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## How does management affect soil C sequestration and greenhouse gas fluxes in boreal and temperate forests? – A review

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### ABSTRACT

The global forest carbon (C) stock is estimated at 662 Gt of which 45% is in soil organic matter. Thus, comprehensive understanding of the effects of forest management practices on forest soil C stock and greenhouse gas (GHG) fluxes is needed for the development of effective forest-based climate change mitigation strategies. To improve this understanding, we synthesized peer-reviewed literature on forest management practices that can mitigate climate change by increasing soil C stocks and reducing GHG emissions. We further identified soil processes that affect soil GHG balance and discussed how models represent forest management effects on soil in GHG inventories and scenario analyses to address forest climate change mitigation potential.

Forest management effects depend strongly on the specific practice and land type. Intensive timber harvesting with removal of harvest residues/stumps results in a reduction in soil C stock, while high stocking density and enhanced productivity by fertilization or dominance of coniferous species increase soil C stock. Nitrogen fertilization increases the soil C stock and N<sub>2</sub>O emissions while decreasing the CH<sub>4</sub> sink. Peatland hydrology management is a major driver of the GHG emissions of the peatland forests, with lower water level corresponding to higher CO<sub>2</sub> emissions. Furthermore, the global warming potential of all GHG emissions (CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O) together can be ten-fold higher after clear-cutting than in peatlands with standing trees.

The climate change mitigation potential of forest soils, as estimated by modelling approaches, accounts for stand biomass driven effects and climate factors that affect the decomposition rate. A future challenge is to account for the effects of soil preparation and other management that affects soil processes by changing soil

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temperature, soil moisture, soil nutrient balance, microbial community structure and processes, hydrology and soil oxygen concentration in the models. We recommend that soil monitoring and modelling focus on linking processes of soil C stabilization with the functioning of soil microbiota.

## 1. Introduction

Global forests removed approximately one third of annual anthropogenic CO<sub>2</sub> emissions from the atmosphere by increasing forest carbon (C) stock in 2020 (Friedlingstein et al., 2022). The global C stock of forests is estimated at 662 Gt of which 45% is in soil organic matter (FAO, 2020). At the European scale within 1 m depth, it is estimated that 22.1, 108 and 578 t C ha<sup>-1</sup> of soil organic C (SOC) are stored in forest floors, mineral soils and peat soils, respectively (De Vos et al., 2015). C stock losses can be driven by deforestation, forest fires, and forest management practices (incl. harvest), which all affect forest soil C sequestration potential in addition to direct effects on the tree stand (Ameray et al., 2021; FAO, 2020; Griscom et al., 2017). Studies on climate change mitigation measures by forest-based activities often focus on tree biomass and wood product sinks and substitution effects. For example e.g. Nabuurs et al., (2017) reported that the EU has the potential to achieve an additional combined mitigation impact through forests and the forestry sector of 441 Mt CO<sub>2</sub> year<sup>-1</sup> by 2050. However, the mitigation potential of forest soils is less often studied. Global analyses on soil C sequestration potential have focused on afforestation, avoided deforestation and peatland restoration concluding that soil C comprises 9% of the mitigation potential of forests and 72% for wetlands (Bossio et al., 2020).

On forest land, soil greenhouse gas (GHG) balance is strongly dependent on management decisions (Jandl et al., 2007; Mayer et al., 2020). The uncertainties in the climate change mitigation potential of European forests are largely due to unknown soil responses to management practices in the current and future climate. A comprehensive understanding of the effects of altered management practices on soil GHG fluxes and vegetation C stock changes is needed to develop forest-based climate change mitigation strategies and to inform expanding C markets.

Despite the importance of forest soils on climate change mitigation capacity, only 11 EU member states report soil GHG emissions and removals in their national GHG inventory<sup>1</sup>. Furthermore, these member states report very large uncertainties for their forest soil GHG estimates (e.g., Finland reports 31.5% and 150% uncertainty for mineral soils and peat soils, respectively) (Finland's National Inventory Report (NIR), 2021; Lehtonen and Heikkinen, 2016). Since the majority of EU member states relies on soil modelling to estimate soil GHG emissions and removals, it is important that the models applied in GHG inventories and scenario analyses accurately represent soil responses to forest management.

Recent review articles on forest management effects on SOC stocks and stock changes have discussed both direct and indirect responses of soil to various management regimes, from afforestation and tree species selection to harvesting intensity and soil mechanical preparation for stand regeneration (e.g. Jandl et al., 2007; Mayer et al., 2020). These reviews studies provide a good synthesis on the effects of land-use changes and management on forest soil properties and C stock changes, but there is a need for further analyses on the potential forest

management practices (incl. post-fire management, peatland hydrology management, wood ash fertilization, etc.) that may enhance climate change mitigation, either by soil C sequestration or reduced CH<sub>4</sub> and N<sub>2</sub>O emissions. Such knowledge together with understanding and modelling of soil processes will support science-based forest and climate policy, evidence-based management recommendations, national GHG inventories, and emerging trading of C sinks. To achieve the goals of the Paris Climate Agreement, the EU has set ambitious targets for C neutrality, which cannot be reached without strengthening the forest C sinks in plant biomass and soil.

A comprehensive understanding of the soil C sequestration potential is needed to plan forest-sector climate change mitigation measures, since incomplete or biased information may lead to inefficient climate policy and non-optimal use of resources. In general, current tools that are used for scenario analyses, which compare different forest use and climate change mitigation strategies, focus on the C sink of growing trees and ignore the GHG emissions and removals of forest soils.

**The objective** of this review is to evaluate forest management practices that may contribute to climate change mitigation by affecting soil C stocks and GHG fluxes in temperate and boreal forests. In addition, we review the effects of management practices with an aim to identify soil processes that affect soil C stock and soil GHG fluxes.

Based on review results we investigate how the effects of forest management on soil C and GHG fluxes are accounted for by models that can be used in GHG inventories and in scenario analyses. First, we will consider the effects of management practices that change organic matter input to the soil (and that can be modelled based on predicted stand development). Second, we assess the effects of management practices that have direct impact on soil characteristics (e.g., temperature, physical and chemical properties) and soil processes (e.g., rate of microbial activity, soil moisture or site hydrology). Finally, we use this review to discuss the implications for model development.

## 2. Methods

We used major databases (e.g. Web of Science, Google Scholar, CAP) to search for published peer-reviewed articles on forest management effects on soil C and GHG fluxes. In this review we consider forest management practices ranging from tree species selections and harvesting practices to fertilization, soil preparation, hydrology management of peatlands, fire management and biodiversity management, which all are widely applied on forest land (details of the search shown in Appendix A). Articles were considered in this study if they were peer-reviewed scientific papers and published in English between 2012–2022. However, a few older papers particularly relevant in the field were then incorporated (e.g. earlier meta-analyses), review on modelling implications was extended beyond searched articles, and the search period related to chapter 3.1 was extended to 2000–2022. This review covered various management practices and specific search terms easily identified a majority of the practices of interest. However, effects of the biodiversity management on soil GHG fluxes were reported in a spectrum of papers that only partly focused on soil GHG fluxes and therefore we did not find search terms that yielded relevant literature. Therefore, for biodiversity management only, the selection of reviewed papers was based on expert knowledge.

<sup>1</sup> According to the EU's NIR countries that are able to report their forest soil carbon stock changes in the mineral soils: EST, FIN, DEU, POL, and SWE reported that their forest soils are a carbon sink for the year 2019; while AUT, IRL, and PRT reported that their forests on mineral soils are a C source; and the rest of the countries were not able to estimate their soil carbon stock changes. Furthermore, peatland soils were reported to be a GHG source in 10 member states of the EU (DNM, EST, FIN, DEU, HUN, IRL, NLD, POL, ROU, SWE), while other countries were not able to estimate the emissions of peat soils.

### 3. Effects of tree stand management on soil C stock and GHG fluxes

#### 3.1. Tree species selection

##### 3.1.1. Rationale for tree species selection

Management can affect tree species composition throughout stand development by selecting tree species during the stand establishment phase and by tending and thinning. The choice of tree species is related to management objectives (e.g., production of high value timber) and can influence forest health and resilience, as well as aesthetic value, recreation, biodiversity, water quality and soil properties. In general, tree species selection in managed forests consider stand ecological conditions and focuses on forest stand productivity, which affects litter input, thus altering biogeochemical processes in the soil and C turnover (Feng et al., 2022).

##### 3.1.2. Impact of tree species selection on soil C stock

Studies in boreal and temperate forests found that coniferous species have similar or larger C stocks in the forest floor compared to broadleaves (Rehshuh et al., 2021; Vesterdal et al., 2013). The larger soil C stock under coniferous forests is likely explained by the more recalcitrant nature of coniferous needles (Scheu et al., 2003; Schulp et al., 2008), with higher lignin content and lower calcium concentration (Hobbie et al., 2006). The lower microfauna activity due to soil acidity (Scheu et al., 2003) can also influence the soil C stock under coniferous species, decreasing the quantity of organic carbon that accumulates in the mineral soil (Thuille and Schulze, 2006). Some studies suggest that labile organic matter is more important than recalcitrant organic matter for generating stable soil C (Cotrufo et al., 2015; Mayer et al., 2020). While in general there is faster litter decomposition of deciduous leaves, the implications for the stabilization of C in broadleaf forests are not yet clear (Mayer et al., 2020). The soil C in coniferous stands can be more susceptible to loss due to weak C stabilization and microenvironmental conditions, which are more vulnerable to climate change (Laganière et al., 2013). Because more C under coniferous species is stored in the organic layer, it is more vulnerable to disturbances such as forest fires and harvesting when compared to broadleaves (Jandl et al., 2021; Mayer et al., 2020).

Tree species mediate plant-microbial interactions (Tedersoo et al., 2016) and therefore dominant tree species change would affect soil microbial composition, understory vegetation, as well as litter and root exudate quality and quantity (Mundra et al., 2022). Replacing birch stands with spruce stands leads to higher fungal biomass in the organic layer correlating with higher SOC and a higher ratio between ectomycorrhizal and saprotrophic fungi (Danielsen et al., 2021; Mundra et al., 2022). Considering that most of stable SOC derives from roots and associated fungi (Adamczyk et al., 2019; Clemmensen et al., 2013), a higher amount of fungal biomass under spruce explains the concomitant increase in SOC. The change in fungal community included higher dominance of basidiomycete *Tylospora* sp. and ascomycete *Wilcoxina* sp. in spruce stands versus basidiomycete *Russula* sp. and ascomycete *Elaphomyces* sp. in native birch stands (Mundra et al., 2022) in line with host tree specificity.

An increase in soil microbial diversity has previously been observed with vegetation change from monocultures of broadleaved species to mixed stands (Šnajdr et al., 2013; Urbanová et al., 2015). In addition to affecting the soil microbial composition, changes in tree species composition can accelerate decomposition and soil C losses, but also increase nutrients available for plant growth, as observed in the replacement of Scots pine to Pyrenean oak (Fernández-Alonso et al., 2018).

Mixing tree species may lead to the expected soil properties based on the proportions of each tree species; however, for several soil properties this is not the case because of antagonistic or synergistic effects instead of pure additive effects (Saetre et al., 1999; Smolander and Kitunen,

2021). Furthermore, biomass over-yielding of mixed forests favours soil C sequestration (Augusto and Boča, 2022). A simulation study showed that in a boreal stand over-yielding was highest in a mixture of the coniferous species Scots pine (*Pinus sylvestris* L.) and Norway spruce (*Picea abies* (L.) H. Karst.), which also advanced soil C sequestration over that of mixtures between one conifer species and the deciduous silver birch (*Betula pendula* Roth.) (Shanin et al., 2014). Indeed, mixed boreal stands composed of Scots pine and Norway spruce have been found to contain higher C stocks in the deeper layer of mineral soil compared to monoculture stands of these same two species (Blaško et al., 2020).

Besides tree species composition, interacting factors such as climate, soil chemical and physical properties, and management also affect soil C stocks.

##### 3.1.3. Impact of tree species selection on GHG fluxes

Tree species composition affects soil respiration through litterfall, litter quality, and root respiration, mediated by the seasonality of environmental drivers (Mazza et al., 2021; Raich and Tufekcioglu, 2000). Some studies find only limited differences in soil carbon cycling between broadleaves and conifers when considering the production of dead organic matter (foliage and fine roots) and its mineralization, or stabilization (Augusto et al., 2015). However, others find differences in fluxes, such as Mazza et al., (2021), who found that fluxes of soil CO<sub>2</sub>, N<sub>2</sub>O, and CH<sub>4</sub> were higher in temperate compared to coniferous forests. They also observed a direct positive relationship between litter quantity and GHG fluxes (Mazza et al., 2021). The effect of tree species composition on soil GHG fluxes will likely be more evident with climate change as the expected changes in net primary production will directly affect litter quantity (Walkiewicz et al., 2021).

#### 3.2. Stand thinning

##### 3.2.1. Rationale for stand thinning

Thinning and harvesting with various methods are two major forest management activities used worldwide (Houghton, 2005; Peres et al., 2006). Thinning is applied to control tree species composition, stand structure and density and to provide early economic income in the early stages of forest rotation cycles (Campbell et al., 2009; Frey et al., 2003; Kolb et al., 1998; McDowell et al., 2008). Furthermore, by reducing competition for nutrients, water and light between the remaining trees, thinning leads to increased tree size and timber quality and therefore economic value of future products (Horner et al., 2010; Martín-Benito et al., 2010). Selection harvesting (i.e., continuous cover forestry) and shelterwood harvesting, which aim to regenerate stand with appropriate post-harvest distribution of large seedling/shelter trees and younger trees in lower canopy layers, can be applied with different intensities (Juutinen et al., 2018; Nieminen et al., 2018), while clear-cutting is followed by regeneration of entire stand.

