Restoration of forestry-drained oligotrophic peatlands can bring climate change mitigation within a few decades

Running head: Climate mitigation by drained peatland restoration

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This is a non-peer-reviewed preprint submitted to *EarthArXiv* of a manuscript submitted for peer reviewed to the journal *Restoration Ecology* (23.09.2025). This version has a few corrections on the submitted version, marked by red font.

Abstract

Introduction: Assessment of climate mitigation of peatland restoration is urgently needed, but data on greenhouse gas (GHG) fluxes from restored forestry-drained peatlands (FDP) is sparse. Using surrogate values from pristine peatlands, some studies have indicated long-lasting warming effect of restoration especially of nutrient-poor FDPs, while studies considering realized conditions and data from restored sites are missing.

Objectives: This study aims at estimation of climate mitigation potential of restoration of FDPs based on post-restoration development of vegetation and hydrology.

Methods: Dynamic trajectories of GHG-fluxes were calculated with process-based models informed by published studies of FDPs and restored peatlands. The model was applied to a sample of 12 restoration sites in Finland with data of carbon sequestration and water-table depth trends. The impact of restoration on global climate forcing was modelled against reference scenario of continued drainage.

Results: Hypothetical restoration scenarios resulted in initial warming effect, but a hummock-level scenario (deep WTD) shifted to a climate cooling effect already after 15 years of restoration. In contrast, a flark-level scenario (shallow WTD) showed increasing warming over the 100-year assessment period. In the empirical data, climate cooling impact was predicted in half of cases already after 10 years, and in most cases within 100 years. Restoration resulted in an average reduction of cumulative absolute global forcing by -1.78 (SD 1.74) t CO_2-equivalent ha^{-1} yr^{-1} over 100 years. Incorporating historical inference from peat inventories and forest management in the drainage scenario indicated even higher mitigation potential for restoration.

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Conclusions: The results predict considerably better climate mitigation potential for restoration of oligotrophic FDPs than suggested by previous modelling studies.

Implications for Practice: Climate mitigation by restoration of nutrient-poor FDPs can be improved with temporarily high CO_2 sequestration and potential dampening of CH_4 emissions by optimizing growth of new *Sphagnum* moss layer. Oligotrophic FDPs have higher mitigation potential than mesotrophic FDPs due to higher moss growth above water level. Drainage scenarios should be considered with alternative management options for climate impact assessment of restoration.

Keywords

carbon storage, emission factors, forest management, GHG emissions, *Sphagnum* peatlands

Introduction

Restoration of peatlands is widely regarded among key land-use strategies to mitigate climate change (Leifeld et al. 2019; Günther et al. 2020; Mander et al. 2024) and climate mitigation is a central motivation for the EU Nature Restoration Law (2022) that introduced the goal to restore 20% of degraded ecosystems by the year 2030. However, the mitigation potential of restoration has been questioned in the case of forestry drained peatlands (FDP) in Finland due to unfavorable soil emissions (Ojanen & Minkkinen 2020; Laine et al. 2024; Launiainen et al. 2025). Although restoration is widely recommended for other benefits, the lack of climate mitigation impetus may hinder wide-scale restoration. Meanwhile, postponing of restoration will likely cause more climate forcing (Günther et al. 2020).

Missing sufficient data from restored sites, Laine et al. (2024) used surrogate data from pristine peatlands, assuming an immediate shift from drained to pristine peatland fluxes after restoration. Their results indicated that restoration of nutrient-poor FDPs into open oligotrophic peatlands caused long-lasting warming impact. Instead, they found mitigation potential in restoration of nutrient-rich FDPs into forested peatlands, due to cessation of high soil emissions of the drained state. According to Launiainen et al. (2025) tree growth balanced out the soil emissions in such FDPs, however, nullifying the mitigation potential. Already earlier, a policy briefing was released stating that restoration of FDPs is unlikely to result in climate mitigation (Kareksela et al. 2021). Such conclusions are premature, however, as studies are lacking process-informed parameterization from empirical studies of restored sites.

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To achieve climate mitigation, effective land-use solutions need to be scaled over large areas. While economic profit will likely keep successfully forested FDPs outside of restoration, unsuccessful FDPs are more readily available. Laiho et al. (2016) estimated that up to one million hectares (20 %) of FDPs are weakly profitable in Finland. One reason behind failures is the weak nutrient regime of oligotrophic peatland types. According to Aapala et al. (2025), approximately 60,000 ha of FDPs have been restored in Finland primarily consisting of nutrient-poor FDPs, while climate mitigation effect remains unrecognized. However, Laatikainen et al. (2025) found that nutrient-poor FDPs had high growth rate of the *Sphagnum* moss layer after restoration, suggesting high CO_2 sequestration and possible dampening of CH_4 emission. To approach this possibility, it is crucial to use available information on post-restoration development of ecosystem processes affecting GHG fluxes.

The development after restoration includes 1) the inundation of peat and litter that were exposed to aeration during drainage phase, and 2) the formation of a new surface layer of moss and litter, resulting in 3) a trajectory of increasing water-table depth (WTD), as the thickening moss layer ascends above the water level (Laatikainen et al. 2025). The short-term dynamics likely includes 4) a delay in the onset of CH_4 production (reviewed by Wilson et al. 2016). Consequently, CO_2 sequestration may temporarily exceed that expected by pristine peatland references and CH_4 emissions remain lower, both favoring the potential for mitigation.

In this study, I attempt to refine the climate mitigation assessment by introducing dynamic trajectories of GHG fluxes, considering the re-established saturation of peat and the formation of new moss layer after restoration. The trajectories are formed with process-based modeling, fitted with published studies of drainage and restoration. The trajectories involve trends of WTD, a key variable for predicting CH_4 emissions. The focus is on the restoration of nutrient-poor FDPs into open oligotrophic fens. In broad terms, this is the commonest category of peatlands in Finland (Eurola et al. 1991), also comprising the bulk of unproductive FDPs (Laiho et al. 2016), and the highest potential among Finnish FDPs for upscaling of restoration.

Methods

The studied case

The nutrient-poor FDPs have developed after the drainage of oligotrophic mire types. The actual nutrient regime is variable, however, and depends on fertilization and alterations by varying effectiveness of drainage. These FDPs are forested by Scots pines (*Pinus sylvestris*), sometimes mixed with downy birch (*Betula pubescens*) and Norway spruce (*Picea abies*). Depending on drainage efficacy, forest mosses dominate but *Sphagnum* mosses commonly prevail with lower frequency. After restoration, *Sphagnum angustifolium* is the most characteristic species to form the revived moss layer,

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and *Eriophorum vaginatum* typically proliferates extensively (Komulainen et al. 1999; Haapalehto et al. 2011; Laatikainen et al. 2025). In general, however, restoration of nutrient-poor FDPs brings relatively little change in species composition (Laine et al. 2011; Haapalehto et al. 2017; Elo et al. 2024). The restoration measures include raising water level by blocking ditches and forming dams with peat, and cutting trees, depending on target peatland type. The aim is to force surface water to spread over the main peat surface. The guidelines of restoration are well-founded for routine application (Aapala et al. 2025).

Process-based models for constructing dynamic CO_2 and CH_4 flux trajectories

To assess climate impact of restoration, process-based models were formulated to calculate dynamic trajectories of CO_2 fluxes for drainage and restoration. The CH_4 flux was treated with a constant rate for drainage scenario following Laine et al. (2024) and with WTD-dependent dynamic trajectories for the restoration scenarios. In addition, fixed N_2O fluxes were included following Laine et al. (2024). I used published studies to inform short-term (10 years) development and set alternative scenarios for long-term (100 years) succession after restoration. Continued drainage is used as the reference state assuming an ideal case of a 50-year-old nutrient-poor FDP. A brief outline of the modeling for the trajectories is given below and more detailed explanation in the Supplementary file and an Excel file with the models is publicly available online (Tahvanainen 2025).

A model for CO_2 flux trajectory in the drainage scenario was based on constant litter input (Ojanen et al. 2013) and decomposition with remaining mass after 2 years following Straková et al. (2012) and after 40 years following Pitkänen et al. (2012) by adjusting decomposition coefficient (k) with age of litter (Fig. S1). A constant rate of decomposition with a minimal coefficient k = 0.005 typical for anaerobic decomposition rate (Scanlon & Moore 2000) was assumed after 50 years. The decomposition estimate of Ojanen et al. (2013) was used as a baseline, with a correction of young litter (3-years) decomposition and with addition of ageing litter cohorts (51 to 150 years) to the total decomposition efflux. This increases soil CO_2 emissions with stand age, while litter input is kept constant. Thus, the model output is conditional to assumption of continued high litter production of mature FDP stand without disturbance from management.

