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# Carbon farming in the European forestry sector

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# Contents

# Executive summary

## What is at stake?

The European Union has made a commitment to become climate neutral by 2050. This ambitious goal will require a drastic reduction in greenhouse gas emissions across its economy, as well as the removal and storage of unavoidable emissions from the atmosphere. Forests occupy almost 40% of Europe's land area and remove atmospheric carbon dioxide through the process of photosynthesis. For this reason, the EU is strongly relying on forests to reduce its emissions through carbon sequestration, and also by carbon storage in wood products and avoiding emissions from wood, etc. These strategies are essential for the EU to reach its carbon neutrality target.

"**Carbon farming**" practices aim to enhance the carbon sequestration potential of forests and soils as well as avoiding or reducing greenhouse gas emissions. European multifunctional forest management aims to simultaneously enhance carbon removal and storage, as well as forest resilience and adaptation to climate change. Management practices such as afforestation (planting trees in an area with no recent tree cover), diversifying forest structure and composition, extending rotation periods or reducing harvesting intensity, site fertilisation as well as agroforestry or peatland management, can all contribute to reaching climate targets while maintaining forest multifunctionality and biodiversity. The EU has put in place a framework to implement carbon farming initiatives and is developing a regulatory framework for the accounting and certification of carbon removals from the atmosphere and wood products.

This report examines EU forest management practices that can potentially improve carbon sequestration in forest ecosystems. It analyses their potential as carbon farming measures and also points out the challenges for monitoring and implementing mitigation milestones in carbon farming practices. The report also explores the policy and economic framework, and recommends key criteria for the successful implementation of carbon farming instruments.

## Which forest management practices are suitable for carbon farming?

This report analyses available practices using the four QU.A.L.ITY criteria established by the EU: **quantification** (carbon removal must be accurately measured), **additionality** (carbon removal must go beyond standard practices), **long-term storage** (carbon must be removed at least for five years) and **sustainability** (multifunctionality, climatic resilience, biodiversity, etc.). These management practices must avoid leakage (transfer of activities resulting in carbon emissions elsewhere).

All of these practices must be carefully tailored to local climatic and forest conditions to balance the potential carbon benefits with the risks of carbon losses due to natural disturbances. These practices not only contribute to carbon sequestration but can also improve forest health, biodiversity and overall ecosystem resilience.

**Afforestation** leads in most cases to carbon sequestration both above- and below-ground. Current literature shows that carbon sequestration rates are highest in the Mediterranean area and decrease in temperate and boreal forests. However, as land use competition increases, land availability for afforestation becomes more limited requiring systematic planning to reconcile management goals (i.e. mitigation, biodiversity conservation and multifunctionality at local and regional scales).

Furthermore, it is crucial to carefully select afforestation sites to avoid displacing agricultural activities. Failure to do so could result in market leakage effects and increase pressure on carbon storage in other ecosystems.

| PRACTICE | | Quantification | Additionality | Permanence | Leakage Prevention |
|---|---|---|---|---|---|
| **Afforestation** | | High | High | High | High |
| **Silvicultural practices** | *Species selection* | Medium | Medium | High | Medium |
| | *Reduced harvest/ Lenghtened rotation* | Low | Medium-High | Medium | Low |
| | *Reduced thinning intensity* | Medium | Medium | Medium | Medium |
| | *Diversification of forest structure* | Medium | Medium | High | High |
| | *No harvesting* | Medium | Medium-High | Medium | Low |
| | *Site fertilisation* | na | Low-Medium | Medium | High |
| **Fire management** | | High | High | Medium | High |
| **Agroforestry** | | Low | High | Medium-high | High |
| **Peatland management** | *Peatland restoration* | Medium | High | High | Medium |
| | *Continuous cover forestry on drained peatlands* | Medium | Medium | High | Medium-high |

**Peatland restoration and management** has the second-highest potential. **Peatland restoration** can reduce greenhouse gas emissions significantly and involves much less land than afforestation. While in Central Europe the reduction is immediate, in Northern Europe benefits are observed in the long-term. **Peatland rewetting** reduces significantly soil greenhouse gas 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 greenhouse gas emission savings for many years. However, this process also has positive long-term effects, including the enhancement of local biodiversity.

**Managing peatland water levels** through continuous cover forestry has the potential to reduce greenhouse gas emissions. Moreover, this practice also excludes treeless periods and associated high $CO_2$ and $N_2O$ emissions. **Wood ash fertilisation** can improve soil fertility and stimulate tree growth especially on drained peatlands, but the impact on upland soils depends on soil nitrogen availability. Effects on soil greenhouse gas emissions remain less understood as there are few measurements in ash-fertilised peatlands. Environmental concerns are linked to heavy metals in wood ash, and the biodiversity impacts are not well-known yet.

The **effects of fire** on soil carbon vary considerably between prescribed burning (lower impact) to wildfires (higher impact). While high-intensity forest fires have negative impacts, low-intensity fires can be positive for the accumulation of soil and biomass carbon.

**Agroforestry practices** improve soil fertility as a result of the increase in litterfall and carbon inputs into soils, but carbon sequestration rates depend on many factors (tree species composition, age, location, management practices etc.). Moreover, the long-term persistence of carbon stocks is challenging.

A set of **silvicultural practices** can be implemented or adapted to increase carbon sequestration in managed forests. The most relevant are:

- A **change in species composition** can increase carbon sequestration in degraded forest areas, but the potential in already sustainable managed forests is limited. It can also have controversial effects on local biodiversity and other forest functions, particularly if non-local species or provenances are used.

- **Reduced harvesting** increases the proportion of older larger trees, and hence forest biomass carbon stocks over time. However, the permanence depends on local disturbances and disturbance resilience. Furthermore, reducing harvesting rates *carries a high risk of market leakage*.

- **Longer rotation periods** allow for significant carbon sequestration during a transition period until the desired (longer) rotation period is reached and the forest landscape can hold higher carbon stock.

- Under certain conditions the **conversion of coppices into forests** can increase carbon sequestration in the long term, but the impact on soil carbon is unclear. With regard to **continuous cover forestry** quantifying additional $CO_2$ sequestration is difficult and requires **long time horizons**, especially to detect changes in soil carbon.

- In contrast to Northern Europe, where forest growth is often nitrogen-limited in upland soils, the use of **nitrogen fertilisers** as a carbon farming practice is questionable in the nitrogen saturated areas of central Europe.

- The **impact** of liming on carbon sequestration **is complex** and depends on the initial soil conditions. Due to the **limited number of studies** on the subject, it is difficult to determine in which regions liming can be considered a carbon farming practice.

## What are the challenges for carbon farming in forests?

(1) The **long and variable timescales** inherent in forestry activities pose a challenge for enhancing forest carbon – in particular how to balance between **short-term/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.

(2) The **non-permanence of forest carbon storage** presents another challenge, as the sequestered carbon can be released back into the atmosphere in the future. Natural disturbances can also lead to rapid carbon losses. Forest management therefore needs to balance the potential benefits with the associated disturbance risks – this depends on local conditions as well as future climate changes, and therefore varies across Europe.

(3) **Setting baselines and verifying carbon removal and gains** resulting from afforestation and reforestation, forest protection and silviculture present important challenges.

(4) **Additionality requires "proof" of a lower carbon sequestration** in the absence of such measures in any form of carbon farming. Carbon farming needs to deliver multiple environmental co-benefits, such as biodiversity conservation, water regulation, or soil health improvement among other ecosystem services. Practices that enhance these co-benefits should be prioritized.

(5) **Methodological (quantification) problems** can affect carbon farming measures – in particular, changes in soil carbon are difficult to measure and quantify, and there is a great need for method harmonization and improvement.

Measuring carbon sequestration in above-ground forest biomass is less demanding and cost intensive than measuring/modelling carbon sequestration in forest soils. However, there are cost-efficient empirical approximations that can be incorporated in forest inventory surveys. **Hybrid approaches** involving remote sensing multiple observation and ground data are one of the most promising approaches to obtain spatialized information on land use changes and forest carbon dynamics.

## Ways forward to implement carbon farming in forests

Forest management practices must take a holistic approach across several forest functions – including carbon storage, biodiversity conservation and other ecosystem services. Environmental, economic and social perspectives and co-benefits should all be considered.

(1) **Uncover conflicting policy goals and resolve them** to support carbon farming in forests. (e.g. national schemes for payments for ecosystem services provide funding for lowering wood harvest, while the EU Renewable Energy Directive promotes the use of biomass for bioenergy).

(2) **Establish reliable systems for monitoring and reporting** carbon sequestration to verify the impact of carbon farming. A **robust monitoring, reporting and verification (MRV) system** can help ensure transparency and accountability, and provide data to support policy and market development.

(3) **Agree on standardized methodologies with transparent guidelines for baseline development** (for example, a business-as-usual scenario). Methods for estimating baselines should align with national-level reporting standards and policy objectives. **Implement tonne-year-accounting** to measure the climate impact of temporary carbon storage more accurately.

(4) **Define the exact scope for removal projects**. This should be determined by the methodology itself and not left to the discretion of individual projects.

(5) **Determine removals conservatively**, rather than using the most accurate estimate. In cases of high uncertainty, approaches should be more conservative and alternative use of credits should be foreseen (not for offsetting greenhouse gas emissions).