Stand thinning may enhance forest resilience as well as drought tolerance (Aldea et al., 2017; Fernández-de-Uña et al., 2015; Mäkinen and Isomäki, 2004; Ruiz-Benito et al., 2013; Sohn et al., 2016) by allowing for increased soil water availability per tree in comparison to un-thinned stands (Bradford and Bell, 2017; D'Amato et al., 2013). Thinning also decreases the intensity of a potential crown fire by reducing crown density and continuity (Agee and Skinner, 2005; Banerjee, 2020). Modelled effects of thinning on stand evapotranspiration show that decreased growing stock and especially decreased amount of deciduous trees increase soil water stock (Leppä et al., 2020b, 2020a). In a meta-analysis, Sohn et al., (2016) pointed out that thinned stands maintained higher growth levels before, during and after drought events and that the benefits increased with thinning intensity. However, higher light availability in heavily thinned stands may promote understory resprout growth (Casals and Rios, 2018), increasing the competition for soil water, reducing the growth of overstorey trees and increasing risk of drought and wildfires (Giuggiola et al., 2018; Vilà-Vilardell et al., 2022, in press). Further, open canopies resulting from thinning may



increase risk of wind damage and allow wind to penetrate more easily to the understory and, together with solar radiation, contribute to drying surface fuels thus increasing surface wildfire spread (Banerjee, 2020).

### 3.2.2. Impact of stand thinning on soil C stock

Recent meta-analyses showed that forest stand thinning may have a slight negative impact on soil C stocks especially in the soil organic layer (Mayer et al., 2020; Nave et al., 2010; Zhang et al., 2018; Zhou et al., 2013). The C stock of the organic layer is reduced if thinning is intense, i. e. up to a 50% reduction of basal area compared to un-thinned stands (Achat et al., 2015; Bravo-Oviedo et al., 2015; Novák and Slodičák, 2004; Powers et al., 2012; Vesterdal et al., 1995). A meta-analysis showed that light thinning ( $\leq 33\%$  removal of stand basal area or stems) increased soil C stocks by 17%, moderate thinning (33–65% removal) did not alter soil C stocks, whereas heavy thinning ( $\geq 65\%$  removal) decreased soil C stocks by 8% (Zhang et al., 2018). They also showed that the soil C increased only in the early stages ( $\leq 2$  years) after thinning but was similar to control stands in later stages. The forest floor and organic layer are more vulnerable to thinning treatments than mineral C stocks (Johnson and Curtis, 2001; Nave et al., 2010; Ruiz-Peinado et al., 2016; Zhou et al., 2013). Indeed, most studies have reported no significant effects of thinning on soil C stocks of mineral soil (Achat et al., 2015; Cheng et al., 2013; Hoover, 2011; Jandl et al., 2007; Jurgensen et al., 2012; Kim et al., 2016; Noormets et al., 2015a; Powers et al., 2011; Ruiz-Peinado et al., 2013, 2016; Skovsgaard et al., 2006; Strukelj et al., 2015; Zhou et al., 2013), although others have documented soil C losses (Chiti et al., 2016; Grosso et al., 2018; Mattson and Smith, 1993; Moreno-Fernández et al., 2015; Mushinski et al., 2019; Strong, 1997).

### 3.2.3. Impact of stand thinning on soil GHG fluxes

To date, numerous experimental studies have examined the impacts of stand thinning on forest soil CO<sub>2</sub> emissions (CO<sub>2</sub> from decomposition and CO<sub>2</sub> from root respiration) with partly conflicting results. Two recent meta-analyses concluded that globally forest thinning significantly increases soil CO<sub>2</sub> emissions (Yang et al., 2022; Zhang et al., 2018). Though it was shown that thinning increases soil CO<sub>2</sub> emissions by 29% (Zhang et al., 2018), another study suggests only a 6.8% increase (Yang et al., 2022). However, these two meta-analyses also reported that the responses of soil CO<sub>2</sub> emissions depend on numerous factors including thinning intensity, post-thinning recovery time, stand type, stand age, measurement season, local climate, thinning-induced changes in litterfall, root biomass, soil nutrients, soil microclimate, and soil microbial community composition and activities (Adamczyk et al., 2015; Gao et al., 2015; Hao et al., 2019; Keller et al., 2005; Zhang et al., 2018).

The increase in CO<sub>2</sub> emissions occurs mainly in light and moderate thinning treatments. From light to heavy thinning, soil temperature increases while litterfall and fine root biomass decrease (Lei et al., 2018; Zhao et al., 2019). The increases in temperature promote soil CO<sub>2</sub> emissions while the decline in litterfall and fine root biomass decrease soil CO<sub>2</sub> emissions, which may result in an increase or decrease in soil CO<sub>2</sub> emissions in response to thinning intensity (Kulmala et al., 2014; Paul-Limoges et al., 2015; Poirier et al., 2014). As an example, light thinning intensities (20% and 40%) significantly increased soil CO<sub>2</sub> emission, while a heavy thinning intensity (60%) showed no impact on soil CO<sub>2</sub> emission (Zhao et al., 2019). Thinning significantly increases soil CO<sub>2</sub> emission in both broadleaved and mixed forests, but not in coniferous forests due to the strong differences in litterfall and woody debris quality.

The increase of soil CO<sub>2</sub> emissions occurs in the early stage of recovery after thinning ( $\leq 2$  years) as the increase in soil temperature, soil disturbance, dead root decomposition and soil nutrients promote microbial enzyme activities leading to higher soil heterotrophic respiration (i.e., CO<sub>2</sub> emission from decomposition). In addition, it is reported that forest thinning significantly increases soil CO<sub>2</sub> emissions during the growing season but not during the non-growing season (Hao et al.,

2019).

## 3.3. Harvesting practices

### 3.3.1. Rationale for stand harvesting practices

Conventional timber harvesting corresponding to stem-only harvesting (SOH, only merchantable stem wood harvested) is the most common harvesting practice worldwide (Mayer et al., 2020). However, due to the considerable interest in using biomass from forest harvesting to bioenergy, the use of whole-tree harvesting (WTH, i.e. harvesting the entire above-ground portion of a tree) and stump harvesting (SH, i.e. pulling out stumps after harvesting the aboveground) practices coupled to shortened rotation length may increase. The harvesting practices have differences in machinery requirements, and in the amount and type of residues that are retained on the site. Stump harvesting is the most intensive practice as it causes additional soil disturbance and reduced root litter input to soil, while SOH is less intensive allowing highest C input to the soil (leaves/needles, branches, twigs, small diameter stems) (Thiffault et al., 2011).

### 3.3.2. Impact of harvesting on soil C stock

Several meta-analyses and reviews have investigated the impacts of harvesting on soil C stocks (Achat et al., 2015; Clarke et al., 2021, 2015; Hume et al., 2018; James and Harrison, 2016, 2016; Johnson and Curtis, 2001; Nave et al., 2010; Thiffault et al., 2011; Walmsley and Godbold, 2010; Wan et al., 2018). Globally, forest harvesting reduces total soil C by an average of 10% with greater losses occurring in soil organic horizons (-30%) whereas the mineral horizons showed no significant or small changes (Clarke et al., 2021; James and Harrison, 2016; Zhou et al., 2013). These meta-analyses also pointed out that soil C losses are greater in broadleaf forests (-36%) than in coniferous or mixed forests (-20%; Nave et al., 2010). Chronosequence studies and meta-analyses suggest that soil C stocks start to recover only 1 to 5 decades following harvest (Achat et al., 2015; James and Harrison, 2016; Nave et al., 2010; Peltoniemi et al., 2004; Sun et al., 2004; Tang et al., 2009).

The retention of harvest residues in stem-only harvesting led to an 8% greater soil C stock compared to whole-tree harvesting, potentially due to a reduced soil disturbance and a higher amount of tree residues left on site. Some meta-analyses also pointed out that soil C loss increases according to the biomass harvesting intensity on both organic and mineral soil layers. Johnson and Curtis (2001) found that whole-tree harvesting led to a decrease (-6%) in soil C stocks whereas an increase was found with stem-only harvesting (+18%), while Clarke et al. (2015) reported that whole-tree harvesting may lead to only a small reduction in soil C compared with stem-only harvesting in the organic horizons. A meta-analysis showed that soil type and texture influence the susceptibility of soils following residue removal: while soil C stock was about 7% higher following stem-only harvesting compared with whole-tree harvesting in coarse- and medium-textured soils, differences between harvesting methods are not significant in fine-textured soils (Wan et al., 2018).

The negative harvesting impacts on soil C stocks can be alleviated or substantially decreased by minimizing machinery-induced soil disturbance (Achat et al., 2015; Laganière et al., 2010), choosing harvesting timing (e.g. harvesting during winter when soils are frozen or snow-covered), or by extending harvest rotations since short rotation lengths are less effective in C sequestration than long ones (Akujärvi et al., 2019; Law and Waring, 2015; Noormets and Nouvellon, 2015; Peng et al., 2002; Pussinen et al., 2002). A recent simulation study showed combination of stem-only harvesting coupled with longer rotation length produced a remarkably higher total C stock than that of whole-tree harvesting and shortened rotation length (Akujärvi et al., 2019).

### 3.3.3. Impact of harvesting on soil CO<sub>2</sub> fluxes

For forests growing on organic soils, such as peatlands, there is still

**Table 1**  
Summary of knowledge gaps and challenges in the modelling of the effects of proposed climate-wise management effects on forest soil.

Management practice		Knowledge and data gaps in assessing the effects on management on forest soils	Soil processes and characteristics needed to be included/improved in soil models
Tree stand management	Tree species selection Stand thinning and harvesting	<ul style="list-style-type: none"> <li>• Gaps in knowledge and long-term monitoring data of mixed-species forests</li> <li>• Lack of data and understanding on the understorey vegetation</li> <li>• Unknown characteristics of initial stand before thinning/harvesting</li> <li>• Inaccurate estimation of litter input to soil after thinning and harvesting, since response of biomass and biomass turnover rate to harvesting intensity vary</li> </ul>	<ul style="list-style-type: none"> <li>• Tree species' effects on soil C stock (e.g. litterfall, decomposability of organic matter, and vertical process)</li> <li>• Changes in environment conditions by thinning and harvesting, and their impact on changes in both biomass growth and decomposition process in soil</li> <li>• Recovery process after thinning and harvesting</li> <li>• Various thinning and harvesting methods</li> </ul>
Nutrient management	Nitrogen fertilization in boreal forests Wood ash fertilization	<ul style="list-style-type: none"> <li>• Interaction between microbial residues (resulting from changed microbial community) and soil C stabilization</li> <li>• Unknown long-term effects of wood ash fertilization on soil biological activity</li> <li>• Great variation in wood ash treatments and partly conflicting results</li> </ul>	<ul style="list-style-type: none"> <li>• Complex N cycle and N<sub>2</sub>O production pathway in soil</li> <li>• Other major nutrition (PK) cycle in forest soils and its impact on soil C stock</li> <li>• Incorporating potential impact of micronutrient</li> <li>• Changes in tree growth and amount and quality of litterfall by nutrient management</li> <li>• Changes in soil microbes by nutrient management</li> <li>• Distribution (e.g. horizontal and vertical) and amount of logging residues</li> <li>• Impact of compaction on soil porosity and hydraulic conductivity and C decomposition</li> </ul>
Site preparation		<ul style="list-style-type: none"> <li>• Huge variation of site preparation methods and their disturbance effects</li> <li>• Limited information on soil temperature and moisture changes after mechanical site preparation</li> </ul>	<ul style="list-style-type: none"> <li>• Impact of compaction on soil porosity and hydraulic conductivity and C decomposition</li> </ul>
Peatland hydrology management (elevated soil water level)		<ul style="list-style-type: none"> <li>• Limited long-term monitoring data</li> <li>• Spatial variation of peat hydraulic conductivity</li> <li>• Variation of peatland micro topography</li> <li>• Responses of microbial communities and their functioning to changed water level</li> </ul>	<ul style="list-style-type: none"> <li>• Hydrology in peatland</li> <li>• Long-term peatland development (thousands years)</li> <li>• Decomposition process at anaerobic condition, which is different from in upland soil</li> <li>• Long-term impact of changes in hydrology on soil C stock and GHG flux (particularly CH<sub>4</sub>) through hydrology and vegetation</li> </ul>
Fire management		<ul style="list-style-type: none"> <li>• Huge variation of intensity of burns</li> <li>• Effect of fire on decay process</li> <li>• Resistance of microbial groups/functioning on fire</li> <li>• Impact of pyrogenic carbon (biochar) on soil C stock and GHG fluxes</li> <li>• Long term data on GHG fluxes after fire management</li> </ul>	<ul style="list-style-type: none"> <li>• Impact on forest soils during and after fire through physicochemical and microbial dynamics</li> </ul>
Biodiversity management		<ul style="list-style-type: none"> <li>• Impact of retention trees and habitats on soils poorly quantified</li> <li>• Impacts of biodiversity on soil</li> <li>• Soil responses to BD management vary according to approaches/management due to local conditions</li> <li>• Interaction (and flow of information) between tree species and their mycorrhizal fungi</li> </ul>	<ul style="list-style-type: none"> <li>• Biodiversity management (e.g. retention forestry, multi-species effects) on soil C stock (particularly dead wood) through environments, litter input, and soil microbes.</li> <li>• Incorporating soil N and soil biological activity</li> </ul>

quite limited information concerning harvesting impacts on soil C stocks or CO<sub>2</sub> emissions. Mäkiranta et al. (2010) and Korkiakoski et al. (2019) observed a large net CO<sub>2</sub> emission from clear-cut sites during the first two to three years, resulting from the decomposition of logging residues and the peat soil. The rise of the soil water-table level, which is the result of reduced evapotranspiration (Leppä et al., 2020b) may decrease the decomposition rate of the peat (Mäkiranta et al., 2010), but with the lack of fresh C inputs following clear cuts, the site C balance becomes negative. How long negative C balance persists has not yet been documented. Selection harvesting effects on soil C depend on the intensity of the thinning cycles, as shown in the simulation study of Shanin et al. (2021). The impacts of machinery-induced soil disturbance and site preparation on peat CO<sub>2</sub> emissions have appeared to be minor (Lepilinen et al., 2022; Pearson et al., 2012).