A dynamic CO_2 flux trajectory for restoration was calculated in a process-based model, fitted with empirical studies, assessing three compartments separately: 1) 'new moss, 2) 'drainage litter', and 3) 'old peat' (Fig. 1). The new moss refers to the *Sphagnum*-dominated moss layer that is established on top of the drainage phase surface, thus, it only occurs in the restoration scenarios. The drainage litter compartment refers to young (3 years) above ground litter from the drainage phase, characterized by litter fall of trees. The old peat withholds older litter and the actual peat formed before drainage.

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In a baseline model of the new moss layer, decomposition rates followed Straková et al. (2012, see also Tarvainen et al. 2013) for two first years (Fig. S1). Subsequently, the decomposition rate was decreased to match the average accumulation of new moss layer after 10 years reported by Laatikainen et al. (2025). The decomposition rate was then set to fall in a linear trend to a minimal constant rate at 50 years (k = 0.005) yielding the recent apparent rate of carbon accumulation estimated for 300-year-old strata in Finnish peatlands (Mäkilä & Goslar 2008). This baseline was adjusted by a vegetation development modifier, which considers the disturbed state after restoration with suppressed production (50 %) and subsequent increase to peak productivity at 6th year (120 %), followed by settling to the average natural *Sphagnum* productivity (Bengtson et al. 2021) after 20 years of restoration (82 % of baseline). This development kept the productivity estimate conservative, not assuming the high baseline of first 10 years after restoration to continue.

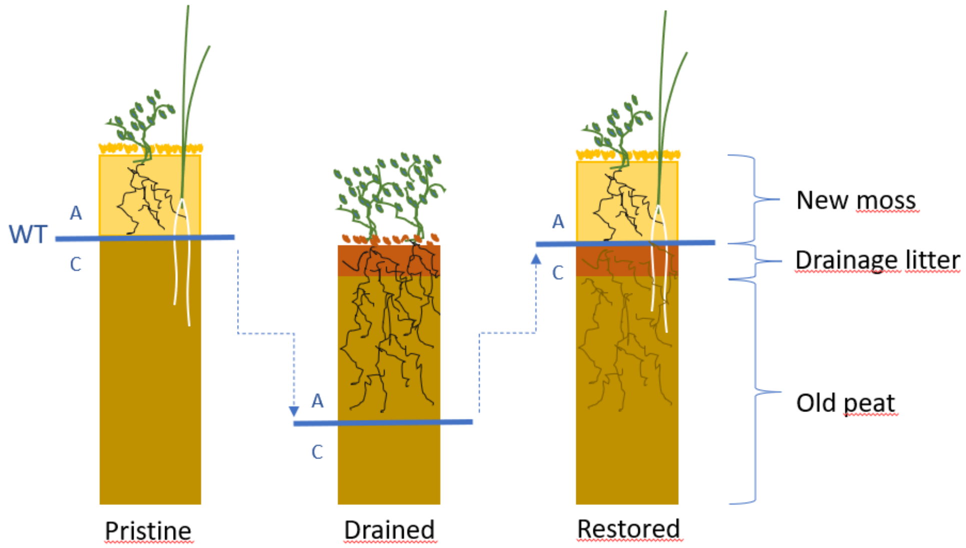

Fig. 1. Main alterations of surface peat strata from pristine to drained and restored peatland. Water table (WT) is the major regulator between prevalence of aerobic (a) and anaerobic (c) conditions in peat. Drainage causes subsidence and alteration of surface stratum into mix of forest litter and peat material. Restoration inundates old peat and drainage litter, and triggers formation of new moss layer.

The CO_2 emission rates from the old peat and drainage litter under restoration were estimated by adjustment of the drainage scenario emissions. This followed the results of Komulainen et al. (1999) of decreasing decomposition rates in FDPs over two years after restoration. The subsequent decline of decomposition rates of old peat and drainage litter were fitted to correspond to the catotelm transition (Frolking et al. 2001; Moore et al. 2007) in 25 years. Thus, it was presumed that the old peat and drainage litter remained in permanently saturated conditions starting from 25 years after restoration. Finally, the CO_2 flux trajectory was calculated by summation of changes in the new moss, drainage litter, and old peat compartments (Table S1). This sequence models development of vegetation and establishment of inundated conditions of peat after restoration.

The CH_4 emission trajectories were estimated for three alternative restoration scenarios based on the dependence of CH_4 flux on WTD according to Wilson et al. (2016) (Table S1). Several studies have found lower CH_4 emissions from restored FDPs than pristine peatlands (Komulainen et al. 1998; Juottonen et al. 2012; Wilson et al. 2016; Urbanová & Bárta 2020). Accordingly, the first 10-years' trajectory was calculated by weighted averaging of results from WTD-models for restored and undrained boreal peatlands obtained from Wilson et al. (2016). The restored peatland model's weight descended linearly from 0.9 to 0.1 in 9 years, and the undrained peatland model was given the opposite weights. Thus, CH_4 emission was assumed to conform to those modelled for undrained peatlands starting from 10th year after restoration, as controlled by WTD. Since different assumptions for the WTD development have great effect on CH_4 emissions, three different scenarios were formulated as informed by WTD monitoring data of the 12 sites studied by Laatikainen et al. (2025). These scenarios describe different long-term vegetation development trajectories, as WTD eventually results from the vegetation succession and new peat formation in tandem with hydrological conditions:

1) In the hummock scenario, WTD started from -9.0 cm and grew to -24.7 cm 10 years after restoration and was kept constant through 100 years. These values are averages of continuous WTD monitoring spanning from the first year after restoration to the last year of available data (6-9 years). The "deepening" of WTD may result either from growth of the moss layer or from lowering of water level, or both (Laatikainen et al. 2025).

2) In the intermediate scenario, WTD was assumed to grow in a linear trajectory from -6.7 cm to -12.7 cm in 10 years after restoration and keep constant through 100 years. In respective order, these values represented averages of the continuous WTD monitoring data over 5 first post-restoration years below the drainage period surface and below the new moss layer surface. Thus, the scenario presumes halted condition after five years of the increase of WTD due to ascending moss surface.

3) The flark scenario repeats the intermediate scenario up to ten years, after which the WTD is set to decrease in a linear trajectory to -2 cm in 100 years. This scenario could result, e.g., from increasing retention of water by the developing moss layer and consequent growth of water storage.

Model application to restoration monitoring sites

After developing the process-based dynamic model for hypothetical scenarios, the model was applied to 12 restoration monitoring sites with data of WTD trends and C accumulation in the moss layer 10

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years after restoration (Laatikainen et al. 2025). Concerning the CO_2 flux estimation, all other parameters of the model were kept fixed, while case-wise iterating the annual litter input to return the observed moss layer mass. WTD was expected to change linearly between the first and last available monitoring years' averages over 10 years and then keep constant. The CH_4 flux trajectories were predicted based on the case-wise WTD data, assuming linear change between first and tenth year after restoration.

Climate impact modelling with REFUGE 4

The climate impacts of the GHG flux trajectories of the hypothetical scenarios and the sample of 12 restoration sites were calculated using the REFUGE 4.1, a user-friendly open-access tool for calculating climate impacts with the IPCC's sixth assessment report methodology (Lindroos et al. 2023). It considers the global land and ocean sinks in the calculation of changes of atmospheric GHG concentrations and can handle both positive and negative fluxes. The results are expressed as Absolute Global Forcing Potential (AGFP), which is a measure of the global warming or cooling impact of the emission scenarios (W m^{-2}). Additionally, a conversion to CO_2-equivalents is used to concretize the results in emission terms. Both results are here adjusted to the effects of one-hectare of peatland and the drainage scenario is used as the control scenario to calculate restoration impact.

To demonstrate the impact of alternative drainage scenarios, additional climate impact modelling with drainage reference following Jauhiainen et al. (2023) and a clearcut forestry rotation model with 60-year cutting interval, informed by empirical studies of CO_2 fluxes after clearcutting (Korkiakoski et al. 2019; Tikkasalo et al. 2025) (Table S2). Detailed descriptions of the alternative drainage scenarios are presented in the Supplementary file.