(6) **Adopt dynamic measures** given the non-permanence of carbon farming activities. This is to address the **risk of carbon reversals** due to natural events or management changes. Recommended approaches are the use of temporal carbon credits, which are periodically verified and adjusted, and dynamic carbon buffers that can be resized based on real-time data.

(7) **Prioritize market leakage prevention** in new regulations along with implementing rigorous and transparent accounting practices for any residual leakage effects.

(8) **Make activities funded by voluntary carbon markets visible in the country's national greenhouse gas inventory**. Greenhouse gas inventories are the main tool for Member States to steer the country towards achieving national targets and assess compliance.

A major factor for the attractiveness of credits from carbon farming in forestry is the question of whether credits can be used for offsetting greenhouse gas emissions. Alternative uses could also be pursued, including compliance use of units for contribution claims, or for getting access to subsidy schemes. However, striving for greenhouse gas-neutral enterprises or products is a major driver for the expansion of the voluntary carbon market.

## Glossary of forest management terms

**Afforestation:** establishing new forest on lands where there was no forest before (for example, abandoned agriculture).

**Agroforestry:** integration of trees and shrubs into crop and animal farming systems to create environmental, economic and social benefits.

**Basal area (BA):** the cross-sectional area of trees at breast height (1.3 m above ground) in m$^2$ per hectare.

**Clear-cutting:** the harvesting in one operation of the whole stand larger than one tree height without any established advance regeneration.

**Continuous cover forestry (CCF):** forest management without clear- cutting, mainly based on natural regeneration and use of the existing understory.

**Coppice:** a type of stand originated from asexual reproduction, such as sprouts or root suckers, in contrast to high forest (regenerated from sexual reproduction).

**Coppicing:** a silvicultural system in which all trees in the previous stand are cut in one operation and most regeneration is from sprouts or root suckers.

**Even-aged forest management:** a planned sequence of treatments designed to create or maintain a stand with predominantly one age class.

**Growing stock:** the sum of volume of all trees above a threshold (such as diameter > 7 or 10 cm) in a forest stand in m$^3$ per hectare.

**High forest:** a type of stand originated from seed or from planted seedlings in contrast to a low or coppice stand.

**Reforestation:** the natural or artificial renewal of a forest ecosystem by establishing trees.

**Rotation period:** in even-aged systems, the period (in years) between regeneration establishment and final cutting.

**Silvicultural system:** a silvicultural system is a planned programme of treatments during the whole life of a stand designed to achieve management goals and sustainability of ecosystem functions.

**Thinning:** reducing tree density to improve growth of the residual trees for enhancing vitality, stability and value of individual trees and stands.

**Uneven-aged forest management:** a planned sequence of treatments designed to create or maintain a stand with three or more age classes.

# 1 Introduction: What is carbon farming and why is it important?

To achieve the targets set in the Paris Agreement in 2015, the European Union (EU) approved a Climate Law in 2021 (EU 2021) that includes a commitment to reduce net carbon emissions by 55% compared to 1990 levels by 2030, the so-called *Fit for 55 plan*, and to become climate neutral by 2050. To meet these ambitious commitments, the European Union countries must drastically reduce their greenhouse gas (GHG) emissions and must adopt activities promoting carbon sequestration. The priority should be to reduce the greenhouse gas (GHG) anthropogenic emissions while adopting measures that increase carbon removal from the atmosphere. To meet these targets, the European Commission adopted a *Communication on Sustainable Carbon Cycles* (EC 2021) in December 2021 as part of the EU *Green Deal* (EC 2019), proposing a series of short to medium-term actions including **carbon farming** (CF). In particular, the EU aims to promote CF initiatives and to develop a regulatory framework for the accounting and certification of carbon removals, i.e. carbon capture from the atmosphere. CF practices include climate-friendly practices implemented by farmers and foresters that enhance carbon sequestration and storage in forests and soils, as well as reducing GHG emissions from soils.

This report aims to critically analyse the role forest-related carbon farming practices can play in meeting the EU's commitment of climate neutrality by 2050, given that forests occupy almost 40% of the European land area. It gathers scientific evidence to examine the effects of forest management on carbon sequestration in forest soils and tree biomass. It then looks at forest management approaches currently applied in the EU and analyses their role as carbon farming measures, as well as challenges for their monitoring and implementation.

Carbon farming activities must also be developed within a policy and economic framework. The report explores the current situation in the EU, which has put in place a framework to implement carbon farming initiatives and is developing a regulatory framework for the accounting and certification of carbon removals from the atmosphere. The report also considers key criteria for the successful implementation of carbon farming instruments.

## 1.1 Defining "carbon farming" in the European Union

According to the European Union (EC 2021, modified in 2024a), *"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 carbon into biogenic carbon pools or the reduction of soil emissions".*

In 2024, the European Parliament and the Council reached a provisional agreement on the first EU-wide voluntary framework for the certification of high-quality carbon removals (EC 2024a), including carbon farming and carbon storage in products.

The certification must follow the so-called QU.A.L.ITY criteria:

(1) **QUantification:** carbon removal activities are measured accurately and deliver unambitious benefits for the climate

(2) **Additionality:** carbon removal activities go beyond standard practices and what is legally required

(3) **Long-term storage:** certificates clearly account for the duration of carbon storage and distinguish permanent storage from temporary storage

(4) **sustainabilITY:** carbon removal activities must support sustainability objectives such as climate change mitigation and adaptation, biodiversity, circular economy, water and marine resources.

Thus, carbon farming now refers to *land management practices that reduce greenhouse gas emissions, especially from soils, and/or increase the sequestration and storage of carbon in vegetation and soils.*

## 1.2 Forests and the carbon cycle

Forests (including soils) are the Earth's main carbon reservoir. They provide important ecosystem services including carbon storage and mitigation of climate change (Bonan 2008; Nabuurs et al. 2017). Forests absorb about 30% of anthropogenic $CO_2$ emissions (approximately 2 Petagrams – billion tons – of C annually) reducing the rate of $CO_2$ increase in the atmosphere (Tian et al. 2016 – Figure 1).

The long-term capacity of forest ecosystems to sequester carbon 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 (see below).

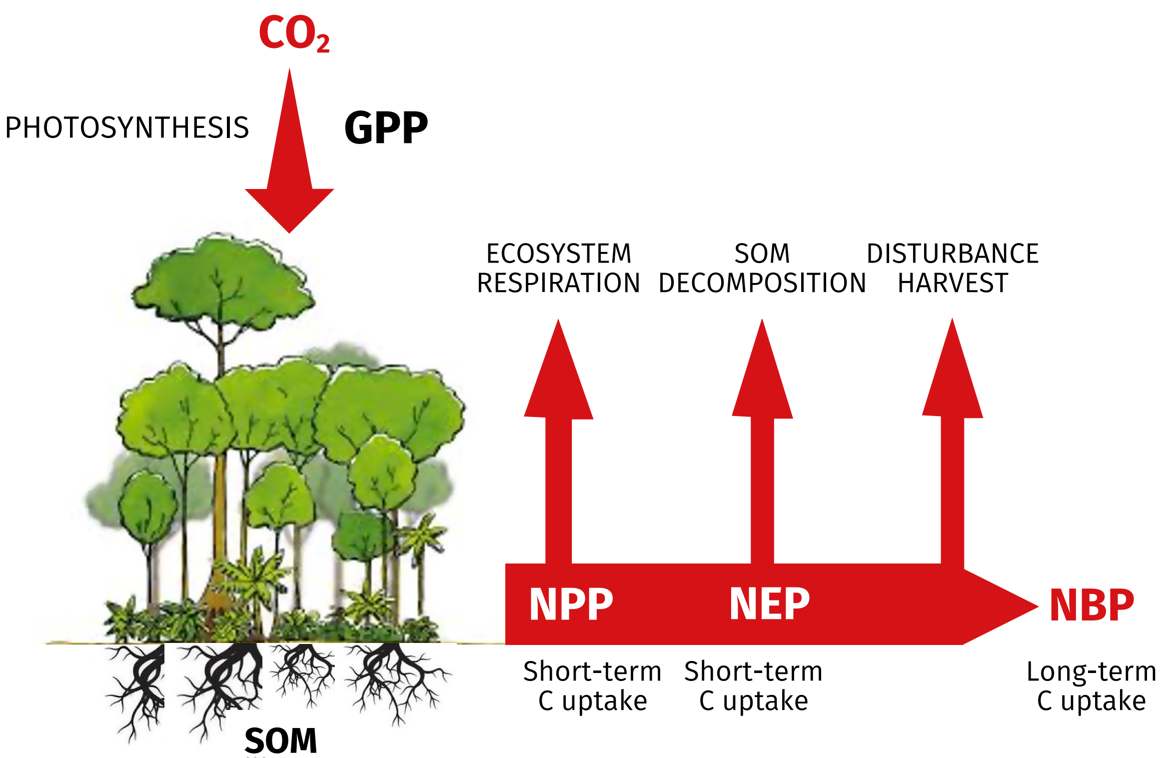


**Figure 1.** Forest carbon balance showing main entrance of carbon through photosynthesis (gross primary productivity) and losses determining the long-term carbon uptake. Gross primary production (GPP) is the amount of carbon captured by plants through photosynthesis over a given period (Pan et al. 2011). GPP minus plant respiration is net primary productivity (NPP). NPP minus soil heterotrophic respiration (primarily resulting from the decomposition of soil organic matter by soil biota) determines net ecosystem productivity (NEP). Moreover, the net carbon sink strength of a forest depends on the occurrence of natural disturbances and tree mortality. This carbon balance is defined as net biome productivity (NBP). NBP also includes loss of carbon through harvest and management activities, when carbon stored in living biomass is transferred to deadwood or harvested wood products. Source: Adapted from IPCC 2000.