### 3.3.4. Impact of stand thinning and harvesting on soil CH<sub>4</sub> and N<sub>2</sub>O fluxes

Thinning and harvesting are generally considered to reduce CH<sub>4</sub> uptake (or increase CH<sub>4</sub> emissions) and increase N<sub>2</sub>O emissions (Yang et al., 2022). However, the effects on N<sub>2</sub>O fluxes are often unclear in European forest soils when fluxes are low (Mazza et al., 2019). These impacts on CH<sub>4</sub> and N<sub>2</sub>O have been attributed to elevated temperature of surface soils due to the removal of vegetation, increased soil moisture caused by reduced evapotranspiration, larger amounts of organic matter input during logging operation, and accelerated soil nitrogen (N) cycling (McVicar and Kellman, 2014; Saari et al., 2004; Wu et al., 2011; Yang et al., 2022; Zerva and Mencuccini, 2005; Zhou et al., 2021).

The effects of thinning and harvesting on CH<sub>4</sub> and N<sub>2</sub>O fluxes are largely dependent on time elapsed after logging and intensity of logging. The logging effects are generally largest immediately after logging, gradually returning to the original level (McVicar and Kellman, 2014; Saari et al., 2004). A recent meta-analysis revealed that thinning suppressed CH<sub>4</sub> uptake (Hedges' *d* was 0.98), and the suppression was enhanced by thinning intensity (slope of meta-regression was significantly higher than zero (Yang et al., 2022)). N<sub>2</sub>O emissions were also influenced by logging intensity. Korkiakoski et al. (2019) reported clear-cutting in a boreal peatland forest caused substantial increase in N<sub>2</sub>O emissions from 1 to 228 ng N<sub>2</sub>O (m<sup>-2</sup> s<sup>-1</sup>), whereas selection harvesting had no significant impacts at the same study site (Korkiakoski et al., 2020). Thus, when evaluating the impact of thinning and harvesting practices on CH<sub>4</sub> and N<sub>2</sub>O fluxes, it is especially important to consider the intensity of logging and to evaluate the impact throughout the entire

forest rotation period, which is difficult to achieve without ecosystem modelling.

### 3.4. Modelling tree species selection, thinning and harvesting effects on soil climate change mitigation potential

Soil C models have been increasingly used to estimate the effects of tree species, stand thinning and harvesting on soil C stock, especially the long-term effects over many decades (e.g., Q model and CoupModel in Eliasson et al. (2013); Yasso model in Peltoniemi et al. (2004); EFIMOD in Shanin et al. (2014) and in Ahtikoski et al. (2022)). The modelled impact of the thinning practices on soil carbon and GHG fluxes depend on the harvesting intensity and interval, since post-harvest stand properties affects litter input (via estimated response of biomass and biomass turnover rates) to the soil, evapotranspiration, soil moisture and temperature (Shanin et al., 2016, 2021).

The accuracy of the modelled effect of tree species selection, stand thinning and harvesting on soil C stock changes depends on the accuracy of the quantity and quality of C input into the litter compartment (Table 1). Some soil C model variables (e.g., tree species, site index, frequency and intensity of thinning, length of rotation and handling of harvest residues) can control the intensity and timing of C input into litter compartment (Eliasson et al., 2013; Kaipainen et al., 2004; Moreaux et al., 2020; Pérez-Cruzado et al., 2012; Wutzler and Mund, 2007). These variables can interact with and be modified by others in the model. For instance, the tree species can determine the C input from harvested plant residue, but is modified by a site index (Pérez-Cruzado et al., 2012; Wutzler and Mund, 2007) and the length of rotation (Moreaux et al., 2020). In addition, the initial soil C stock, prior to thinning and harvesting, may also affect the results of simulation (Table 1). Thus, the users of soil C models should consider the disturbance and management history to reduce the uncertainty on initial soil C stock (Wutzler and Reichstein, 2007). In particular, harvesting impacts on soil C depend on multiple factors that are not fully accounted for by the current models including harvesting method, soil type and soil moisture at the time of harvesting (Table 1; Fig. 1).

## 4. Effects of nutrient management on forest soil C stock and GHG fluxes

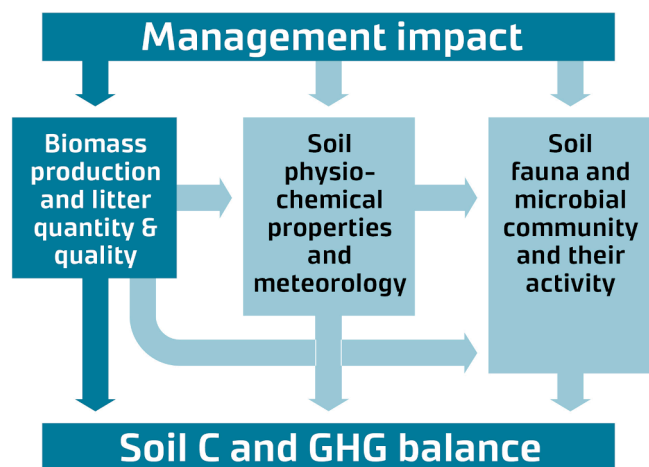
### 4.1. Nitrogen fertilization

#### 4.1.1. Rationale for forest nitrogen fertilization

Nutrient management such as fertilization with nitrogen (N), phosphorus (P) or wood ash (WA) can increase the availability of growth-limiting nutrients and therefore enhance tree growth. N is commonly the most deficient nutrient for tree growth, especially in the boreal region where atmospheric N deposition is low. In central Europe, N fertilization should be avoided since the high atmospheric N deposition may result in negative effects on forest ecosystems such as N leaching and soil acidification (de Vries et al., 2014). In northern Europe, N fertilizers are currently used to improve forest productivity particularly on stands that grow on nutrient-poor mineral soils (Fox et al., 2007; Moilanen et al., 2005; Noormets et al., 2015b; Saarsalmi et al., 2012). In certain regions, N fertilizer and boron are recommended to be applied together to balance the deficiency of boron (Saarsalmi and Mälkönen, 2001).

#### 4.1.2. Impact of forest fertilization on soil C stock

A recent literature review by Mayer et al. (2020) reported an overall positive effect of N fertilization on soil C stocks across different forest ecosystems. The effects of N fertilization on soil C stocks varied with soil layers. For example, a meta-analysis by Nave et al. (2009) found that N fertilization alone in North American and European temperate forests increased the C stocks in mineral soils by 23.5% but had no effect on the C stocks in forest floor; an experimental study in a European boreal



**Figure 1.** Forest management affects soil C and GHG balance by changing (i) stand biomass production and litter quality and quantity, (ii) soil physio-chemical properties, and (iii) soil organisms and their activity. Management impacts that are currently accounted for by the models used in the GHG inventories and scenario analyses are shown in dark blue and impacts that are not yet included are shown in light blue.

forest revealed that long-term N fertilization increased C stocks in both the mineral layer (15–167%) and organic layer (14–87%) (Mäkipää, 1995). The estimated accumulation rate of soil organic horizon C caused by added N was 10 kg C kg<sup>-1</sup> N for a boreal forest at Sweden (Maaroufi et al., 2015), and 17 to 23 Mg ha<sup>-1</sup> in the organic layer and 21 to 24 Mg ha<sup>-1</sup> in the top 10 cm of mineral soil in 30 coniferous stands under repeated N fertilization (Saarsalmi et al., 2014).

In addition to the main effects of N fertilization, the combined effects of N fertilization and tree species on soil organic C stock were reported by Hyvönen et al. (2008), with approximately twice the C sequestration rate at *P. abies* stands (13 kg C kg<sup>-1</sup> N) than that at *P. sylvestris* stands (7 kg C kg<sup>-1</sup> N). With 29 forest sites distributed over a latitudinal gradient in Sweden, Jørgensen et al. (2021) reported an increase in soil organic horizon C accumulation with N fertilization, with a larger increase at high latitudes. In contrast to using N fertilization alone, applying N fertilization together with wood ash can further increase soil C stock (Saarsalmi et al., 2012) while applying NPK together had no greater effect than N fertilization alone on soil C sequestration rate (Hyvönen et al., 2008).

N fertilization increased both the above- and below-ground litter input (Leppälampi-Kujansuu et al., 2014). In contrast to above-ground litter input, the below-ground litter input (e.g. roots, mycelia) contributes more to the soil stable C (Berhongeray et al., 2019). Positive effects of fertilization on below-ground litter input are mainly related to the increase in fine root turnover (King et al., 2002; Leppälampi-Kujansuu et al., 2014) and changes in the mycelia production of ectomycorrhizal fungi (Ekblad et al., 2013). Interestingly, the increase in soil C sequestration (47%) exceeded the increase in tree biomass C sequestration (3.7%) when N addition levels increased from 50 to 150 kg ha<sup>-1</sup> year<sup>-1</sup> (Frey et al., 2014). Application of P had an impact on the mycorrhizal community but not on fungal biomass and additional P was allocated to aboveground photosynthetic biomass rather than to forest soil (Zavišić et al., 2018).

The increase of soil C stocks in relation to N fertilization is mainly attributed to the overall decrease in organic matter decomposition rate ( $\geq 70\%$ ) and the increase in litter input ( $\leq 20\%$ ) (Franklin et al., 2003; Frey et al., 2014; Marshall et al., 2021). Consistent with the “microbial N mining” hypothesis, soil decomposition is decreased when N fertilization relieves the microbial population from the need to decompose organic matter to release nutrients (Craine et al., 2007). Fertilization with fast-release N increases net N mineralization but decreases (1) aerobic C mineralization, (2) C and N concentrations in the microbial biomass, and (3) fungal-to-bacterial biomass ratio (Maaroufi et al., 2015; Martikainen et al., 1989; Smolander et al., 1995, 1994; Treseder, 2008, 2004). Fungal-to-biomass ratio decrease may be driven by the loss of ectomycorrhizal fungi (Högberg et al., 2010). The decrease in decomposition is mainly linked to the inhibition of lignin-degrading enzymes and reductions in fungal biomass and activity. For example, Frey et al. (2004) observed a reduction in the activity of phenol oxidase accompanied by a significantly lower fungal-to-bacterial biomass ratio in fertilized temperate forest plots; Bonner et al. (2019) observed a reduction in the activity of peroxidase accompanied by a low ratio of enzymatic to nonenzymatic oxidation in fertilized boreal forest plots. A meta-analysis by Chen et al. (2018) found that the suppression in lignin-degrading enzymes is the main contributing factor to N-induced C accumulation in soil. Furthermore, a recent study by Hasegawa et al. (2021) emphasized that the N-induced shift in organic matter to contain increasingly lignin-derived compounds plays an important role in the accumulation of C.

The degree of N-induced changes in decomposition appears to be dependent on decomposition stage, litter quality (e.g. lignin content, C:N ratio), site condition (e.g. vegetation type, climate) and N addition level (Fog, 1988; Frey et al., 2004; Knorr et al., 2005). A recent meta-analysis by Gill et al. (2021) found that N fertilization stimulates the early-stage but slows down the late-stage decomposition rate. In addition to the changes in microbial demand for N, another significant factor

affecting decomposition rate is the dynamic change in litter chemical fractions, with a high proportion of soluble materials and non-lignified compounds at early stages and a high proportion of lignin-bound compounds at late stages.

#### 4.1.3. Impact of fertilization on soil GHG fluxes

As much as 40% of reduction in total soil respiration was observed with short-term (1–3 years) N fertilization in boreal and temperate forests (Franklin et al., 2003; Olsson et al., 2005; Sitaula et al., 1995). A meta-analysis by Janssens et al. (2010) reported that N-addition decreased heterotrophic respiration by 15% in an average of 36 N-manipulation forest studies. A 10-year daily measurement with automated chambers estimated that the soil CO<sub>2</sub> efflux was decreased with N input annually by 21% (Oishi et al., 2014). Despite the generally negative response of soil CO<sub>2</sub> efflux to N fertilization, a positive response was reported from studies where N fertilization potentially stimulated photosynthesis, soil microbial activity and rhizosphere respiration (Janssens et al., 2010). Taken together, N-induced reductions in soil respiration are mainly attributed to reductions in below-ground C allocation, shifts in the saprotrophic community and increased abiotic stabilization of soil organic matter (Janssens et al., 2010).

