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Review

# A review of forest management practices potentially suitable for carbon farming in European forests

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## ABSTRACT

To meet the European Union's climate neutrality targets by 2050, carbon farming (CF) has emerged as a key strategy to enhance carbon (C) sequestration in managed ecosystems. This review assesses a broad set of forest management practices with potential to sequester carbon in aboveground biomass (AGB) and soil organic carbon (SOC) in European forests, while considering co-benefits and trade-offs. The analysis, based on a literature review covering boreal, temperate, and Mediterranean regions, evaluates practices such as afforestation, species selection, changes in rotation periods, reduced harvest intensity, continuous cover forestry, and peatland management. Results show that afforestation on croplands offers the highest short-term carbon sequestration potential, while agroforestry and peatland rewetting provide significant long-term benefits, particularly for SOC. Reduced or no harvest also offers short term sequestration potential, but the risk of leakage is potentially very high. However, the success of CF practices is highly context-dependent, influenced by forest type, disturbance risk, and future climatic conditions. This review highlights the urgent need for future studies considering both above and belowground carbon sequestration as well as co benefits. Furthermore, the importance of integrating sustainability, permanence, leakage prevention and additionality into CF initiatives and underscores the need for long-term, site-specific studies to inform policy and carbon certification frameworks.

## 1. Introduction

The European Union (EU) has committed to becoming climate neutral by 2050, requiring both reductions in greenhouse gas (GHG) emissions and enhanced carbon (C) sequestration. An intermediate target is foreseen in 2030 with a reduction of 55 % in emissions compared to the 1990 levels (EU, 2021). To meet these targets, the EU adopted a *Communication on Sustainable Carbon Cycles* (European Commission, 2021) as part of the EU *Green Deal* (European Commission, 2019), proposing a series of short to medium-term actions, one of which, is the carbon farming (CF) initiative. CF involves climate-friendly practices used by farmers and foresters that improve C sequestration and storage in agricultural and forest lands, while reducing GHG emissions. Moreover, the CF framework represent an opportunity for forest owners/managers to generate C credits representing an additional income. According to the EU (European Commission, 2021; modified in European Commission, 2024), "*Carbon farming can be defined as any practice or process, carried out over an activity period of at least five years, related to terrestrial or coastal management and resulting in capture and temporary storage of atmospheric and biogenic C into biogenic C pools or the reduction of soil emissions*".

Forest ecosystems are the earth's main C reservoir, providing

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important functions such as C sequestration and storage and, consequently, contributing to climate change mitigation (Nabuurs et al., 2017). Within forest ecosystems, soils store more than half of the forest carbon pool (Scharlemann et al., 2014). The long-term capacity of forest ecosystems to sequester C from the atmosphere depends on their productivity, age, health and resilience (Janowiak et al., 2017), as well as on forest management activities and the occurrence of natural disturbances. In addition to capturing C, forest ecosystems support biodiversity and offer crucial ecosystem services essential to societal and human well-being. These services include the production of timber and non-wood forest products, soil formation, erosion protection, water purification and storage, local climate regulation, and recreational opportunities (Thompson et al., 2014). The CF concept recognises and aims to preserve these natural values, ensuring that biodiversity and ecosystem services are not compromised.

Despite extensive research over recent decades, the impact of forest management on long-term C sequestration remains unclear (Keenan et al., 2013; Kutsch and Kolari, 2015; Hyyrynen et al., 2023). Models predicting the C balance of global forests by 2100 show conflicting results, with some forecasting a positive outcome and others a negative one (Austin et al., 2020; Beillouin et al., 2022). At EU level, recent trends have shown a decline in the C sink capacity of forests, a situation expected to worsen due to climate change impacts (Seidl et al., 2017; Senf and Seidl, 2021; Roebroek et al., 2023). Some countries (e.g., Germany, Austria, Finland) have started to report, based on national inventories, that their forest sink capacity is declining (EEA, 2025). The great uncertainty on the observed trends is due to the increasing impact of forest disturbances like pest outbreaks, fires, and wind events. The impact of these disturbances on EU forests greatly increased in recent decades (Patacca et al., 2023), adversely affecting forest health and consequently, their ability to sequester C. Another important aspect related to EU forests, is the fact that they are aging, with tree mortality accelerating by 1.5 % annually (Senf et al., 2021), a double rate than the one observed in the late 20th century, equivalent to 1 % of the EU-27 forest area dying yearly.

Thus, how effectively CF practices in the forestry sector can generate C removals is still debated (Linser et al., 2018; Roebroek et al., 2023). For this reason, this review analyses different forest management practices that can potentially be implemented as CF practices in European forests providing the most updated C mitigation potential connected to different practices for aboveground biomass (AGB) and soil organic carbon (SOC) pools.

### 1.1. *Literature review*

A systematic review of the scientific literature was conducted with the aim of identifying the mitigation potential offered by different forest management practices in Europe (EU-27 plus Norway) in terms of C sequestration in soils and aboveground biomass. Studies were divided in three broad geographic regions to represent Boreal, Temperate and Mediterranean forests. For this purpose, the "Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) Protocol for systematic reviews and meta-analysis was applied to collect available data from publications (Moher et al., 2009). The systematic literature review, was limited to publications from the last 10 years (2013–2023), and was completed in February 2024. A combination of keywords was used both within Scopus database of peer-reviewed literature and on the Google Scholar web research engine. The terms (a) forest management, (b) carbon sequestration, (c) soil organic carbon, (d) carbon farming, (e) climate mitigation, (f) European forests, (g) biodiversity, (h) greenhouse gas emissions were combined with each of these terms identifying forestry practices: 'afforestation', 'tree species selection', 'longer rotation period', 'conversion of coppice to high forests', 'continuous cover forestry', 'site fertilisation', 'fire management', 'peatland fertilisation', 'peatland restoration'.

Studies, with respect to baseline SOC or to the control plot, had to report the ΔSOC rate (Mg C $ha^{-1}$ $yr^{-1}$) or, alternatively, the experimental difference in either SOC stocks (Mg C $ha^{-1}$) or the concentration of SOC (g $kg^{-1}$) along with the bulk density. The literature cited by review articles, meta-analyses and experimental studies was in turn reviewed whenever it met the inclusion criteria. In the case of longitudinal studies performed by the same research group at the same experimental site, the publication with the longest study duration was selected. Further, studies analyzing SOC in shallow topsoil were excluded in line with the minimum depth of 0–30 cm required by the Intergovernmental Panel on Climate Change (IPCC) guidelines (IPCC et al., 2006) to report changes in SOC. Twelve relevant studies published before 2013 were added to increase the number of studies. Consequently, the identified management practices were classified according to their expected C sequestration potential and alignment with EU climate goals. The key-word research yielded 19 publications in the boreal zone, 18 in the temperate and 29 in the Mediterranean zone. These publications included a total of 349 case studies reporting changes in aboveground and/or soil organic carbon. Only 14 studies included both estimates for ABG and SOC. See Table 1 for all the studies included. The results were discussed in view of the additional environmental co-benefits provided by the practices, considering the Carbon Removal and Carbon Farming regulation (European Union, 2024), which introduces clear and uniform criteria for certification of C removals based on four key principles: a) *Quantification*: removals must be measured by reliable scientific methods; b) *Additionality*: certified practices must go beyond what is already required by legislation or current practices; c) *Permanence*: removed carbon must remain stored over time, preventing it from being released too soon; d) *Sustainability*: removals must not harm other environmental objectives, such as biodiversity or soil quality.