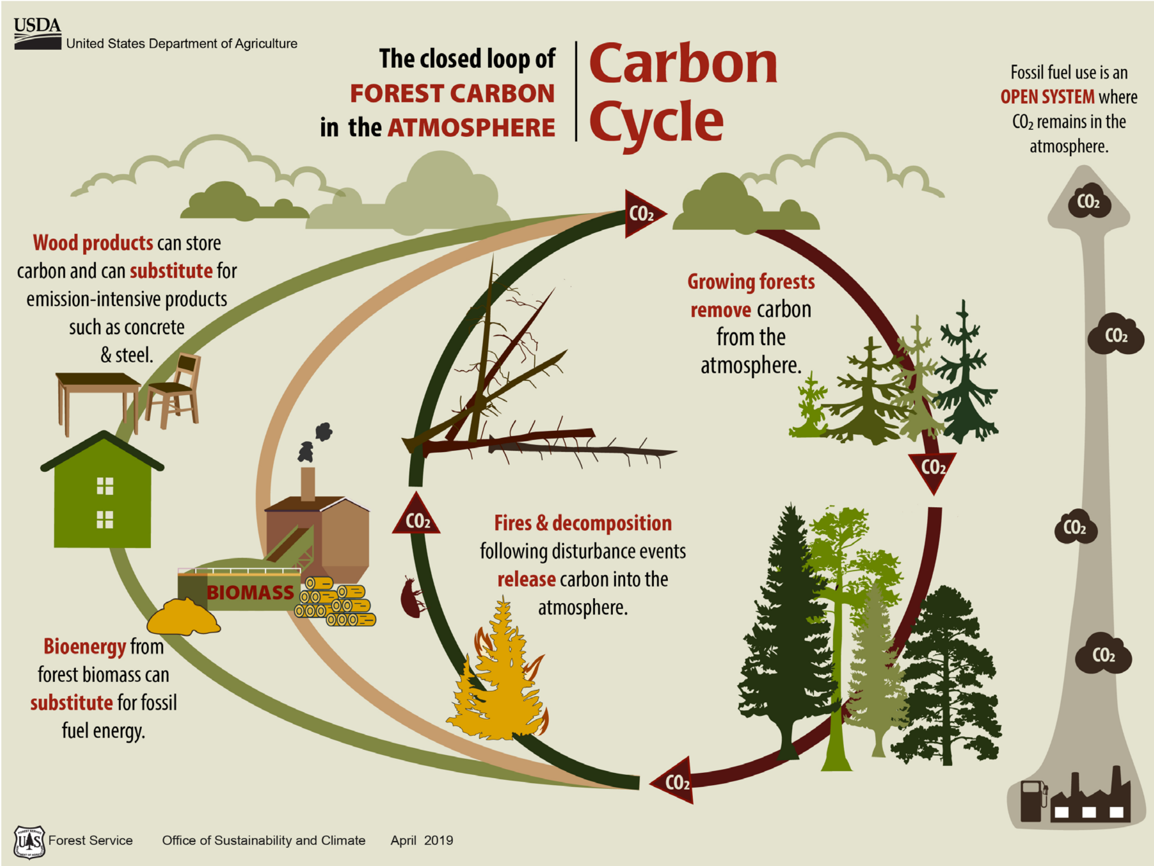


**Figure 2.** Forest carbon cycle with forest management. Source: USDA 2019[1].

The use of timber products will determine whether the carbon stored in wood will be returned to the atmosphere through decomposition or combustion or stay stored in wood products for longer times. Thus, sustainable carbon management must also consider the use of wood products, which in many cases will determine the net sink capacity of managed forests (Figure 2).

Currently, forests and wood products remove approximately 380 Mt$CO_2$ eq per year, which compensates for about 10% of EU-27's total GHG emissions (Verkerk et al. 2022).

Besides carbon sequestration, forest ecosystems harbour biodiversity and provide many other important ecosystem services that are vital to society and human wellbeing. These include provision of timber and non-wood forest products, soil formation and protection against erosion, water purification and retention, local climate regulation, and provision of recreational use (Thompson et al. 2014). The concept of carbon farming accounts for these services and includes provisions to avoid any harm to these "*natural values*".

Despite much research over the last decades (Keenan et al. 2013; Kutsch and Kolari 2015; Hyyrynen et al. 2023), the role of forest management on long-term carbon sequestration potential remains uncertain. Consequently, predictions are contradictory among existing models, which disagree on whether the carbon balance of global forests will be positive or negative in 2100 (Austin et al. 2020; Beillouin et al. 2022). It is, therefore, generally challenging to derive robust conclusions about the potential of carbon farming practices in the long term.

1 www.fs.usda.gov/managing-land/sustainability-and-climate/carbon

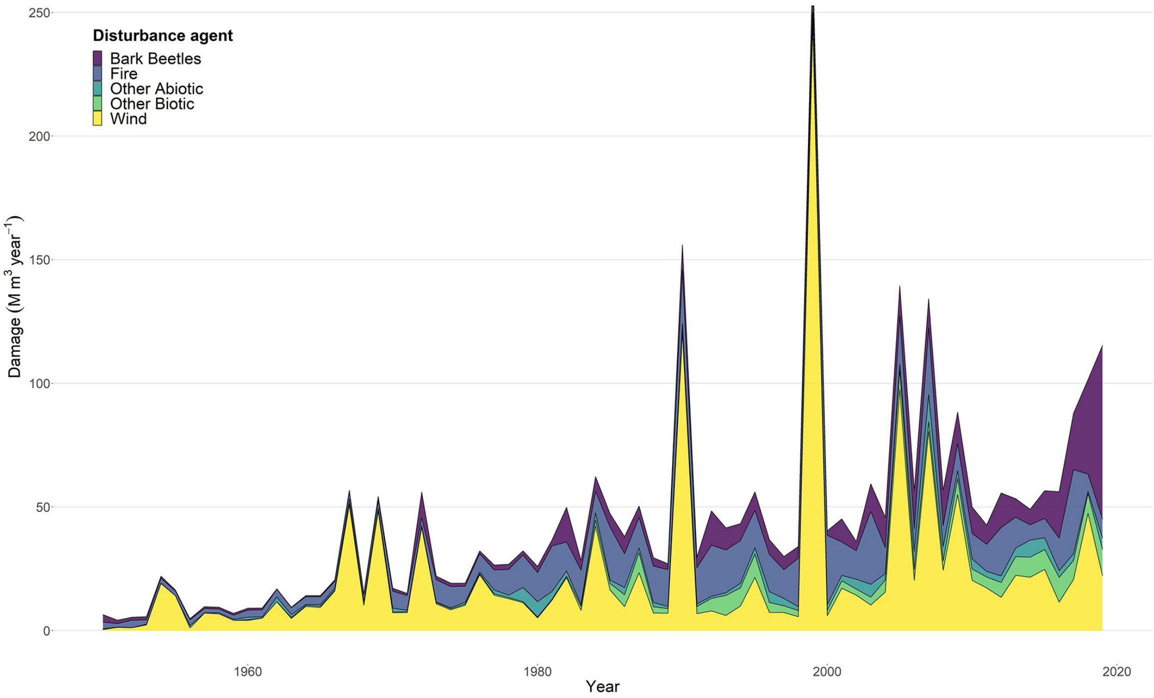


**Figure 3.** Disturbance trends in Europe over the last decades. Source: Patacca et al. 2023.

A major uncertainty is how carbon farming interacts with forest disturbances (such as pest outbreaks, fires, winds). Indeed, the observed positive trend in the carbon sink capacity of European forests has declined in the last decades and is predicted to decrease even more in the future as a result of climate change (Seidl et al. 2017; Senf and Seidl 2021; Roebroek et al. 2023). Disturbances have already increased over the last decades in Europe (Patacca et al. 2023) compromising forest health and carbon sequestration capacity (Figure 3). Several studies point at a reduction in the forest net carbon sink capacity and even foresee a transition to a net forest $CO_2$ source (Peñuelas et al. 2017; Wang et al. 2021) with the most vulnerable regions located in southern and northern Europe (Forzieri et al. 2021).

## 1.3 Soils are key carbon stocks

**Soils are the largest terrestrial carbon reservoir** (IPCC 2021), with European soils hosting more than half of the forest carbon pools (Figure 4. However, in Europe more than 60–70% of soils are degraded as a direct result of unsustainable management practices (EC 2020), and are a net emitter of $CO_2$ (EEA 2022a). Improved soil management for carbon sequestration is therefore particularly relevant for any carbon farming practices.

European soils (down to 1m), including forest floor, mineral and peat soils, store on average 22, 108 and 578 tons of soil organic carbon (SOC) per hectare, respectively (De Vos et al. 2015). Climate-friendly soil management practices, including rewetting of organic soils and changes in land management practices and/or land use, offer an annual mitigation potential estimated at 71–115 Mt $CO_2$eq (Frelih-Larsen et al. 2022). Such practices would turn EU soils from their current status as a net emitter of around 64 Mt $CO_2$eq yr$^{-1}$ to a net carbon sink (EEA 2022b).

