen et al. 2023). Hence, decreasing harvests tends to decrease the loads. However, on the other hand, increasing the amount of standing timber tends to increase the loads through its effect on evapotranspiration. More trees consume more water and lower the water table during the growing season (Nieminen et al. 2018). This allows more peat to dry, decompose, and release nutrients (Ojanen and Minkkinen 2019) which are then washed away by seasonal water level fluctuations. The same process affects GHG emissions from peat soils. Lowering the water table increases $CO_2$ and $N_2O$ emissions from the aerobic decomposition of dry peat (Ojanen and Minkkinen 2019, Minkkinen et al. 2020), but decreases $CH_4$ emissions from anaerobic decomposition (Ojanen et al. 2010). Hence, there is a trade-off between the GHGs and—here too—the policy's net effect depends on which effect dominates.

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Examining a carbon rent policy's effects on nutrient export and soil GHG emissions is important for two reasons. First, peatlands are common in Finland. Undrained mires and drained peatlands together make up more than a third of the country's productive *forestry land* (Kulju et al., 2023). Due to their large extent, even relatively modest changes in nutrient loads in runoff and soil GHG emissions per peatland hectare could amount to notable impacts within a catchment area or at the national level. Furthermore, Finland is not the only country with extensive peatlands. As controlling peatland GHG emissions globally could play an important role in climate change mitigation efforts (Leifeld and Mechetti, 2018), our results are—to a certain extent—also relevant in a broader geographical context. Second, nutrient loads in runoff and soil GHG emissions are difficult and expensive to monitor at the stand or forest estate-level. Hence, their regulation by result-based policy instruments that require accurate measurement (e.g. taxes on soil GHG emissions) seems unfeasible at present. If a carbon rent policy is implemented, it may be therefore practical to disregard these side effects: i.e. subsidize carbon storage in living trees and wood products (which is easier to measure)—and leave nutrient export and emissions from decomposing peat unregulated. In this context, it is important to quantify the policy's effect on soil emissions and nutrient export from forests. Notable negative side effects could speak against the policy, whereas positive ones could provide an argument in its favor.

We conduct our analysis using the Finnish Forest and Energy Policy model (FinFEP) (Lintunen et al. 2015). Previously, Pohjola et al. (2018) have examined the effects of a carbon rent policy using an earlier version of the same model. Here, we focus on the side effects of the carbon rent policy (i.e. nutrient loads and soil GHG emissions)—which is a new, previously unaddressed, topic. To this end, we have augmented FinFEP by a peatland water table model (Sarkkola et al. 2010), a description of processes driving nutrient loads in runoff from organic and mineral soils (Nieminen et al. 2023) and peatland soil GHG emission models (Ojanen et al. 2010, Ojanen and Minkkinen 2019, Minkkinen et al. 2020).

# Methods

## FinFEP model

This section provides a general overview of the Finnish Forest and Energy Policy model (FinFEP). The aim is to outline the big picture of how the model works and what mechanisms drive the results presented in this article specifically. A detailed and more extensive model description is provided in Lintunen et al. (2015).

FinFEP is a partial equilibrium model of the Finnish forest and energy sectors. Wood supply is determined endogenously in the model. The initial state of forest resources is based on national forest inventory data. Annual harvests are determined by rules depicting (profit or utility maximizing) landowner behavior. Harvests and growth determine how forest resources develop over time. Likewise, the demand for wood (of different timber grades) is determined endogenously in the model. It is driven by the input use decisions of the producers of final goods in the forest and energy sectors. The demands for the produced final goods, in turn, are depicted by exogenous demand functions. The production technologies relating input use to the production of final goods are described in detail. The forest and the energy sector are both included in the model, as they both use wood and as they are partly integrated (e.g. pulp mills may also generate heat and power). Material flows in the model are shown in Fig. 1.

Forest resources at the regional level are stratified by soil type, site quality, species, age, and owner type. Total harvests (of each timber grade of each species) in the market-level model are determined by harvesting rules that depict aggregate landowner behavior. The rules have been developed by first defining preferences for different landowner types, then simulating a large quantity of management decisions of individual landowners under uncertainty of future timber prices, and finally aggregating data on individual decisions into rules that consistently depict the combined behavior of all forest owners in alternative circumstances. The landowner types—and the effects of carbon rent on their management decisions—is described below in detail.

All landowners are assumed to apply even-aged management (which is currently the predominant practice in Finland). The owners choose when to thin or fell a stand, and if stands are thinned, the owners also choose how intensively. Each owner aims to maximize the combined present value of profits (net of taxes and subsidies) and private amenity values that they derive from their forest (Hartman 1976). FinFEP includes three landowner types who differ in terms of their valuation of amenities. The first type is the *Faustmannian* landowner, who does not value amenities—and, thus, maximizes profit only (as assumed in Faustmann (1849) and Samuelson (1976)). *Hartmanian* forest owners, on the contrary, value both profits and amenities (as in Hartman 1976). There are two Hartmanian types in the model: strong and mild Hartmanians. The former give more weight to amenity values than the latter.

The landowner's optimization problem is set up as follows. Let $V(s_t)$ denote the land value function. Its argument, $s_t$, is a state vector that contains current timber prices, i.e. pulpwood and sawlogs, and variables that characterize the state of the stand, i.e. stand age, number of trees, average tree volume, and volume distribution width. Prices follow exogenous mean-reverting random process, whereas the stand characteristics have endogenous dynamics. Let $R_f(s_t, x_t)$ denote the landowner's payoff function, where $f$ denotes the owner's type (i.e. each owner type has a different payoff function). The payoff function depicts net income and amenity value derived from the stand in period $t$. The function's second argument, $x_t$, is a control vector that depicts management options, i.e. thinning intensity (which may be 0, if the stand is not thinned) and binary final-felling decision variable (i.e. whether the stand is felled or not). Let $\beta = (1+r)^{-1}$ denote the discrete-time discount factor (where $r$ is the interest rate 2.5%). Last, let $E_t$ denote the conditional expectation operator (as the development of timber prices is depicted by a random Markov process, the expectations depend on information that is currently available in each period). The landowner's optimization problem is summarized by the Bellman equation

$$V(s_t) = \max_{x_t} \left( R_f(s_t, x_t) + \beta E_t V(s_{t+1}) \right). \tag{1}$$

constrained by state dynamics $s_{t+1} = g(s_t, x_t)$ (Lintunen et al. 2015). The landowner chooses management activities, $x_t$, that maximize the sum of current net income and amenity value, $R_f(s_t, x_t)$, and expected land value, $\beta E_t V(s_{t+1})$. The expected land value depends on the future state of the world, $s_{t+1}$, which includes the future state of the stand (which is affected by current management activities, $x_t$) and future prices (which follow a random process). The current land value, $V(s_t)$, is the present value of the infinite future stream of income and amenity value obtained from the stand under optimal management.

The payoff function, $R_f(s_t, x_t)$, comprises three elements: net timber income, $\pi(s_t, x_t)$, carbon income, $\sigma(s_t, x_t)$, and amenity value, $\alpha(s_t, x_t)$. The amenity value is based on the after-harvest

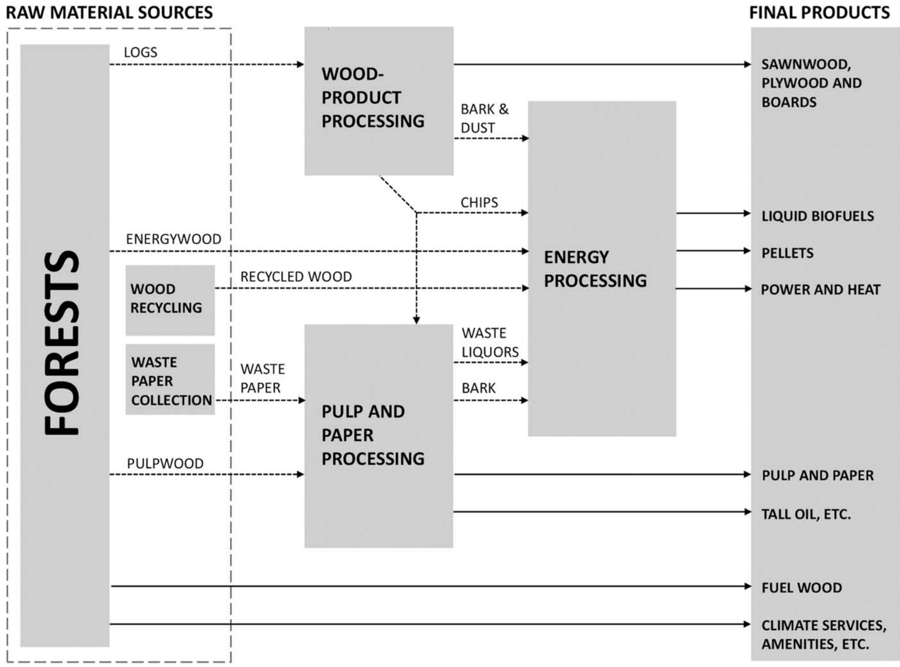


**Figure 1.** Material flows in the FinFEP model.

number of trees times the square of average tree volume. The weight assigned to amenity value, $w_f$, depends on the landowner type, $f$. For Faustmannian owner's $w_f = 0$; for Hartmanian owners $w_f > 0$. The weights were chosen in model calibration to rationalize the age distribution observed in NFI. Due to the calibration process, the weights also vary between regions, site-classes, and tree species. Typically, $w_f \approx 1$. Combining these elements, we obtain

$$R_f(s_t, x_t) = \pi(s_t, x_t) + \sigma(s_t, x_t) + w_f \alpha(s_t, x_t). \tag{2}$$