### 1.2. *Forest management practices*

#### 1.2.1. *Afforestation*

Afforestation, namely planting trees on former croplands and grasslands, holds promise as a climate change mitigation strategy with long-term benefits, and, in principle, it is clearly a CF activity. In addition to sequestering C in both, soil and above-ground biomass, afforestation can, sometimes, provide many other environmental co-benefits. Depending on previous land use, local climate, stand age and tree species, estimates for AGB variations due to afforestation ranged between 5 and 25 Mg $CO_2$ $ha^{-1}$ $yr^{-1}$ (Vesterdal et al., 2002; Thuille and Schulze, 2006; Hiltbrunner et al., 2013; Cukor et al., 2022; Vacek et al., 2022; Zeidler et al., 2022). According to our own review, AGB carbon sequestration rates for afforestation in Europe can be even higher in some cases, ranging from 2 to 35 Mg $CO_2$ $ha^{-1}$ $yr^{-1}$ (Table 1). Nevertheless, these high rates are achieved until a peak is often reached after 10–20 years and after remain constant or decline (Chen et al., 2020; Yang et al., 2025). Exceptionally high rates are obtained using fast growing non-native species that do not to comply with other CF requirements such as positive impact on the water cycle or an increase in biodiversity (see Fig. 1). This is particularly evident for the Mediterranean area (e.g. Spain, Table 1) where the exceptional sequestration rates are due to the use of fast growing species (e.g. *Eucaliptus spp*., *Pinus radiata*) when the rotation period is elonged. These plantations, whose wood is mainly used by the pulp and paper industry and for plywood purposes, are however subject to high disturbance risks, such as the growing incidence of fire affecting the Mediterranean area posing problems for the long term retention of the C removed from the atmosphere. When these species are excluded, afforestation generally enhances AGB carbon sequestration across various biomes, with an average rate of 5–10 Mg $CO_2$ $ha^{-1}yr^{-1}$, the lowest rates occurring in the boreal zone (Table 1). SOC sequestration rates are significantly lower due to the slow accumulation process, which peaks at 10–30 years after planting, and then often declines (Liu et al., 2021; Yang et al., 2025). In boreal regions, afforestation on bare fallow with conifers may initially

**Table 1**
Summary of the studies included in the article for the different biomes in Europe: boreal, temperate and mediterranean. Positive numbers indicate $CO_2$ removal, while negative numbers $CO_2$ emissions.

| Management | Country | Species | Age[a] | Native | Type[b] | AGB[c] rate | SOC[d] rate | Reference |
|---|---|---|---|---|---|---|---|---|
| **BOREAL** | | | Years | | | Mg $CO_2$ $ha^{-1}yr^{-1}$ | Mg $CO_2$ $ha^{-1}yr^{-1}$ | |
| ***Afforestation cropland*** | Finland | *Picea abies* | 10 | | C | 2.53 | −0.46 | Ťupek et al., 2021 |
| | Finland | *Picea abies* | 50 | | C | 5.93 | 0.73 | Ťupek et al., 2021 |
| | Finland | *Betula* spp. | 10 | | B | 0.40 | −0.37 | Ťupek et al., 2021 |
| | Finland | *Betula* spp | 50 | | B | 4.68 | 1.32 | Ťupek et al., 2021 |
| | Sweden | *Betula spp.* | 9 | | B | 3.04 | −2.53 | Rytter and Rytter (2020) |
| | Sweden | *Picea abies* | 9 | | C | 2.82 | −3.51 | Rytter and Rytter (2020) |
| | Sweden | *Populus spp.* | 9 | | B | 4.21 | −2.12 | Rytter and Rytter (2020) |
| | Denmark | *Picea abies* | 29 | | C | 15.32 | – | Vesterdal et al. (2002) |
| | Denmark | *Oak spp* | 29 | | B | 9.89 | – | Vesterdal et al. (2002) |
| ***Afforestation grassland*** | Finland | *Picea abies* | 10 | | C | 2.38 | 0.40 | Ťupek et al., 2021 |
| | Finland | *Picea abies* | 50 | | C | 4.61 | 0.48 | Ťupek et al., 2021 |
| | Finland | *Betula spp.* | 10 | | B | 0.48 | 0.44 | Ťupek et al., 2021 |
| | Finland | *Betula spp.* | 50 | | B | 5.49 | 0.81 | Ťupek et al., 2021 |
| | Norway | *Picea abies* | 50 | | C | – | 0.07 | Strand et al. (2021) |
| ***Peatland management*** | Global | | | | C/B | – | 3.40 | Mander et al. (2024) |
| | Finland | *Picea abies* | | | C | – | 0 | Peltoniemi et al. (2023) |
| | Finland | *Picea abies* | | | C | – | – | Rissanen et al. (2023) |
| | Finland | *Picea abies* | | | C | 0.15 | 0.11 | Lehtonen et al. (2023) |
| | Finland | *Picea abies* | | | C | 10.65 | – | Aro et al. (2020) |
| | Scandinavia | *Pinus sylvestris* | | | C | – | 0.37 | Wilson et al. (2016) |
| | Scandinavia | *Pinus sylvestris* | | | C | – | 0.26 | Wilson et al. (2016) |
| | Scandinavia | *Pinus sylvestris* | | | C | – | 5.38 | Wilson et al. (2016) |
| ***Ash fertilisation*** | Finland | *Picea abies/Pinus sylvestris* | | | C | 4.76 | −0.48 | Ojanen et al. (2019) |
| | Sweden | *Picea abies* | | | C | 11.45 | – | Van Sundert et al. (2021) |
| | Finland | *Pinus sylvestris* | | | C | 3.44 | – | Moilanen et al. (2015) |
| | Finland | *Pinus sylvestris* | | | C | 5.49 | – | Moilanen et al. (2013) |
| | Norway | *Picea abies* | | | C | 3.70 | – | Hanssen et al. (2020) |
| | Finland | *Pinus sylvestris* | | | C | 3.88 | – | Hytonen 2016 |
| ***Longer rotation period*** | Sweden | *Pinus sylvestris/Picea abies* | | | C | 4.21 | – | Peichl et al. (2023) |
| | Finland | *Pinus sylvestris* | | | C | 0.66 | – | Akujärvi et al., 2019 |
| | Norway | *Picea abies* | | | C | 4.58 | – | Stokland, 2021 |
| | Finland | *Pinus sylvestris/Picea abies* | | | C | 2.34 | – | Triviño et al., 2017 |
| **TEMPERATE** | | | | | | | | |
| ***Afforestation cropland*** | Latvia | *Picea abies/Pinus sylvestris* | 15 | | C | 6.41 | −0.84 | Petaja et al. (2023) |
| | Latvia | *Betula spp.* | 15 | | B | 6.22 | −1.83 | Petaja et al. (2023) |
| | Czech Republic | *Fagus/Quercus/Acer/Tilia* | 14 | | B | 16.03 | 6.22 | Cukor et al. (2022) |
| | Czech Republic | *Picea abies* | 14 | | C | 18.67 | 5.49 | Cukor et al. (2022) |
| | Czech Republic | *Populus/Alnus/Acer* | 52 | | B | 9.15 | – | Vacek et al. (2022) |
| | Czech Republic | *Spruce/Larix* | 52 | | C | 8.60 | – | Vacek et al. (2022) |
| | Czech Republic | *Spruce/Larix* | | | C | 12.27 | – | Zeidler et al. (2022) |
| | Poland | *Pinus sylvestris* | 10–50 | | C | – | 1.24 | Smal et al. (2019) |
| ***Afforestation grassland*** | Italy/Germany | *Picea abies* | 93–112 | | C | 10.06 | 1.06 | Thuille and Schulze (2006) |
| | Germany | *Pinus sylvestris* | 120 | | C | – | 0.91 | Heinsdorf (1994) |
| | Ireland | *Fraxinus/Alder/* | 4 | | B | 2.30 | 0.91 | Peichl et al. (2010) |
| | Ireland | *Fraxinus* ssp. | 12 | | B | 9.95 | −1.02 | Wellock et al. (2014) |
| | Ireland | *Fraxinus* ssp | 20 | | B | 8.60 | −3.04 | Wellock et al. (2014) |
| | Ireland | *Fraxinus* ssp | 27 | | B | 5.85 | −3.18 | Wellock et al. (2014) |
| | Ireland | *Fraxinus* ssp | 40 | | B | 7.39 | −1.97 | Wellock et al. (2014) |
| | Switzerland | *Norway spruce* | 40 | | C | – | 0.37 | Speckert et al. (2023) |
| | Switzerland | *Picea ssp.* | 25–120 | | C | 12.63 | 0 | Hiltbrunner et al. (2013) |
| ***Fertilisation*** | Germany | *Fagus sylvatica* | 0–145 | | B | – | −0.77 | Bauhus et al. (2004) |
| | Germany | Several | | | Various | – | 0.66 | Grüneberg et al. (2019) |
| | Centra/North Europe | Several | | | Various | 2.01 | 0.51 | De Vries et al. (2006) |
| ***Peatland restoration*** | Global | | | | Forested | – | −0.87 | Wilson et al. (2016) |
| | Germany | | | | Forested | – | −5.71 | Tiemeyer et al. (2020) |
| | Germany | *Pinus mugo/Picea abies* | | | C | – | −1.68 | Hommeltenberg et al. (2014) |
| | Germany | *Alnus glutinosa* | | | B | 47.43 | – | Schweiger et al., 2021 |
| **MEDITERRANEAN** | | | | | | | | |
| ***Afforestation cropland*** | Spain | *Pinus sylvestris* | 52 | N | C | 13.58 | – | Ruiz Peinado et al. (2016) |
| | Spain | *Pinus pinaster* | 59 | N | C | 8.23 | – | Ruiz Peinado et al. (2013) |
| | Italy | *Pseudotsuga menziesii* | 11 | P | C | 10.58 | – | Coletta et al. (2016) |
| | Spain | *Ceratonia siliqua* | 26 | N | B | – | 2.60 | Palacios Rodríguez et al. (2022) |
| | Spain | *Quercus suber/ilex* | 11 | N | B | – | 0.66 | Renna et al. (2024) |
| | Spain | *Pinus halepensis* | 18 | N | C | – | 2.01 | Segura et al. (2016) |
| | Spain | *Populus* ssp. | 10 | P | B | – | 6.73 | García-Campos et al., 2022 |