Moreover, a wide adoption of carbon farming practices, such as peatland restoration, agroforestry, or substituting fodder crops with grass, could additionally mitigate 150–350 Mt $CO_2$eq yr$^{-1}$ by 2050 for mineral and organic soils combined (Bellassen et al. 2022). These carbon farming practices should aim to sequester carbon in the soil or reduce soil carbon losses and GHG emissions and have important co-benefits.

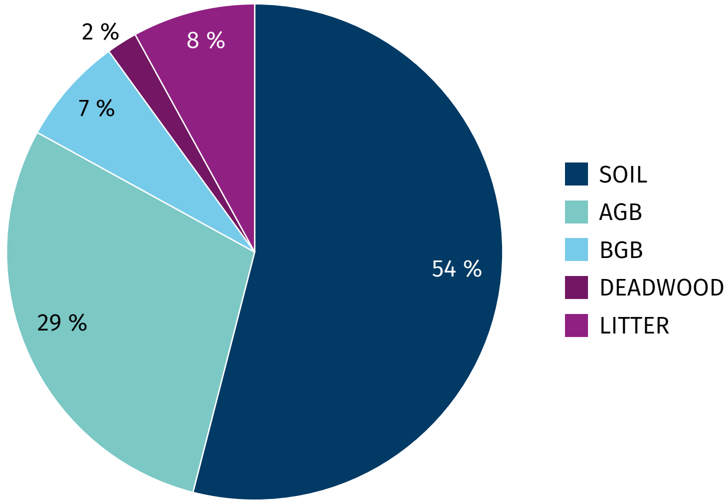


**Figure 4.** Proportion of forest carbon pools in Europe 2020: ABG, above-ground biomass; BGB, below-ground tree biomass. Data source: Forest Europe 2020.

However, trade-offs can occur, including an increase in non-$CO_2$ GHG emissions, with negative consequences for biodiversity and food production in some cases. Thus, careful consideration of co-impacts should be considered when planning and implementing different practices.

## 1.4 Achieving EU's climate targets

To achieve climate neutrality, the EU relies on the *Land Use, Land Use Change and Forestry* (LULUCF) sector to compensate for the hard-to-abate residual emissions by 2050, with the LULUCF sector required to achieve a target of net removals of -310 Mt of $CO_2$-eq $yr^{-1}$ by 2030 (EU 2018; EU 2023a). According to the European Commission scenarios for 2050, a net sink of 333 Mt $CO_2$ eq $yr^{-1}$ by 2050 is expected (EC 2024b). Since the current net uptake is estimated to be around 286 Mt $CO_2$ eq $yr^1$ (EEA 2023), the LULUCF sector, and forests, need to be managed to increase their capacity to absorb $CO_2$ from the atmosphere (Figure 5).

However, the expected contribution of the LULUCF sector to reach climate neutrality should not be taken for granted, given that the capacity of European forests to sequester carbon has decreased over the last decade and is expected to decrease even more by 2050 because of increasing harvest along with forest ageing and climate change (Forzieri et al. 2021). Moreover, the positive impact of tree planting on carbon sequestration at EU level can be counteracted by direct radiative impacts, emissions from the extraction and use of wood products or non-suitable site and climatic conditions (Kirschbaum et al. 2024).

The need to reduce GHG emissions by replacing fossil fuels and energy intensive construction materials, such as cement, with other materials, can lead to increasing wood demand, representing further pressures that may counteract the net carbon sequestration potential of European forests (Wernick et al. 2021; Korosuo et al. 2023). A recent analysis shows that European energy policies aimed at replacing fossil fuels with biofuels might be compromising carbon storage in forests in some cases (Searchinger et al. 2022).

Thus, how effectively carbon farming practices in the forestry sector can generate carbon removals is debated (Linser et al. 2018; Roebroek et al. 2023). The forestry sector can contribute to climate change mitigation by avoiding deforestation and forest degradation and by adapting forest management to enhance carbon stocks in biomass and soils (IPCC 2019, 2022).

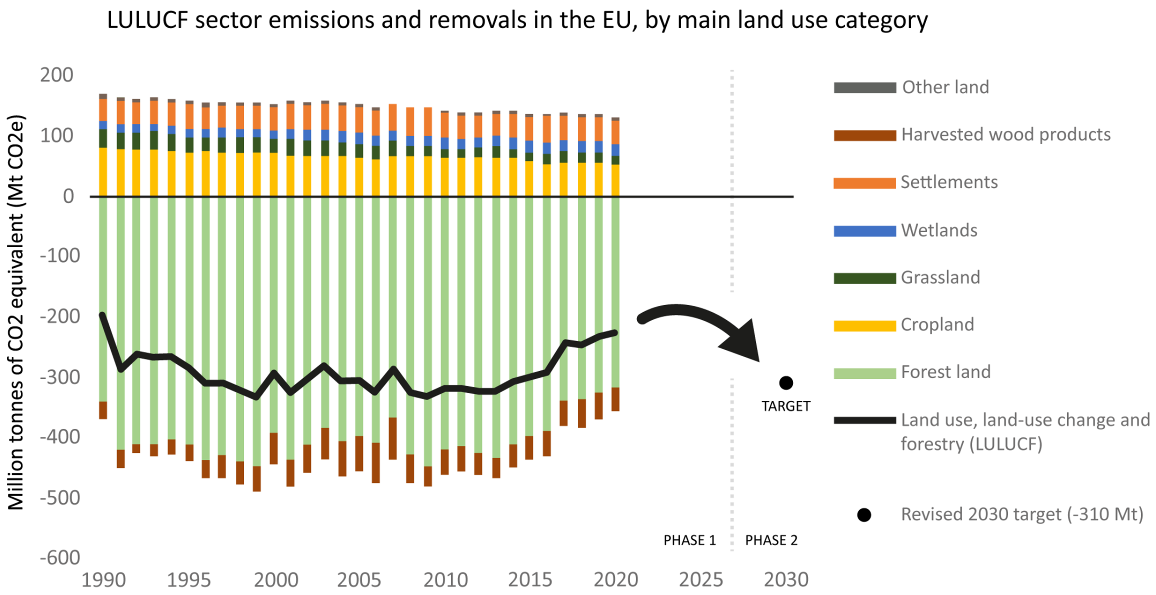


**Figure 5.** LULUCF sector emissions and removal in the EU by land use category. Source: EEA 2023.

Therefore, in principle, there is potential for carbon farming practices through adapted forest management activities (Erb et al. 2018; Sha et al. 2022) with potential co-benefits for ecosystem restoration and biodiversity conservation. Their applicability and performance strongly depend on forest types and site conditions, as well as on national management systems. For this reason, it is necessary to **analyse the potential of different forest management practices that can contribute to carbon farming in European forests**.

# 2 Forest management practices

European forests cover more than 159 million hectares, which is approximately 38.6% of the EU area (Eurostat 2023a). Several different forest biomes are present in Europe, reflecting its climatic and pedoclimatic diversity (such as boreal, alpine forests with conifers, temperate broadleaved and Mediterranean). Most European forests are privately owned (60% of forested land) and the rest are publicly owned (40%).

**Forest management practices vary substantially across Europe**, from places in which conservation legislation precludes all forest management activities to intensive short-rotation monoculture forests for energy-related biomass production. Europe has been a net exporter of wood products in the past four years with more than 80% of the forest area managed for timber extraction (Eurostat 2023b). Nevertheless, only 10% of the total forest area of Europe is intensively managed and an increasing proportion (currently 30%) is managed as multiple-use forest (Forest Europe 2020).

Past management strategies were mostly designed to maximise timber production while recent EU-level policies are directed toward sustainable and climate-resilient forests (FAO and PlanBleu 2018; Forest Europe 2020). **Sustainable forest management practices not only involve carbon storage but biodiversity conservation, hydrogeological protection, and social and economic aspects** (Siry et al. 2005). These benefits are considered in the European definition of carbon farming and are often called *forest ecoservices* (Deal et al. 2017; Diaz-Balteiro 2017) and are included in the European climate agenda (*Nature Restoration Law* 2024[2]). Approximately 30-70% of most major European forest taxa have a "poor" or even lower conservation status (Muys et al. 2022). One of the main co-benefits of carbon farming activities is to enhance ecosystem sustainability, including biodiversity conservation. This was also identified by the United Nations, which recognised this decade as the *Decade on Restoration* (www.decadeonrestoration.org).

This chapter examines the scientific evidence on how forest management practices can contribute to carbon farming. To assess how European forests can be managed to increase their carbon sequestration potential and simultaneously fulfil other functions, as well as adapt to climate change, we conducted a literature review. We reviewed publications from the last 10 years (2013–2023) concerning various management practices and reporting carbon accumulation rates in biomass and/or forest soils (for methods and table of studies see Annex 1).

The forest management practices examined in this chapter include afforestation, silvicultural practices, diversification of species and forest structure, site fertilisation, agroforestry and peatland management.

Table 1 summarises the results that are further explained in the text.