Net timber income, $\pi(s_t, x_t)$, depends on harvest volume. Hence, thinning and felling contribute to it positively. Carbon income, however, is negatively affected by harvests, as they remove carbon from the forests. Likewise, also amenities are negatively affected by harvests.

Carbon income, $\sigma(s_t, x_t)$, consists of *carbon rent* payments to the landowner. Carbon rent is an annual subsidy for maintaining forest carbon stocks. In principle, these stocks may be defined broadly (to include carbon in standing timber, other tree biomass, deadwood, and soils) or narrowly (to include, e.g. only carbon contained in standing timber). In FinFEP, the stocks are defined narrowly. Hence, the subsidy depends on the stock of carbon in standing timber ($tCO_2$), $S(s_t, x_t)$, the carbon price (€ $tCO_2{}^{-1}$), $p_c$, and the interest rate, $r$, so that

$$\sigma(s_t, x_t) = r p_c\, S(s_t, x_t). \tag{3}$$

The above formulation of carbon rent is valid when the carbon price is time-invariant, as it is in FinFEP. (If a time-variant carbon price is used, the rate of change in price must also be taken into account, see Lintunen et al. [2016]).

Carbon rent policies (e.g. Pohjola et al. [2018], Sohngen and Mendelsohn [2003]) differ from flux-pricing policies (e.g. van Kooten et al. [1995], Lintunen and Uusivuori [2016], Rautiainen et al. [2017]) in terms of how the incentives for carbon storage are formulated. Carbon rent policies subsidize stocks but do not price fluxes. Flux-pricing policies price fluxes, but do not subsidize stocks. Nevertheless, under standard assumptions, both approaches can be applied to incentivize identical landowner behavior (Lintunen et al. 2016). Subsidizing carbon storage in either way tends to usually lighten harvests (Niinimäki et al. 2013, Pihlainen et al. 2014, Pohjola et al. 2018) and reduce the share of timber from final felling at the market level (Pohjola et al. 2018). The carbon rent policy in this article is implemented in the same way as in Pohjola et al. (2018).

The policy function $x_f(s_t)$ characterizes landowner type $f$'s optimal solution to the optimization problem (1) in state of the world, $s_t$. As $s_t$ develops over time, the function therefore describes the landowner type's optimal thinning and final-felling decisions in any conceivable circumstances (i.e. with all combinations of stand state and timber prices). The dynamic program (1) is solved numerically using discretized state-space with linear interpolation. The problem is large as the policy function has six dimensions. Hence, it is difficult to illustrate or directly apply in the forest sector model. Instead, we approximate the policy function using a separate parametric functions for thinning and clearfelling decisions. We simulate the stand-level model to generate data on decisions, $x_t$, given the states, $s_t$. Then we separately regress thinning intensity and the probability to clearfell on the state. In the timber market-model, we assume that all the stands in the same state are thinned equally, as indicated by the regressed function. In the case of clearfelling, we interpret the probabilities as area shares of stands that are clearfelled. Hence, the regression functions depict a type of aggregation of behavioral rules. Using these stand level thinning and clearfelling relations together with the current state of forest stands, we construct micro-founded timber supply functions for the market-level model. The details of this procedure are described in Lintunen et al. (2015).

The demand for final goods is represented by exogenous demand functions. The goods are produced by representative firms that maximize the sum of current and future profits by optimizing their input use and capacity investments. Input use in each period is optimized within the limits of existing production capacity. Capacity investments compensate for depreciation and expand capacity over time. The investment decisions are made subject to restricted planning horizon (rather than under perfect foresight).The derived demand for timber (of different species and timber grades) and other intermediate goods (such as energy) arises from the input use optimization of the domestic representative firms. In addition, the model has exogenous demand components, such as, export demand and unmodelled domestic demand, that are depicted by demand functions. Similarly, the supply of intermediate goods is a result of endogenous production decisions and exogenous supply functions, e.g. imports. The markets of both intermediate and final goods are depicted in the model. Hence, equilibrium prices and traded quantities are determined endogenously, by balancing the sums of all supply and demand components.

The climate policy examined in this article affects the input use of representative firms through two mechanisms. The first mechanism is the pricing fossil $CO_2$ emissions, which affects fossil fuel use decisions. We assume that firms need to pay the same price for all fossil $CO_2$ emissions. The price is equal to the price of the EU-ETS allowance. (In addition to the carbon price, the wood and energy industries depicted in the model are subject to the standard taxes and regulations currently applied in Finland; for example, fuel use in heat production is subject to excise taxes). The second mechanism is a subsidy for storing carbon in HWP. The subsidy is paid to firms that produce long-lived wood products. Its size, $\sigma_{\mathrm{HWP}}$, depends on the amount of carbon stored into the product, $\rho_{\mathrm{HWP}}$, the decay rate of the HWP fraction, $\delta_{\mathrm{HWP}}$, the carbon price, $p_c$, and the discount rate, $r$, so that

$$\sigma_{\mathrm{HWP}} = \frac{r}{r+\delta_{\mathrm{HWP}}} p_c \rho_{\mathrm{HWP}}. \tag{4}$$

Notably, the HWP carbon subsidy is based on the same time-invariant carbon price, as the carbon rent paid to the forest owners. The carbon rent policy applied to the forests, Eq. (3), incentivizes the same behavior as a flux-pricing policy in which harvests are taxed as if the $CO_2$ contained in the wood were oxidized immediately (Lintunen et al. 2016). Hence, to avoid the double counting of emissions, wood fuels are treated as emission free in energy use. Likewise, storing carbon in HWP is seen as a carbon sequestration activity that is subsidized (4), as it prevents some of the $CO_2$ that 'has already been paid for' at harvest from entering the atmosphere (Lintunen and Uusivuori 2016, Lintunen et al. 2016b). From the point-of-view of incentivizing optimal carbon storage, the HWP subsidy is needed to counteract a part of the carbon rent policy's effect. That is, if only carbon storage in standing timber is subsidized, the policy decreases harvests and carbon storage in HWP too strongly. Subsidizing also storage in HWP softens the effect on harvests, as it implies that carbon storage in both forests and wood products is valued in the same way. Additionally, the HWP subsidy directs wood use towards products that provide long-lived carbon storage.

Data structure and sources of the FinFEP model are described in Lintunen et al. (2015) and Pohjola et al. (2018). However, the model has also been updated in many ways after the publication of the descriptions. The forest data is updated to correspond NFI data from years 2018–2022. The forest and energy sector installations are now based on years 2023–2024 and new energy technologies have been added. The model calibration is based on the years 2019–2023, but given the rapid change in the energy sector, the emphasis is on the latest years. In addition, wood imports have been reduced permanently to match up with the current trade ban situation between Finland and Russia.

The following section summarizes the specific policy scenarios that are analyzed in this study. The sections thereafter describe how runoff processes and soil GHG emissions were added to the FinFEP model. Capturing these effects requires modelling how the water table is affected by forest management. Hence, a water table model is introduced in the next subsection. Runoff and emissions are discussed thereafter. In FinFEP, the water table model was implemented in detail for sites at the same representative locations that are used to calibrate the model's forest growth descriptions. Nutrient loads in runoff and GHG emissions based on the water table model were then generalized to the regional level in the same way as the growth descriptions. (In Lintunen et al. (2015), the representative sites, for which growth descriptions are available, are listed in Table 4 and the growth descriptions applied for different site types in each NUTS3-region are shown in Table 5).