Although methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O) fluxes are much lower than the amount of CO<sub>2</sub> uptake in forest ecosystems, the global warming potentials (GWP) of CH<sub>4</sub> and N<sub>2</sub>O are almost 30 and 300 times higher than that of CO<sub>2</sub>, respectively. Siljanen et al. (2020) emphasized the importance of evaluating soil N<sub>2</sub>O fluxes because the cooling effect of N<sub>2</sub>O uptake was on average 35% of that of CH<sub>4</sub> uptake in a spruce forest.

A long-term experiment in a Swedish forest by Håkansson et al. (2021) reported that repeated N fertilization decreased the soil CH<sub>4</sub> uptake over time. An 8-year long experiment in a temperate deciduous forest by Chan et al. (2005) found that N-fertilization decreased soil CH<sub>4</sub> uptake by 35%. At the global scale, N-induced changes in CH<sub>4</sub> uptake is biome-specific and dose-dependent, with the effect shifting from positive to negative when the N addition dose increases in boreal and temperate forests (Xia et al., 2020).

The release or sink of N<sub>2</sub>O is mainly associated with the nitrification and denitrification process, where soil microorganisms such as nitrifiers and denitrifiers play an important role in the biological transformation of N. The effects of N fertilization on soil N<sub>2</sub>O emission or uptake are highly dependent on the soil environment. For example, the uptake of N<sub>2</sub>O was favored by high soil silt and water content (Siljanen et al., 2020). The N induced increase in N<sub>2</sub>O emission through nitrification or denitrification appears to be soil pH dependent with increases in emissions not always observed in acid boreal forest soils (Saarsalmi and Mälikönen, 2001; Smolander et al., 1995). A long-term experiment in Sweden showed that N<sub>2</sub>O emissions increased in the fertilization years but not during the subsequent years (Håkansson et al., 2021).

Overall, the effects of fertilization on GHG emission or uptake appears to be dependent on soil water content, soil silt content, soil pH, fertilization addition rate, and the time since fertilization (Brumme and Beese, 1992; Håkansson et al., 2021; Jassal et al., 2011; Siljanen et al., 2020; Sitaula et al., 1995; Xia et al., 2020).

## 4.2. Wood ash fertilization

### 4.2.1. Rationale for wood ash fertilization

Wood ash (WA) fertilization is recommended for peatland forests, where tree growth is mainly limited by P or potassium (K) rather than N (Moilanen et al., 2005). In some countries, WA is used to compensate for the nutrient losses caused by harvesting and in forests growing on mineral soils. In addition to alleviating nutrient deficiency, buffering soil acidification is another focus in forest nutrient management. The WA fertilization and liming (addition of Ca and Mg) have been adopted for this purpose, particularly in Northern Europe, where the forest soils are naturally acidic, and in Central Europe, where atmospheric N and



sulphur deposition increase the risk of soil acidification. However, liming is not recommended as a growth increasing measure after [Derome et al. \(1986\)](#) reported a long-time (18-year) decrease in tree growth after liming at the rate of  $2\text{ t ha}^{-1}$ . Therefore, liming is not further discussed as a climate change mitigation measure in this review. In addition to WA fertilization and liming, rock dust (stone meal) as the by-product of mining, was occasionally applied to increase the pH of acidified soil and to provide micronutrients (e.g. S, Ca, K, Mg) for plants ([Mersi et al., 1992](#); [Szmidski and Ferguson, 2004](#)).

Results on WA fertilization effects on soils vary depending on the quantity and quality of the applied ash, soil type, site type, and time passed since fertilization. The quality varies regionally based on both source materials and incineration methods ([Augusto et al., 2008](#); [Marsca et al., 2017](#); [Pitman, 2006](#); [Vassilev et al., 2013](#)). The effects of WA on soil chemical properties and microbial processes in C and N cycling may last decades (e.g. [Rosenberg et al., 2010](#); [Saarsalmi et al., 2014, 2012](#)). In addition to increasing mineral nutrient concentrations, WA may decrease soil acidity, the response depending on both dose and ash type. According to the review by [Huotari et al. \(2015\)](#), application of WA of  $3\text{--}5\text{ Mg ha}^{-1}$  generally decreased the acidity by  $0.5\text{--}3$  pH units in both surface peat and the organic forest floor on top of mineral soils. Granulated ash, especially, does not always raise the pH ([Huotari et al., 2015](#); [Maljanen et al., 2014](#)), probably because the granules dissolve gradually.

#### 4.2.2. Impact of wood ash fertilization on soil microbes

The higher pH and nutrient concentrations due to WA fertilization shape the soil microbial communities and their activity in both mineral and peat soils. They may, e.g., increase the fungi to bacteria ratio by promoting the abundance of both mycorrhizal and wood-decomposing fungi directly or via increased tree growth ([Peltoniemi et al., 2016](#)). [Cruz-Paredes et al. \(2017\)](#) in turn found WA leading to a reduced importance of fungi and shifts in the bacterial community in the forest floor. The dose (quantity) of WA shapes the responses; e.g., [Bang-Andreasen et al. \(2017\)](#) found bacterial numbers increasing up to a WA dose of  $22\text{ t ha}^{-1}$  followed by a detrimental decrease at an extreme dose of  $167\text{ t ha}^{-1}$ , which is far above doses used in practice. Also the bacterial community composition changed with copiotrophic bacteria responding positively up to the dose of  $22\text{ t ha}^{-1}$  while an adverse effect was seen for oligotrophic bacteria. Few general patterns have been reported so far, possibly because of the great variation in the applied WA treatments, methods for analysing the soil communities, and site characteristics (also see the review by [Huotari et al. \(2015\)](#)). Yet, it has been concluded that WA influences ectomycorrhizal fungal species composition, but the belowground mycorrhizal biomass or species richness are not generally affected ([Kjoller et al., 2017](#)). Overall, bacteria seem to be more responsive than fungi to WA-induced changes in pH ([Cruz-Paredes et al., 2021](#)). WA impacts on soil fauna and the decomposer food web have overall been deemed minor ([Huotari et al., 2015](#); [Mortensen et al., 2020](#); [Qin et al., 2017](#)). Potential Cd bioaccumulation risk is highest in systems with many earthworms, isopods and snails ([Mortensen et al., 2018](#)). WA further shapes the ground vegetation composition, often leading to reduced abundance of shrubs and mosses, and increased abundance of forbs and graminoids ([Ethelberg-Findsen et al., 2021](#); [Maljanen et al., 2014](#)). These changes will affect the litter inputs to the soil, as well as the soil communities.

#### 4.2.3. Impact of wood ash fertilization on soil GHG fluxes

In earlier research reviewed by [Huotari et al. \(2015\)](#), increased GHG emissions were not observed in the short term ( $< \sim 5$  years) following WA fertilization in peatland forests (also, e.g., [Rütting et al. \(2014\)](#)). Methane emissions are not likely to increase at any time scale, as WA usually leads to lowered water-table levels (WTL) due to the increased growth – and evapotranspiration capacity – of the tree stands on drained peatlands. For example, methane emissions were shown to be marginal when the WTL is lower than 30 cm below the peatland surface (e.g.,

[Ojanen et al., 2010](#)). As a result of the interaction between plant productivity and WTL, WA may increase the soil  $\text{CH}_4$  sink on drier sites ([Maljanen et al., 2014](#); [Ojanen et al., 2019](#)).

Increased  $\text{N}_2\text{O}$  emissions from peatlands following WA fertilization have not been observed either ([Huotari et al., 2015](#); [Maljanen et al., 2014](#); [Ojanen et al., 2019](#)). In some laboratory incubations, WA has actually decreased the peat soil  $\text{N}_2\text{O}$  production rate, especially when pH did not increase simultaneously ([Bornø et al., 2020](#); [Liimatainen et al., 2014](#); [Maljanen et al., 2014](#)). In some field studies on peatlands, a reduction in  $\text{N}_2\text{O}$  emission has also been observed ([Rütting et al., 2014](#)). In mineral soils that have previously received N fertilization, WA fertilization may increase in  $\text{N}_2\text{O}$  emissions ([Bornø et al., 2020](#)).

Reported WA effects on  $\text{CO}_2$  emissions from peatlands are more variable, and seem to depend on time passed since fertilization. In spite of the improved conditions for organic matter decomposition due to increased pH and nutrient status, WA has not led to increased decomposition rates or soil  $\text{CO}_2$  emissions in the short term ( $< \sim 5$  years; [Huotari et al., 2015](#)). This may be partly because ash application initially disturbs the soil microbial community ([Björk et al., 2010](#)). In the longer term ( $> \sim 5$  years), however, increased decomposition rates for, e.g., needles and cellulose, and increased heterotrophic soil respiration have been observed ([Maljanen et al., 2014](#); [Moilanen et al., 2012](#); [Ojanen et al., 2019](#); [Saarsalmi et al., 2014](#)). Soil  $\text{CO}_2$  emissions seem to increase especially in sites with high soil N concentrations, while sites with low soil N may show a  $\text{CO}_2$  sink ([Moilanen et al., 2012](#); [Ojanen et al., 2019](#)). The most N-rich drained peat soils, which are often a net source of C prior to WA fertilisation ([Ojanen et al., 2013](#)), are thus likely to become even greater sources thereafter. The increasing wood production may more than compensate for the loss of C from the soil for several decades ([Moilanen et al., 2012](#); [Ojanen et al., 2019](#)), but if the net soil emissions continue, N-rich organic soil sites will undoubtedly become net sources of  $\text{CO}_2$  to the atmosphere in the long term ([Ojanen et al., 2019](#)).

In addition to direct soil responses to WA fertilization, it is critical to evaluate the changes in the litter input quantity and quality over the time since fertilization. Increased tree litter inputs, especially, may cause high soil  $\text{CO}_2$  fluxes to the atmosphere, but the overall balance may result in net soil C inputs ([Straková et al., 2012](#)). The increased production should thus be considered in addition to the changes in the decomposition of (and heterotrophic respiration from) the peat soil.

In mineral soil forests as well as in peatlands, the effect of WA fertilization on soil  $\text{CO}_2$  emissions is dependent on the fertility and acidity of the site. In mineral soil forests rich in N, wood ash amendment in high doses may have a negative effect on the C balance ([Rosenberg et al., 2010](#)). Also, stimulated SOM turnover and an increase in the labile fraction of SOM has been observed for coniferous forests in Denmark and Finland ([Hansen et al., 2016](#)). In coniferous forests WA fertilization may increase C mineralization and soil respiration ([Rosenberg et al., 2010](#); [Saarsalmi et al., 2012](#)). This C source seems rather minor compared to other C fluxes in forest ecosystems; however, the net effect on soil C balance has not been thoroughly evaluated. Interpretation of changes in total soil respiration is challenging whenever major changes in vegetation composition occur as well, since those may affect root production and the proportion of autotrophic root respiration. Overall, the effects of WA fertilization on GHG emission or uptake appears to be dependent on soil water content, soil silt content, soil pH, fertilization addition rate, and the time since fertilization ([Brumme and Beese, 1992](#); [Håkansson et al., 2021](#); [Jassal et al., 2011](#); [Siljanen et al., 2020](#); [Sitaula et al., 1995](#); [Xia et al., 2020](#)).

### 4.3. Chemical agents

#### 4.3.1. Rational for use of chemical agents

While chemical agents are commonly used in commercial forestry globally, their use in European forests appears to be limited ([McCarthy et al., 2011](#)). Pesticides and herbicides have been applied in Europe in afforestation, during soil preparation for seed sowing and in young

forest stands as a means to control competing vegetation and insects (Karmilowicz, 2019; Östlund et al., 2022). Nowadays, application of chemical agents is not common in European forests. Herbicides are still used in some boreal and temperate regions, especially during the first few years of seedling establishment in planted forests.

According to an experimental study with young loblolly pine (*Pinus taeda* L.) herbicides tended to decrease soil C and in some areas the soil C pool was reduced by about 0.5 kg C·m<sup>-2</sup> (Sartori et al., 2007). Also, plots where fertilizers and herbicides were applied showed higher annual cumulative resin-extractable N (372 kg N·ha<sup>-1</sup>) than plots treated only with herbicides (13 kg N·ha<sup>-1</sup>; Sartori et al., 2007). The use of herbicides resulted in a decrease in fine root biomass pool at 540 kg C·ha<sup>-1</sup> and in an increase in forest floor biomass at 10,050 kg C·ha<sup>-1</sup> (Sartori et al., 2007).

In a study with white spruce (*Picea glauca* [Moench] Voss) seedlings, soil C in the forest floor was reduced from 2.13 to 1.53 kg C·m<sup>-2</sup> and available P decreased from 3.49 to 0.70 g P·m<sup>-2</sup> due to herbicide application (Burgess et al., 1995). Very little effect on C and N stocks were observed in the forest floor when both herbicide and fertilizer were used, and no significant changes were observed in the mineral soil (Burgess et al., 1995). Similar results were found in a study with white pine (*Pinus strobus* L.) seedlings (Burgess et al., 1995).

Other studies found that the litter decomposition rate was not affected by herbicide application. Fletcher and Freedman, (1986) did not observe significant change in the litter decomposition rate of red maple (*Acer rubrum* L.) and white spruce foliage when herbicides were used. A reduced speed of litter degradation was observed only when herbicide concentrations were over 50 times higher than residue concentration normally found after silvicultural herbicide treatments.