2 www.europarl.europa.eu/pdfs/news/expert/2024/2/press_release/20240223IPR18078/20240223IPR18078_en.pdf

**Table 1.** Main carbon farming practices and co-benefits

| PRAC-TICES | Afforestation | Silvicultural practices | Diversification of species and forest structure | Site Fertilisation | Agroforestry | Peatland Management |
|---|---|---|---|---|---|---|
| **Carbon farming activi-ties** | Plantation of forests on land that have not been forested within the last 50 years:<br>• From cropland to forest<br>• From grassland to forest | • Tree species selection<br>• Extending rotation period<br>• Reduced harvest intensity | • Conversion of cop-pice to high forests<br>• Continuous forest cover<br>• No harvesting | • Liming<br>• Wood ash | • Establishment of agrosilvopastoral systems<br>• Management options | • Ash fertilisation<br>• Water table management trough CCF<br>• Peatland restoration (rewetting) |
| **Zone** | Boreal, Temperate, Mediter-ranean | Boreal, Temperate, Mediterranean | Temperate, Mediter-ranean | Boreal, Temperate | Temperate, Mediter-ranean | Boreal, Temperate |
| **Type of activity** | Carbon sequestration | Carbon sequestration | Carbon sequestration | Carbon sequestra-tion | Carbon sequestra-tion | Carbon sequestration<br>Emission reduction |
| **Impact on AGB** | • Positive | • Positive on the long term | • Positive | • Positive | • Positive | |
| **Impact on SOC** | • Positive on cropland in the long term<br>• Possible loss of carbon on grassland | • Null in the short term<br>• Positive in the long term | • Null | • Null | • Positive | • Positive |
| **Co-benefits** | • Improved soil health<br>• Increased biodiversity<br>• Increase water<br>• Reduce erosion<br>• Potential income<br>• Improved microclimate<br>• Climate adaptation<br>• Biodiversity (on cropland only) | • Greater stability<br>• Improves resilience<br>• Reduced risk of disturbances<br>• Increase biodiversity<br>• Potential income<br>• Improves forest structure<br>• Regenerates forests<br>• Reduce impact of drought<br>• Reduced pest attacks<br>• Fire prevention | • Improved soil health<br>• Improved water<br>• Biodiversity<br>• Reduce erosion<br>• Increase soil fertility<br>• Aesthetics | • Increase soil fertility<br>• Potential benefit for biodiversity<br>• Potential reduc-tion of GHG | • Improved soil health<br>• Improved water retention<br>• Biodiversity<br>• Climate adaptation<br>• Diversification of revenues<br>• Improved micro-climate | • Enhance soil fertility<br>• Increase tree growth in low fertile soils<br>• Reduction in GHG emissions<br>• Improved biodiversity<br>• Reduced nutrient leaching with CCF<br>• higher economic benefits to landowners<br>• Water quality |
| **Risks** | • Increase of fire risk<br>• Biodiversity loss in grass-lands<br>• Use of non-native species<br>• Loss of land for food pro-duction | • Possible SOC losses connected to harvesting operations<br>• Impact on wildlife<br>• Very long periods decrease growth<br>• Pests attacks<br>• Fire damage<br>• Soil erosion | • Need for conserva-tion measures | • Nutrient runoff<br>• Heavy metal accumulation | • Biodiversity loss in grasslands<br>• Use of non-native species | • Methane emissions<br>• Decrease in production in fertile soils<br>• Nutrient runoff when applied near water ways |
| **Limita-tions** | • Scarcity of non-forested land<br>• Displacement of produc-tion | • Increase costs in rugged terrain<br>• Non-economic revenues from products | • Non-economic revenues<br>• from products | • Increased costs<br>• Growth decrease in time | | • In some cases CFF is not enough and needs to be combined with hydrological measures<br>• Displacement of production |

## 2.1 Afforestation

Afforestation on former croplands and grasslands holds promise as a climate change mitigation strategy with long-term benefits, and, in principle, it is clearly a carbon farming activity. In addition to sequestering carbon in both, soil and above-ground biomass, afforestation can provide many other environmental co-benefits. Depending on previous land use, local climate, stand age and tree species, estimates for afforestation range 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 of more recent studies, carbon sequestration rates resulting from afforestation in Europe can be even higher in some cases, ranging from 2 to 35 Mg $CO_2$ $ha^{-1}$ $yr^{-1}$ (see Figure 6).

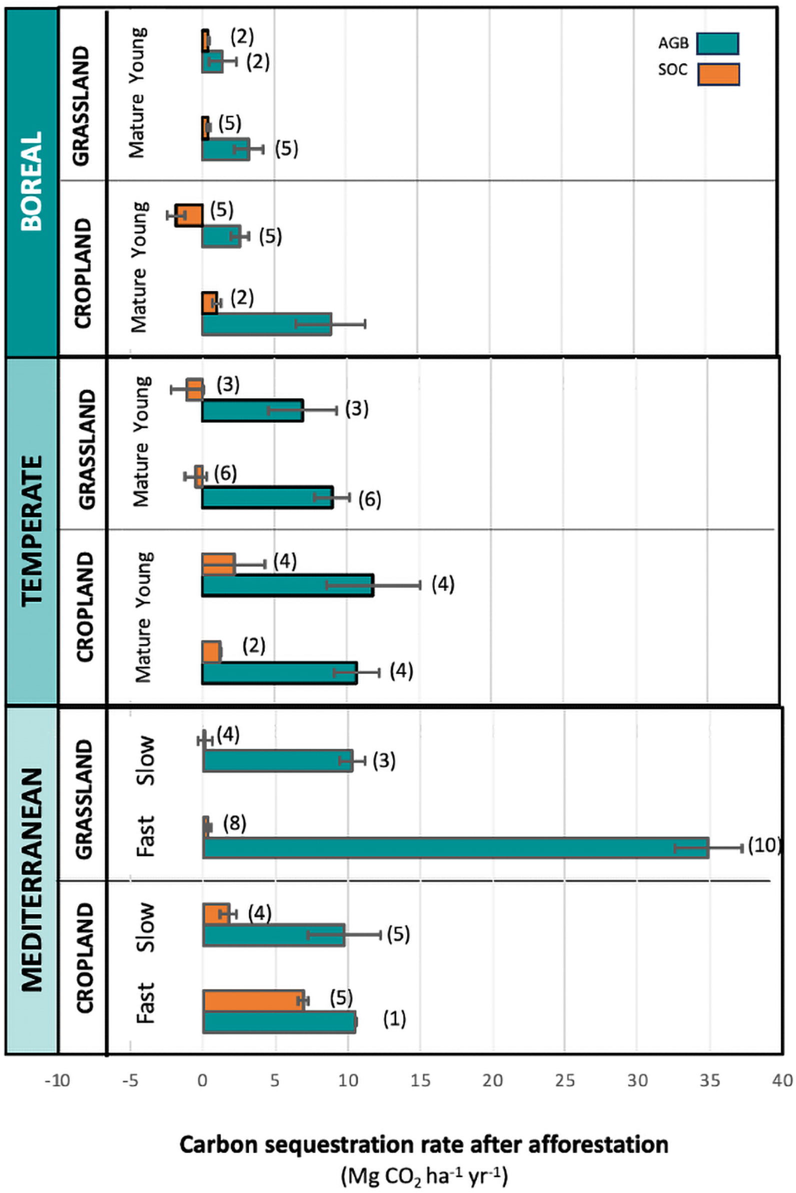


**Figure 6.** Carbon sequestration rates in tree above ground biomass (AGB) and soil organic carbon (SOC) of afforestation on grassland and croplands based on the literature review for different regions in Europe: boreal, temperate and Mediterranean. We have divided the studies into mature and young forest stands (more or less than 25 years old) for boreal and temperate, and in plantations (fast growing species) and native (slow growing species) in the Mediterranean. Bars represent the mean plus the standard error considering the number of case studies reported in brackets. Source: authors' own calculations.

The very high sequestration values correspond to intensive plantations of fast-growing species used in the Mediterranean countries. Excluding these very fast-growing species, afforestation promotes carbon sequestration in biomass in all biomes with an average rate of 5–10 Mg $CO_2$ $ha^{-1}$ $yr^{-1}$, being lowest in the boreal zone. Carbon sequestration rates in soils are much lower as it takes much longer to accumulate.

In **boreal regions**, afforestation on bare fallow using conifers may induce small carbon sequestration or even soil carbon losses in the short term (Tupek et al. 2021), while in the long term, an increase is likely for both soil carbon and above-ground biomass. In this northern region, the impact of afforestation on former grasslands differs from that on croplands, where soil carbon often increases shortly after afforestation (Figure 6).

In contrast, in **temperate regions**, afforestation of agricultural land induces carbon sequestration mainly in new tree biomass, while changes in soil carbon are less obvious and vary depending on previous agricultural use but soil carbon losses are rarely observed (Mayer et al. 2020).

A similar trend in carbon sequestration is observed in **Mediterranean areas**, where fast-growing tree species (such as *Eucalyptus* and *Populus*), known for their high potential for carbon sequestration, are frequently used. The impact of these plantations needs to be evaluated at landscape and local scales while also considering socioeconomic aspects. For example, there may be trade-offs with other ecosystem services such as water provision, soil properties and fire risk as well as biodiversity preservation. In Mediterranean areas, the response of grassland soils to afforestation is variable, with soil carbon losses in humid areas sometimes offsetting decades of biomass carbon accumulation. However, in dry areas, there is a positive response in terms of soil carbon sequestration as shown in Figure 6. Particularly promising is the impact on soil carbon of afforestation on mine spoil banks using native species, and even more if nitrogen-fixing species are used (Box 1).