## Policy scenarios

We demonstrate the effects of a carbon rent policy by comparing two scenarios: a policy scenario, in which carbon rent and a HWP subsidy are implemented, and a business-as-usual (BAU) scenario, in which they are not. The two scenarios are otherwise similar. The demand for final goods and the regulation of fossil emissions develop in the same way. All fossil $CO_2$ emissions are priced according to the emission allowance price in the EU Emissions Trading Scheme (EU ETS). The $CO_2$ price starts at 90€ $tCO_2^{-1}$ in the base year (2020) and rises at a rate of 2% $yr^{-1}$. In addition to the uniform $CO_2$ price, the industries are subject to all other relevant climate and energy -related taxes and regulations that are currently applied in Finland.

In the policy scenario, the carbon rent (paid to landowners) and the HWP subsidy (paid to the users of wood) are both based on a carbon price of 15€ $tCO_2^{-1}$. The policy is implemented in the same way as in Pohjola et al. (2018). However, unlike Pohjola et al. (2018) we apply different $CO_2$ prices to fossil emissions (90€ $tCO_2^{-1}$) and forest carbon (15€ $tCO_2^{-1}$). We do so, as we assume that the adoption of the carbon rent policy is motivated by the need to meet EU targets for LULUCF sector removals. Our results suggest that even a low subsidy suffices to strengthen the sink enough for this purpose. Furthermore, basing the carbon rent on high $CO_2$ price could cause large market disruptions (Pohjola et al. 2018). Applying a low carbon price can help avoid market disruptions and thereby increase the social acceptability of the scheme (Assmuth et al. 2024).

## Water table

Forests on organic and mineral soils are kept track of separately in FinFEP. The water table model is only applied to forests on organic soils, where it is needed for the calculation of nutrient loads in runoff and soil GHG emissions from the decomposition of peat (as described in the following sections). It is not applied to forests on mineral soils, as modelling nutrient loads in runoff there does not require a water table model, and the decomposition of peat is not a relevant issue on mineral soils.

Water table depth affects the decomposition of peat on drained peatlands. Trees regulate the water table by controlling evapotranspiration. Growth and harvests determine the standing volume—and, hence, the water consumption—of trees. Through this channel, forest management decisions affect the water table depth and, hence, nutrient loads in runoff and GHG emissions from drained peatlands. In addition to forest management, also ditch network maintenance affects the water table, as it affects the flow of water out of the peatland.

Water table depth of drained peatlands in late summer (cm below ground), $W_L$, depends on standing timber volume ($m^3\ ha^{-1}$), $V$, average monthly precipitation during the summer (mm, June–August), $P$, ditch depth (cm), $D$, and latitude (degrees north), $L$. Following Sarkkola et al. (2010), and incorporating the modifications made by Nieminen et al. (2023), we assume

$$W_L = \alpha_1 + \alpha_2 \left(1 - 0.3^{\left(\frac{V}{90}\right)}\right) - \alpha_3 \left(1 - 0.3^{\left(\frac{V}{90}\right)}\right) \ln (P + \alpha_4) + \alpha_5 \ln (D + 1) - \alpha_6 L - \alpha_7 P, \quad (5)$$

where $\alpha_1, \ldots, \alpha_7$ are parameters (see supplementary material, Table S1). Positive values of $W_L$ indicate how far below ground the water table is. $W_L$ is increasing in $V$ and $D$ and decreasing in $P$ and $L$. Following Nieminen et al. (2023), we apply an average ditch depth of 60 cm in our calculations, and model precipitation based on the temperature sum of the growing season (degree days), $T$. I.e.,

$$P = \alpha_1 + \alpha_2 T, \quad (6)$$

where $\alpha_1$ and $\alpha_2$ are parameters (see supplementary material, Table S1). Water table depth in the late summer, $W_L$, is as such utilized as an explanatory variable when modelling nutrient (Nieminen et al. 2023). However, another water table variable (i.e. average water table depth during the growing season (cm), $W_G$) is needed for modelling peatland GHG emissions (Ojanen et al. 2010, Ojanen and Minkkinen 2019, Minkkinen et al. 2020). The water table is at its lowest in late summer. We assume that $W_G = W_L - 10cm$, i.e. that during growing season the water table is on average 10 cm higher than in the late summer.

## Nutrient loads

N and P loads in runoff can be divided into three annual components: baseload, load caused by harvests, and load caused by other management activities. The components are modelled separately for organic and mineral soils.

Baseload is modelled for organic soils only. The baseload is triggered when a peatland site is drained for the first time and nutrients start to leak out at an accelerated rate (Nieminen et al. 2023). A similar effect does not occur on mineral soils. We model baseload N and P loads (kg $ha^{-1}\ yr^{-1}$), $bl_{\rho\tau}$, as function of late-summer water table depth (cm), $W_L$, following Nieminen et al. (2023). That is,

$$bl_{\rho\tau} = \max \left[0, \alpha_{1\rho\tau} \left(\frac{W_L}{100}\right)^2 + \alpha_{2\rho\tau} \left(\frac{W_L}{100}\right)\right], \quad (7)$$

where $\rho$ is the nutrient type (i.e. **N or P**), $\tau$ is the site type, and $\alpha_{1\rho\tau}$ and $\alpha_{2\rho\tau}$ are site-type specific parameter values provided in the supplementary material (Table S2). *(While the above formula is not explicitly provided in* Nieminen et al. [2023], *it is utilized in the calculations therein. The function and its parameters were obtained from the authors.)* The parameters are based on the SUSI model (Laurén et al. 2021). As FinFEP has a 5-year timestep, the baseload during one period, $BL_{\rho\tau} = 5bl_{\rho\tau}$.Harvests trigger additional nutrient loads in runoff on both organic and mineral soils. The harvest type, $\psi \in \{T, F\}$, may be thinning, $T$, or final felling, $F$. Loads from organic soils depend on the amount of removed wood ($m^3\ ha^{-1}$), $H$, regardless of whether the stand is thinned or felled. On mineral soils, only final felling triggers loads; thinning does not. A harvest in period $t$ affects loads in periods $\tilde{t} \geq t$. On organic soils, the effect lasts for only one period (i.e. $\tilde{t} = t$). On mineral soils the effect lasts for two periods (i.e. $\tilde{t} \in (t, t+1)$). Let $HL_{\psi\rho\tau\tilde{t}}$ denote the load (kg $ha^{-1}$) of type $\rho$ caused by a harvest of type $\psi$ in a forest of type $\tau$ during the five-year period $\tilde{t}$. For organic soils

$$HL_{\psi\rho\tau\tilde{t}} = \frac{\alpha_{\rho\tau 1}}{1 + e^{\alpha_{\rho\tau 2} - \alpha_{\rho\tau 3} H}}, \quad (8)$$

where $\psi \in \{T, F\}$, $\rho \in \{N, P\}$, $\tilde{t} = t$ and $\alpha_{\rho\tau 1}$, $\alpha_{\rho\tau 2}$ and $\alpha_{\rho\tau 3}$ are parameters. For mineral soils,

$$HL_{\psi\rho\tau\tilde{t}} = \begin{cases} \alpha_{\rho\tau 1} & when\ \psi = F\ and\ \tilde{t} = t \\ \alpha_{\rho\tau 2} & when\ \psi = F\ \tilde{t} = t + 1 \\ 0 & when\ \psi = T \end{cases}, \quad (9)$$

where $\rho \in \{N, P\}$ and $\alpha_{\rho\tau 1}$ and $\alpha_{\rho\tau 2}$ are parameters. The parameters for Equations [E08] and [E09] were obtained from Nieminen et al. (2023) and are provided for reference in Table S3 in the supplementary material.

Besides harvests, other management activities that increase the export of nutrients include fertilization and ditch network maintenance (Nieminen et al. 2023). However, as these activities are not endogenously optimized in FinFEP, their effects on nutrient export are omitted from the analysis. Fertilization and ditch network maintenance under carbon rent are assumed to remain at the same level as in the baseline. Hence, they can be omitted from analyses, in which the effects of the carbon rent policy are measured as deviations from the baseline (as in Fig. 3 and Fig. 4 in Results).

## Soil GHG emissions

Soil carbon emissions in FinFEP arise from two distinct processes: (i) the decomposition of peat on organic soils, and (ii) the decomposition of dead organic matter in all forests. Below, we discuss how these processes are modelled in our calculations.