(*continued on next page*)

**Table 1** (*continued*)

| Management | Country | Species | Age[a] | Native | Type[b] | AGB[c] rate | SOC[d] rate | Reference |
|---|---|---|---|---|---|---|---|---|
| **BOREAL** | | | Years | | | Mg $CO_2$ $ha^{-1}yr^{-1}$ | Mg $CO_2$ $ha^{-1}yr^{-1}$ | |
| | Spain | *Populus* ssp. | 5–10 | P | B | – | 7.10 | Sierra et al. (2013) |
| | Spain | *Populus* ssp. | 20–30 | P | B | – | 3.30 | Sierra et al. (2013) |
| | Spain | *Populus* ssp. | 50–100 | P | B | – | 1.98 | Sierra et al. (2013) |
| | Italy | *Quercus/Fraxinus/Salix/ Populus* | 16 | N | B | 9.52 | – | Magnani et al. (2005) |
| | Italy | *Fraxinus/Prunus/Quercus* | 9 | N | B | 12.44 | – | Alberti et al. (2006) |
| | Italy | *Juglans regia* | 34 | N | B | 3.73 | – | Certini et al. (2023) |
| | Italy | *Quercus robur/Alnus* | 27 | N | B | – | 1.68 | Chiti et al. (2011) |
| | Italy | *Eucalyptus* ssp. | | P | B | – | 1.79 | Novara et al. (2012) |
| ***Afforestation Grassland*** | Spain | *Eucalyptus* globulus | 10 | P | B | 28.29 | −0.87 | Pérez-Cruzado et al., 2012 |
| | Spain | *Eucalyptus* globulus | 15 | P | B | 36.12 | 0.62 | Pérez-Cruzado et al., 2012 |
| | Spain | *Eucalyptus* globulus | 20 | P | B | 40.00 | 0.66 | Pérez-Cruzado et al., 2012 |
| | Spain | *Eucalyptus nitens* | 10 | P | B | 37.29 | −2.05 | Pérez-Cruzado et al., 2012 |
| | Spain | *Eucalyptus nitens* | 15 | P | B | 44.07 | −0.18 | Pérez-Cruzado et al., 2012 |
| | Spain | *Eucalyptus nitens* | 20 | P | B | 47.58 | 0.18 | Pérez-Cruzado et al., 2012 |
| | Spain | *Pinus radiata* | 30 | P | C | 35.17 | 1.02 | Fernández-Núñez et al., 2010 |
| | Spain | *Pinus radiata* | 35 | P | C | 36.20 | 0.99 | Pérez-Cruzado et al., 2012 |
| | Italy | *Pseudotsuga menziesii* | 11 | P | C | 10.58 | – | Coletta et al. (2016) |
| | Spain | *Pinus radiata* | 10 | P | C | 28.80 | – | Fernández-Núñez et al., 2010 |
| | Spain | *Betula* ssp. | 10 | P | B | 5.60 | – | Fernández-Núñez et al., 2010 |
| | Spain | *Pinus sylvetsris* | 49 | N | C | – | −0.58 | Nadal Romero et al. (2016) |
| | Spain | *Pinus nigra* | 49 | N | C | – | 0.10 | Nadal Romero et al. (2016) |
| | Spain | *Pinus sylvetsris* | 55 | N | C | – | −0.55 | Campo et al. (2019) |
| | Spain | *Pinus nigra* | 55 | N | C | – | 1.57 | Campo et al. (2019) |
| | Italy | *Pinus nigra* | 35 | N | C | 13.36 | – | Iovino et al. (2021) |
| | Italy | *Castanea sativa/Quercus* spp. | 10–25 | N | B | 8.97 | – | Iovino et al. (2021) |
| ***Agroforestry*** | Italy/Spain | *Malus/Pyearus/Prunus* | | N | B | 19.40 | – | Kay et al. (2019) |
| | Spain | *Prunus ssp.* | | N | B | 4.98 | – | López-Bellido Garrido et al. (2016) |
| | Italy | *Olea europaea* | | N | B | 8.16 | – | Proietti et al. (2012) |
| | Spain | *Quercus ilex* | | N | B | 0.47 | – | Kay et al. (2018) |
| | Spain | *Quercus suber* | | N | B | 3.00 | – | Kay et al. (2019) |
| | Spain | *Quercus suber* | | N | B | 3.00 | – | Palma et al. (2014) |
| | Med | *Paulownia ssp* | | P | B | 12.44 | – | Kay et al. (2019) |
| | Italia | *Quercus suber* | | N | B | – | 1.65 | Francaviglia et al. (2012) |
| | France | *Juglans regia* | | N | B | – | 1.06 | Cardinael et al. (2017) |
| ***Longer rotation*** | Spain | *Pinus sylvestris* | | N | C | 10.87 | – | Moreno-Fernandez et al. (2015) |
| | Spain | *Pinus sylvestris* | | N | C | 13.73 | – | Moreno-Fernandez et al. (2015) |
| | Spain | *Pinus sylvestris* | | N | C | 6.88 | – | Pérez Cruzado et al. (2012) |
| | Spain | *Eucalyptus/Pinus radiata* | | P | C | 36.75 | 0.07 | Pérez Cruzado et al. (2012) |
| ***Harvest Intensity*** | Spain | *Pinus sylvestris* | | N | Control | 6.04 | 4.24 | Bravo-Oviedo et al. (2015) |
| | Spain | *Pinus sylvestris* | | N | Light | 5.78 | 4.06 | Bravo-Oviedo et al. (2015) |
| | Spain | *Pinus sylvestris* | | N | Moderate | 5.74 | 4.61 | Bravo-Oviedo et al. (2015) |
| | Spain | *Pinus sylvestris* | | N | Control | 13.58 | 6.44 | Bravo-Oviedo et al. (2015) |
| | Spain | *Pinus sylvestris* | | N | Moderate | 10.43 | 6.37 | Bravo-Oviedo et al. (2015) |
| | Spain | *Pinus sylvestris* | | N | Heavy | 9.04 | 6.40 | Bravo-Oviedo et al. (2015) |
| | Spain | *Pinus pinaster* | | N | Control | 8.24 | 7.21 | Ruiz-Peinado et al. (2013) |
| | Spain | *Pinus pinaster* | | N | Moderate | 6.11 | 7.03 | Ruiz-Peinado et al. (2013) |
| | Spain | *Pinus pinaster* | | N | Heavy | 10.58 | 6.44 | Ruiz-Peinado et al. (2013) |
| | Italy | *Pseudotsuga menziesii* | | P | Light | 16.25 | – | Coletta et al. (2016) |
| | Italy | *Pseudotsuga menziesii* | | P | Moderate | 16.72 | – | Coletta et al. (2016) |
| | Italy | *Pseudotsuga menziesii* | | P | Heavy | 17.31 | – | Coletta et al. (2016) |