### Box 1. Afforestation of degraded soils.

Post-mining sites offer an excellent opportunity for afforestation with potential optimum results in soils given that presumably these soils are far from saturation. Initial rate of C sequestration in post-mining sites can reach up to 2.4 Mg C ha-1yr$^{-1}$ while afforestation of agriculture land closed to saturation is about -0.3 Mg C $ha^{-1}yr^{-1}$.

- Afforestation of heavily degraded soils where soil C stock is far from saturation would speed up C storage in coming decades.
- Using trees producing litter with low CN ratio (N fixers) will speed up initial C sequestration but soils will reach saturation sooner. The opposite pattern is expected in trees producing litter with high CN ratio (conifers).
- 0.5% of EU land is affected by mining and quarrying (2.1 million ha). 35 million hectares are affected by moderate to severe erosion, which is 17.8% of the European arable soils. Focusing on most severe affected areas (-5 million ha) would produce the fastest carbon accumulation in relative terms.

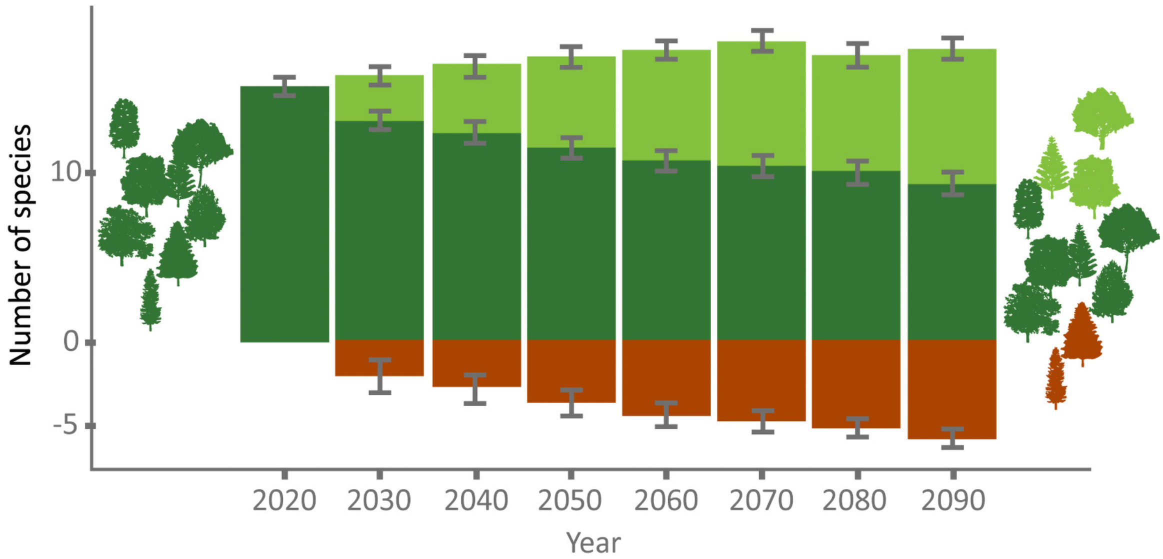


**Figure 7.** Average number of tree species climatically suitable across Europe according to Wessely et al. (2024). Dark green bars are suitable tree species for planting considering future climate, light green number of tree species that will become suitable in the future according to current climate predictions (IPCC, RCP 4.5) and brown ones show the number of species lost in this decade.

In **central Europe**, a significant amount of alpine pasture areas are being abandoned. Afforestation of such alpine pastures or unmanaged tree line areas could be promoted to accelerate the carbon sequestration process compared to the much slower natural forest succession, thus ensuring "*additionality*." In the long term, a significant sequestration rate (~3.5 Mg $CO_2$ $ha^{-1}$ $yr^{-1}$) can be achieved, with most of the additional carbon stored in new tree biomass (Hiltbrunner et al. 2013; Speckert et al. 2023). However, high altitude afforestation is demanding in terms of time, labour and cost and, in the case of alpine meadows, could negatively affect biodiversity.

In **afforestation plans**, considering the "*Do no significant harm*" principle, priority should be given to native tree species adapted not only to current climate conditions but to future climate. However, in certain cases, introducing non-native, non-invasive species can make sense, if they are better adapted to future climatic conditions. A significant challenge for afforestation is the identification of suitable areas as well as tree species.

A recent species distribution modelling analysis of the suitability of tree species in Europe for future climate shows that the number of species is smaller than under current conditions and, thus, there is a limited choice for forest management (Figure 7, Wessely et al. 2024). This bottleneck could negatively impact timber production, carbon storage and biodiversity conservation (McFadden 2024).

In view of food security and the land demand for other uses (for example, urbanisation, solar power generation) across Europe, it remains **questionable how much agricultural land can be used for afforestation** (Van de Ven et al. 2021; Mo et al. 2023; Zheng et al. 2023). Urban and peri-urban areas (Box 2) emerge as particularly effective sites for afforestation, as they not only contribute to climate mitigation but also generate a range of additional ecosystem services, such as improved air quality, water retention, mitigation of extreme events, cooling, increased biodiversity and recreational services (Haase et al. 2014).

### Box 2. Trees outside forests.

Forest inventories in most countries record closed canopy forests (Tomppo et al. 2008). However, in many European countries, in particular northern countries, a substantial number of trees can be found in hedgerows, gardens, parks, urban areas, grasslands and agricultural lands that are not included in national-level carbon stocks. Since many European countries comprise large agricultural and urban landscapes, the exclusion of trees outside forests from systematic carbon stock and uptake assessments will potentially underestimate national inventories and affect climate models. Indeed, a recent estimation of the contribution of trees outside forests, based on high-resolution nanosatellite imagery across Europe, estimated a contribution of 0.8 petagrams of carbon in Europe (Liu et al. 2023). Although this represents only 2% of the total tree biomass of national inventories, in some countries such as UK it can reach up to 10% and, for the Netherlands, trees in urban areas contribute 8% to national carbon stocks.

Moreover, beyond carbon sequestration, trees outside forests may have important benefits by protecting land, providing resources for local communities, regulating the local climate, contributing to habitat networks, affecting the hydrological cycle and improving air quality and so represent an important economic and social value (Thomas et al. 2021). Thus, planting in urban areas, parks and agricultural lands may be also considered as a potential carbon farming practice that should be incentivised.

**To sum up**, although afforestation results in carbon sequestration in biomass, the **impact on soils depends on the previous land use and occurs only after several years**. Moreover, in some cases, it can lead to losses in biodiversity (such as in the case of former grasslands) or even increase the risk of disturbances such as forest fires in the southern part of Europe (Stevens and Bond 2024). Therefore, afforestation projects should be assessed with caution.

## 2.2 Silvicultural practices

Carbon sequestration can be promoted directly and indirectly through silvicultural measures. The potential for indirect promotion of carbon sequestration is present in most silvicultural practices, as the promotion of site-adapted, vigorous, long-lived trees and disturbance-resistant and resilient stands is consistent with most management objectives.

The **additionality of indirect measures** is more difficult to verify. For example, guidelines for selection of trees for felling can include recommendations for carbon farming. The measures presented below represent direct contributions to carbon sequestration and can be often verified using forest databases. This is particularly the case of countries where detailed forest inventories and ongoing forest management planning are regularly carried out (see Forest Europe 2020).

### 2.2.1 Tree species selection

The effects of **changes in tree species composition** on carbon sequestration are difficult to quantify because they **depend heavily on the species** used. Slower growing species (such as deciduous species) may have higher wood densities and therefore will not necessarily offer lower carbon sequestration potentials than faster growing coniferous species. For example, Förster et al. (2021) found that Scots pine stands in northern Germany had a lower carbon sequestration potential than naturally developing beech forests.

Furthermore, **adapting tree species composition** can have diverging effects on carbon sequestration in tree biomass and soil, and the effects on soil carbon may also depend on the mycorrhizal symbiont (reviewed in Mayer et al. 2020, and in Schindlbacher et al. 2022). Moreover, current tree species-specific carbon sequestration rates may change in the future depending on individual tree species responses (growth, mortality) to global change (Kasper et al. 2022; Diers et al. 2023). Vospernik et al. (2024), for instance, showed that mixing oak and pine can mitigate, but not fully compensate, for productivity losses by climate change.

Under certain conditions, **promoting mixed species stands** could be a possible carbon farming practice by adapting forests to climate change and at the same time increasing productivity and carbon sequestration (Augusto and Boča 2022). In addition to a possible higher productivity and the provision of other ecosystem services (Huuskonen et al. 2021), more diverse forest stands often show greater stability (Pretzsch et al. 2015; Liang et al. 2016; et al. 2016a). Species mixing can further **increase resistance and resilience to extreme drought** (Pretzsch et al. 2013) and reduce the risk of biotic and abiotic disturbances (Guyot et al. 2016).

**Tree species richness** also often **positively affects soil carbon** through changes in litter quality, nitrogen fixation and rooting patterns (Díaz-Pinés et al. 2011). However, a recent literature review (Huuskonen et al. 2021) suggests that mixing **tree species can promote growth and carbon sequestration in central and southern Europe**, while **no positive effect** was evident **in northern boreal forests**.

In general, the **'additionality'** generated by mixing tree species would be difficult to identify/verify and it can be context specific. Conversion of monocultures to mixed stands is also considered a climate adaptation measure.