#### 4.4. Modelling effects of nutrient management on soil C stock and GHG fluxes

The increase of soil C stocks in relation to N fertilization is mainly attributed to the overall decrease in organic matter decomposition rate, with less than 20% of the increase attributed to litter input ( $\leq 20\%$ ) (Franklin et al., 2003; Frey et al., 2014; Marshall et al., 2021). One-dimensional soil C models, e.g., Yasso or soil C module of CENTURY (Sierra et al., 2012), account for the increase of litter input with soil fertilization via increased biomass production and constant death rates of the biomass compartments (leaves, branches roots, etc.). However, C-only models are generally unable to modify the decomposition rate according to soil fertility (or added nutrients) and therefore underestimate C stocks in soils with higher nutrient status and insufficient drainage (Dalsgaard et al., 2016; Tupek et al., 2016). Ågren et al. (2001) found that the decomposition rate decreased when the decomposer efficiency increased, encouraging rapid formation of recalcitrant compounds. Furthermore, soil C stabilization is influenced by the interaction between microbial residues, mineral surfaces, and complex polymers with varying stoichiometry, and these interactions are not explicitly considered in traditional soil models (Table 1). In light of the Microbial Efficiency-Matrix Stabilization (MEMS) framework by Cotrufo et al. (2013), integrating more flexibility in microbial substrate use efficiency (rather than a fixed parameter) and specifying the chemical structure of soil and litter in modelling would largely improve our understanding and predictions in long-term soil C and N cycling. For WA fertilization as a supply of other limiting nutrients than N e.g., P in peatlands, models often lack explicit representation of N and P cycling and when included their representation can be overly simplified.

For example, in the original version of the soil module of the CENTURY model (Metherell et al., 1993), the topsoil N is defined as linearly related to lignin content, which drives the decomposition rate of the slowest (i.e., passive) SOC pool. Modeled SOC pools are also surprisingly insensitive to topsoil N; e.g. a 20% increase in topsoil N and litter resulted in a 0.5% and a 15% increase of equilibrium SOC, respectively (Tupek et al., 2016). Because N fertilization has been associated with lower fungal/bacterial ratio, reduction of lignin degrading enzymes,

phenoxides and peroxidase (Bonner et al., 2019; Chen et al., 2018; Frey et al., 2004), model advances to capture N fertilization impacts could also include modifying decomposition rates of fast and slow pools or explicitly simulating nutrient pools along with C pools.

Baskaran et al. (2017) incorporated ectomycorrhizal fungi (ECM) into a soil C-N model for a N-limited forest site and showed that ECM growth promoted the tree growth but also increased SOM decomposition thus reducing the soil C stock. There are many ways to represent mechanisms such as microbial adaptation to soil nutrient conditions variations in models; thus multiple types of measurements (e.g., litter decomposition rates, microbial and enzyme activities, soil pools) that correspond to modeled processes are critical to distinguish between different model representations (Manzoni et al., 2021).

Some models consider the N effect of one or two microbial pools (Abramoff et al., 2017), but it is not yet clear whether models that explicitly represent microbial community structure can be scaled up to represent any biogeochemical fluxes at field scales (Kaiser et al., 2014; Marsland et al., 2020). Linking microbial enzyme production and decomposing substrate is common in microbial models (Abramoff et al., 2018, 2022; Manzoni et al., 2021). However, these models need further development to better account for multi-nutrient cycling between plants, microorganisms and soil which may vary, especially across hydrologically different ecosystems e.g., forests vs peatlands.

## 5. Mechanical site preparation, vehicle movements and stump harvesting

### 5.1. Effect of mechanical site preparation on forest soil C stock and GHG fluxes

#### 5.1.1. Rationale for forest soil preparation

Site preparation aims to improve regeneration success by both improving the germination of tree seeds and promoting the survival and growth of tree seedlings. The site preparation decreases the competition by ground vegetation; increases soil aeration, temperature and moisture conditions, increases nutrient availability; and reduces damaging effects by small mammals and insects (that avoid bare soil), especially pine weevil attacks (reviewed by Mayer et al. 2020; Sikström et al. 2020).

The average proportion of disturbed soil surface area following mechanical site preparation was 37% for mounding, 52% for disc trenching and 62% for ploughing (Sikström et al., 2020). However, the variation between sites with regard to surface disturbance is very large, for example for mounding disturbance can vary between 17% and 67%. In Northern Europe mounding and disc trenching are currently by far the most common site preparation treatment. In Finland mounding comprised about 70% of total site preparation area in 2021 (Suomen virallinen tilasto 2022). The environmental conditions that decomposing microbes experience can be very different for undisturbed soil, exposed mineral soil surfaces, and buried organic or double organic layers (Palviainen et al., 2007).

In addition to site preparation, forest soil is mechanically affected by vehicle movements during the harvesting and other forest operations having partially similar effects as mechanical site preparation. The impacts of soil disturbance by mechanical harvesting depend largely on the intensity of the harvest but also on soil type and soil properties as well as slope. In Mediterranean studies, the forest floor layer was decreased or was even absent for several years after heavy mechanized forest operations and the heavy machinery increased soil compaction and had negative effects on topsoil aggregate formation and increased soil compaction (Gartzia-Bengoetxea et al., 2011, 2009a, 2009b). Simultaneously, the disruption of soil aggregates and exposure of previously protected soil organic matter to microbial attack reduced the amount of resistant soil C pool (Gartzia-Bengoetxea et al., 2011). The overall effects of mechanized forest operations on the soil C are partly due to introduced changes in the ground vegetation and tree seedling cover, erosion risks and eventually tree stand development.

### 5.1.2. Impact of site preparation to soil C stock

Both coniferous needles, root litter, and logging residues are degraded faster when buried in the soil than when left on the soil surface (Johansson, 1994; Lundmark-Thelin and Johansson, 1997; Mjöfors, 2015; Mjöfors et al., 2015). In addition, mechanical site preparation may break litter and residues, further enhancing their degradation. Therefore, nutrients availability increases after soil preparation. However, organic matter decomposition of the buried humus layer is not necessarily faster than that of the undisturbed soil profile humus layer. Instead, the decomposition of the 'old' C in the humus layer may be similar or even lower in the buried double humus layer in the mounds and in the single humus layer after soil inversion than in the humus layer of undisturbed soil surface (Smolander et al., 2000; Smolander and Heiskanen, 2007). This may be explained by the lack of plant C inputs which can enhance the degradation of existing organic matter (i.e., priming). This is supported by the observation that the planting of tree seedlings stimulated C mineralization in the double humus layer of the mounds (Smolander et al., 2000). Still more information from different forest sites is needed on the rate of decomposition of the surface organic matter when buried in mineral soil at depth. Its degradation and, as stated by Mayer et al. (2020), its stabilization mechanisms probably depend on the site preparation practice, forest type and soil type.

Studying site preparation effects on soil C stock is challenging since all types of soil surfaces created by site preparation should be sampled and their proportion of the total area should be estimated, as in gas flux measurements. In addition, soil sampling should extend deep enough to take into account the organic matter accumulation such as the buried organic layer in the site preparation plots. Response of forest soil C stock to mechanical site preparation in the long term has been the focus of several studies performed in Sweden where plots with different types of site preparation treatments have been compared to plots without mechanical site preparation. Ten years after deep soil cultivation, soil C stock in the uppermost 50 cm of the soil profile did not differ from that of the manual patch scarification treatment (Nordborg et al., 2006). Ploughing combined with stump harvest had slightly decreased the C stock determined after 22–24 years in the uppermost 70 cm as compared to light, manual patch scarification, but the total C stock was similar in both treatments because of enhanced tree growth in the ploughing treatment (Egnell et al., 2015). Mjöfors et al. (2017) compared the effects of disc trenching, mounding and ploughing on the C stocks of different ecosystem compartments when ca. 25 years had elapsed after site preparation in three coniferous stands in Sweden. All site preparation treatments increased the C stock in the trees and in the whole ecosystem but did not significantly affect the C stock in the uppermost 30 cm of soil. The average C stocks ( $\text{Mg ha}^{-1}$ ) in the whole ecosystem were 129 for control without soil preparation, 157 for ploughing, 152 for mounding, 138 for disc trenching treatments, and the respective total C stock for the humus layer and mineral soil 49.5, 47.2, 50.3 and 50.1  $\text{Mg ha}^{-1}$ . Reductions in total soil C contents were reported 60–70 years after mechanical site preparation (Örlander et al., 1996), but the methods that were used disturbed close to 100% of the soil surface and are not used anymore. Mjöfors et al. (2017) concluded that any potential losses in soil C stock are small compared to the gains in the tree C stock and recommended the moderate soil preparation methods, mounding or disc trenching, since they disturb ecological, aesthetic and recreational ecosystem services less than ploughing.

Stump harvesting for bioenergy is another method of site preparation. Stump removal significantly increases soil disturbance when compared to conventional more superficial site preparation (Kaarakka et al., 2018; Saksa, 2013; Strömngren et al., 2012; Strömngren and Mjöfors, 2012). It affects soil C pools directly by reducing soil organic matter formation due to removal of stumps and coarse roots and more indirectly, by mixing the topsoil and subsoil materials. Thus, stump harvesting may reduce soil organic matter in the short term. However, results from long-term experiments available (32–39 years) could not verify any soil C decline caused by stump removal (Persson and Egnell,

2018).

### 5.1.3. Impact of site preparation on soil GHG fluxes

Fluxes of C from the soil surface are affected by the entire soil profile. Efficient mixing of the organic layer with the uppermost mineral horizons increased soil respiration at a boreal clear-cut site (Mallik and Hu, 1997).  $\text{CO}_2$  fluxes in the forest regeneration areas vary; fluxes from tilts and mounds are typically larger than from undisturbed soil during the first one or two years after soil preparation, although at some sites, the opposite is true (Mjöfors et al., 2015; Pumpanen et al., 2004; Strömngren et al., 2017; Strömngren and Mjöfors, 2012). This variation has been explained by the different moisture and temperature conditions. From the exposed mineral soil surface,  $\text{CO}_2$  emissions are typically low. When the proportions of the different surfaces in the whole soil surface were taken into account, disc trenching did not affect soil  $\text{CO}_2$  emissions in the first year but increased the emissions by 10% in the second year (Strömngren and Mjöfors, 2012). In another Swedish study where the emissions were monitored in 14 regeneration sites after disc trenching, patch scarification, and control with no disturbance, soil preparation decreased emissions slightly but not significantly in the first two years (Strömngren et al., 2017). To summarise, the additional effect of soil preparation on mineral soil  $\text{CO}_2$  emissions after clear-cutting seems to be minor and is at least partially controlled by soil moisture conditions and the presence of fresh plant material.

Site preparation of organic soils strongly impacts soil GHG fluxes. E.g., partial soil scarification/ploughing before planting the seedlings results into bare peat exposure on mounds/ridges and microscale variation in peat temperature and moisture. Both factors are crucial in controlling  $\text{CO}_2$ ,  $\text{CH}_4$  and  $\text{N}_2\text{O}$  emissions in an altered peatland's microtopography after the clear cut (Korkiakoski et al., 2019).  $\text{CO}_2$  emissions are expected to increase in warmer soil but decrease under moist or dry extremes. Dry peat patches may be slightly more efficient in oxidation of atmospheric  $\text{CH}_4$  than moist patches, with higher  $\text{CH}_4$  and  $\text{N}_2\text{O}$  emissions in water saturated peat areas and ditches.

Strömngren et al. (2016) showed that in the first two years after site preparation on mesic sites (mounding on two sites and disc trenching on one site), the soil was predominantly a methane sink with the exception of wheel ruts and other water-covered pits that emitted methane. When proportions of different soil surfaces of the study area were taken into account, the soil was either a source or a sink for methane depending on the site, but site preparation did not have a significant effect (Strömngren et al., 2016).

Strömngren et al. (2016) found that oxygen controls both nitrification and denitrification; nitrification being an aerobic process and denitrification anaerobic. Therefore, the depth of ground water affects  $\text{N}_2\text{O}$  emissions strongly, as well as the availability of N.  $\text{N}_2\text{O}$  fluxes were affected by the type of soil disturbance but in different ways at different sites, depending on the ground water level and soil compaction. However, as with methane emissions, when the mean fluxes of  $\text{N}_2\text{O}$  were scaled according to the cover of disturbance types, site preparation had no significant effect (Strömngren et al., 2016).

Persson and Egnell (2018) recently reviewed the effects of stump removal on GHG emissions and soil C stocks in northern Europe and America. The general conclusion was that stump harvesting on podzolic soils had no effect on  $\text{CO}_2$ ,  $\text{N}_2\text{O}$  and  $\text{CH}_4$  emissions.

### 5.2. Modelling site preparation effects on soil C stock and GHG fluxes

Effects of mechanical site preparation are not currently considered in the forest soil models that are applied to estimate forest soil C dynamics and GHG fluxes (Fig. 1)(Table 1). Currently applied soil models could account for the effects of site preparation that are driven by soil moisture and temperature changes, if representative data for such changes are made available. Mechanical site preparation may also redistribute soil organic matter. Since several models consider a vertical distribution of soil C and N, simulating effects of site preparation is within a reach of the



current models. Furthermore, site preparation can change the succession of forest vegetation with implications for soil microbiota; representing these processes would require more complex model developments, including vertical transport of organic matter and dynamic vegetation.