The decomposition of peat gives rise to emissions of three GHGs: $CO_2$, $CH_4$, and $N_2O$. The GHGs are distinguished by the subindex $g$. The annual emissions of each gas from organic soils ($g\ m^{-2}\ yr^{-1}$), $E_{g\tau}$, depend on the average depth of the water table during the growing season, $W_G$. (Emissions expressed in $g\ m^{-2}$ can be converted into $t\ ha^{-1}$ by multiplying by the factor 100 $t\ ha^{-1}$ $(g\ m^{-2})^{-1}$) Equation (10) summarizes the $CO_2$, $CH_4$, and $N_2O$ emission functions obtained from Ojanen and Minkkinen (2019), Ojanen et al. (2010), and Minkkinen et al. (2020), respectively:

$$E_{g\tau} = \begin{cases} -\alpha_{g\tau 1} + \alpha_{g\tau 2} W_G & when\ g = CO_2 \\ -\alpha_{g\tau 1} + \alpha_{g\tau 2} e^{-\alpha_{g\tau 3} W_G} & when\ g = CH_4 \\ \left(\alpha_{g\tau 1} + \alpha_{g\tau 2} {W_G}^2\right) e^{-\alpha_{g\tau 3}\Gamma} + \alpha_{g\tau 4} & when\ g = N_2O \end{cases}. \quad (10)$$

$\Gamma$, in the $N_2O$ formula, is the carbon-to-nitrogen ratio. We apply the value $\Gamma = 30$, which is the mean value for drained peatlands from Minkkinen et al. (2020). The values of parameters $\alpha_{g\tau 1}, \ldots, \alpha_{g\tau 4}$ for each gas are provided in Table S4 of the supplementary material. Notably, $CO_2$ emissions depend on site type, i.e. whether the site is nutrient-rich or nutrient-poor.

$CH_4$ and $N_2O$ are converted into equivalent $CO_2$ based on their Global Warming Potential (GWP) (Lashof and Ahuja 1990, IPCC 2021). $GWP_{100}$ is used as the metric in the conversions. $GWP_{100}$ expresses the cumulative radiative forcing caused by 1 tonne of

GHG emitted to the atmosphere relative to that caused by 1 $tCO_2$ over the time horizon of 100 years. The $GWP_{100}$ value of $CH_4$ (of non-fossil origin) is 27 (IPCC 2021). $CH_4$ is a much more potent GHG than $CO_2$, but also decays much faster. The $GWP_{100}$ value of $N_2O$ is 273 (IPCC 2021). $N_2O$ is even more potent than $CH_4$ and remains in the atmosphere for a longer time. Its decay profile is somewhat comparable to that of $CO_2$.

The decomposition of dead organic matter was modelled using the Yasso2007-simulator (Tuomi et al. 2011a, Tuomi et al. 2011b). The decay of foliage and woody fractions in different diameter classes (1 cm, 3 cm, 10 cm, 30 cm, and 50 cm) were modelled separately. Stylized decay profiles, $\sigma_{wt}$, describing the decay of each fraction, $w$, over time, $t \leq 0$, were then developed by fitting a model with three constant decay rate nodes, distinguished by subindex $i$. Letting $a_{wi}$ denote the weight of node $i$ in the decay profile ($\sum_i a_{wi} = 1$), and letting $\delta_i$ denote the decay rate associated with node $i$, the decay profile of fraction $w$ can be expressed as

$$\sigma_{wt} = \sum_i a_{wi}(1 - \delta_i)^t \tag{11}$$

where $\sigma_{wt} \epsilon [0, 1]$ denotes the share of carbon remaining in dead organic matter of type $w$, $t$ years after being deposited into the soil carbon pool. The dead organic matter pools receive inputs from harvesting residues and natural mortality. The inputs are tree components: roots, stumps, stems, branches, bark, and foliage. Inputs of each kind are allocated into the decay pools, $w$, according to size (e.g. branches are allocated between 1 cm and 3 cm fractions). (Notably, however, the soil litter flux generated by living trees and other vegetation is currently not included in the model and, therefore, the soil carbon stocks in the model do not cover the entire soil carbon pool. Hence, the effect of the carbon rent policy on soil carbon stock (Fig. 6) is the effect caused by changes in natural mortality and the generation of logging residues. The potential effects of forest management changes on the litter component of soil carbon stocks on mineral soils are not accounted for.)

## Sensitivity analysis

As the values of certain variables used in the modelling of the water table (e.g. ditch depth and the difference between $W_L$ and $W_G$) are based on expert judgment rather than actual measurement, the accuracy of the modelled water table level is somewhat uncertain.

To test how this uncertainty affects our results, we assess the carbon rent policy's effects on nutrient export and soil GHG emissions in three alternative conditions. That is, in addition to the default conditions (described in the previous sections), we examine two extreme scenarios depicting notably wetter and drier conditions. Both scenarios are plausible, but unlikely.

The extreme scenarios were designed assuming that the models (i.e. the functions and their parameters) depicting the water table, nutrient export, and GHG emissions are correct—but that our assumptions regarding certain key inputs to water table model may be wrong. These key input variables are (i) average ditch depth, (ii) precipitation and, (iii) the difference between $W_L$ and $W_G$. Variables (i)–(ii) affect nutrient export. Variables (i)–(iii) affect GHG emissions from decaying peat.

In the wet scenario, we assume that (i) the average ditch depth is 30 cm (instead of 60 cm, as in the baseline), (ii) precipitation that is 15% higher than in the baseline, and (iii) the difference between $W_L$ and $W_G$ is 20 cm (rather than 10 cm difference, as in the baseline). In the dry scenario, we assume that (i) the average ditch depth is 90 cm (instead of 60 cm, as in the baseline), (ii) precipitation that is 15% lower than in the baseline, and (iii) the difference between $W_L$ and $W_G$ is the same as in baseline (as it would be unrealistic to assume no difference between the growing season average and late summer water table level).

# Results

Implementing the carbon rent policy reduces total annual harvests compared to BAU (Fig. 2A). Relatively, harvests decrease more in Northern Finland than in Southern Finland. In both regions, timber volume harvested by thinning (Fig. 2C) decreases less than that obtained by final felling (Fig. 2D). As harvests decrease throughout the country, forests accumulate more growing stock than they would without the policy (Fig. 2A). I.e., the policy increases carbon storage in forests—as intended. The big picture aligns with the findings of Pohjola et al. (2018).

Carbon rent increases the baseloads of N and P throughout the country (Fig. 3A–D). The gradual increase is due to the gradual accumulation of growing stock (Fig. 2B). More trees mean greater evapotranspiration and drier peat than in the BAU scenario. Nutrients released by the decomposition of the additional dry peat make up the increased baseloads. The effect on the baseload is stronger in the North, where peatlands are more common, and accumulation of growing stock is relatively stronger than in the South (Fig. 2B).

Carbon rent decreases variable N and P loads from organic and mineral soils (Fig. 3A–D). The immediate decrease in the variable loads is caused by the immediate drop in harvests, that occurs when the policy is implemented (Fig. 2A). The effect wears off over time (Fig. 3A–D) as harvests levels converge towards BAU. Harvests can be increased, as greater share of stands start to reach their optimal rotation length under the carbon rent policy. The maturing of the stands shows as an increase the volume of timber obtained from final felled sites (Fig. 2D).

Altogether, carbon rent decreases N and P loads in the short run (Fig. 4). In the case of both nutrients, the strong and immediate decrease in variable load first outweighs the gradual increase in the baseload. However, the net effect is eventually reversed, as the baseload remains above BAU and the difference in the variable load diminishes in the long run.

Carbon rent increases GHG emissions ($CO_2$ eqv.) from the decomposition of peat (Fig. 5). $CO_2$ and $N_2O$ emissions increase, as the water table sinks lower, and more peat decomposes aerobically. At the same time $CH_4$ emissions decrease, as less peat decomposes anaerobically. The change in $CO_2$ emissions is notably larger than the changes in $N_2O$ and $CH_4$ emissions. The differences in the size of the effects can be understood by examining the functions that are used to calculate $CO_2$, $N_2O$ and $CH_4$ based on water table level during the growing season (10). $CO_2$ emissions respond linearly to changes in the water table. That is, the effect of a change in the water table is the same regardless of its initial level. $CH_4$ emissions, on the contrary, are only sensitive to changes in the level of the water table when the water table is very close to the surface. If the water table is low, changes in its level have little effect on $CH_4$ emissions. Hence, a situation in which a moderate change in the water table alters $CO_2$ emissions notably more than $CH_4$ emissions can arise if the water table is already at a relatively low level in the baseline. This effect is most clearly visible in Southern Finland (Fig. 5B), but—to a certain degree—also applies to Northern FInlanf (Fig. 5A), and the country as a whole (Fig. 5C).