Overall, a **change of tree species** dominance by forest management is a **long-lasting process and can result in initial carbon losses**. Carbon sequestration at the forest scale can only be expected in the long term.

- Therefore, **species conversion or species mixing** may not apply in carbon crediting schemes which aim at short timeframes (for example, five years), but only in long-term approaches of a decade or more.
- With regard of **tree species selection**, it generally would be advisable to avoid implementing carbon market mechanisms that incentivise the use of fast growing, but climate un-adapted tree species instead of slow growing, but better climate-adapted tree species.

### 2.2.2 Extending rotation period

Extending rotation periods (the number of years between complete harvest cycles) have traditionally been defined by economic objectives, neglecting the potential of mature stands to sustain tree growth and carbon sequestration. Extending the rotation period is therefore proposed as a **strategy to increase carbon sequestration**, and is thus a potential carbon farming practice.

Here are some examples:

In **Norway**, Stockland et al (2012) investigated the impact of extending rotation periods from 100 to 120 years in Scots pine and Norway spruce stands, finding an increase in biomass carbon sequestration ranging from 2.1 Mg $CO_2$ $ha^{-1}$ $yr^{-1}$ to 8.1 Mg $CO_2$ $ha^{-1}$ $yr^{-1}$.

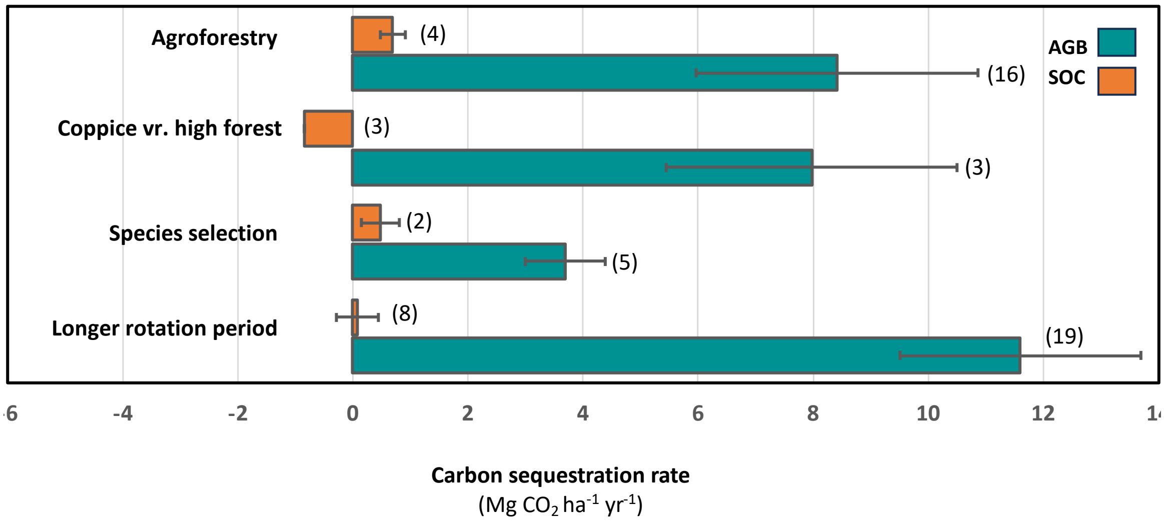


**Figure 8.** Carbon sequestration rates in tree above-ground biomass (AGB) and soils (SOC) of different practices in the **Mediterranean region**. Bars represent the mean plus the standard error of the number of studies reported in brackets.

Trivino et al (2017) explored optimal management strategies for a landscape in **Central Finland**, observing that longer rotation periods could lead to an increase in annual carbon sinks by 1.5 Mg $CO_2$ ha$^{-1}$ yr$^{-1}$ to 2.9 Mg $CO_2$ ha$^{-1}$ yr$^{-1}$ over 10 and 30-year periods, respectively.

In a **pan-European** modelling simulation with various tree species, Kaipainen et al (2004) suggest an additional biomass carbon sequestration between 2.1 and 4.4 Mg $CO_2$ ha$^{-1}$ yr$^{-1}$ for a 20-year increase in rotation length.

For **Mediterranean ecosystems**, our literature review also shows high potential for additional biomass carbon sequestration (on average ~12 Mg $CO_2$ ha$^{-1}$ yr$^{-1}$; Figure 8) by prolonging rotations, although this was partly achieved in fast-growing non-native tree plantations, which are often unsuitable for carbon farming due to their adverse effects on other ecosystem functions.

In general, longer rotation periods allow for **significant carbon sequestration only during a transition period until the desired (longer) rotation period is reached** and the forest landscape (management unit) enters a new stable state. This time frame usually extends over a few decades and can therefore, at best, buy time for the implementation of measures to reduce fossil $CO_2$ emissions.

As **litter production** is positively related to above-ground biomass (Jevon et al. 2022), higher carbon input in older stands with delayed harvests likely positively affects soil carbon stocks (Feng et al. 2022a). Modelling suggests a strong influence of tree species and climate on potential soil carbon sequestration, for example between zero and 2 Mg $CO_2$ ha$^{-1}$ yr$^{-1}$ for prolonging rotation by 20 years across European sites (Kaipainen et al. 2004). Such positive indirect effects on soil carbon are likely to be longer lasting than the direct effects on forest biomass.

Extending rotation periods offers **additional ecosystem co-benefits**, such as promoting biodiversity and increased recreational service in forested areas (Baskent and Kaspar 2023). However, when considering forest management strategies with longer rotation periods, it becomes crucial to assess **potential disturbance risks**. Areas prone to high levels of disturbances may not be suitable candidates for longer rotation periods. Furthermore, extending the rotation period may **reduce the timber harvest and, consequently, lead to leakage** (see Chapter 4). Therefore, careful attention must be given to balancing the advantages of extended rotations with the associated risks and suitability of the environment.

### 2.2.3 Reduced harvest intensity

Reduced harvesting **increases the proportion of older larger trees**, and hence forest biomass carbon stocks over time.

Under **even-aged** forest management, harvest can be reduced during stand thinning and during final harvest. As with even-aged forestry, reduced harvesting can also increase biomass carbon stocks in uneven-aged or continuous cover forestry.

Studies and experience show that it is possible to achieve sustainability of **uneven-aged** stands with different levels of growing stock, depending on the management objective (Schütz 2002). Many uneven-aged stands are managed with higher growing stocks, as required for sustainability for objectives other than carbon farming (for example, for forest regeneration), serving as examples of best practice for other owners (see examples in Krumm et al. 2020).

However, in both even-aged and uneven-aged forests, the permanence of carbon sequestration depends on the **local disturbance** regime **and** the disturbance **resilience** of the forest.

Larger trees are generally more vulnerable to disturbances such as storms, fire, drought or bark beetles (Brienen et al. 2020; Korolyova et al. 2022). Therefore, increasing the proportion of **mature trees** also **increases the vulnerability** to disturbance-related carbon losses. Such losses can easily outweigh or even overcompensate for carbon gains, especially if disturbance increases in the future (Senf and Seidl 2021; Brèteau-Amores et al. 2023). A related issue is the risk of **leakage**, where reducing the intensity of timber harvesting in one area might simply shift the demand to another location, leading to potential deforestation or degradation elsewhere (Kallio and Solberg 2018).

An **extreme approach** is to **halt harvesting** or to **abandon forest management** altogether. However, this poses serious **risk of leakage** besides all the economic and ecological consequences. Forest management abandonment has been put forward as a climate change mitigation strategy with **biodiversity co-benefits** (Nagel et al. 2023; Langridge et al. 2023).

An argument for management abandonment is that **mature forests store more carbon** in biomass and soils than managed forests; i.e., forest dynamic models predict highest carbon sequestration in tree biomass of unmanaged forests (see, for example, Seidl et al. 2007; Krug 2019; Schwaiger et al. 2019; Štraus et al. 2023). Stopping tree harvesting would also allow for the build-up of a significant **deadwood carbon pool** in the longer term (Schulze et al. 2020). This can be beneficial for additional carbon sequestration, but the long-term benefits for mitigation can be context specific. For example, in fire prone and dry ecosystems such as the Mediterranean, build-up of deadwood could significantly increase the risk of fire (Mantero et al. 2023) and lead to substantial unintended carbon losses. Yet, under specific, stable conditions (such as in slow-growing subalpine pine forests or productive beech forests), non-management can simultaneously enhance forest climatic resilience and mitigation potential (Jandl et al. 2019).

While forest management cessation can be an adequate carbon farming practice in very specific contexts, **in general it is not a valid mitigation strategy** for many European forests. It can increase the risk of disturbance, particularly in the context of climate change, and therefore carry a high risk of carbon release. In **some Central European countries**, **legislation** requires certain management measures, such as preventive measures in the case of pest infestation (removal of infested trees) or active regeneration and maintenance of protection forests, which makes it difficult to abandon some degree of active forest management.

### 2.2.4 Conversion of coppice to high forests

Coppicing represents one of the oldest forms of systematic and sustainable use of forests. It is a very flexible system requiring low energy input to deliver small size wood primarily for energy (for example, firewood and charcoal), agriculture and small-scale businesses. Coppice forests characterise the mountainous areas of **central, east and southern Europe** in particular.