[a] N: Native; P: Plantation.
[b] C: Coniferous; B: Broadleaves.
[c] AGB: aboveground biomass.
[d] SOC: soil organic carbon.

lead to minimal SOC sequestration or even induce losses, though long-term outcomes typically show increases in both SOC and AGB (Ťupek et al., 2021). This effect varies depending on whether the afforestation occurs on former grasslands or croplands, with the latter often showing quicker SOC increases after tree planting (Ťupek et al., 2021). Conversely, in temperate regions, afforesting agricultural land mainly boosts an increase in new tree biomass, with less consistent and smaller SOC changes (Mayer et al., 2020). In Mediterranean regions, fast-growing tree species like *Eucalyptus spp.* and *Populus spp*, known for their robust carbon sequestration capabilities, are commonly planted. As discussed earlier, the broader impacts of these plantations should be assessed not just environmentally but also socioeconomically at both landscape and local levels. Potential trade-offs exist with other vital ecosystem services such as water supply, soil quality, fire risk, and biodiversity conservation (Lautenbach et al., 2017).

The effect of afforesting grasslands varies with climate; in humid areas, SOC losses can negate many years of biomass carbon gain (Fino et al., 2020). Conversely, more arid areas often see positive effects on SOC sequestration (Pellis et al., 2019). Notable are the benefits of planting native species, particularly nitrogen-fixing, on reclaimed mine

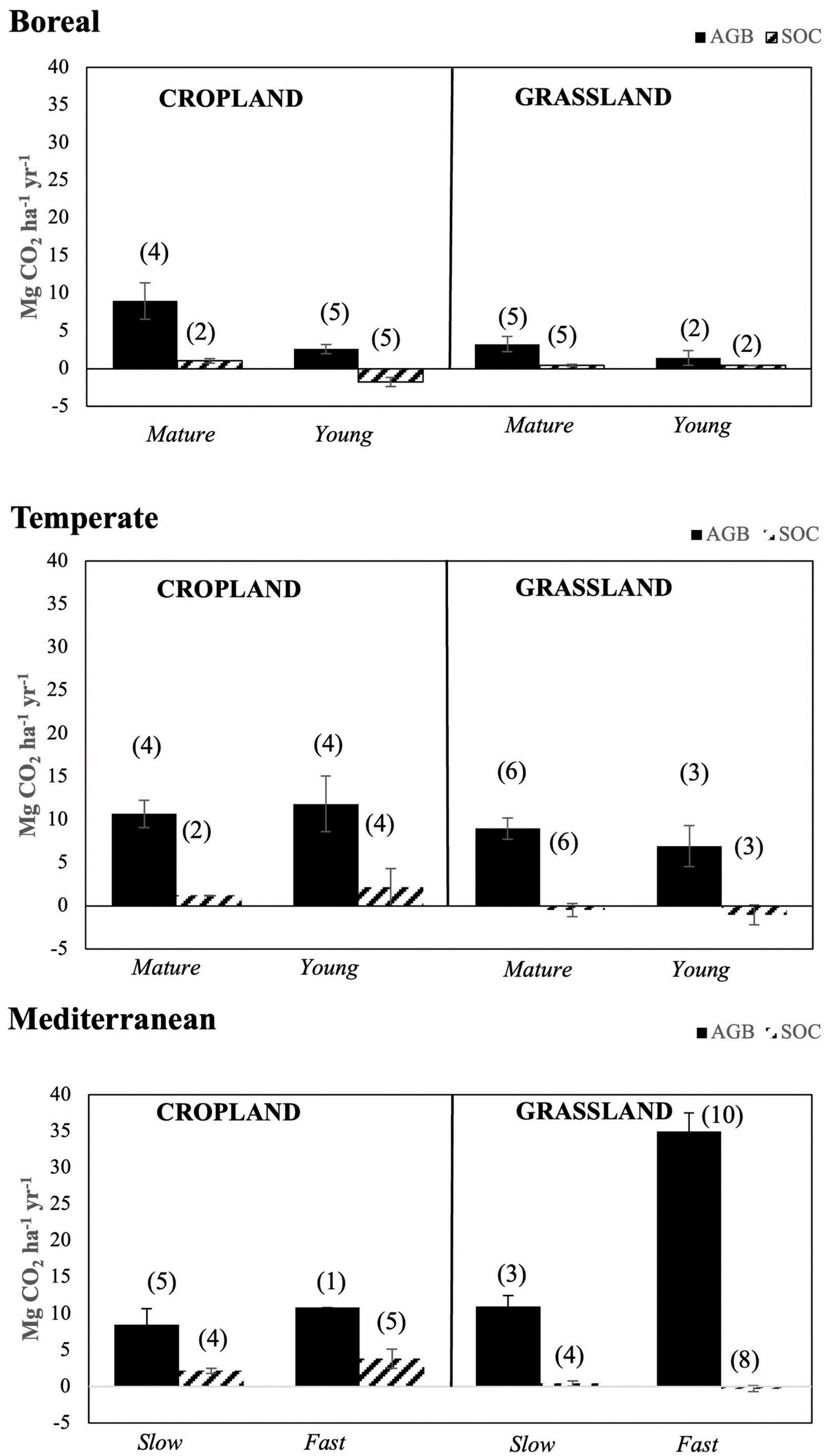


**Fig. 1.** Carbon sequestration rates in tree above ground biomass (AGB) and soil organic carbon (SOC) for afforestation on grasslands and croplands based on the literature review for different regions in Europe: boreal, temperate and mediterranean. The studies were divided into mature and young forest stands (more or less than 25 years old) for boreal and temperate zones, and in plantations (i.e. fast-growing species) and native (i.e. slow growing species) in the mediterranean zone. Bars represent the mean plus one standard error considering the number of study cases reported in brackets (corresponding to sites within single publications as specified in Table 1).

sites, which can significantly enhance SOC levels (Chiti et al., 2011). In Central Europe, many alpine pastures and unmanaged treeline areas are being abandoned (Estel et al., 2015). To achieve carbon sequestration, which occurs naturally but slowly through the forest secondary succession process, afforestation in these regions may be encouraged, enhancing "additionality". Over the long term, afforestation can secure significant C capture rates, approximately 3.5 Mg $CO_2$ $ha^{-1}yr^{-1}$, primarily in new tree biomass (Hiltbrunner et al., 2013; Speckert et al., 2023). However, afforestation at high altitudes involves considerable time, labour, and costs, and could potentially reduce biodiversity in alpine meadows. When planning afforestation, the principle of "*do no significant harm* (DNSH)" should be upheld, with a preference for planting native species that are adapted to both current and future climatic conditions. In some instances, introducing non-native, non-invasive species may be advantageous if they are better suited to anticipated climate changes. Identifying appropriate locations and tree species for afforestation remains a significant challenge and it is essential for a successful and efficient long-term carbon sequestration. Recent studies on the suitability of tree species for future climates in Europe suggest a narrowing range of suitable species, presenting a limited set of options for forest management (Wessely et al., 2024), potentially negatively impacting timber production, carbon storage and biodiversity conservation (McFadden, 2024). Given concerns about food security and competing demands for land, such as urbanization and solar power generation, it is debatable how much agricultural land in Europe can be dedicated to afforestation (Van de Ven et al., 2021; Zheng et al., 2023). Urban and peri-urban areas present promising opportunities for afforestation, offering not just climate mitigation benefits but also enhancing air quality, water retention, resilience against extreme weather, cooling effects, biodiversity, and recreational opportunities (Haase et al., 2014). Thus, although afforestation offers an opportunity for carbon sequestration, mainly in AGB and on croplands, careful examination of local characteristics, species suitability, and potential negative effects on other ecosystem services should be assessed.