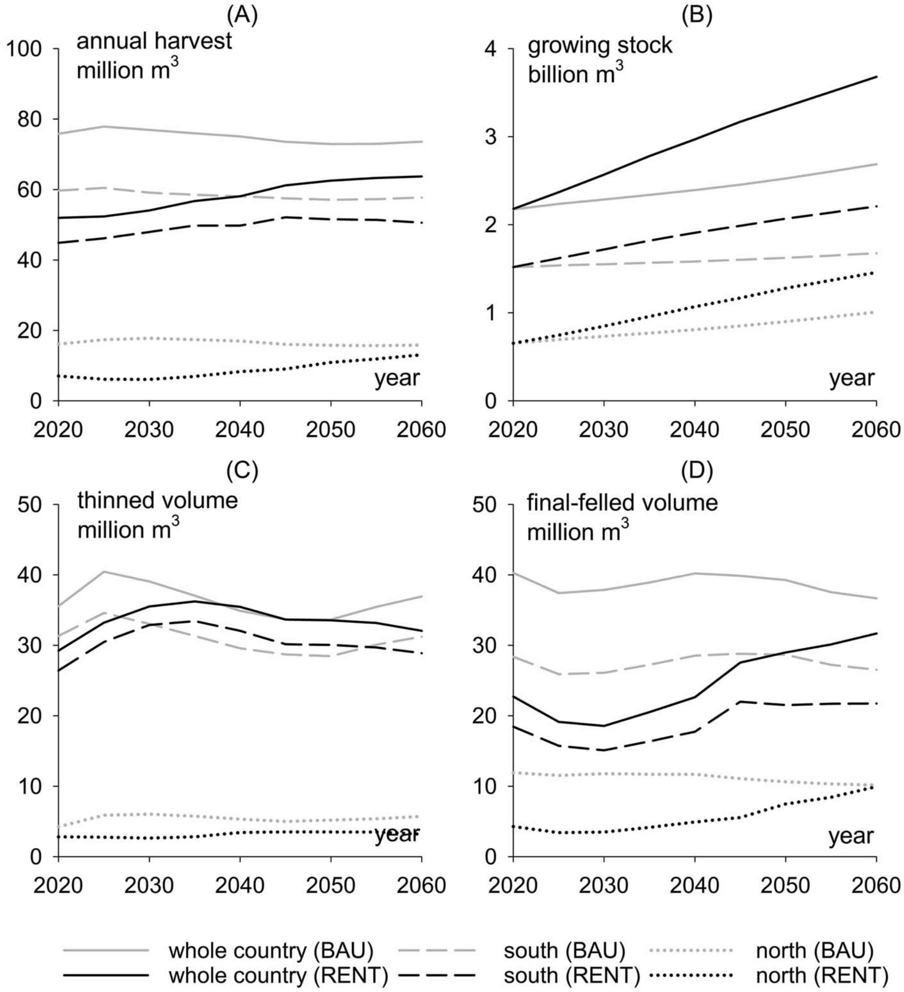


**Figure 2.** Annual wood harvest in Finland (panel A) and the development of growing stock volume in Finnish forests (panel B) in two scenarios: business-as-usual (BAU, gray lines) and a carbon rent scenario in which carbon storage in forests is subsidized based on a 15€ $tCO_2^{-1}$ carbon price (RENT, black lines). The values are shown for the whole country (solid lines), southern Finland (dashed lines), and northern Finland (dotted lines). Total annual harvest (panel A) is disaggregated into harvest volumes by thinning (panel C) and final felling (panel D).

$CO_2$ equivalent conversions in Fig. 5 are based on the $GWP_{100}$ metric. The use of other common metrics could alter the comparisons slightly, but it would not essentially change the qualitative conclusions. For example, the cooling effect of reduced $CH_4$ emissions would be roughly three times stronger, if the $GWP_{20}$ value of $CH_4$ (79.7) were used instead of its $GWP_{100}$ value (27). However, even tripling the strength of $CH_4$ would not make the cooling effect of the decreased CH4 emissions greater than the warming effect of the increased $CO_2$ emissions. The net warming effect of the three GHGs together would of course be smaller in total. In the case of $N_2O$, using its $GWP_{20}$ value (273) would not change the calculations as it is the same as its $GWP_{100}$ value (273) (IPCC 2021). Alternatively, one might suggest using a GWP value based on a longer examination horizon: e.g. $GWP_{500}$ instead of $GWP_{100}$ or $GWP_{20}$. This, however, would only further dampen the role of $N_2O$ emissions—as the $GWP_{500}$ value of $N_2O$ (130) is lower than its $GWP_{100}$ value.

All in all, the net effect of carbon rent on total GHG emissions from decomposing peat is small compared to the policy's overall effect on the forest $CO_2$ sink (Fig. 6). Carbon rent most strongly affects carbon storage in standing trees and dead organic matter. Storage in standing trees increases strongly, as more trees are left standing. At its greatest, the annual sink in standing trees is 40 $MtCO_2yr^{-1}$ stronger than in the BAU scenario. The sink in dead organic matter decreases initially by 16 $MtCO_2yr^{-1}$, as fewer trees are felled and, hence, less residue feeds into the pool.

For the same reason, the sink in HWP decreases initially by 5 $MtCO_2yr^{-1}$, as fewer trees are felled and thus less timber is processed into wood products. In comparison, the policy's effects on GHG emissions from decomposing peat are relatively small. Hence, the policy's net climatic impacts are roughly the same regardless of whether emissions from the decomposition of peat are taken into account.

Finland's target is to exceed its historical LULUCF sink of 14.865 Mt $CO_2$ $yr^{-1}$ by 2.889 Mt $CO_2$ $yr^{-1}$ (European Parliament and Council of the European Union 2023), implying total annual removals of 17.754 Mt $CO_2$ $yr^{-1}$ in the LULUCF sector. As other parts of the sector beside forests emit approximately 10 Mt $CO_2$ $yr^{-1}$, forests and HWP should therefore remove at least 28 Mt $CO_2$ $yr^{-1}$ to enable reaching the target. Currently, the sink in forests and HWP sink is close to zero (Natural Resources Institute Finland 2025). It would therefore need to strengthened by 28 Mt $CO_2$ $yr^{-1}$. In Fig. 6, this level of increase relative to the baseline is achieved in the second period, and maintained thereafter for several periods. Hence, a carbon rent policy based on a 15€ $tCO_2^{-1}$