The climate modelling starts in 2020 and results of CO_2-equivalent emissions are presented for 16-, 31-, 50-, and 100-year horizons. These timeframes were selected following Laine et al. (2024) to relate results to the carbon neutrality targets of Finland by 2035 (Finnish Climate Act 423/2022), and the EU by 2050 (European Climate Law 2021/1119).

Results

Hypothetical model scenarios

The restoration scenarios had a total net emission of 1177 g CO_2 m^{-2} over the first three years after restoration. After this, a CO_2 sink was indicated that peaked at -303 g CO_2 m^{-2} yr^{-1} the 12th year after restoration (Fig. 2). The average CO_2 flux was -48 g CO_2 m^{-2} yr^{-1} over the first 10 years and -204 g CO_2 m^{-2} yr^{-1} over 100 years after restoration. At 100 years the CO_2 flux rate was -172 g CO_2 m^{-2} yr^{-1} and at 300 years -55 g CO_2 m^{-2} yr^{-1}. The drainage scenario showed an increasing efflux from 3 to 105 g CO_2 m^{-2} yr^{-1},

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with an average net emission of 58 g CO_2 m^{-2} yr^{-1} over 100 years. In the restoration scenarios, the total accumulated CO_2 flux amounted to a sink of -20368 g CO_2 m^{-2} in 100 years, which translates to a mean annual rate of 56 g C m^{-2} yr^{-1}. The drainage scenario indicated a cumulative emission of 5848 g CO_2 m^{-2} in 100 years, with a carbon loss rate of 16 g C m^{-2} yr^{-1}.

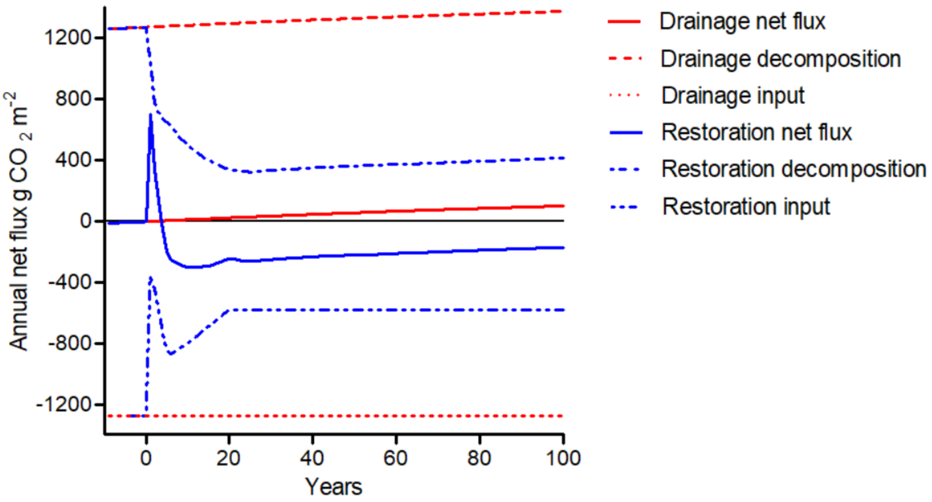

Fig. 3. Trajectories of the net annual CO_2 fluxes and total flux components of decomposition and litter input in the drainage and restoration scenarios.

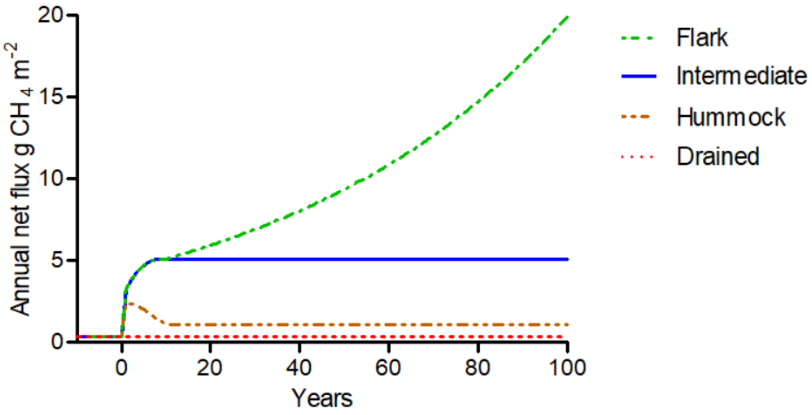

Fig. 4. Trajectories of annual CH_4 emissions of the three hypothetical restoration scenarios.

All restoration scenarios showed drastic rises of CH_4 emissions after restoration. The intermediate and flark scenarios shared the same WTD trajectory over the first 10 years and their CH_4 emissions rose from the first year's 3.3 g CH_4 m^{-2} yr^{-1} to 5.1 g CH_4 m^{-2} yr^{-1} by the 10th year. After this the flark scenario emission rose to 19.9 g CH_4 m^{-2} yr^{-1} in 100 years. The hummock scenario CH_4 emission rose to a maximum of 3.4 g CH_4 m^{-2} yr^{-1} in the second year and descended to 1.1 g CH_4 m^{-2} yr^{-1} in the 10th year after restoration (Fig. 3).

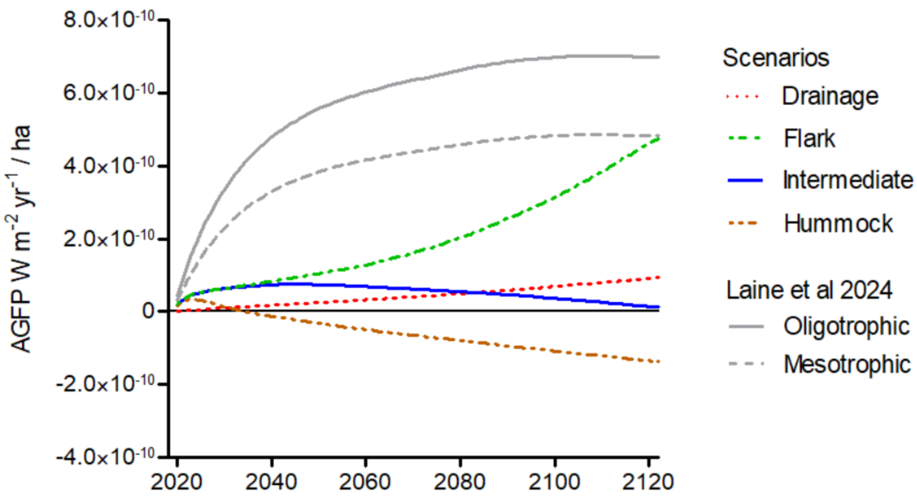

Fig. 5. Annual absolute global forcing potential (AGFP) of one hectare of drained or restored oligotrophic peatland. In addition to the scenarios of this study, results are presented with input from Laine et al. (2024) for mesotrophic and oligotrophic open peatland restoration.

The drainage scenario resulted in a steady growth of AGFP (Fig. 4). Restoration had higher AGFP than drainage until 64th year after restoration in the intermediate scenario and through the assessment period in the flark scenario. The hummock scenario also had higher AGFP than drained scenario, but only for 11 years and it resulted in a negative AGFP in the 16th year after restoration. The hummock scenario's AGFP amounted to -0.1336 nW m^{-2}/ ha at 100 years. The intermediate scenario showed a descending trend of AGFP down to near zero level at 100 years. The flark development scenario showed an increasingly ascending trend with circa five times stronger forcing than the drainage scenario at 100 years.

Climate impact of restoration fitted with moss growth and water level data

When the process-based dynamic model was fitted with data of post-restoration new moss layer C accumulation and WTD trends, nine out of twelve sites (75 %) were indicated with negative AGFP and all sites had lower AGFP than the drainage scenario at 100 years (Fig. 5). The average AGFP of the sites was -0.117 nW m^{-2}/ ha at 100 years. This cooling effect was significantly stronger in oligotrophic than in mesotrophic peatlands (df = 11, t = 2.419, p = 0.034).

The cumulative absolute forcing converted to constant annual CO_2 eq. emissions indicated a warming effect in 16-year assessment with 2.0 t CO_2-eq ha^{-1} yr^{-1} emissions, a neutral effect in 31-year assessment, and a -0.93 t CO_2-eq ha^{-1} yr^{-1} sink effect in 50-year assessment, on average. In these timeframes, however, the 95 % confidence interval of mean did not indicate significant difference from zero (no effect). The 100-year assessment indicated a significant cooling impact with an average -1.78 (SD1.74) t CO_2-eq ha^{-1} yr^{-1} sink effect, compared to the drainage scenario (Fig. 6).