#### 1.2.2. Tree species selection

Selecting the right species for a specific location that are adapted to local and future climatic conditions is another potentially interesting CF practice. The impact of changes in tree species composition on C sequestration varies significantly by species, making it challenging to measure precisely. Species with slow growth rates, like deciduous trees, often have denser wood, which can sequester C at rates comparable to faster-growing coniferous species. Förster et al. (2021) demonstrated that Scots pine stands in northern Germany sequestered less C than naturally regenerating beech forests, highlighting the complex relationship between tree growth rates and C sequestration capabilities. Altering the composition of tree species can have varied impacts on the C sequestered in both AGB and SOC. This effect on SOC can be influenced by mycorrhizal symbionts, as explored in studies by Mayer et al. (2020) and Schindlbacher et al. (2022). Additionally, species-specific C sequestration rates might change in the future due to different growth and mortality responses to global changes, as noted by Kasper et al. (2022) and Diers et al. (2023). Vospernik et al. (2024) observed that combining oak and pine could partially offset productivity losses caused by climate change. Promoting mixed species stands could adapt forests to climate change while enhancing productivity and C sequestration (Augusto and Boča, 2022). Beyond potentially high productivity and the provision of other ecosystem services (Huuskonen et al., 2021), diverse forest stands tend to be more stable and resilient to extreme conditions and disturbances (Pretzsch et al., 2013, 2015; Guyot et al., 2016). Mixing tree species can enhance not only aboveground productivity but also key soil-related co-benefits. Diverse rooting systems and litter inputs improve soil structure, organic matter accumulation, and microbial diversity, fostering greater nutrient cycling and carbon sequestration as also observed by Díaz-Pinés et al. (2011). As a result, mixed forests often display higher resilience and long-term sustainability compared to monocultures (Seddon et al., 2021). However, Huuskonen et al. (2021) reviewed that while mixing species can enhance growth and carbon storage in central and southern Europe, it shows no beneficial effects in the northern boreal forests, despite only few studies compare boreal mixed forests to respective monocultures. The review concluded that mixed forests offer greater ecosystem services than monocultures in boreal areas—such as biodiversity, soil benefits, and risk management—though browsing by ungulates remains a major threat to young stands. In general, only few studies report data on the impact of species selection and they mainly focus on the Mediterranean area, with rates of about 3.5 Mg $CO_2$ $ha^{-1}yr^{-1}$ for AGB and about 1 Mg $CO_2$ $ha^{-1}yr^{-1}$ for SOC (Fig. 2).

Mixing tree species to enhance 'additionality' in C sequestration is complex and highly dependent on specific conditions. While converting monocultures to mixed stands serves as a climate adaptation strategy, changing the dominance of tree species through forest management is a long-term endeavour and may initially result in C losses. Carbon sequestration at the forest scale is generally a long-term process, making species conversion or mixing less suitable for carbon crediting schemes that target short timeframes, such as five years, but more applicable to schemes planning for a decade or more. Furthermore, in selecting tree species for these efforts, it is recommended to prioritise slow-growing, climate-adapted species over fast-growing ones that may not be well-suited to changing climatic conditions.

#### 1.2.3. Longer rotation period

Traditionally, rotation periods, the intervals between complete harvesting cycles, have been mostly determined by economic factors related to timber production and regeneration period, overlooking the ability of mature forests to continue growing and sequestering carbon. Proposing longer rotation periods as a strategy to enhance C capture is therefore advocated as a viable CF practice.

In Norway, Stockland et al. (2021) investigated the impact of extending rotation periods from 100 to 120 years in Scots pine and Norway spruce stands, finding an increase in AGB sequestration ranging from 2.1 Mg $CO_2$ $ha^{-1}$ $yr^{-1}$ to 8.1 Mg $CO_2$ $ha^{-1}$ $yr^{-1}$ (Table 1). Triviño et al. (2017) investigated the effects of extending rotation periods on C sequestration in Central Finland, finding that such extensions could enhance annual carbon sinks significantly over decades. Similarly, a Europe-wide study by Kaipainen et al. (2004) predicted increases in biomass carbon sequestration from extended rotation lengths.

While extending rotations in Mediterranean regions shows potential for substantial C gains, especially in fast-growing non-native plantations (Fig. 2), these are generally not ideal for CF due to their potential negative impacts on other ecosystem services (e.g. water management, biodiversity). Extending rotation periods can provide significant short-term C sequestration benefits and serve as a transitional strategy to mitigate fossil $CO_2$ emissions. Additionally, prolonged rotations may improve SOC stocks indirectly through increased litter production (Feng et al., 2022) and provide ecological co-benefits like enhanced biodiversity and recreational opportunities in forests (Başkent and Kašpar, 2023). However, the feasibility of longer rotations must be weighed against the potential for increased disturbance risks and the implications for timber harvest, which could lead to leakage issues, thus necessitating a balanced approach to forest management strategies. Lengthening rotation periods reduces the annualised harvested volume per unit area over time by lowering harvest frequency, all else equal. Such reductions in one region typically shift unmet demand elsewhere, so on-site gains are frequently offset by market leakage, which empirical and modelling syntheses broadly place in the 60–100 % range across contexts and product categories (Murray et al., 2004; Gan and McCarl, 2007; Jonsson et al., 2012; Kallio and Solberg, 2018; Meyfroidt et al., 2020). The mechanism is straight-forward: a local supply cut raises prices; with relatively inelastic demand and positive supply elasticities elsewhere, replacement is expected (Murray et al., 2004).

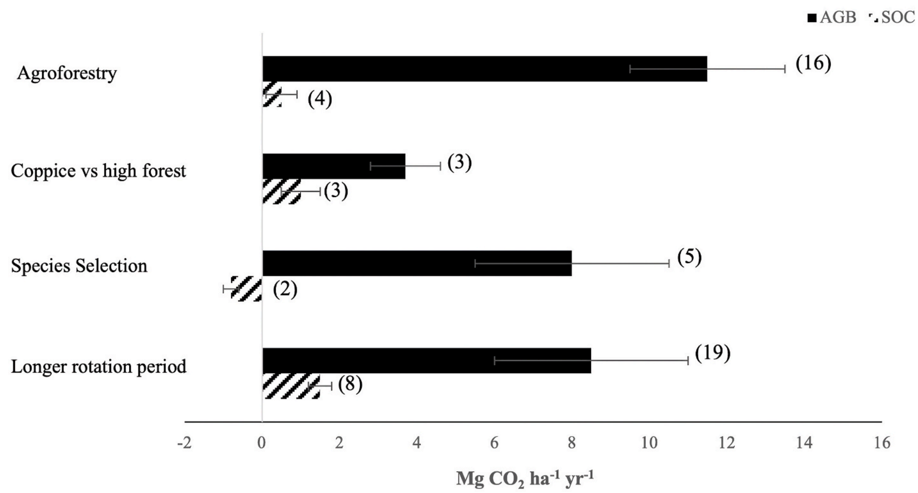


**Fig. 2.** Carbon sequestration rates in tree above-ground biomass (AGB) and soil organic carbon (SOC) of different practices in the Mediterranean region. Bars represent the mean plus one standard error and the number of stdy cases reported in brackets (corresponding to sites within single publications as specified in Table 1).