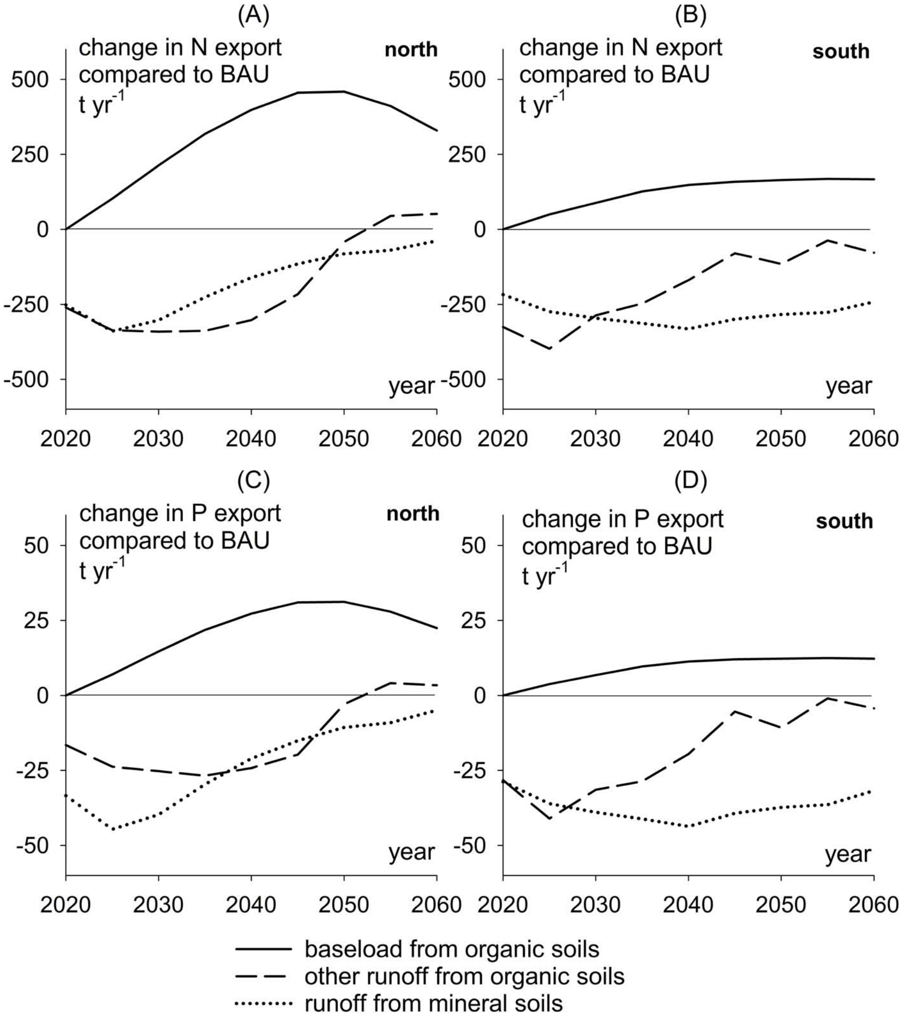


**Figure 3.** Change in annual export of nitrogen (N) and phosphorus (P) in runoff from forest soils Finland in response to the implementation of a carbon rent policy based on a 15€ $tCO_2^{-1}$ carbon price. Annual values are expressed relative to the baseline provided by the BAU scenario. Panels on the top row show N loads in northern Finland (panel A) and southern Finland (panel B). Panels on the bottom row show P loads in northern Finland (panel C) and southern Finland (panel D).

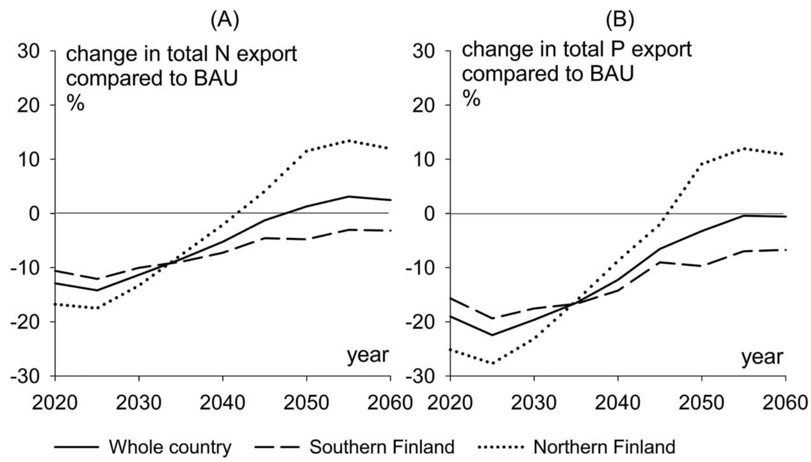


**Figure 4.** Relative change in total annual export of nitrogen (N) (panel A) and phosphorus (P) (panel B) from forest soils in Finland in response to the implementation of a carbon rent policy (based on a 15€ $tCO_2^{-1}$ carbon price) compared to the BAU baseline.

price strengthens the sink sufficiently in the medium term. The policy's implementation might be meaningful if the EU extends its LULUCF regulation to continue beyond 2030. Notably, however, the full effect is not yet achieved in the first period, due to the adjustment of the carbon stocks in dead organic matter and HWP. Hence, the policy does not provide an immediate fix to the problem, but rather should be implemented with a long-term goals in mind.

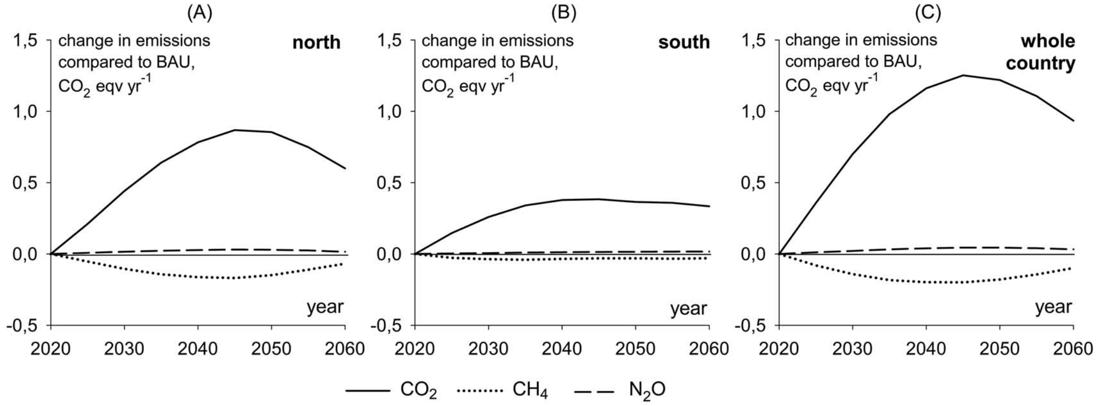


**Figure 5.** Absolute changes in $CO_2$, $N_2O$ and $CH_4$ emissions from decomposing peat after the implementation of a carbon rent policy, expressed relative to the BAU baseline for northern Finland (panel A), southern Finland (panel B), and the country as a whole (panel C). $CO_2$ equivalent conversions based on $GWP_{100}$.

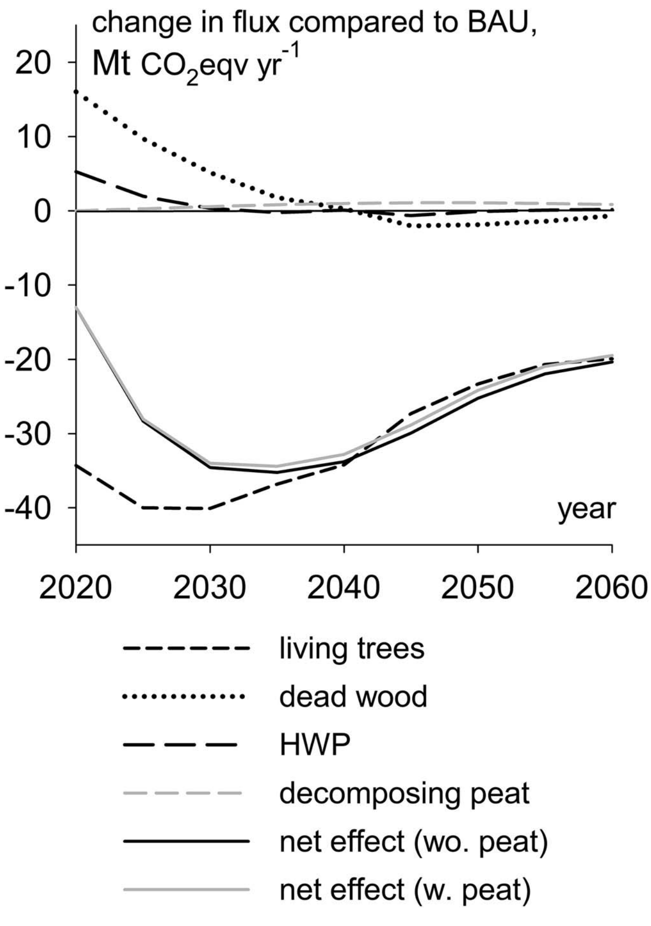


**Figure 6.** Effects of the carbon rent policy compared to the BAU scenario on carbon sink in forests and wood products. The comparison is made as follows. Let *A* denote the net flux from the stock into the atmosphere in the carbon rent policy scenario and let *B* denote the net flux from the stock into the atmosphere in the BAU scenario. The net fluxes are calculated using the stock change method. Positive values are emissions, negative values are removals. Let *E*, denote the effect of the carbon rent policy, shown in the figure. $E = A - B$.