## 6. Peatland hydrology management and peatland restoration

### 6.1. Effects of peatland hydrology management or restoration on soil GHG fluxes

#### 6.1.1. Rationale for peatland management

Peatlands have been subject to artificial drainage for centuries due to land-use changes, including forestry or bioenergy plantations (Tapio-Biström et al., 2012). In boreal and temperate zones, around 15 Mha of peatlands have been drained for forestry to achieve economically wood production (Paavilainen and Päivänen, 1995). Drainage leads to a rapid drawdown of water table level (WTL) that is over the subsequent decades which is exacerbated by increased evapotranspiration of shrubs and trees. Consequently, the oxic layer above the WTL enables more efficient microbial decomposition, which enhances the CO<sub>2</sub> emissions. At a global scale, CO<sub>2</sub> emissions from drained peatlands have increased by more than 20% from 1990 to 2008 (from about 1058 to 1298 Tg including agricultural use; (Joosten, 2009). According to the national GHG inventories, CO<sub>2</sub> emissions from drained organic forest soils, which are often peatlands, have a substantial impact on forest C sinks (e.g. National Inventory Report (NIR) Finland, 2021). When under forest cover, drained sites may be C sinks at the ecosystem level due to the enhanced C accumulation in forest biomass (e.g., Hommeltenberg et al., 2014; Ojanen et al., 2013). However, it is evident that soil of nutrient-rich drained peatlands are C sources (Ojanen et al., 2013; Ojanen and Minkkinen, 2020), and harvesting by clear-cutting will turn peatland ecosystems into a large source of GHG emissions (Korkiakoski et al., 2019). Thus, novel management practices that can decrease emissions are needed to reach the climate targets of the LULUCF sector.

In forestry-drained peatlands used for timber production, continuous-cover forestry (CCF) has been suggested as a management practice with both economic and environmental benefits (Nieminen et al., 2018). It relies on the selection harvesting instead of the clear-cutting performed in even-aged management (Nieminen et al., 2018). The continuously maintained tree stand with consistent evapotranspiration rates enables the WTL to be optimized for tree growth without ditch network maintenance (Leppä et al., 2020a, 2020b).

#### 6.1.2. Impact of peatland management on soil GHG fluxes

The common management practice on drained peatlands is rotation forestry (incl. ditch network maintenance, stand thinning, clear-cutting and regeneration after site preparation with mounding). Over the rotation period such a management regime is a net source of GHG emissions, at least on fertile peatland sites, since soil emissions are larger than the average C sink of trees (Shanin et al., 2021). Continuous cover forestry may result in lower emissions, since high emissions that follow clear-cutting (Korkiakoski et al., 2019) are avoided and frequent thinning reduces growing stock and evapotranspiration, which can elevate WTL.

WTL rise after harvesting could decrease the CO<sub>2</sub> emissions (Ojanen et al., 2013, 2010; Ojanen and Minkkinen, 2019) while a more moderate WTL could reduce the CH<sub>4</sub> emissions that would otherwise be caused by waterlogging after clear-cutting. The ideal WTL of peatland forests is lower than 30 cm depth to sustain the tree growth and simultaneously maintain a CH<sub>4</sub> sink function of soil, as CH<sub>4</sub> emissions increase only when WTL is higher than -30 cm (Ojanen et al., 2013, 2010).

On drained peatlands that are not used for timber production, restoration has been conducted by various techniques like damming or infilling the ditches (Lindsay, 2010), which could efficiently recover the WTL to restore the original ecological functions and biodiversity (Jauhainen et al., 2002; Worrall et al., 2007). Restoration by clear-cutting trees and blocking ditches on boreal forestry-drained peatlands

decreased the CO<sub>2</sub> emissions and enhanced soil C sequestration (Komulainen et al., 1999; Laine et al., 2019). However, the rewetted sites did not recover their CO<sub>2</sub> sink 8 or 10 years after restoration compared to pristine sites, with C balances ranging from a net CO<sub>2</sub> source to a smaller CO<sub>2</sub> sink (Purre et al., 2019). The N<sub>2</sub>O emissions of rewetted boreal peatlands were lower than forestry-drained and even undrained peatlands (Minkkinen et al., 2020). However, differences in N<sub>2</sub>O emissions were only observed on nutrient-rich sites but not on nutrient-poor sites, suggesting that N<sub>2</sub>O emissions are controlled by the interactions between soil C:N ratio and WTL (Minkkinen et al., 2020). Following the recovery of WTL, the oxic layer where the CH<sub>4</sub> oxidation occurs will decrease which would cause the re-establishment of CH<sub>4</sub> emissions. The increase of CH<sub>4</sub> emissions 2 years after rewetting of boreal forestry-drained peatlands has been observed (Komulainen et al., 1998). However, Urbanová et al. (2012) found the restoration of a drained bog forest in central Europe did not show a significant effect on CH<sub>4</sub> emissions during the first year of rewetting compared with the undrained bog after long-term drainage (50 years). Even though the methanogen abundance and community composition would have recovered to a near-pristine level, the recovery of anaerobic microbial processes may take time, and CH<sub>4</sub> production may remain low due to the substrate limitation caused by the highly decomposed organic matters and slow recovery of vegetation after the long-lasting drainage (Urbanová and Bárta, 2020).

In addition, the GHG emissions of forest-to-bog restored peatlands in the UK were also examined. Results showed that the C sink function was successfully recovered after long-term restoration by felling trees and blocking ditches (16–17 years post-restoration; Creevy et al., 2020; Hambley et al., 2019; Mazzola et al., 2021). Whereas in the short-term, the restored sites were still a C source (less than 10 years after restoration), because while some were CO<sub>2</sub> sources (Hambley et al., 2019) and others sinks, the restored sites were consistently a strong source of CH<sub>4</sub> (Creevy et al., 2020) or source of both GHGs (Rigney et al., 2018). Meanwhile, the C emissions of restored bogs largely depends on the vegetation composition (Creevy et al., 2020; Mazzola et al., 2021). A model of peatland photosynthesis at different stages after restoration suggested that the C assimilation capacity of a peatland can reach a near-pristine level 5–10 years after restoration (Lees et al., 2019).

Water-table rise in peatlands following rewetting can be rapid, but the recovery of hydrochemistry and vegetation does not always follow. In particular, microbial communities only recover if sufficient substrate is available (Emsens et al., 2020). Therefore, the recovery of the peatland ecosystem as a whole depends on the time after restoration as well as the degree of change caused by drainage, which tends to be higher if the site was more nutrient-rich and wetter before drainage (e.g. Laine et al., 1995). In addition, the effect of restoration varied largely depending on the geographical location (e.g., temperate vs. boreal peatlands) and peatland type (e.g., nutrient-poor bogs vs. nutrient-rich fens or swamps).

When considering only the GHGs emissions/uptake by soil, Günther et al. (2020) showed that rewetting of drained peatlands of various land use categories eventually has a net cooling effect based on the radiative forcing of GHGs. They found that CO<sub>2</sub> was the dominant forcing which determined the radiative forcing difference between drained and rewetted sites, and the potential CH<sub>4</sub> emissions would not undermine the climate change mitigation potential of peatland rewetting (Günther et al., 2020). But if the C stock from vegetation biomass was taken into account, the ability of peatland rewetting to mitigate climate change highly depends on the land use and climate conditions. Ojanen and Minkkinen (2020) showed that rewetting provides rapid climate benefits, more so for agricultural peatlands and in the tropics than for forestry-drained peatlands in the temperate and boreal zones. This is due to the decrease of wood C storage caused by rewetting temperate and boreal forests, which delays the onset of the cooling impact of restoration. Thus, it may be more beneficial to simply abandon tree stands in forestry-drained peatlands.



## 6.2. Modelling peatland hydrology effects on soil GHG fluxes

The existing models for pristine (undrained) peatlands simulate development of vegetation and element fluxes over decades under reconstructed hydrological conditions (e.g., Frolking et al., 2010, 2001; Qiu et al., 2018; Wu and Blodau, 2013). In addition, some models account for stand management and are applied on managed peatland forests (e.g., Kasimir et al., 2018; Shanin et al., 2021).

Empirical research on boreal drained peatlands has shown that the water table level (WTL) and the peat nutrient status drive CO<sub>2</sub>, N<sub>2</sub>O and CH<sub>4</sub> emissions by altering oxic conditions of the peat. Currently, there are models that are able to estimate changes in the WTL at daily time step, e.g. SpaFHy-peat model by Launiainen et al. (2019) and SUSI model by Laurén et al. (2021). The SpaFHy-peat model uses canopy cover, dominant height, leaf area indices (LAI), peat type, distance to ditch, ditch depth and meteorological data as drivers. The SpaFHy-peat model integrates a leaf-scale stomatal conductance model with a simple canopy radiation transfer scheme allowing the model to estimate vegetation-specific water use and its impact on soil water balance. The model produces estimates for daily WTL values that can be used to estimate mean WTL for May-October, which is a driver of the empirical GHG exchange models (Minkkinen et al., 2020; Ojanen et al., 2010; Ojanen and Minkkinen, 2020). Peat soil GHG exchange can be estimated by combining the hydrological model with the empirical GHG exchange models (Table 1).

Similarly to SpaFHy-peat, the SUSI model (Laurén et al., 2021) also simulates WTL based on aboveground hydrology (infiltration and evaporation) and couples that with process representation of vertical and horizontal water movement. The strength of the SUSI model is the ability to estimate nutrient availability based on the stoichiometric ratios of decomposing organic matter. The SUSI model also estimates CO<sub>2</sub> emissions from decomposing peat as driven by variation in WTL. These relationships are based on the empirical models by Ojanen et al. (2010), and CH<sub>4</sub> emissions relationships based on empirical models by Minkkinen et al. (2007). The modular structure enables constant development of new features (Laurén et al. 2021).

The Estimating Carbon in Organic Soils - Sequestration and Emissions (ECOSSE) model, was developed from concepts originally derived for mineral soils in the RothC (Coleman and Jenkinson, 1996) and SUNDIAL models. The ECOSSE description of decomposition differs from RothC and SUNDIAL in the response of aerobic decomposition to soil water content and the incorporation of a pH rate modifier. Below field capacity, the response of aerobic decomposition to soil water content follows the relationship given for SUNDIAL by Bradbury et al. (1993). Between field capacity and saturation, the response of aerobic decomposition to soil water content includes a linear reduction in the rate of decomposition. When the water content of the soil is above field capacity, an aerobic rate modifier is estimated from the soil water content. Anaerobic decomposition is not included in RothC or SUNDIAL but is described in ECOSSE using rate modifiers that are set according to relationships formulated for the anaerobic decomposition process. A proportion of the CH<sub>4</sub> produced is oxidised back to CO<sub>2</sub> depending on transportation of CH<sub>4</sub> in plants, the rate of diffusion through the soil and the thickness of the aerobic region crossed by the CH<sub>4</sub>. The main applications of the ECOSSE-model are to simulate the impacts of land-use and climate change on GHG emissions from mineral and peat soils (Smith et al., 2010).

Shanin et al., (2021) used a process based Romul\_Hum model (Komarov et al., 2017) for decomposition of litter in combination with empirical WTL models by Ojanen and Minkkinen, (2020) for peat decomposition. In a modeling study along the forest-peatland transect in Finland with gradually increasing soil C stocks and soil water content, Tupek et al. (2022, submitted manuscript) accurately modelled mineral and organic soil C stocks and soil CO<sub>2</sub> emissions by replacing the original Yasso07 precipitation-based environmental modifier with the optimized bell-shaped soil moisture function of Davidson et al. (2012), which

represents both the substrate and oxygen limitation on decomposition. The moisture modifiers used in the most common biogeochemical models either do not use a bell-shaped function or have a default maximum decomposition rate defined for moist soils (Sierra et al., 2012) which limits their usability for peatlands. The standard version of the ORCHIDEE model which originally used Van Genuchten hydraulic conductivity specific to non-peat (coarse-, medium-, and fine-textured) and peat soils (Ducoudré et al., 1993) was modified to a peatland version (ORCHIDEE\_PEAT) by improving peat soil hydraulics (Qiu et al., 2018). In ORCHIDEE\_PEAT, modified soil C modelling accounts for the impact of estimated WTL, limited soil drainage, and soil water saturation controlling decomposition in addition to the modification of the number of soil C pools and flows between them. In this model, the CENTURY model structure is used for the decay of non-peat plants, while the decay of peat vegetation is based on C pools dynamically controlled by WTL (acrotelm - peat layer above WTL and catotelm - peat layer below WTL) (Qiu et al., 2018). The WTL-dependent dynamic peat modeling enables modelling impact of variation in climate (Frolking et al., 2010, 2001) and landscape inundation (Kleinen et al. 2012).

Assessment of the complete GHG exchange of managed peatland forests should also consider CH<sub>4</sub> emissions from ditches (e.g. Minkkinen et al 2020) and assessment should cover all development stages including relatively high N<sub>2</sub>O and CO<sub>2</sub> emissions from clear-cut areas (Korkiakoski et al., 2020; Mäkiranta et al., 2010).