### *1.2.4. Conversion of coppice to high forests*

Coppicing, an ancient and sustainable forest management practice, is primarily used to produce small wood sizes for energy, agricultural, and local business needs, especially in the mountainous regions of central, east, and southern Europe. This method, which involves periodic cutting to ground level to stimulate growth, has historically simplified tree species composition by favouring species that regenerate well from stumps. Many coppice forests have been neglected or abandoned due to rural depopulation and economic challenges, representing a significantly underused natural resource (Unrau et al., 2018). Nevertheless, coppices remain crucial for biodiversity because of their indigenous species, continuous forest cover, and structural diversity. While coppicing is traditionally more productive on dry sites, converting coppice forests to high forests on more moist sites could enhance long-term C sequestration in forest biomass under certain conditions (Bruckman et al., 2011; Lee et al., 2018). How the conversion of coppice to high stands affects SOC is still unclear (Camponi et al., 2022). In term of C sequestration rates, converting coppice to high forest in the Mediterranean area results in about 8 Mg $CO_2$ ha$^{-1}$, while for SOC the rates are around zero or even negative (Fig. 2).

### *1.2.5. Reduced harvest intensity*

Reducing harvest intensity increases the number of older, larger trees, enhancing forest biomass C storage over time. In even-aged stand management, this can be achieved through adjusted thinning regimes and delayed final harvest. Similarly, in uneven-aged or continuous cover forestry, carbon stocks can be enhanced by maintaining higher growing stock. Research and practical experiences indicate that uneven-aged stands can be sustainably managed with varying stock levels depending on management goals, offering a model for CF and other objectives like regeneration (Schütz, 2002; Krumm et al., 2020). Yet, the long-term C storage in both forest management types depends on maintaining these practices consistently, despite the challenges posed by potential disturbances. Large trees are often more susceptible to disturbances like storms, fires, drought, or pests (Brienen et al., 2020; Korolyova et al., 2022). As such, increasing the number of mature trees can elevate the risk of significant C losses due to these disturbances, potentially offsetting any gains in C storage, especially if such disturbances become more common in the future (Senf and Seidl, 2021; Brèteau- Amores et al., 2023). Additionally, as with longer rotations, reducing harvest intensity can shift harvesting to other regions (market leakage), effectively offsetting most of its local environmental gain (Murray et al., 2004; Gan and McCarl, 2007; Jonsson et al., 2012; Kallio and Solberg, 2018; Meyfroidt et al., 2020).

One point often made for forest management abandonment is that unmanaged forests accumulate more C in AGB and SOC compared to managed ones, with forest dynamic models usually indicating peak in C sequestration for biomass of unmanaged forests (e.g., Seidl et al., 2007; Krug, 2019; Schwaiger et al., 2019; Štraus et al., 2023). In term of SOC sequestration, increasing the intensity of harvesting reduce the sequestration rates (Table 1).

Not harvesting trees could also foster a significant increase in deadwood C pool over time, which could be beneficial for additional C capture (Schulze et al., 2020). However, the effectiveness of this approach can vary depending on the context; for instance, in fire-prone ecosystems such as the Mediterranean, accumulating deadwood could significantly increase the risk of fire (Mantero et al., 2023), leading to potential extensive C losses. Nonetheless, in certain stable environments like slow-growing subalpine pine forests or productive beech forests, abandoning management could improve both the resilience of forests to climate change and their C mitigation potential (Jandl et al., 2019).

The cessation of forest management might be considered an appropriate CF strategy in very specific circumstances (Gregor et al., 2022); however, for most European forests, it is not a viable mitigation strategy. The likelihood of disturbances, exacerbated by climate change, could result in significant C releases. Moreover, in regions like Central Europe, legislation mandates specific management actions such as pest control and maintaining protective forests, which necessitates some level of active forest management; and where local harvesting is curtailed, unmet demand can leak to other regions, diluting net climate gains.

### *1.2.6. Continuous cover forestry on upland soils*

In Central and Northern Europe, adapting forests to climate change often involves transitioning from clear-cutting to continuous cover forestry (CCF) and diversifying species compositions from single-species stands to mixed-species forests. According to Seidl et al. (2007), in an Austrian Alps Forest unit, CCF can outperform traditional even-aged management with clear-cutting in long-term C sequestration. This approach leads to forests that are less vulnerable to disturbances such as insects and diseases, reducing the likelihood of unexpected C losses (Mohr et al., 2024). CCF also maintains a steady addition of C to the forest floor, unlike clear-cutting, which can disrupt C inputs and temporarily increase soil $CO_2$ (Mayer et al., 2014; Kobler et al., 2015) and $N_2O$ emissions in drained peatlands (Korkiakoski et al., 2023; Tikkasalo et al., 2025).

CCF maintains consistent root C input into the soil, preserving soil functionality, unlike clear-cutting which introduces bursts of dead root litter C. While thinning in CCF marginally and temporarily reduces C uptake by forests (Lindroth et al., 2018; Vesala et al., 2005), transitioning from clear-cutting to more natural forest management like CCF may initially lead to C losses, requiring a long-term perspective to appreciate the benefits (Hilmers et al., 2020). The transition is complex, influencing biodiversity and soil health, which may enhance forest productivity before C sequestration benefits can be fully realised.

#### 1.2.7. Site fertilisation

Fertilizing nutrient-deficient soils not only boosts tree growth but also typically slows the decomposition of soil organic matter, enhancing C sequestration in forest ecosystems (Melikov et al., 2023). In Northern European forests, where tree growth is commonly limited by nitrogen availability, adding nitrogen enhances biomass production and promotes SOC accumulation, particularly in the organic layer (Saarsalmi et al., 2014; Richy et al., 2024). Conversely, in Central Europe, the effectiveness of nitrogen fertilizers for CF is debatable due to high atmospheric nitrogen levels that have led to nitrogen saturation in many forest soils (Schmitz et al., 2024). This saturation can result in undesirable outcomes like nitrate leaching into water bodies (Mäkipää et al., 2023). Consequently, countries like Switzerland, Slovenia, and parts of Germany restrict or prohibit forest soil fertilization. Additionally, forest fertilization impacts biodiversity variably across different species, with some showing increases in abundance and others decreases, following changes in undergrowth biomass (Sullivan and Sullivan, 2018).

#### 1.2.8. Liming

Liming forest soils is primarily aimed at reducing soil acidification and improving fertility, thereby enhancing tree growth and increasing $CO_2$ capture in tree biomass (Reid and Watmough, 2014). In temperate forests, the impact of liming on tree growth varies by species and can either be negligible, positive, or negative (Table 1). A study on six spruce stands in southern Germany found that although stem growth wasn't enhanced by liming, it did boost fine root production and increased drought tolerance (Kohler et al., 2019). Conversely, in the boreal forests of Northern Europe, liming generally restricts tree growth (Derome et al., 2000), and lead to a long-term decline in SOC and nitrogen pools on acidic nitrogen-rich soils (Persson et al., 2021). Given these findings, liming has not been widely endorsed as a CF strategy in boreal environments. The interaction between liming and SOC is complex and varies depending on the soil's initial state. Liming can transform thick forest floor layers into more active humus layers, potentially reducing their C content by enhancing the activity of decomposers and earthworms (Bauhus et al., 2004; van Straaten et al., 2023). On mineral soils, the impact of liming on SOC is influenced by the soil's level of acidification and its initial organic C and clay content. Moreover, liming may increase nitrate leaching due to accelerated soil nitrogen cycling rates (Kreutzer, 1995; Gundersen et al., 2006). The sparse research available makes it challenging to definitively classify liming as a beneficial CF practice without a site-specific evaluation of its potential additional C sequestration benefits, which are highly dependent on the tree species involved and local soil conditions.