Assumptions regarding the development of the water table in the baseline affect the results. In the wet scenario, N and P export is reduced more than in default conditions, when the carbon rent policy is implemented (Fig. 7A & B). In the dry scenario, the export of both nutrients is reduced less than in default conditions. Nevertheless, in both cases, the qualitative impact of the carbon rent policy remains the same, and differences in conditions only affect the overall trends slightly. Qualitative differences are observed regarding soil GHG emissions. In default and dry conditions, implementing carbon rent policy increases soil GHG emissions measured in equivalent $CO_2$ (Fig. 7C). However, in wet conditions, the policy decreases these emissions. When the water table is high, the cooling effect of reduced $CH_4$ emissions is stronger than the warming effect of increased $CO_2$ emissions and, hence, the policy has a negative net effect on soil GHG emissions. Despite the qualitative difference, the total effect on soil GHG emissions from decaying peat remains small compared to the overall effect of the policy (Fig. 6). Hence, variation in the baseline water table does not notably alter the big picture.

## Discussion

The aim of this study is to evaluate the expected side effects of a carbon rent policy in advance. The evaluation must rely on modelling methods, as the effects cannot be assessed empirically before the policy is implemented. Models are always simplified representations of reality. Thus, it is useful to discuss how well the modelled results can be expected to emulate reality.

Our results suggest that carbon rent decreases N and P loads in the short run but increases them in the long run (Fig. 4). The trend is driven by changes in annual harvests and growing stock volume compared to the baseline. Annual harvests decrease immediately when the policy is implemented. The drop in annual harvests decreases the variable component of the nutrient loads (which is affected by logging activity). Growing stock volume, on the contrary, accumulates over time. The greater accumulation of growing stock gradually increases the baseload. Hence, as long as these two effects are qualitatively correct (i.e. '*decreased logging activity increases variable loads*' and '*an increased amount of standing timber in forests increases the baseloads*'), then a similar trend is likely to occur in the real world as well—because carbon rent can be expected to decrease harvests (and, hence, variable nutrient loads) in the short run, and increase the accumulation of growing stock (and, hence, the baseloads) in the long run. The strength of the effects (Fig. 3)—and the timing of the reversal of the net effect (Fig. 4)—could differ from what we report, as our model provides a relatively coarsely simplified depiction of reality. Qualitatively, however, the trends should be similar.

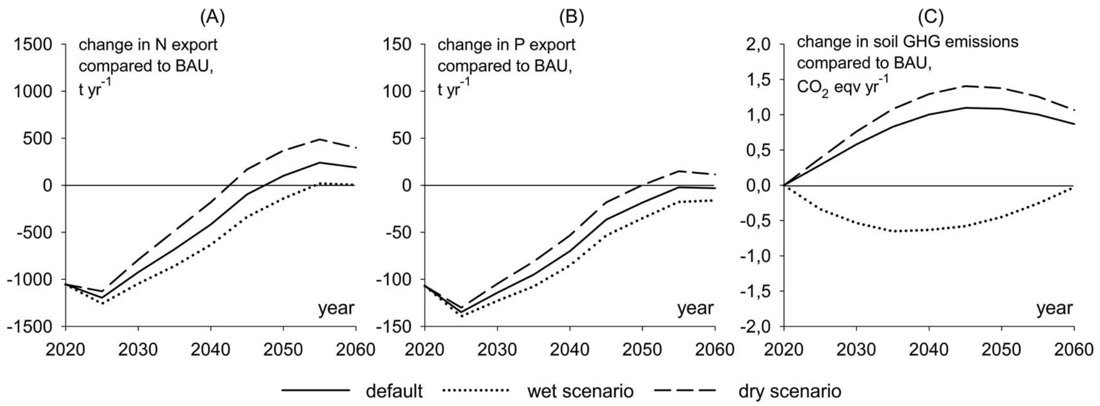


**Figure 7.** The sensitivity of the changes in N export (panel A), P export (panel B), and soil GHG emissions from decaying peat (panel C), to the water table level in the baseline. In the wet scenario, the water table is systematically higher than in the default scenario. In the dry scenario, it is systematically lower.

We can thus infer that implementing a carbon rent policy would be unlikely to worsen eutrophication caused by the export of nutrients from forest soils in the short run. Co-benefits would be more likely. However, the policy would not completely fix the forestry-driven component of the eutrophication problem either. Hence, if decreasing the export of nutrients is the goal, then regulating the loads separately is necessary—regardless of whether a carbon rent policy is implemented. This is in line with the Tinbergen Rule stating that for each distinct policy target there must be at least one separate policy instrument (Tinbergen 1952).

A limitation of this study is that we omit organic carbon in runoff from our analysis. Organic carbon affects water quality, as—along with increased iron concentrations—it causes the browning of water bodies (Finér et al. 2021, Härkönen et al. 2023). The export of organic carbon from forest soils to water bodies may also lead to increased GHG emissions from lakes to the atmosphere (Bayer et al. 2019). Drained peatland forests have been shown to play a significant role in organic carbon loads (Nieminen et al. 2021), and the mechanisms driving them include fluctuations in hydrology, peat decomposition and peat exposure (Härkönen et al. 2023). As the understanding of the links between forestry and organic carbon in runoff is only emerging within the natural sciences, we lack suitable functions that could be used to depict the processes in our forest sector model. However, it is likely that the effects of a carbon rent policy on organic carbon in runoff—mediated through an initial decreasing of harvests and a gradually increasing growing stock—would somewhat resemble its effects on nutrient loads.

Previous forest carbon pricing studies have not examined how GHG emissions from organic soils are affected by the changes in forest management induced by the policies. In a recent interdisciplinary review of forest carbon payments, Assmuth et al. (2024) therefore advise caution. The authors suggest that, until there is a better understanding of carbon payments' side effects on soil GHG emissions on peatlands, the scope of the policy should be restricted to forests on mineral soils, where the effects on the induced management changes on soil carbon are better understood. Their concern is that forest carbon rent payments could incite management changes which (as their unintended side effect) could increase soil GHG emissions significantly. In principle, the problem could be avoided if soil GHG emissions were also covered by the carbon payments (i.e. if soil carbon fluxes were also priced). This would provide landowners an incentive to account for the value of soil carbon in the optimization of forest management (and the resulting optimal management could look different than in this study). In practice, however, GHG emissions from peatlands are difficult to monitor and, therefore, difficult to include in forest carbon payment policy. As policies that target only storage in living biomass are more likely to be implemented, concerns regarding '*unintended soil GHG emission side effects*' are justified.

In this study, we address these concerns. We examine a policy in which carbon rental payments are only attached to carbon stored in standing timber. Our results suggest that implementing such a policy (based on 15€ $tCO_2^{-1}$ carbon price) would increase net GHG emissions from decomposing peat (Fig. 5), but the increase would be small compared to the overall effect of the policy on the forest carbon sink over the studied timeframe (Fig. 6). Hence, emissions from decomposing peat would not compromise the policy's intended effect. Moreover, the policy examined in this paper strengthens the forest carbon sink slightly more than is needed to reach the national LULUCF emission target. If a carbon rent policy were implemented, it might be therefore be milder (i.e. based on a lower carbon price) than the one examined here. A milder policy's effects on soil GHG emissions would be smaller than what we present.

Our findings regarding soil GHG emissions should be interpreted as tentative. Their validity depends on three questions, which cannot be answered exhaustively at present. The questions are: (1) '*Is the water table modelled correctly in the BAU baseline?*', (2) '*How strongly does intraregional variation in the water table affect peatland GHG emissions?*', and (3) '*How well do the applied functions (Equation 10) describe average regional $CO_2$, $CH_4$, and $N_2O$ emissions based on the average regional water table level?*'. Notably, Questions 2 and 3 are interrelated. In this study, emissions from decomposing peat are modelled based on the regional average of the water table level. Large intraregional variation in the water table could result in GHG emission patterns that differ from those reported in this study.

Correctly modelling the baseline water table is important, as it affects inferences about how the carbon rent policy changes soil GHG emissions. Changes in $CH_4$ and $N_2O$ emissions in response to changes in the water table are sensitive to its initial level, whereas changes in $CO_2$ emissions are not (see discussion related to Fig. 5 in Results). Hence, an error in the level of the water table in the BAU scenario could affect conclusions regarding $CH_4$ and $N_2O$ emissions from peat decomposition, and their relative importance compared to $CO_2$. The BAU water table baseline cannot be validated, as extensive and geographically explicit measured data on the seasonal variation of the water table throughout the country is not available. We model our baseline following Nieminen et al. (2023) and rely on expert estimates for parameter values for which data are not available (e.g. ditch depth, and difference between the water table in late summer and the average water table during the growing season). While the modelling of the water table is based on a valid approach that has been recently applied for similar purposes by experts in the field of modelling peatlands—we acknowledge that the applied approach is crude and potentially subject to error. The results of our sensitivity analysis nevertheless suggest, that our qualitative conclusions regarding the effects of a carbon rent policy on nutrient export and soil GHG emissions appear to be robust to reasonable variation in the water table baseline (Question 1). Although assuming wetter conditions does change the soil GHG results qualitatively (i.e. the relative importance of $CO_2$ and $CH_4$ is reversed and the cooling effect of decreased $CH_4$ emissions overpowers the warming effect of increased $CO_2$ emissions [Fig. 7C]), the scale of the effects compared the policy's effects on other $CO_2$ fluxes (Fig. 6) remains small and does not change the big picture.