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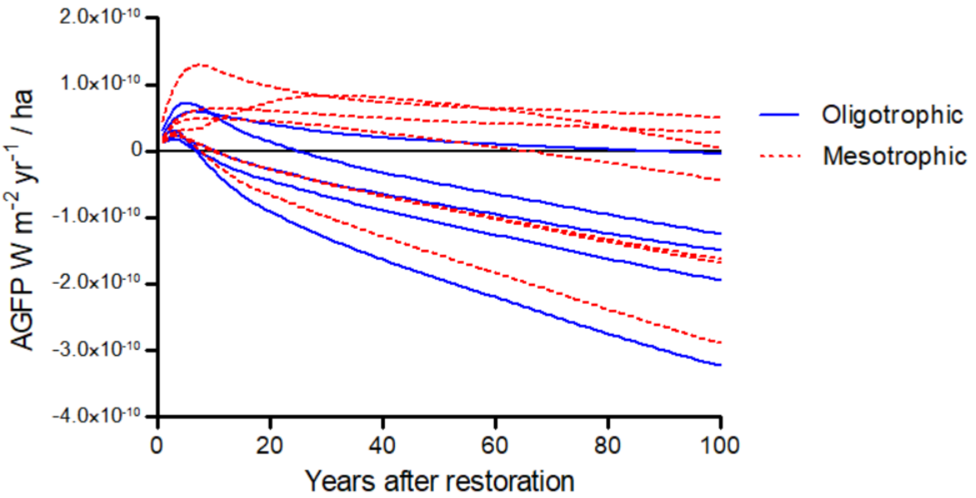

Fig. 6. Annual absolute global forcing potential (AGFP) per one hectare of 12 restored peatlands, as estimated by the process-based dynamic model fitted with data of 10-year growth of new moss layer and WTD.

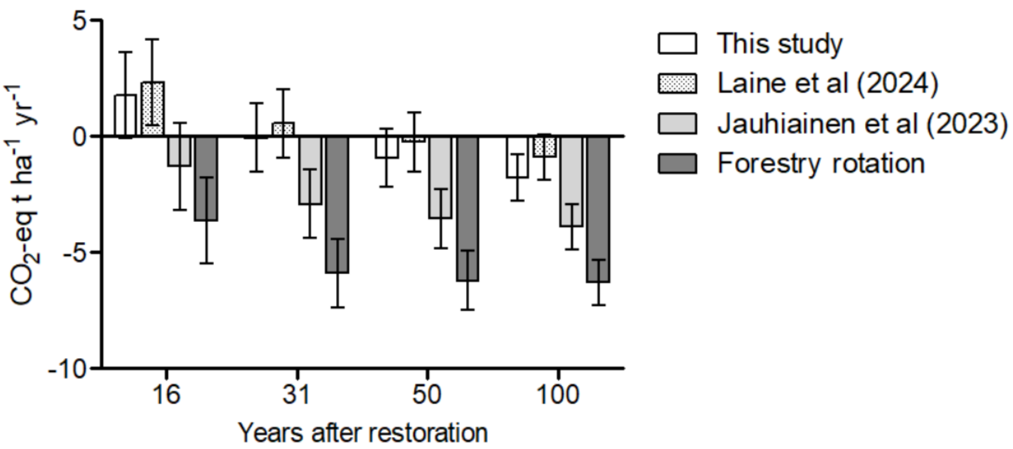

Fig. 7. Average effects and 95 % confidence intervals of restoration on cumulative absolute forcing converted to constant annual CO_2 eq. emissions (n = 12). The cumulative effects are calculated for 16, 31, 50, and 100-years timespans relative to the drainage scenario. Effect of alternative reference scenarios of continued drainage are shown for nutrient-poor FDPs according to this study, Laine et al. (2024), and for average trajectories following Jauhiainen et al. (2023), and forestry rotation. Negative values indicate climate cooling impact of restoration.

Among the alternative drainage scenarios, the Laine et al. (2024) scenario indicated a nonsignificant average mitigation potential of -0.88 (SD 1.74) t CO_2-eq ha^{-1} yr^{-1} sink effect in 100-year assessment for the 12 sites sample (Fig. 6). The Jauhiainen et al. (2023) reference with combined inference from GHG-flux studies and peat inventory studies indicated a significant sink effect of -3.88 (SD 1.74) t CO_2-eq ha^{-1} yr^{-1}, and when amended with forest rotation with 60-year clearcut interval, the mitigation potential grew to -6.29 (SD 1.74) t CO_2 eq/ha annual sink effect. The forest rotation reference indicated a significant climate mitigation potential for restoration already in the 16-year assessment with -3.61 (SD 1.74) t CO_2 eq/ha annual effect.

Discussion

The results indicated a significant potential for climate mitigation by restoration of FDPs into open *Sphagnum* peatlands with an average cooling impact of -1.78 t CO_2-eq. ha^{-1} yr^{-1} in the 100-year assessment. After an initial warming impact of restoration, a shift to climate cooling effect was indicated for half of the studied cases after 10 years of restoration. This result is in stark contrast with recent studies that assumed an immediate shift to pristine mire GHG fluxes after restoration (Laine et al. 2024; Launiainen et al. 2025). In this study, dynamic GHG trajectories were adjusted with short-term (10 years) developments after restoration and alternative scenarios of long-term (100 years) succession. The mitigation potential was significantly stronger in oligotrophic than in mesotrophic FDPs, which also contradicts the findings of Laine et al. (2024). This difference is hardly conclusive, however, since the process-based models for the flux trajectories did not differentiate between peatland types. On the other hand, the classification between nutrient-poor vs. nutrient-rich FDPs is not accurate and there is overlap between the classes in nutrient concentrations, while their distinction more accurately reflects pH (Menberu et al. 2017). The classification of FDPs is focused on potential for tree growth and it may not be optimal for assessment of restoration. Laatikainen et al. (2025) found highest growth rate of *Sphagnum* moss layer after restoration of acidic and nitrogen-poor FDPs, aligning with the general pattern of dominance of *Sphagnum* in acidic bogs and poor fens.

The results demonstrated the effects of higher CO_2 sequestration and lower CH_4 emissions in restored peatlands than anticipated by previous studies (Laine et al. 2024; Launiainen et al. 2025), both resulting from the rapid formation of new *Sphagnum* moss layer and consequently increasing WTD (Laatikainen et al. 2025). Indeed, when site specific data of moss layer growth and WTD were applied, the alternative drainage scenario from Laine et al. (2024) also resulted in negative forcing and a small climate mitigation potential in the 100-year assessment. Alternative drainage scenario from Jauhiainen et al. (2023) resulted in -3.88 t CO_2-eq. ha^{-1} yr^{-1} mitigation and application of forestry rotation nearly doubled the potential to -6.29 t CO_2-eq. ha^{-1} yr^{-1}. These alternative scenarios did not include effects of tree growth and wood products, however, which can change the mitigation potential (Launiainen et al. 2025). Indeed, the climate mitigation assessment of restoration is highly dependent on the reference scenario of continued drainage and forestry practices.

Drainage scenario considerations

Increased litter production rates are expected after successful drainage from increased productivity especially of trees (Straková et al. 2010; Ojanen et al. 2013; Minkkinen et al. 2018). Such high litter

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production, reaching levels of upland forests (Vucetich et al. 2000; Starr et al. 2005) is not representative of the up to 1 million hectares of weakly productive FDPs in Finland (Laiho et al. 2016), however. I used the litter input data of Ojanen et al. (2013) for the drainage scenario, which means that the CO_2 sequestration rate likely represented the upper bounds of what could be expected for FDPs directed to restoration, making the estimation of climate mitigation potential by restoration conservative.

The drainage scenario resulted in near-zero soil CO_2 flux, differing slightly from the earlier estimate by Ojanen et al. (2013), who found a weak CO_2 sink (-70 g CO_2 m^{-2} yr^{-1}) for nutrient-poor FDPs. The difference was caused by estimation of young litter decomposition. The model used by Ojanen et al. (2013) resulted in approximately 85 % of mass remaining in the annual balance of young litter. Such high remaining mass has been found for woody debris (Vávřová et al. 2009), but this decay resistant material comprises only 30 % of above ground litter (Straková et al. 2010; Ojanen et al. 2013). Straková et al. (2012) found 75 % and 66 % of mass remaining of composite above ground litter after one and two years in a nutrient-poor FDP. Higher decomposition rates were found by Laiho et al. (2004) for pine needles with 64 % and 51 %, and roots with 70 % and 60 % of mass remaining after one and two years. Results of He et al. (2020) were closely similar, on average, for multiple below ground litter qualities. I used the decomposition rates of Straková et al. (2012), which among the available references was a conservative choice against overestimating decomposition in FDPs. The model still resulted in a 38 % higher decomposition rate of young litter than estimated by Ojanen et al. (2013), who used the Yasso07 model. This difference agrees with the indication that models (Yasso07, Yasso15, Century) tended to underestimate litter decomposition in mineral soil forests by 43 % using default parametrization (Ťupek et al. 2019).