## 7. Effects of fire management

### 7.1. Fire management and soil C stock and GHG fluxes

#### 7.1.1. Rationale for fire management

Wildfires consume around 0.5M ha of forest every year in southern Europe (<https://www.eea.europa.eu/ims/forest-fires-in-europe>) and the importance of this perturbation is unquestionable. However, fire can also be used as an ecologically sound management practice at fire-prone locations to create more resistant and resilient forests to future disturbances. Fire has been traditionally used in Europe to manage forest residues or to prepare forest sites for restocking. More recently, prescribed burning formalised the planned use of fire to achieve precise and clearly defined management objectives. Reducing fire hazard was the initial motivation across many regions of southern Europe, although prescribed burning has spread to temperate-boreal and neighbouring Eurasian regions and includes a variety of objectives, from biodiversity conservation to pest management (Fernandes et al., 2013; Goldammer, 2013). Burning the forest understory can also be a management option to increase the resistance of forest stands to extreme drought events and wildfires (Vilà-Vilardell et al. 2022, in press). Prescribed burning and stand thinning are used for wildfire hazard management, by reducing the predicted extent of simulated fires through the strategic placement of fuel treatments (e.g., Finney et al., 2007; Piqué and Domènech, 2018). Wildfire management (i.e., wildfires that are allowed to burn under appropriate weather conditions) can also be considered a fire management option and is now a subject of debate in landscape adaptation to extreme wildfires. However, social, technological and political constraints currently preclude its application.

All fires affect soil biota directly when soil heats to killing temperatures, or indirectly through immediate changes in the soil properties or long-term changes in plant communities and the biochemical characteristics of organic matter inputs (Certini, 2005). Logically, prescribed fires cause smaller effects on soil microbial biomass (average reductions of about 25%) than wildfires (~40%) (Wang et al., 2012). In fact, in low intensity fires, the direct effect of heating can be little or null (Dooley and Treseder, 2012), but it can be significant in the case of burning slash or wood piles. After the initial decline in biological activity, pioneer microorganisms usually take advantage of fire-induced release of readily metabolisable organic compounds, allowing for the quick recovery of microbial activity. However, given that resistance and resilience to soil

**Table 2**Effect of forest management practices on the soil C stock, CO<sub>2</sub>, CH<sub>4</sub>, N<sub>2</sub>O emissions and soil biota, the key drivers of these effects, and some identified limitations.

Management practice		Soil C stock	CO <sub>2</sub> emissions	CH <sub>4</sub> emissions	N <sub>2</sub> O emissions	Soil biota	Key drivers	Limitations
Tree stand management	Tree species selection	Increase with coniferous or mixed-coniferous species	-	-	-	Change from deciduous to coniferous trees increases fungal biomass and mixed-stands have higher microbial diversity than monocultures	Tree species identity and diversity; site condition; climate; soil type (organic/mineral soil)	Soil C stock increases with conifers, but greater vulnerability of coniferous SOM to disturbance and warming climate.
	Stand thinning and harvesting	Decrease	Increase	Decreases CH <sub>4</sub> uptake	Increase	-	Soil type (organic/mineral soil); tree species and age; harvesting method (whole tree harvesting or stem-only harvesting); thinning intensity; recovery time; soil temperature and moisture	Soil CO <sub>2</sub> (and N <sub>2</sub> O on peat soils) fluxes increase after clearcutting, but this is quantified by direct measurements only on few sites.
Nutrient management	Nitrogen fertilization in boreal forests	Increase	Decrease	Decrease	Increase	Lower fungal-to-bacterial ratio. Lower microbial N and C use efficiency.	Decomposing stage; litter quality; site condition (vegetation and climate); N addition level	Microbial mechanisms for soil C stabilization and interaction between microbial residues and mineral surfaces and complex polymers are not quantified.
	Wood ash fertilization	Uncertain	Increase	Decrease	Decrease or no change	Change in fungal-to-bacterial ratio, bacterial community composition and fungal species composition	Quantity and quality of ash; soil type and condition (e.g. silt content, pH, water content); site type; fertilization rate; time after fertilization	Vegetation composition affects root production and respiration, which may change at the same time.
Site preparation	Chemical agents	Decrease	-	-	-	-	-	-
		No effect	No effect on mineral soils, but increased emissions on organic soils.	No effect	No effect	-	Stand regeneration success and enhanced biomass production compensate accelerated soil respiration.	Very high variation in the soil properties after site mechanical preparation and therefore challenging to measure differences between treatments
Forested peatland hydrology management (elevated soil water level)	Reduces rate of peat C loss	Decrease	Decrease	Increase	Increase	Elevated water table level affects CH <sub>4</sub> producing (increase) and consuming (decrease) microbes	Time after restoration; drainage degree before restoration; vegetation composition, nutrients level; geographical location and peatland type	Hydrological restoration may increase the CH <sub>4</sub> emissions and decrease the C stock of tree biomass in peatland forests.
Fire management		Decrease-No effect	Decrease	Decrease	-	Decrease microbial biomass	Fire severity on soil components and recovery time of litter inputs, and plant and soil diversity	Positive net C balance in comparison to wildfire emissions, but higher C loss than after clear-cutting. Prescription window restricting burning when litter is dry will reduce the impacts on soil.
Biodiversity management		Increase	Decrease	Restoration of peatlands may increase CH <sub>4</sub> emissions	Restoration of peatlands may increase N <sub>2</sub> O emissions	Increase in diversity	Soil type (mineral/organic soil), restoration/conservation intensity, retention of dead trees, and introduced shifts in tree species composition.	Retention trees may increase biological activity and soil respiration.

heating differs between microbial groups (Table 2), intense fires can result in changes in the microbial community diversity and structure (Goberna et al., 2012). For instance, it is known that soil heating increases the predominance of soil bacteria while reducing the fungal abundance (Fultz et al., 2016; Rutigliano et al., 2007). In a review by Taudière et al. (2017), negative effects of prescribed fires on ectomycorrhizal (ECM) fungal richness are reported, but the authors acknowledge that further studies are necessary to understand how this practice could impact ECM fungal diversity. Furthermore, Tomao et al. (2020) shows that there is not much consistency in the literature on the effect of prescribed burning on fungal species diversity. What seems clear is that shifts in fungal community composition occurs (Oliver et al., 2015; Reazin et al., 2016), and that the recovery of the pre-burning community can take more than a decade (Hart et al., 2018; Oliver et al., 2015). These community changes could have potential consequences at a functional level and on the C and nutrient cycling. Then, the recovery of soil microbial functions, such as the capacity to degrade compounds of different biochemical characteristics, may depend on the capacity of plant communities to regrow and provide different organic compounds through root turnover and exudation (Garcia-Pausas et al., 2022).

### 7.1.2. Impact of fire management on soil C stock and fluxes

Even in low intensity burns, the losses of ecosystem C and N can be significant (Certini, 2005; Homann et al., 2011). Although most of the studies agree unequivocally that untreated stands release more emissions to the atmosphere during a wildfire than treated stands (Restaino and Peterson, 2013), when considering the emissions derived from treatments, the relative effectiveness of fuel treatments in mitigating C losses is less clear, and highly variable (Restaino and Peterson, 2013: -33 Mg C ha<sup>-1</sup> to +3 Mg C ha<sup>-1</sup>). For instance, in *Picea abies* forests of Finland, CO<sub>2</sub> efflux during the two years after clear-cutting and burning of slash was comparatively lower than the clear-cut only stand (difference ~10.1 Mg C ha<sup>-1</sup>), but the total amount of CO<sub>2</sub> released was higher due to the immediate CO<sub>2</sub> emissions during burning (~28.0 Mg C ha<sup>-1</sup>, Kulmala et al., 2014). Furthermore, a sink capacity of CH<sub>4</sub> is reduced during at least the first year after burning (Virkkula et al., 2014). In the long-term, however, the decay of logging residuals at the clear-cut stand may reduce the differences between clear-cutting with and without burning of slash (Table 1).

As litter and soil organic layers contain an important proportion of surface C stocks, the consumption of ground fuels during a prescribed burning can represent the largest fraction of emissions caused by fire management. Consequently, in ecosystems with deep organic horizons (e.g. boreal forests, peatlands) prescribed burning can negatively impact C storage (Pellegrini et al., 2022). In a meta-analysis on temperate forests, Nave et al. (2011) indicated that prescribed burning caused, on average, 46% and 35% reductions of the forest floor C and N stocks, respectively.

Depending on the temperatures reached, fires can also potentially alter the physical, chemical, and biological properties of the uppermost mineral horizons, which may affect key processes of the C cycling (Alcañiz et al., 2018; Certini, 2005). However, the heat transfer to the soil surface and into the soil is an uncertainty in modelling fire impact on soil processes (Table 1). The amount, characteristics and arrangement of fuels determines the proportion of released energy by fire that impacts forest floor and mineral layers. Thus, under slash piles soil temperature can reach more than 100°C at 10 cm depth, and even more intense and longer-lasting heating can occur in soils under wood piles (200-300°C) (Busse et al., 2013), causing stronger C and nutrient losses, soil physical alterations, and mortality of faunal and microbial communities. The heat transfer into organic and mineral layers during the burning of shredded debris is higher than during the burning of standing understorey. In both managements, intense heating of mineral soil is generally restricted to the 2-5 cm soil depth, and diminishes steeply with depth (Cawson et al., 2016). Soil moisture limits temperature increase during a

fire (Busse et al., 2005); therefore, burning when organic and mineral soils are moist is outlined in the prescription guidance to prevent soil impacts (Table 1).

Overall, fire could eventually affect both C inputs to and outputs from the soil. Understorey burning alters the quantity or quality of the organic matter inputs through shifts in plant community composition, abundance and vigour, which in turn may affect the microbial activity and organic matter mineralisation. However, mid-term changes in soil C and N after low-to-medium intensity prescribed burnings seem to be more related to plant community dynamics than to the direct effects of fire (Martí-Roura et al., 2011). Although litterfall increases during the first months after a prescribed burning (Espinosa et al., 2018), recovery of the forest floor mass to pre-fire stocks could take several years. For instance, in *Pinus ponderosa* forests of Oregon (US), Busse and Gerrard (2020) recorded faster rates of litter mass accumulation in burned than in unburned sites, but the recovery of organic horizons to pre-burning levels took about 15-20 years. Increase of litter N content, at least in places with a high frequency of burning, may contribute to litter decomposition (Ficken and Wright, 2017) and slow down the recovery of C stocks in organic horizons.

Despite the growing consensus among researchers and managers concerning the benefits of fire as a management tool for ecosystem issues such as wildfire hazard reduction, forest productivity, and biodiversity conservation of endangered species, the benefits of this practice for key soil components and functions are less obvious. Compared to the reduced and transient effects of low-intensity prescribed burns, burning shredded debris or slash piles have greater potential to negatively impact soil C stocks. In these cases, burning when organic and mineral soils are moist is mandatory to reduce soil C emissions. Minimizing soil C emissions during burning together with soil organic matter stabilization processes triggered by fire supports fire management as a smart forest tool to mitigate the effects of stand-replacing wildfires and enhancing forest biodiversity. We have extensive knowledge about the effects of fire management on soil physical and biogeochemical variables such as C and nutrient dynamics (e.g., Alcañiz et al., 2018; Martí-Roura et al., 2014, 2011 and references there-in), but less understanding about the cumulative effects of repeated burnings on soil functions. Thus, more knowledge is needed on a mechanistic understanding of plant and soil biota-mediated responses to different fire intensities and recurrences to avoid negative impacts on soils (Table 1).

### 7.1.3. Impact of fire management on soil C stability

Short-term organic C stock reduction after fire can be compensated in the long term by multiple transformations that increase the stability of organic matter and potentially buffer C emissions due to heterotrophic respiration (Pellegrini et al., 2021). Fire can alter the stability of soil organic matter by changing its physical protection and its biochemical quality. Partially charred organic material may have different biochemical quality than uncharred organic matter, altering their decomposition dynamics (Martí-Roura et al., 2014; Pellegrini et al., 2021). On the other hand, the rapid rise of soil temperature even under low intensity fires cause structural degradation of soil aggregates due to the steam pressure when there is a rapid vaporisation of soil pore water (Albalasmeh et al., 2013; Jian et al., 2018). This leads to a loss of C by microbial mineralisation of previously physically protected organic matter (Jian et al., 2018), and may also compromise the capacity of soils to stabilise new organic matter inputs. Structural degradation can also occur in the case of high intensity fires, if the organic matter that binds aggregates is burnt (Alcañiz et al., 2018).

On average, about 5% of the burned biomass is converted to pyrogenic C (PyC) (DeLuca et al., 2020), which is considered a stable form of C resistant to microbial degradation (Singh et al., 2012 but see Ghosal et al. 2016). In a data synthesis, DeLuca et al. (2020) found no significant differences in the amount of PyC produced by wildfires and prescribed fires (2.2±1.7 Mg ha<sup>-1</sup>), with less variability observed in prescribed fires due to their usual low fire intensity. PyC not only represents a form of

long-term C storage, but also alters microbial processes and nutrient cycling (Pingree and DeLuca, 2017) and can adsorb organic compounds (i.e., exudates, microbial byproducts or other products derived from litter decomposition) (Gundale and DeLuca, 2006). The presence of PyC reduces the amount of soluble organic matter and increases its aromaticity (Hobley et al., 2019). However, the increase in SOM recalcitrance in the long term after a fire is not always related to an increased proportion of aromatic C, but could also be related to an increased abundance of condensed forms (Rovira et al., 2012). This increased biochemical recalcitrance may reduce the microbiological activity and nutrient cycling in soils. For instance, Speratti et al. (2018) showed that the addition of PyC (biochar) to forest soils reduced the mineralisation of native organic matter (negative priming). How relevant the interactions between fire intensity, soil characteristics and microorganisms' dynamics are for soil organic matter stabilization remains unclear (Table 1).