Addressing Questions 2 and 3 is more difficult, as we do not have data on (i) the intraregional variation of the water table, or (ii) the variation between sites in how emissions and nutrient export respond to changes in the water table. We therefore cannot validate the accuracy of our GHG emission estimates in these respects. Here too, the functions used to calculate the emissions are based on state-of-the-art knowledge (Ojanen et al. 2010, Ojanen and Minkkinen 2019, and Minkkinen et al. 2020), but we do not know how large the estimation error is when the method is applied to calculate GHG emissions at the regional level based on average water table level. Hence, our results are crude and may change as data and methods improve.

That said, if the average water table in the baseline is roughly correct and the results are not compromised by intraregional variation, the effects of the carbon rent policy on GHG emissions from decomposing peat observed in this study (Fig. 5) are logical. Carbon rent is implemented to increase carbon storage in forests and wood products (compared to the baseline). Increased carbon storage means more living biomass in forests, which in turn means greater evapotranspiration—and drier peat. $CO_2$ emissions increase, as the dry peat decomposes. This chain of events could occur in the real world.

The increased understanding of the harmful water quality effects and soil GHG emissions related to forestry on peatlands has motivated a growing body of literature on possible mitigation measures. The measures for decreasing nutrient and organic carbon loads can be divided into those that aim to halt runoff before it reaches water bodies—e.g. overland flow fields—and those that aim to reduce the nutrient loads in runoff in the first place by avoiding extreme fluctuations of the water table level and by decreasing the exposure of the peat layer (Miettinen et al. 2020, Härkönen et al. 2023). The latter type of measures typically involve transitioning from rotation forestry to continuous cover forestry (CCF) (in the form of either selection harvesting or strip harvesting), where clear-felling operations are avoided and ditch network maintenance may be unnecessary due to the constant presence of some growing stock and hence evapotranspiration (Leppä et al. 2020). CCF also likely entails climate benefits on fertile sites through decreased soil GHG emissions (Lehtonen et al. 2023). While the economic profitability of CCF in peatland forests may not be massively inferior to that of rotation forestry (Juutinen et al. 2021), some sort of policy interventions are probably needed if a large-scale transition into CCF is desired by society. On the other hand, as CCF may lead to a lower average growing stock due to the need to enable natural regeneration (cf. Tahvonen et al. 2024), a carbon rent policy that omits soil GHG emissions may make CCF in peatland forests less attractive to forest-owners. This hypothesized effect could be perceived to contribute indirectly to the effects of a carbon rent policy on nutrient loads and soil GHG emissions. However, verifying these effects would require the use of a different type of model—as FinFEP does not include CCF, and the value of soil GHG fluxes are not taken into account in the optimization of forest management.

In this study, we focus on the effects of a carbon rent policy. However, various other policies to strengthen the forest sink have also been proposed. (For an extensive review in the Finnish case, see Soimakallio and Pihlainen (2023)). Other alternatives include, e.g. harvest taxes and auctioned harvest quotas. The proposed policies differ from each other, e.g. in terms of their effects on the distribution of income and the government budget. However, one thing that they have in common is that—to strengthen the forest carbon sink—the policies usually need to decrease harvests and increase the accumulation of growing stock in forests. Notably, these two effects are two main drivers of the trends in nutrient export and soil GHG emissions examined in this paper. Hence, while we here examine a carbon rent policy, the results can also be used to make qualitative inferences about other policies aimed at strengthening the forest sink.

Our harvest and carbon sink results concur both qualitatively and quantitatively with results obtained using stylized, theoretically consistent market-level models in which the timing of clear-felling is freely optimized (e.g. Lintunen and Uusivuori 2016, Rautiainen et al. 2018). However, the forest carbon sink in our results reacts to carbon pricing more strongly than in the results of forest-sector modelling studies from Sweden (Guo and Gong 2017) and Norway (Lopez et al. 2024). Lopez et al. (2024) also note the difference in results obtained using the Norwegian forest sector model, NorFor, and results obtained by Pohjola et al. (2018) using FinFEP.

As the models differ in many respects, it is difficult to pinpoint to an exact reason for the differences with full certainty. However, several potentially contributing factors can be identified. First, both Guo and Gong (2017) and Lopez et al. (2024) impose a set of fixed silvicultural programs from which forestowners must choose how to manage their forests. It is not clear if the sets of programs based on current practices are flexible enough to describe forest owners' optimal harvest behavior under carbon pricing. Furthermore, Guo and Gong (2017) also have a myopic (i.e. not forward-looking) supply function as a control, which may not be flexible enough to describe the optimal harvest behavior in every contingency. In the FinFEP, on the contrary, the incentives created by carbon pricing are fully incorporated into harvest decisions that can adapt to changed market conditions quite freely, while maintaining their forward-looking nature in a consistent manner. Second, the elasticity of wood demand affects the changes in harvest levels. In Guo and Gong (2017) demand is based on constant elasticity formulation and in Lopez et al. (2024) export volumes are fixed. In FinFEP, the demand for forest industry products is very elastic, as the relevant markets are global. This allows harvest levels to react more strongly. (However, the firms will use wood as long as revenues from its use exceed the variable costs). Lastly, in addition to these differences in the modelling of demand and supply, there can also be other, more subtle, differences that are difficult to identify.

## Conclusion

We examine the side effects of a carbon rent policy. The analysis is conducted using FinFEP, which is a partial equilibrium model of the Finnish forest and energy sectors. The examined policy is based on a 15€ $tCO_2^{-1}$ carbon price and it is implemented so that only carbon in standing timber and HWP is subsidized. Hence, storage in other stocks, such as soils, is not subsidized. The examined side effects include N and P export from forest soils and soil GHG emissions from decomposing peat, namely, $CO_2$, $N_2O$, and $CH_4$.

We find that the policy decreases harvests in the short run and increases the accumulation of growing stock in the long run. Through these mechanisms, the policy affects evapotranspiration and raises the average water table on peatlands in the short run, but then again lowers it in the long run. In response to these dynamics, N and P loads in runoff decrease first, but then recover. The initial decrease (of 10%–20% for N and 15%–30% for P at its highest) is caused by the decrease in harvests. However, the effect wears off and is reversed over time as growing stock accumulates and harvests recover. Geographically, both effects are more pronounced in Northern Finland where the peatlands' share of all *forestry land* is higher (40%) than in Southern Finland (28%) (Kulju et al., 2023).

The policy slightly increases soil GHG emissions from peatlands. Its effect on $CO_2$ emissions is notably larger than on $N_2O$ and $CH_4$. The net effect of all three GHGs (measured as equivalent $CO_2$) is nevertheless small compared to its effect on the forest sink in total. Due to uncertainties related to the modelling of the water table and to the models used to estimate GHG emissions from decaying peat, our conclusions regarding soil GHGs should be interpreted as tentative, rather than conclusive. Nevertheless, based on our tentative sensitivity analyses, our inferences about the scale of the policy's effect on soil GHG emissions (relative to the scale of the policy's effects on other $CO_2$ fluxes) appear to hold even if assumptions affecting the water table level are varied.

Our findings have two main policy implications. First, implementing a carbon rent policy would be more likely to generate co-benefits than tradeoffs in terms of preventing eutrophication caused by nutrient export from forest soils. At the very least, the policy is unlikely to worsen the situation in the short run. However, the policy cannot be expected to fix the issue either. If nutrient loads are to be reduced, a separate policy is needed to regulate them. Second, (if our tentative results are correct, then) carbon rent is unlikely to notably affect soil GHG emissions at the national level. It could therefore be safe to implement a carbon rent policy on peatlands—even if soil carbon stocks cannot be covered by it. However, as the result is tentative, we recommend conducting further research to validate the claim.

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## Supplementary data

Supplementary data are available at *Forestry* online.

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## Data availability

The data underlying this article will be shared on reasonable request to the corresponding author.

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