#### 1.2.9. Fire management

Increasing temperatures will increase the number of wildfires in the absence of specific fire management (Turco et al., 2014). The recent increase in the severe wildfires in parts of the Mediterranean region directly links to climate and land use changes (Prichard et al., 2017). Decreases in biomass production as a result of climate change may limit fire incidence over parts of the Mediterranean but other parts with high biomass accumulated due to land use changes are prone to severe wildfires (Lecina-Diaz et al., 2014). Prescribed fire is a valuable forest management practice that can be used to reduce fire risk by decreasing fuel load, which is growing due to increasing forest area (Pausas and Keeley, 2019). The effects of prescribed fires on SOC vary considerably, although the impact is generally lower than that of wildfires and can even increase SOC (Alcañiz et al., 2018).

The effects of low or high-frequency fire occurrence on SOC stocks differ (Agbeshie et al., 2022). While high-intensity forest fires have severe negative effects on forest soils and result in nutrient losses, the breakdown of soil aggregate stability and hydrophobicity, low-intensity forest fires can lead to increased fertility and pH. Less severe fires should be established as the most sustainable regime to stabilise SOC pools (Fernandes et al., 2013). The establishment of prescribed burning and fire management baselines as well as additionality indicators are challenging, yet it should be on a future research-policy agenda as fire risks are increasing, jeopardising the European forest carbon sink.

#### 1.2.10. Agroforestry

Silvopastoral agroforestry systems are by far the most widespread agroforestry land-use type in Europe, covering 15.1 million ha, 3.5 % of European land area (den Herder et al., 2017). Maintaining current European cultural agroforestry landscapes and promoting the conversion of new agricultural land to agroforestry systems could enhance biodiversity and multiple ecosystem services, including C sequestration. This strategy could also be employed on cropland, where trees could be used as windbreaks, buffers or for shade provision (Nair et al., 2010). Agroforestry systems enhance soil fertility through increased litterfall and rhizodeposition, helping to reduce soil erosion and improve water quality (Moreno Marcos et al., 2007; Jose, 2009; Kay et al., 2019). The rate of C sequestration in these systems varies depending on the composition, age, and geographic location of tree species, as well as environmental factors and management practices, including the soil type and the historical management's legacy effect (Nerlich et al., 2013). The AGB pool shows positive sequestration rates ranging between 0.5 and 19.4 Mg $CO_2$ $ha^{-1}$ $yr^{-1}$, influenced by the choice and number of tree species (Table 1 - Kay et al., 2019). Lower values are observed for the SOC sequestration rates ranging between 1.0 and 1.6 Mg $CO_2$ $ha^{-1}$ $yr^{-1}$ (Table 1). However, it has to be considered that some methodological factors, such as the sampling design, can influence these estimates due to the complex array of agroforestry systems (e.g. samples collected under the influence of the trees vs. those in tree free areas). Ensuring the long-term viability of these systems involves careful management planning to support ongoing forest regeneration.

### 1.3. Peatland management

#### 1.3.1. Water table management in drained peatlands by continuous cover forestry

In boreal regions, employing continuous cover forestry (CCF) for managing peatland water levels significantly reduces nutrient discharge into water systems, notably in the Nordic countries, Baltic states, and Poland, and also aiding in the restoration of native peatland flora. CCF, by averting the extensive GHG emissions associated with clear-cutting, presents immediate climatic advantages (Tanneberger et al., 2021). CCF on fertile drained peatlands has been proposed as an alternative to even-aged forestry (Nieminen et al., 2018). The CCF maintains a continuous biomass presence, thus eliminating the high C dioxide and nitrous oxide emissions linked with the barren phases following clear-cuts. This practice also involves adjusting the water table to higher levels than those in densely forested and well-drained areas, contributing to reduced GHG emissions and aiding in climate adaptation during extended dry spells. According to Lehtonen et al. (2023), transitioning to CCF in fertile drained peatlands, could cut annual emissions substantially while preserving economic returns for landowners (Juutinen et al., 2021), enhancing water quality by reducing nutrient runoff (Palviainen et al., 2022), and maintaining optimal water table levels (Leppä et al., 2020). However, the difference in water table levels between CCF management and even-aged management has been less than expected in some sites, suggesting that additional water level management, such as dams and ditch blocking, may be necessary to ensure sufficient reductions in soil-related GHG emissions (Peltoniemi et al., 2023). Restoration and rewetting have also been considered to reduce GHG emissions from drained peatlands (Gü;nther et al., 2020), despite in Central Europe a SOC loss is observed (Table 1).

#### 1.3.2. Wood ash fertilisation of drained peatlands

Wood ash fertilization, aimed at boosting tree growth by enhancing soil fertility, presents an effective C sequestration strategy on drained peatlands (Hytönen et al., 2016). This technique alters peat chemistry by increasing soil pH and making nutrients like phosphorus and potassium more available, especially beneficial on low pH, low fertility sites (see Jansone et al., 2020). While it creates a long-term C sink in nutrient-poor locations, its impact varies on more fertile sites, ranging from negligible C uptake to potential $CO_2$ emissions, indicating possible long-term C losses (Ojanen et al., 2019). Nonetheless, understanding the effects of ash fertilization on GHG emissions from soil remains limited, with few studies conducted on ash-fertilized peatlands (e.g., Ernfors et al., 2010; Maljanen et al., 2014; Ojanen et al., 2019). Typically, ash fertilisation increases tree growth, but the extent depends on soil nitrogen (Moilanen et al., 2013, 2015; Lehto and Ilvesniemi, 2023). According to Lehtonen et al. (2021), increasing ash fertilisation radically in Finland by 30,000 ha for 2022–2025 and after that by 100,000 ha annually would bring an additional increment of 1.2 Tg $CO_2$ $yr^{-1}$. Current $CO_2$ emissions from drained forested peatlands in Finland are 11.6 Tg (EEA, 2025).

In terms of SOC sequestration, the impact of ash fertilisation in upland soils has yielded conflicting results. Some studies report a positive effect with an increased carbon sink in trees of 2.5 Mg $CO_2$ $ha^{-1}yr^{-1}$ (Hanssen et al., 2020), others have found no effect (Moilanen et al., 2013). Consistent with liming, wood ash application often increases SOC mineralisation (Zimmermann and Frey, 2002; Rosenberg et al., 2010), which may offset C sequestration in AGB. The utilization of wood ash as a soil amendment in forested peatlands is hindered by concerns regarding the accumulation of heavy metals and the absence of definitive regulations, which limits its broader application in Central Europe. Regarding biodiversity, the effects of ash fertilization are largely undocumented, though some studies suggest it may positively influence biodiversity (Silvan and Hytönen, 2016; Zuševica et al., 2022). The IPCC (Hiraishi et al., 2013) recommends using ash from bioenergy plants and avoiding ash fertilization near waterways to prevent nutrient runoff and leaching, promoting sustainable land management by enhancing soil fertility and tree growth.

#### 1.3.3. Peatland restoration

Restoring peatlands significantly reduces soil GHG emissions, and rewetting drained agricultural peat soils is becoming common in Central Europe. Such measures require comparatively less land and should be prioritized due to their efficiency in mitigating emissions (Leifeld and Menichetti, 2018). In Northern Europe, the climate benefits of restoration are not immediate and will only materialise in the long run, while in Central Europe climate benefits from peatland restoration are more immediate (Ojanen and Minkkinen, 2020). Forested peatlands are less widespread in Central Europe than in Northern Europe but still cover significant areas in certain countries (e.g., ~270,000 ha in Germany and 150,000 ha in Poland - Peters and Unger, 2017). GHG emissions from drained forested peatland soils are high and range between 12 and 29 Mg $CO_2$-eq $ha^{-1}$ $yr^{-1}$ in temperate climates (Tiemeyer et al., 2020; Wilson et al., 2016). Rewetting can significantly reduce the soil GHG emissions by on average 8 Mg $CO_2$-eq $ha^{-1}$ $yr^{-1}$ (Wilson et al., 2016) or in some cases even turn them back into a net GHG sink in the longer-term (Gü;nther et al., 2020). A side-effect of rewetting is increased methane emission, which can offset much of the cooling in the early decades after rewetting (Ojanen and Minkkinen, 2020). In this context, the total GHG reductions achieved can also depend on the nutrient content of the degraded peat. Rewetting of nutrient-poor temperate peatland soils has a significant GHG reduction potential, whereas the rewetting of nutrient-rich soils does not reduce soil GHG emissions (Wilson et al., 2016). In the boreal region, reduced $CO_2$ emissions due to restoration varied between 2.2 and 11.5 Mg $CO_2$-eq $ha^{-1}$ $yr^{-1}$, while simultaneously the increase in $CH_4$ emissions varied between 0.7 and 6.0 Mg $CO_2$-eq $ha^{-1}$ $yr^{-1}$. However, the net GHG balance still provides climate benefits (Wilson et al., 2016).