## 7.2. Modelling of post-fire C dynamics

Although several models have been applied to predict post-wildfire erosion (Wittenberg and Pereira, 2021), to our knowledge, no specific models have been developed to simulate post-fire soil physicochemical and microbial dynamics (Table 1). In the short-term, soil modelling should consider the effects of heating on soil communities and their recovery capacity. Forest fires cause a notable reduction in the microbial biomass in the top 10 cm soil (Wang et al., 2012). The magnitude of the effect mostly depends on the maximum temperature reached and the duration that temperatures exceed the lethal threshold, which depends on the fuel distribution and on soil water content. Soil water is volatilized under high temperature, and the vapor kills soil biota. At a similar temperature, mortality is greater in a humid soil than in a dry soil (Barreiro et al., 2020). Models should also account for the sudden increase of available nutrients, which may reduce microbial activity (Treseder, 2008), and the release of labile organic matter (Choromanska and DeLuca, 2002), which may prime the mineralisation of labile and recalcitrant organic pools in the uppermost mineral soil layers. Under intense fire, models should consider the pace at which soil functioning recovers, which in part is likely to depend on the recovery of litter inputs and vegetation regrowth (Garcia-Pausas et al., 2022). To our knowledge no specific simulation routines exist to account for the short-term impacts of fire on soil dynamics. In the mid- and long-term, as long as litterfall and vegetation recover, the dynamics of soil C can be modelled with some of the currently applied soil models, taking into account the effects of the reduction of soil organic layers caused by fire on the dynamics of uppermost soil layers.

## 8. Biodiversity management

### 8.1. Effects of biodiversity management on soil carbon stocks and GHG fluxes

#### 8.1.1. Rationale for biodiversity management

Protection of key habitats or establishment of forest conservation areas that have high biodiversity value are major methods applied to maintain forest biodiversity. At forest stand scale, characteristics that enhance biodiversity include presence of old trees and broadleaf tree species, and availability of different categories of dead wood (Felton et al., 2016). Retention forestry, which creates these characteristics, suggests that forest stands are only partially cut, leaving single trees or groups of trees that age and die on site. Thus, the retention increases the forest dead wood C pool, but decreases the production of merchantable timber (Nuney and Keeton, 2010; Roberge et al., 2015; Santaniello et al., 2017). On the other hand, retention trees have positive impact on diversity of mycorrhiza, which determine the capacity of seedlings to take up water and nutrients (Korkkama et al., 2006; Sterkenburg et al., 2019).

#### 8.1.2. Impact of biodiversity management on soil C stock

Protection of key habitats or establishment of forest conservation areas that have high biodiversity value can have a positive impact on forest C stocks at the stand level. Old-growth forest conservation results in greater C stock in soils than does extensive and intensive forest management (Ameray et al., 2021), but measured differences in the soil C stocks between managed and unmanaged forests can be small or negligible partly due to differences in the site history and lower fertility level of the set-aside forest (Tamminen and Ilvesniemi, 2012).

The overall impact of tree retention on soil C stock has not been quantified. In general, retention increases input of coarse woody debris to the forest floor. Furthermore, it is known that fungi inhabiting wood in advanced decay stages transfer N from soil to dead wood (Mäkipää et al., 2017), which decreases the amount of N in the organic layer and can likely decrease the organic layer decomposition rate. On the other hand, retention of broadleaf species may increase decomposition, since as compared to conifers, broadleaf species generally have higher soil biological activity and decomposition rates, although these tree species effects are also mediated by site properties (Augusto et al., 2015).

Restoration of degraded ecosystems may provide co-benefits for climate change mitigation and biodiversity, e.g., C stock of dead wood and habitat availability for saproxylic species. However, restoration of boreal peatlands with positive biodiversity impacts may be an inefficient method to mitigate climate change (Juutinen et al. 2020). Peatland restoration is discussed in Chapter 6 and fire management in Chapter 7.

### 8.2. Modelling effects of biodiversity management on soil C and GHG fluxes

Forest conservation can be accounted for in forest ecosystem and soil models, which derive soil C stock changes from stand growth and soil edaphic and climatic conditions (e.g., Mäkipää et al. 2011). The changes in the dead wood and soil C stocks can be modelled with some of the currently applied soil models (e.g., YASSO07, Tuomi et al. 2011; CENTURY, Parton et al. 1987; ROMUL, Chertov et al. 2001), but these models do not fully account for changes in soil biological activity (Table 1). Modelling of the peatland hydrology restoration is discussed in chapter 6.2. and modelling of post-fire soil carbon dynamics in chapter 7.3.

## 9. Synthesis

The soil is the largest C pool in the forest (FAO, 2020) and potentially a large sink or source of GHGs (e.g., Ameray et al., 2021) that is affected by forest management decisions. In this review, we identified some highly likely impacts of commonly applied forest management practices on soil C stock or GHG emissions, and also potential impacts with high uncertainty and knowledge gaps (Table 2). We identified several management practices that can contribute to climate change mitigation by increasing soil C stock (e.g. transition from broadleaved to coniferous trees or mixed-stands, and N fertilization of boreal forest on mineral soil) or by reducing soil C loss or GHG emissions (e.g. sustainable fire management and elevated water level on peatland forests) (Table 2). On the other hand, soil carbon losses may be accelerated by intensive thinning regimes and harvesting practices (e.g. intensive stand thinning on mineral soils sites, whole tree harvesting and short rotation length), which are followed by mechanical site preparation that may accelerate soil C losses. In general, the direction of soil C and GHG emissions change due to various management practices can be derived from current knowledge, but further experimental studies, long-term monitoring and scenario modelling are needed to quantify and upscale the rate and magnitude of change (Table 2).

Forests are complex and dynamic ecosystems, where the responses of soil GHG fluxes to management often diverge from the responses of tree stand. This emphasizes the importance of soil measurements and development of soil models applicable to target ecosystems where



climate change mitigation potential is evaluated or mitigation measures are planned. Incomplete or biased information may lead to inefficient climate policy and non-optimal use of resources. For instance, drainage of forested peatlands can enhance the C sequestration of growing stock in a Norway spruce dominated stands, but the continuous GHG emissions of nutrient-rich peat soils are larger than the temporary C sink of the tree stand in the long term (over a rotation period or longer) (Ahtikoski et al., 2022; Shanin et al., 2021). Furthermore, reported high CO<sub>2</sub> and N<sub>2</sub>O emissions after clear-cutting of the drained fertile peatland sites indicate that more sustainable management regimes are needed, such as continuous cover forestry as an alternative to rotation forestry (Korkiakoski et al. 2019; Nieminen et al. 2018). Since the direction of change in the soil carbon stock can be opposite to that of trees, at least in short term, assessment of the forest C sink that considers only trees is incomplete and likely to be invalid and we strongly recommend moving towards comprehensive GHG inventories, forest monitoring methods and scenario analyses that quantify the total GHG balance of the forest including the forest soils. Thus, further improvements in forest inventories/forest soil surveys, soil models and data analysis are needed. Reliable soil models, GHG scenario tools and monitoring methods will contribute to the design and evaluation of climate policy targets in the EU and globally (National Inventory Report (NIR) European Union 2021; Forsell et al., 2018).

Effects of forest management practices are not yet widely integrated into scenario analyses that support climate and forest policy, since forest management affects soil processes in multiple ways, effects in different conditions are not well quantified, and many have not yet been included in the modelling frameworks (Fig. 1, Table 1). In general, the management effects that change biomass production and litter quantity and quality are represented by soil models. Changes in environmental conditions such as soil temperature and moisture are sometimes represented in soil models, and depend on relevant and representative measurements that describe the effects of various management interventions (Table 1). The climate change mitigation potential of forest soils, as estimated by modelling approaches, depends on stand biomass driven effects (such as quantity and quality of litter input) and climate factors that affect the decomposition rate. Thus, management practices that change tree species composition, volume of the growing stock or natural mortality can be accounted for when litter input to the soil is estimated. However, forest C modelling based on the biomass change without consideration of soil characteristics cannot account for loss of nutrients in removed biomass and feedbacks that reduced nutrient availability have on the future growth of trees (e.g. Palosuo et al., 2008). Dynamic biomass production without considering the feedback of changing soil properties may be biased, as variation in forest growth and productivity is determined by soil conditions, such as soil fertility and water content; therefore any changes in these conditions may affect the future C sink of vegetation (Mäkipää et al., 2015; Shanin et al., 2016).

The dynamic changes of soil preparation or management, which can directly modify soil temperature, moisture, nutrient balance, microbial processes, erosion, hydrology and oxygen concentration, are not explicitly represented in the models (Table 1). For example, models like Yasso07 (Tuomi et al., 2011) and CENTURY (Parton et al., 1987) are used for the analyses of forest C sinks and applied in national GHG inventories (IPCC 2019) but exclude effects of forest management, although some more comprehensive ecosystem models include them. For example, SUSI (Laurén et al., 2021) and ORCHIDEE (Chang et al., 2013; Levis et al., 2012; Wu et al., 2016) can simulate impacts of management on hydrology, soil temperature, and soil C cycling for peatlands and croplands, respectively. Terrestrial biosphere models including landscape scale models have historically been developed to understand and make predictions for undisturbed ecosystems, with land management representation added later. As a result, estimation of land

management effects on soil C, nutrient and water cycles are often based on few studies (Levis et al., 2012; Zhou et al., 2021) and primarily considers the ways in which various land management practices modify existing processes rather than adding new processes. Parsimonious soil C models with less input data requirements e.g., Yasso and CENTURY as formulated in Sierra et al., (2012) are more spatially applicable e.g., on regional and global scale than more complex models which contain more site-specific parameters and are computationally demanding. However, complex models account for more processes affecting the decomposition and soil C stabilization, thus including interactions between soil properties, microbiota, nutrients, and management. In general, soil C stabilization processes, e.g. organic-mineral interactions, microbial decomposition, or the chemical complexity or organic compounds, are not included in the current models and require further measurements and new generation models. The most challenging task in modelling is to integrate management effects that change soil microbial communities and their functioning. Here further understanding on the variation of microbial community structure and functional roles of fungi and bacteria are needed. We emphasize that both measurements from empirical studies and modelling supported by development of theory need to work together, and incorporating more processes in modelling may enhance our ability to reliably predict the future.

We need to develop and efficiently implement more ambitious mitigation and adaptation to tackle climate change and the unforeseen risks that changing conditions introduce to ecosystems, provision of ecosystem services and human wellbeing. Since soil is a large C stock, it is essential to understand effects of planned management practices on soil processes and to avoid risks of turning soil from a C and GHG sink to a source. Meanwhile, supply of sustainably produced timber and other ecosystem services need to be maintained. This review explores the potential of soil management practices to mitigate climate change, as well as the current capacity of modelling and scenario analysis tools to support planning of forest operations and climate policy. We stress that current models and analysis frameworks do not fully account for the relevant soil processes described in this review and encourage the parsimonious development of multiple models and analysis frameworks to support the estimation and optimization of soil GHG balance for climate policy.

#### Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

#### Data availability

This is review article and published peer reviewed articles are used as a data in this study

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## Appendix A. Literature search on effects of different management practices on forest soil C and GHG fluxes.

Management practice	Searching strings (used for title, abstract or keywords)	Databases	Number of studies screened	Number of studies included
Stand thinning and harvesting practices	soil AND (carbon OR respiration) AND forest AND (thinning OR clear cut* OR logging OR harvest*)	Web of Science, Google Scholar	341	95
Tree species selection	(soil carbon OR soil organic carbon) AND (tree species OR tree species identity OR tree species diversity)	Google Scholar, JSTOR	57	41
Fire management	((Prescribed & (fire* or burn*)) or (slash & burn*)) & Soil* & Forest*.	Web of Science	165 (of which 30 were reviews and 6 were meta-analyses)	44
Nutrient management, incl. N fertilization, wood ash fertilization and liming	“nitrogen fertili*ation” OR “nitrogen addition” OR “wood ash fertili*ation” OR “wood ash addition” OR “liming” OR “limestone” (Title) and “forest* ” (Title) and “soil*” (Title) and “carbon stock*” OR “carbon storage*” OR “organic matter decomposition” OR “organic matter minerali*ation” OR “enzyme*” OR “litter” OR “lignin” OR “soil respiration” OR “greenhouse gas” OR “greenhouse gas *flux*” OR “CO2” OR “carbon dioxide” OR “CH4” OR “methane” OR “N2O” OR “nitrous oxide” (Topic) and English (Language) and 2012.01.01-2022.12.31 (publication date)	Web of Science, Google Scholar, CAB,	N fertilization, wood ash and liming: 47	N fertilization, wood ash and liming: 47
Chemical agents	soil carbon AND forest AND (herbicide* OR pesticide* OR chemical agent*)	Google Scholar, JSTOR	22	12
Peatland hydrology and restoration	“peatland” or “drained peatland” or “forestry-drained peatland” or “restored peatland” or “drained peatland forest” with “hydrology management” or “hydrology restoration” or “rewetting” or “water table management” or “forest-to-bog”	Google Scholar	96	33

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