The drainage scenario described an ideal case of an average nutrient-poor FDP 50 years after drainage without effects of forest management. This kept the modeling simplified and comparable to earlier estimation (Laine et al. 2024). However, while Laine et al. (2024) kept the drainage CO_2 flux at a constant rate, I applied a trajectory with continuous addition of litter cohorts. Although the decomposition rate of old litter is slow, the cumulative effect amounted to 103 g CO_2 m^{-2} yr^{-1} of additional emission at 100 years. The average net CO_2 flux was a 58 g CO_2 m^{-2} yr^{-1} emission in the drainage scenario, as opposed to the -45 g CO_2 m^{-2} yr^{-1} sink used by Laine et al. (2024). In an extensive review, Jauhiainen et al. (2023) found slightly higher average emissions for typical nutrient-poor FDPs (79 g CO_2 m^{-2} yr^{-1}), and substantially higher emissions for low-productive sites (269 g CO_2 m^{-2} yr^{-1}). Thus, the drainage scenario in this study likely underestimates emissions, acting against climate mitigation potential of restoration. When the emission factors of Jauhiainen et al. (2023) were used, the modeling resulted in -3.88 t CO_2-eq. ha^{-1} yr^{-1} climate mitigation, on average. The emission factors

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from Jauhiainen et al. (2023) included peat inventory studies in addition to gas flux results. This complicates the comparison but also demonstrates important aspects of model assumptions connected to FDP age and management.

The historic effects of drainage are relevant to potential future effects of forestry rotation, which will repeatedly reintroduce afforested state, ditch clearance, and fertilization – the same factors that contributed to the carbon loss observed by peat inventories (Simola et al. 2012; Pitkänen et al. 2013). Instead, using GHG flux data at the mature state with high litter production may underestimate soil emissions under future management of FDPs. To incorporate forestry rotation effects, Launiainen et al. (2025) expected smaller CO_2 emissions from peat after clearcutting due to rising water level, but decomposition of harvest residues raised the total emission to approximately 1000 g CO_2 m^{-2} yr^{-1} after clearcutting. Empirical studies have found higher emissions after clearcutting. Korkiakoski et al. (2019) observed CO_2 emissions amounting to 3086 g and 2072 g CO_2 m^{-2} yr^{-1} in the first and second year after clearcutting of a nutrient-rich FDP, despite of 23-cm rise of water level. They also found CH_4 emissions of 4 and 6 g CH_4 m^{-2} yr^{-1}, respectively, i.e. an order of magnitude higher than expected for FDPs by Laine et al. (2024) and in this study. Tikkasalo et al. (2025) found a 2330 g CO_2 m^{-2} yr^{-1} emission from another nutrient-rich FDP one year after clearcutting. These results indicate that forestry management can have a remarkable role in soil GHG balance of FDPs that calls for further attention.

The study sites of Korkiakoski et al. (2019) and Tikkasalo et al. (2025) were nutrient-rich FDPs that likely have higher emissions than nutrient-poor FDPs (Laine et al. 2024). Applying forestry rotation in an alternative drainage reference with moderate two-year CO_2 emission rates (1500 and 900 g CO_2 m^{-2} yr^{-1}) after clearcutting followed by a 40-year descent of emissions (Menichetti et al. 2025) to baseline level of Jauhiainen et al. (2023) indicated a -5.9 t CO_2-eq. ha^{-1} yr^{-1} climate mitigation potential already after 31 years of restoration. This has immense implications within the timescale of EU’s climate neutrality target by 2050 (European Climate Law 2021/1119). However, only a few short-term studies are available on soil emissions after clearcutting from limited types of FDPs. Furthermore, emissions can be adjusted by different forestry practices (Lehtonen et al. 2023).

Carbon sequestration in restored peatlands

Restoration of FDPs re-establishes the functional acrotelm with thick moss layer, sequestering carbon with a known rate that may temporarily exceed that of pristine peatlands (Kareksela et al. 2015; Laatikainen et al. 2025). At the same time, decomposition rates fall with reintroduced saturated conditions in the old peat and litter that had been exposed to aerobic decomposition during drainage. These main effects of restoration can increase CO_2 sequestration despite lower litter production. However, a time lag before the onset of efficient growth is expected. Tong et al. (2025) found

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decreasing CO_2 emission from restored nutrient-poor FDPs with 147 g CO_2 m^{-2} yr^{-1} emission rate after three years of restoration and a total emission of 693 g CO_2 m^{-2} over the first three years after restoration. In this study, a range from 267 to 1936 g CO_2 m^{-2} total emission over 3 years was indicated, after which the CO_2 sequestration was enough in half of the sites to result in negative AGFP after 10 years. I used the average of new moss layer mass reported by Laatikainen et al. (2025) in the modeling of CO_2 sequestration. This was a conservative choice because the sample included mesotrophic sites apparently unsuitable for establishment of oligotrophic *Sphagnum* vegetation. The average for oligotrophic sites in Laatikainen et al. (2025) was 37 % higher than for all sites and nearly the same as found by Kareksela et al. (2015) for restored *Sphagnum*-dominated oligotrophic pine mires.

While long-term results of GHG fluxes are missing from restored FDPs, surrogate values have been obtained from pristine peatlands. Laine et al. (2024) calculated the net CO_2 sequestration as the sum of CH_4 emission and the long-term apparent rate of carbon accumulation (LORCA) in peat, considering also minor contributions of deposition and a proportion of leached DOC. They estimated a fixed rate of -124 g CO_2 m^{-2} yr^{-1} for oligotrophic open mires, withholding a LORCA value of 17 g C m^{-2} yr^{-1} (-62 g CO_2 m^{-2} yr^{-1}). The hypothetical restoration scenario had clearly higher average net sink (-204 g CO_2 m^{-2} yr^{-1}) over 100 years. However, the modelled CO_2 sequestration rate at 300 years was only -55 g CO_2 m^{-2} yr^{-1}. This highlights the significance of short-term development after restoration.

Although the long-term model result for CO_2 sequestration remains admittedly speculative, the use of LORCA as a surrogate data source is not satisfactory. The actual carbon balance of any peatland, restored sites included, depends on the history and condition of the whole peat deposit. Even the oldest peat cohorts continue to decay, although at extremely slow rates, and their combined effect will eventually limit the capacity of peatland carbon sink (Clymo 1984). Although peat thickness varied greatly in the FDPs studied by Ojanen et al. (2010, 2013), it did not have explanatory power on soil heterotrophic respiration that was related to tree volume, site type, temperature, and water level. This underlines that saturated deep peat remains in relatively inert state also in FDPs and has a minor contribution to the CO_2 flux. Restoration effects on decomposition are also largely limited to surface peat strata, where the water level is again adjusted. Therefore, it is assumed that the lack of consideration of peat thickness in the modeling has little significance to potential mitigation.

Methane emission rate of restored oligotrophic Sphagnum peatlands

Increased CH_4 emissions are expected after restoration, due to increasingly anaerobic conditions caused by raising water level, while FDPs have negligibly low CH_4 emissions (Ojanen et al. 2010; Wilson et al. 2016). Since CH_4 emissions have strong short-term warming impact, the expected CH_4 emission after restoration has a decisive effect on the impact assessment. Laine et al. (2024) used a

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high CH_4 emission factor of pristine oligotrophic fens (22.0 g CH_4 m^{-2} yr^{-1}) as a surrogate value for restoration of nutrient-poor FDPs into open peatlands. Data from other reviews have indicated only slightly lower emission rates (Saarnio et al. 2007; Abdalla et al. 2016). The use of surrogate emission factors neglects the realized WTD conditions of restored peatlands and the observed suppression of CH_4 production in restored peatlands. I used WTD data from restored sites and a conservative application of model of Wilson et al. (2016) for restored peatlands' CH_4 emissions. This resulted in remarkably lower CH_4 emissions than anticipated by earlier studies (Laine et al. 2024; Launiainen et al. 2025). In the long-term, re-established natural dynamics are expected to govern CH_4 emission, calling for consideration of the ecosystem succession concerning WTD.