Overall, peatland restoration measures can have a long-lasting (centuries+) reductive effect on soil GHG emissions and hence can be considered as a measure with high permanency. Another clear co-benefit of peatland restoration is the positive impact on biodiversity (Rana et al., 2024). However, it should not be forgotten that rewetting of drained forested peatlands can substantially negatively affect C sequestration in tree biomass. Water tables near or even above the peat surface create anoxic conditions in the tree root zone, significantly reducing growth or killing the existing trees (Kozlowski, 1986). If productive forests on drained peatland soils are rewetted, the reduced tree $CO_2$ uptake can offset the GHG emission savings from the rewetted soil for decades. According to Laine et al. (2024) peatland restoration may produce cooling or warming depending on the chosen pathway, showing that restoration from nutrient fertile site to mire with trees produced cooling, while warming occurred when peatlands with poor soils were restored to open mires. The results of Laine et al. (2024) were discussed by Launiainen et al. (2025), where in addition to GHG exchange also albedo and harvested wood products was accounted for. Results of Launiainen et al. (2025) show that restoring nutrient rich peatlands provide climate benefits in long term time horizon (<200 years). On the other hand, global warming potential based GHG exchange estimations with peatlands is problematic, as those exclude the climatic benefit from the historical carbon storage in peatland soils (Rinne et al., 2025). To sum up, in the longer term (centuries) the positive effects of GHG emission reductions from the rewetted soil prevails (Hommeltenberg et al., 2014; Schwieger et al., 2021). Given the limited time horizons in C crediting schemes, the initial response of tree biomass to rewetting and different restoration pathways needs to be carefully considered.

## 2. Challenges for implementation

Although about 40 % of the European Union's land area is occupied by forest, to what extent they will contribute to reach the EU climate neutrality by 2050 is questionable. Among all the practices, afforestation is the one leading to the higher C sequestration rates in the short term in the AGB. In the case of soil, the C accumulation takes much longer and when afforestation is performed on grasslands it can even lead to C and biodiversity losses. Moreover, main challenges are related to species availability for future climate and land competition with other land uses. In this sense, tree species selection is required prioritising climate-adapted species, although it may lead to initial C losses and slow C sequestration rates in the short term, as most of these species are slow growing.

Agroforestry is a system to be promoted to increase the C both in the soil and the AGB, and at the same time generate a series of additional ecosystem services. Some forest management practices, involving longer rotation periods as well as reducing harvest intensity, needs to be carefully evaluated considering the local risk related to climate, such as increase of fire risk in the Mediterranean region or pathogen risk in eastern Europe. Besides, leakage issues in relation to a decrease in wood harvest need to be considered in relation to the increasing wood demand.

Another challenge to consider is the non-permanence of forest carbon storage as the sequestered carbon can be released back into the atmosphere in the future. This can occur either because of disturbances such as wildfires, droughts or pests, which will increase in the future, or as a result of management practices and wood processing. In this context, the use of prescribed fires appears as a promising practice to preserve the forest C stocks, considering the increasing impact of this natural disturbance not only in the Mediterranean area.

Peatland management in north European countries is a promising CF practice with reasonable mitigation rates for both AGB and SOC. Peatland restoration and management has a high potential reducing GHG emissions significantly and involving much less land than that of other CF practices (e.g., afforestation). While in Central Europe the reduction is immediate, in Northern Europe benefits are observed in the long-term. Peatland rewetting reduces significantly soil GHG emissions, even turning the ecosystem back into a net carbon sink in the long term. However, its side effect of increased methane emissions can offset the benefits from $CO_2$ sequestration. Rewetting productive forests on drained peatland can also lead to a decrease in tree $CO_2$ uptake, which may counterbalance the GHG emission savings for many years. However, this process also has positive long-term effects, including the enhancement of local biodiversity. While the review revealed a large number of studies assessing the impact of forest practices on carbon storage, it also highlighted substantial variability in results, largely due to differences in studied forests and their environmental conditions. To provide a solid foundation for designing policy instruments such as carbon credit markets, further research is needed.

## 3. Conclusions

The long and variable timescales inherent in forestry activities pose a challenge for enhancing forest carbon, particularly in balancing short and long-term climate goals. Forest policy instruments are needed to reverse the downward trend of the carbon sink and to support forest adaptation to ongoing climate change. However, this might require measures that, in the short-term, reduce the net forest sink to increase forest resilience and thus generate more sustained carbon storage in the long term.

Even though forest carbon uptake can be tangible in the long run, given that European forests are subject to a densification process, forest management is required to enhance forest resilience against hazards such as drought. Management must therefore balance the potential benefits (e.g. increased carbon sequestration) with the associated disturbance risks. This will depend on local climatic and forest conditions, as well as future climate and environmental changes, and will therefore vary regionally. The success of such measures may critically depend on regional forest risk exposure (e.g., fire, wind, pests) and forest responses to climate change. Carbon farming practices will be more effective in regions that are less affected, or even benefited, by climate change, because of higher productivity under new climatic conditions. In contrast, adaptation rather than mitigation policies should be prioritized in regions where forests are already on the brink of mortality. Furthermore, leakage risk must be addressed wherever harvesting is reduced: without safeguards, higher wood demand or local supply cuts can shift pressure to other regions, undermining net reductions. Mitigation should couple demand-side measures (material efficiency, reuse and longer-lived products, and curbs on energy-wood) with sustainable intensification on already managed forests where ecologically appropriate, tighter import safeguards and traceability to avoid shifting pressure to high-deforestation regions, and explicit, evidence-based leakage deductions in crediting so reported gains reflect system-wide outcomes.

Finally, this review highlights the need for further long-term studies reporting and quantifying both the C in the AGB and SOC for specific management practices, as well as including the evaluation of other ecosystem services to ensure that forests contribute sustainably to achieving European emission reduction targets.

## CRediT authorship contribution statement

**Tommaso Chiti:** Writing – original draft, Data curation. **Ana Rey:** Writing – original draft, Data curation. **Jens Abildtrup:** Writing – review & editing. **Hannes Böttcher:** Writing – review & editing. **Jurij Diaci:** Writing – review & editing, Writing – original draft. **Oliver Frings:** Writing – review & editing. **Aleksi Lehtonen:** Writing – review & editing, Writing – original draft. **Helga Pülzl:** Writing – review & editing. **Andreas Schindlbacher:** Writing – review & editing, Writing – original draft. **Miguel A. Zavala:** Writing – review & editing, Writing – original draft.

## Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Tommaso Chiti reports financial support was provided by European Forest Institute (EFI). Ana Rey reports financial support was provided by European Forest Institute (EFI). Jens Abildtrup reports financial support was provided by European Forest Institute (EFI). Hannes Bottcher reports financial support was provided by European Forest Institute (EFI). Jurij Diaci reports financial support was provided by European Forest Institute (EFI). Aleksi Lehtonen reports financial support was provided by European Forest Institute (EFI). Andreas Schindlbacher reports financial support was provided by European Forest Institute (EFI). Miguel A. Zavala reports financial support was provided by European Forest Institute (EFI). If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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## Data availability

Data will be made available on request.

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