In the short-term, restoration of FDPs has proven successful in returning water level and storage functions (Menberu et al. 2016), although a legacy effect of ditches may prevail causing spatial variation (Haapalehto et al. 2014). Studies extending to 10 years after restoration have indicated, however, that WTD tends to increase as the moss layer develops and ascends higher above the water level (Haapalehto et al. 2011; Laatikainen et al. 2025). In the sample of 12 restored sites, the average WTD was -9.0 cm in the first year after restoration, similar to pristine poor fens (Menberu et al. 2016), but grew to -24.7 cm in the last available data (6 to 9 years). WTD has major control on CH_4 emissions (Ojanen et al. 2010; Abdalla et al. 2016; Wilson et al. 2016; Evans et al. 2021) and while high emissions are expected from the restoration target type of wet oligotrophic fens due to low WTD, the realized WTD levels of restored sites do not support such expectation.

Several studies have found lower CH_4 emissions from restored than from pristine peatlands (Komulainen et al. 1998; Juottonen et al. 2012; Rey-Sanchez et al. 2019; Urbanová & Bárta 2020; Tong et al. 2025). Wilson et al. (2016) reported an average emission rate of 6.3 g CH_4 m^{-2} yr^{-1} for nutrient poor boreal restored peatlands. Their models of CH_4 emissions against WTD indicated that restored peatland emissions reached between 14 to 27 % of the emissions of undrained peatlands for the same WTD levels. This suggests that suppression of CH_4 emissions in restored peatlands is not caused merely by deeper WTD. Juottonen et al. (2012) found low CH_4 emission rates from restored Finnish FDPs reaching only 2 % of their pristine control sites 10 years after restoration, explained by low population densities of methanogenic microbes. Urbanová & Bárta (2020) found recovery of methanogenic microbial communities after 7-13 years of restoration of bogs and spruce swamp forests, while the CH_4 production was still lower than in pristine peatlands. Recently, Tong et al. (2025) reported low emissions of 3.1 to 5.8 g CH_4 m^{-2} yr^{-1} during the first three years after restoration of a boreal oligotrophic fen, amounting 32 to 49 % of their pristine control sites. Tyystjärvi et al. (2024) conducted a process-based simulation study, with calibration data from restored FDPs, where they found 6 g CH_4 m^{-2} yr^{-1} initial emissions that decreased to 1 g CH_4 m^{-2} yr^{-1} in 100 years after restoration.

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In FDPs, Rissanen et al. (2023) found low CH_4 emissions (2.6 g CH_4 m^{-2} yr^{-1}) from ditches with spontaneous infilling by *Sphagnum* mosses, while ditches without moss cover had nearly tenfold emissions (20.6 g CH_4 m^{-2} yr^{-1}). They found some negative CH_4 fluxes from moss-covered ditches on dry occasions, contributable to methanotrophy. Indeed, rapidly establishing methanotrophy may effectively prevent CH_4 emissions. Putkinen et al. (2018) found that methanotrophy was independent of succession stage in restored peat mining area, instead depending on the thickness of aerobic *Sphagnum* moss layer.

The anaerobic CO_2:CH_4 production ratios in *Sphagnum* peat tend to be far greater than predicted by electron balance models (1:1) and one mechanism to cause this is likely the hydrogenation of organic terminal electron acceptors (TEAs) (Wilson et al. 2017). Blodau & Deppe (2012) found that the addition of peat humic acid suppressed CH_4 but not CO_2 production. This may explain observation of Juottonen et al. (2012), who found a negative relationship between DOC concentration and CH_4 emission. A further mechanism to increase CO_2:CH_4 ratio is the non-enzymatic release of CO_2 from Maillard reactions that can contribute about 10 % of anaerobic CO_2 release from *Sphagnum* peat (Cory et al. 2025). Interestingly, high availability of both organic TEAs and Maillard agents can be expected after restoration. Menberu et al. (2017) reported a high average DOC concentration of 75 mg/l of pore water soon after restoration and a decrease after 6 years to pristine peatlands level (33 mg/l). They also reported elevated specific UV-absorbance (SUVA) after restoration, indicating high aromaticity of DOC, which can suppress microbial activity. The onset of high growth rate of *Sphagnum*, on the other hand, produces galacturonic acid that can act as a Maillard reagent (Cory et al. 2025). These conditions created by restoration may partly explain the observed low CH_4 emissions, thus, further supporting the use of restoration-specific trajectories of CH_4 emissions in climate impact modeling, instead of surrogate values from pristine peatlands. In this study, the suppression of CH4 emissions in restored peatlands was included with a conservative weighting up to 10 years after restoration. It is possible, however, that CH_4 emission begin more readily after restoration of unsuccessfully drained FDPs with close to pristine microbial dynamics.

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SUPPLEMENTARY INFORMATION

Restoration of forestry-drained oligotrophic peatlands can bring climate change mitigation within a few decades

Teemu Tahvanainen

<u>Process-based dynamic models of soil CO_2 flux trajectories for drainage and restoration scenarios</u>

This supplement describes details of process-based models constructed for calculations of GHG-trajectories for drained and restored oligotrophic peatlands used for climate forcing modelling. An Excel-file with all models is published online in Zenodo (Tahvanainen 2025).

Drainage scenario

The main source for the drainage scenario CO_2 flux model input was Ojanen et al. (2013). The total litter input (L_0) was obtained as the average 1274 g CO_2 m^{-2} yr^{-1} of nutrient-poor types, consisting of 601 g CO_2 m^{-2} yr^{-1} aboveground litter and 673 g CO_2 m^{-2} yr^{-1} belowground litter, following the mean proportions reported by Ojanen et al. (2013). In a model for litter accumulation in the drainage scenario, the remaining mass at year *t* of litter cohort *i* equals the addition to litter stock in year *t*, and it was calculated with the function

(Equation 1)

$$iL_t = L_{t-1} \times e^{-k_t}$$

where the previous year's litter addition L_{t-1} is decayed by decomposition rate -k_t. The cumulative soil litter stock is confined as

(Equation 2)

$$L_t = \sum_{i=1}^{t} iL_t$$

To account for decreasing decomposition rate of each litter cohort with age, k_t was adjusted in a descending trajectory (Fig. 2). For aboveground litter, the values k_1 = 0.288 and k_2 = 0.128 were used to fit the 75 % and 66 % of remaining mass following results of Straková et al. (2012) for mixed litter in FPDs. After this, k_t was decreased to a minimum at k_{50} = 0.005 following Scanlon & Moore (2000), with an exponential phase from k_3 to k_{10}, followed by a linear decrease between k_{11} to k_{50}. The exponential phase was fitted with an annual multiplier 0.8317 to yield an above ground litter stock of 8340 g CO_2 m^{-2} at 40 years, conforming to the litter stock estimate following Pitkänen et al. (2007), who sampled above ground litter of 47 Finnish FDPs. The 3-year litter decomposition totaled at 245 g CO_2 m^{-2} yr^{-1}

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efflux, i.e. 38 % higher than in Ojanen et al. (2013). Belowground litter was modelled with the same parameters, observing that the remaining mass did not fall below the trajectories found for belowground litter by He et al. (2025). The cumulative addition to CO_2 efflux from decomposition of ageing litter cohorts ($iL_{51} – iL_{150}$) was added to the baseline decomposition (iL_{50} = 1271 g CO_2 m^{-2}) to approximate the trajectory of efflux starting from the first year of the drainage scenario (50-year-old forest stand).

The CH_4 and N_2O flux rates were kept constant following Laine et al. (2024), with 0.34 g CH_4 m^{-2} $yr^{-1}$and 0.08 g N_2O m^{-2} yr^{-1}. The model describes constant soil processes, and it does not account for the tree stand dynamics or management.

Restoration scenarios

The trajectory of net CO_2 flux for restoration scenarios was estimated by the summation of the flux components of moss layer, drainage litter, and old peat. The same CO_2 flux trajectory was used for three alternative long-term scenarios. The new moss CO_2 sequestration was estimated so that the cumulative mass at 10 years conformed to the average of 4855 g CO_2 m^{-2} observed by Laatikainen et al. (2025) for 18 FDP restoration sites. The k_t was iterated together with a constant annual CO_2 sequestration in biomass. This was necessary because of unknown decomposition of the moss layer litter accumulated in 10 years. With k_1 = 0.198 and k_2 = 0.076 the mass remaining was fitted to 82 % and 76 % after first two years of decomposition, following Straková et al. (2012) results from oligotrophic *Sphagnum* mires (Fig. 2). The k_t was then adjusted with an annual modifier of 0.800 to fit the average 10-year moss layer stock with a 727 g CO_2 m^{-2} yr^{-1} input, conforming to net primary production (NPP) with 60 % contribution from *Sphagnum* mosses that would grow at about the upper quartile rate of approximately 440 g CO_2 m^{-2} yr^{-1} of *Sphagnum* productivity (Bengtson et al. 2021). The minimum k = 0.005 was set similarly as in the drainage scenario following Scanlon & Moore (20000). This iteration resulted in a baseline model of the new moss layer litter accumulation. The trajectory was then adjusted by a modifier to account for the vegetation development (vM_t), while yielding the same 10-year stock. The sequence of vM_t started from 0.5, peaking at 1.2 in the 6th year, and then descended to a constant level of 0.802 in the 20th year after restoration (Fig. 2). This sequence considers 1) the disturbed state after the restoration, 2) the growth peak within a few years, and 3) a descent to average natural *Sphagnum* productivity of 350 g CO_2 m^{-2} yr^{-1} (Bengtson et al. 2021) contributing 60 % of total NPP of 583 g CO_2 m^{-2} yr^{-1}. The sequence of vM_t was adjusted in gross accordance with reports of development in restored peatlands (Haapalehto et al. 2011; Kareksela et al. 2015, Anderson et al. 2016). The model was continued to 300 years to observe that the recent

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apparent carbon accumulation (RERCA) of new moss was 45 g C m^{-2} yr^{-1}, equaling the average 300-year RERCA for pristine mires in Finland (Mäkilä & Goslar 2008).

Remaining mass in the new moss layer (*nm*) of each cohort *i* at time *t* of litter (iL_t) was calculated as

(Equation 3)

$$iL_t = (L_0 \times vM_t) \times e^{-k_{t-i+1}}$$

where the baseline litter production L_0 = 727 g CO_2 m^{-2} yr^{-1} is adjusted with vM_t and $-k_{t-i+1}$ which repeats the same trajectory of k_t for each cohort *i*.

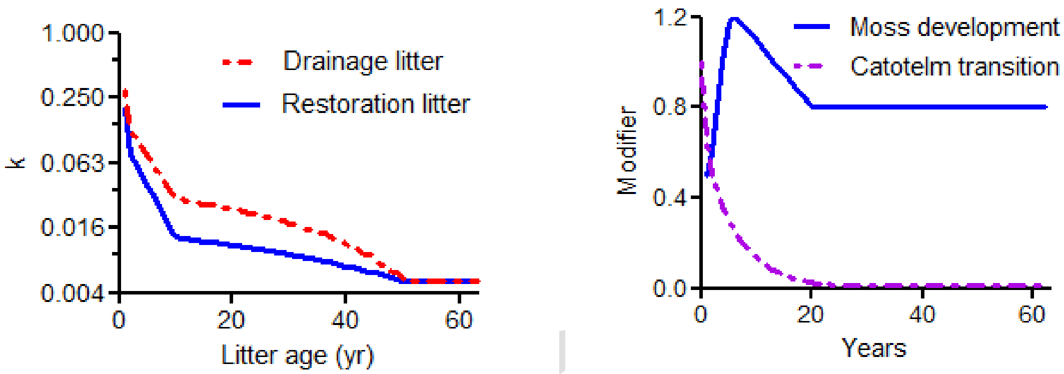

Fig. S1. Post-restoration trajectories of A) the decomposition coefficient k for above ground litter in drainage and restoration scenarios (notice log-2 scale), and B) temporal modifiers for CO_2 flux components.

The CO_2 efflux from old peat (*op_R*) was estimated by adjusting the baseline from the drainage period value op_R_0 = 1273 g CO_2 m^{-2} yr^{-1}. The drainage litter stock (*dl_L*) before restoration was estimated with the model for drainage scenario aboveground litter (3-year stock). An anaerobic transition modifier (aM_t) was introduced to account for reduced decomposition. The modifier was set to aM_1 = 0.680 and aM_2 = 0.500 following results of Komulainen et al. (1999), who measured decomposition in drained and restored sites. The modifier was adjusted by multiplier 0.85 each year until a set minimum aM_{25} = 0.01 (Fig. 2). Thus, heterotrophic respiration was expected to settle in 25 years at 1 % (13 g CO_2 m^{-2} yr^{-1}) of the drainage scenario baseline. This can be reflected with similar chance in the acrotelm-catotelm transition in natural peatlands (Frolking et al. 2001; Moore et al. 2007). Finally, the net CO_2 flux of the restoration scenarios was calculated as

(Equation 4)

$$Net\ CO_2 = -(nm\_L_t - nm\_L_{t-1}) + (opR_0 + dl\_L_{t-1} - dl\_L_t) \times aM_t - C_{DOC} + C_{dep}$$

with all components in units of g CO_2 m^{-2} yr^{-1}. The growth of litter stock to nm_L_t from nm_L_{t-1} represents accumulation in the new moss layer. The drainage litter stock dl_L decreases from dl_L_{t-1} to dl_L_t and adds to heterotrophic respiration of old peat op_R_0, both adjusted by aM_t. A 10 % share of leaching dissolved organic carbon (C_{DOC} = 3.48 g CO_2 m^{-2} yr^{-1}) and carbon deposition (C_{dep} = 1.83 g CO_2 m^{-2} yr^{-1}) are accounted following Laine et al. (2024).

Table S1. GHG-trajectories of hypothetical restoration scenarios used for REFUGE4 climate impact modelling.

Scenario:	**Drainage**			**Hummock**			**Intermediate**			**Flark**		
Year	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O
2020	3.2	0.34	0.08	700.0	2.37	0.03	700.0	3.27	0.03	700.0	3.27	0.03
2021	4.5	0.34	0.08	340.1	2.38	0.03	340.1	3.77	0.03	340.1	3.77	0.03
2022	5.8	0.34	0.08	131.4	2.30	0.03	131.4	4.17	0.03	131.4	4.17	0.03
2023	7.1	0.34	0.08	-53.9	2.17	0.03	-53.9	4.48	0.03	-53.9	4.48	0.03
2024	8.4	0.34	0.08	-206.0	2.00	0.03	-206.0	4.72	0.03	-206.0	4.72	0.03
2025	9.7	0.34	0.08	-255.2	1.82	0.03	-255.2	4.90	0.03	-255.2	4.90	0.03
2026	10.9	0.34	0.08	-273.0	1.63	0.03	-273.0	5.02	0.03	-273.0	5.02	0.03
2027	12.2	0.34	0.08	-282.6	1.45	0.03	-282.6	5.10	0.03	-282.6	5.10	0.03
2028	13.5	0.34	0.08	-295.8	1.28	0.03	-295.8	5.13	0.03	-295.8	5.13	0.03
2029	14.7	0.34	0.08	-300.8	1.12	0.03	-300.8	5.12	0.03	-300.8	5.12	0.03
2031	17.2	0.34	0.08	-304.7	1.12	0.03	-304.7	5.12	0.03	-304.7	5.20	0.03
2033	19.7	0.34	0.08	-300.1	1.12	0.03	-300.1	5.12	0.03	-300.1	5.44	0.03
2035	22.1	0.34	0.08	-287.6	1.12	0.03	-287.6	5.12	0.03	-287.6	5.61	0.03
2039	26.9	0.34	0.08	-247.5	1.12	0.03	-247.5	5.12	0.03	-247.5	5.95	0.03
2043	31.6	0.34	0.08	-261.3	1.12	0.03	-261.3	5.12	0.03	-261.3	6.33	0.03
2049	38.5	0.34	0.08	-253.8	1.12	0.03	-253.8	5.12	0.03	-253.8	6.92	0.03
2059	49.6	0.34	0.08	-236.3	1.12	0.03	-236.3	5.12	0.03	-236.3	8.05	0.03
2079	70.1	0.34	0.08	-214.5	1.12	0.03	-214.5	5.12	0.03	-214.5	10.89	0.03
2099	88.6	0.34	0.08	-193.3	1.12	0.03	-193.3	5.12	0.03	-193.3	14.72	0.03
2119	105.4	0.34	0.08	-173.8	1.12	0.03	-173.8	5.12	0.03	-173.8	19.91	0.03
2169	140.7	0.34	0.08	-132.9	1.12	0.03	-132.9	5.12	0.03	-132.9	19.91	0.03

<u>Effect of alternative drainage scenarios on climate mitigation by restoration</u>

Constant emission factor scenarios follow Laine et al. (2024) and Jauhiainen et al. (2023) reports for nutrient-poor FDPs. Jauhiainen et al. (2023) presented separate emission factors for typical and low productive FDPs. They also reported results of peat inventory and GHG-flux studies separately and in combination. In addition, a simple model for forestry rotation with clearcutting is presented.

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Table S1. GHG-trajectories used in REFUGE4 for forestry rotation scenarios. All fluxes g m^{-2} yr^{-1} (multiplier 1E +12 in REFUGE). Baseline values according to Jauhiainen et al (2023) are given in bold.

Baseline:	**NuP typical comb**			**NuPlow comb**				**NuP typical comb**			**NuPlow comb**		
Year	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O	Year	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O
2020	**79.2**	**0.34**	**0.08**	**269.2**	**0.34**	**0.08**	2020	**79.2**	**0.34**	**0.08**	**269.2**	**0.34**	**0.08**
2021	1500	0.34	0.08	1500	0.34	0.08	2030	**79.2**	**0.34**	**0.08**	**269.2**	**0.34**	**0.08**
2022	900	0.34	0.08	900	0.34	0.08	2031	1500	0.34	0.08	1500	0.34	0.08
2060	**79.2**	**0.34**	**0.08**	**269.2**	**0.34**	**0.08**	2032	900	0.34	0.08	900	0.34	0.08
2080	**79.2**	**0.34**	**0.08**	**269.2**	**0.34**	**0.08**	2070	**79.2**	**0.34**	**0.08**	**269.2**	**0.34**	**0.08**
2081	1500	0.34	0.08	1500	0.34	0.08	2090	**79.2**	**0.34**	**0.08**	**269.2**	**0.34**	**0.08**
2082	900	0.34	0.08	900	0.34	0.08	2091	1500	0.34	0.08	1500	0.34	0.08
2140	**79.2**	**0.34**	**0.08**	**269.2**	**0.34**	**0.08**	2092	900	0.34	0.08	900	0.34	0.08
2169	**79.2**	**0.34**	**0.08**	**269.2**	**0.34**	**0.08**	2130	**79.2**	**0.34**	**0.08**	**269.2**	**0.34**	**0.08**
							2169	**79.2**	**0.34**	**0.08**	**269.2**	**0.34**	**0.08**

The forestry rotation reference scenario was calculated using CO_2 emission trajectories with clearcuts in 60-year intervals starting immediately (2020 and 2080) or ten years after start of modelling (2030 and 2090). The baseline emissions followed Jauhiainen et al. (2023) but two years' emissions following clearcut were 1500 g CO_2 m_{-2} yr_{-1} and 900 g CO_2 m_{-2} yr_{-1}, followed with 40-year linear descent to the baseline. This timeline of fading clearcut impact follows findings from mineral-soil conifer forests (Menichetti et al. 2025). The two-year emission rates after clearcut were adjusted to be intermediate between those of the restoration scenario of this study (702 and 342 g CO_2 m^{-2} yr^{-1}) and results of Korkiakoski et al. (2019) and Tikkasalo et al. (2025) for clearcut impacts in nutrient-rich FDPs. The model of Launiainen et al. (2025) indicated an approximate 1000 g CO_2 m^{-2} yr^{-1} emission after clearcut (NEE mainly comprising of soil and harvest residue emissions) conforming to the estimate here. Only CO_2 emissions were modified with the forestry rotation, although increasing CH_4 emissions could also be expected (Korkiakoski et al. 2019).

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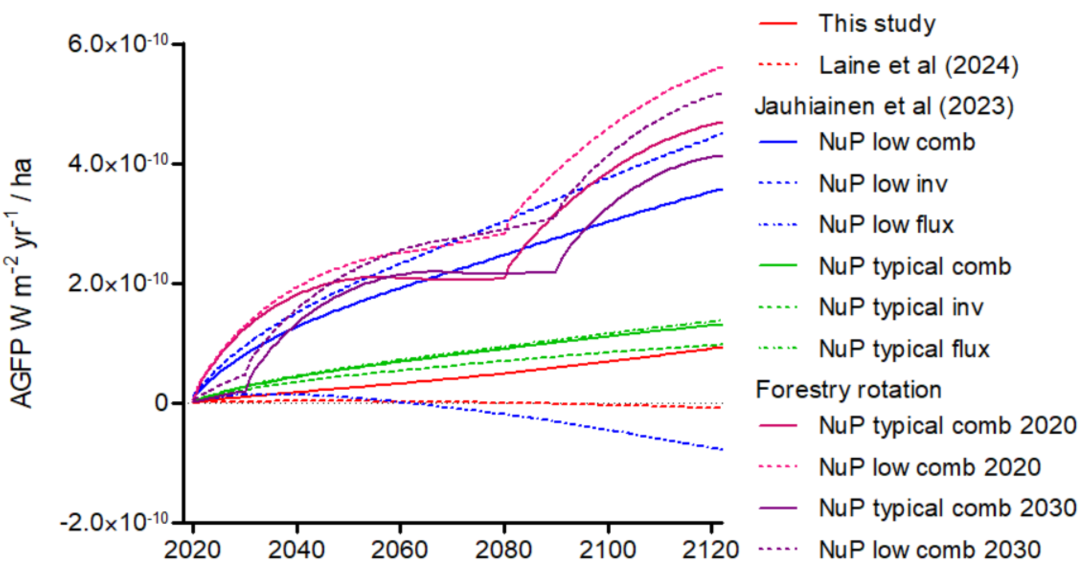

Fig. S2. Absolute global forcing potential of alternative drainage scenarios for nutrient-poor FDPs. NuP = nutrient poor, low = low productive, typical = productive, flux = GHG-flux studies, inv = peat inventory studies, comb = combined flux and inv. Forestry rotation trajectories with 60-year clearcut intervals start from 2020 or 2030. Positive values indicate climate warming impact of drainage.

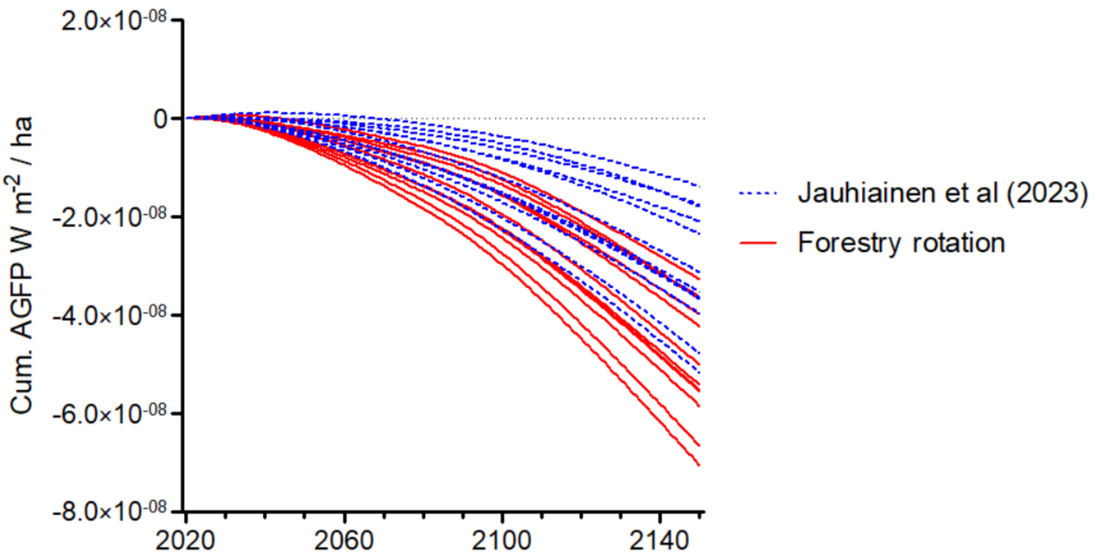

Fig. S3. Cumulative absolute global forcing potential calculated for the 12 restored FDPs studied by Laatikainen et al. (2025) with reference to averages of the alternative drainage scenarios following Jauhiainen et al. (2023) and the forest clearcut rotation. Negative values indicate climate cooling effect relative to drainage.

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