Tree species composition has long been simplified by coppicing, causing species with low stump sprouting ability to disappear from coppice forests (Fabbio 2016). Due to rural abandonment and technical and economic limitations, most of the **coppice forests are neglected or abandoned**, representing a significantly underused natural resource (Unrau et al. 2018). Coppice forests are important for biodiversity because of the native tree species, the continuity of the forest cover over time, the diversity of forest structures and the age of the trees, which is why this form of management is promoted in some countries (Kirby et al. 2017).

Under specific preconditions, the **conversion of coppices into high forests** could be considered a carbon farming practice. Gradual conversion of coppices to high forests can **increase carbon sequestration** in forest biomass on mesophilic sites in the long term, while coppice forests can be more productive on dry sites (Bruckman et al. 2011; Lee et al. 2018). How the conversion of coppice to high stands affects **soil carbon is as yet unclear** (Camponi et al. 2022).

### 2.2.5 Continuous cover forestry

In **Central and Northern Europe**, forest adaptation to climate change is frequently accompanied by a shift from clear-cutting to continuous cover forestry (CCF) and a change in tree species composition from monocultures to mixed forests. In this context, Seidl et al (2007) showed that within a forest unit in the Austrian Alps, CCF can achieve higher carbon sequestration in the longer term than even-aged management with clear-cutting (spruce monocultures, 100-year model simulation).

- **CFC leading to uneven-aged stands** are, for example, less susceptible to disturbance, insects and disease, so unexpected carbon losses are likely to be lower than in even-aged systems (Mohr et al. 2024).

- Another advantage of CCF is the **continuous input of carbon** to the **forest floor**. Clear-cutting can interrupt the input of carbon to the soil and alter the soil microclimate, making the soil and the ecosystem a temporary net source of $CO_2$ (Mayer et al. 2014; Kobler et al. 2015) and of $N_2O$ in the case of drained peatlands (Korkiakoski et al. 2023).

- CCF also provides a constant **input of root carbon** into the soil (see Box 3), maintaining soil functionality, whereas clear-cutting results in pulses of dead root litter carbon into the soil.

- Moreover, it seems that **thinning**, as done in CCF, reduces carbon uptake by forests only marginally and by a few years (Lindroth et al. 2018; Vesala et al. 2005).

Like changing tree species composition, the transition from clear-cutting to closer to nature approaches such as CCF is a long-term process that may initially result in carbon losses from the ecosystem (Hilmers et al. 2020).

Quantifying additional $CO_2$ sequestration is difficult and requires **long time horizons**, especially to detect any changes in soil carbon. However, the effects on biodiversity, including soils, and, in turn, on forest productivity should be more immediate (Box 3).

### Box 3. Soil benefits of continuous cover forestry.

Around half the carbon fixed by trees is transferred below ground to maintain and grow roots responsible for water and nutrient uptake. A significant proportion of this is exuded through the rhizosphere into the soil where it fuels soil microbiota responsible for nutrient cycling.

Recognition of the key role of soils in the global C cycle has fueled a scientific effort to understand the processes leading to the formation and retention of soil organic matter, and the roles of the soil biota in these processes. According to Prescott and Grayston (2023) rhizodeposition is recognised as fundamental for sustaining life below ground and replenishing SOM demands. The biological transformation of exudates into microbial metabolites and necromass is a major source of soil organic matter (SOM). Forest management can strongly affect below-ground carbon and rhizodeposition.

Moreover, roots are also essential as they represent important carbon stocks that have longer residence times than above-ground biomass.

1. Clear-cut harvesting affects much of the below-ground forest biodiversity and strongly impedes SOM and C stocks formation.
2. Continuous cover forestry, where trees are retained, sustain and support the re-establishment of below-ground life and function following forest harvest and may mitigate post-harvest soil C losses. Sustaining the below-ground ecosystem via inputs from living roots is an underappreciated benefit of continuous cover and retention forestry.

### 2.2.6 Site fertilisation

Fertilisation of nutrient-poor soils increases tree growth and often slows down the decomposition of soil organic matter. It can therefore accelerate the sequestration of carbon in forest stands (Melikov et al. 2023). In **northern European forests**, tree growth is often nitrogen limited in upland soils and so the addition of nitrogen increases biomass production and facilitates soil carbon build up, especially for the organic layer (Saarsalmi et al. 2014).

In contrast to northern Europe, the use of **nitrogen fertilisers as a carbon farming practice is questionable** in **central Europe**. Atmospheric N inputs have brought many central European forest soils to a state of N saturation and atmospheric N deposition, although recently at lower rates (Schmitz et al. 2024). Fertilisation of N-saturated soils carries the risk of negative side effects such as nitrate leaching to ground and surface waters (Mäkipää et al. 2023). In some central European countries (such as Switzerland, Slovenia and parts of Germany) fertilisation of forest land is therefore prohibited.

Based on the findings from Sullivan et al (2018), forest fertilisation has varying effects on biodiversity depending on the species group. Based on their review, herbs increased in abundance, while bryophytes and dwarf shrubs declined. Fertilisation increased foliage biomass and so, afterwards, species such as mule deer, moose and hares benefited from fertilisation.

### 2.2.7 Liming

Liming of forest soils is primarily aimed at **reversing soil acidification**, but it also **improves fertility**. Accordingly, liming can stimulate tree growth and $CO_2$ sequestration in tree biomass (Reid and Watmough 2014).

In **temperate forests**, liming has various species-specific effects on tree growth, ranging from no effect to increased or decreased growth (Long et al. 2022). In six limed spruce stands in southern Germany, stem growth was not stimulated, but liming increased fine root production and improved tree tolerance to drought (Kohler et al. 2019).

In **northern European boreal forests** liming has been shown to limit tree growth (Derome et al. 2000), and according to Persson et al (2021) liming on acidic N-rich soil resulted in a long-term decline in soil C and N pools. Therefore, liming has not been promoted as a carbon farming practice in the boreal conditions.

The **effects** of liming on soil carbon sequestration **are complex** and depend on the initial soil conditions. Soils with thick forest floor layers are particularly affected, as liming transforms them into more 'active' humus layers, reducing their carbon content through increased activity of decomposer microbes and earthworms (Bauhus et al. 2004; van Straaten et al. 2023). The effects of liming on mineral soils depend on soil acidification and initial soil organic carbon and clay content. Recently, Van Straaten et al (2023) found that the more lime applied, the higher the soil carbon losses. In addition to affecting soil carbon dynamics, liming can also increase nitrate leaching by accelerating soil N cycling rates (Kreutzer 1995; Gundersen et al. 2006).

Given the **limited number of studies** on the subject, it is difficult to say whether liming can be considered a carbon farming practice. If it is considered as such, a case-specific assessment of the additional $CO_2$ sequestration should be carried out, as the effects on tree growth could be highly species specific and the effects of liming on soil carbon depend on the local soil preconditions.

## 2.3 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 others 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 soil carbon **vary** considerably, although the impact is generally lower than that of wildfires and can even increase soil carbon (Alcañiz et al. 2018).

The effects of low or high-frequency fire occurrence on soil carbon 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 soil carbon 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.

## 2.4 Agroforestry

Livestock 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 carbon sequestration. This strategy could also be employed on crop land, where trees could be used as windbreaks, buffers or for shade provision (Nair et al. 2010).

**Soil fertility** will be improved as a result of the increase in litterfall and rhizodeposition in agroforestry systems and this strategy will also contribute to reducing soil erosion and improve water quality (Moreno et al. 2007; Jose 2009; Kay et al. 2019).

The carbon sequestration rate depends on tree species composition, age of the different species, geographic location, environmental factors and management practices (Nerlich et al. 2013) as well as on soil type and legacy effect of historical management.

Soil carbon **sequestration rates related to different agroforestry systems are positive** (0.4 Mg $CO_2$ $ha^{-1}yr^{-1}$-1.7 Mg $CO_2$ $ha^{-1}yr^{-1}$; Cardinael et al. 2017). For above-ground biomass, sequestration rates are also positive (0.5 Mg $CO_2$ $ha^{-1}yr^{-1}$ to 19.4 $CO_2$ $ha^{-1}yr^{-1}$) and depend on the tree species selected and their number (Kay et al. 2019). A major challenge remains to ensure the **long-term persistence** of these systems by ensuring forest regeneration through adequate management planning.

## 2.5 Peatland management

### 2.5.1 Water table management in drained peatlands by CCF

Managing peatland water levels through continuous cover forestry (CCF), plays a crucial role in reducing nutrient loading into waterways over the long term, especially in **Nordic countries, but also in the Baltic states and Poland**. Additionally, these practices help recover original peatland vegetation. The immediate climate benefits of CCF in boreal conditions relate to the avoidance of clear-cuts that have significant GHG emissions.

CCF is a management practice potentially **relevant for carbon farming in drained peatlands** as it has the potential to reduce GHG emissions (Tanneberger et al. 2021). CCF on fertile drained peatlands has been proposed as an **alternative to even-aged forestry** (Nieminen et al. 2018). The rationale behind CCF lies in maintaining growing biomass on peatland sites, thus avoiding the need for maintenance ditching. Additionally, CCF **excludes the treeless period** after clear-cutting, which is associated with high $CO_2$ and $N_2O$ emissions (Figure 9). CCF also manages the water table, raising the level compared to situations where tree stands are fully stocked and well-drained. This higher water table level reduces $CO_2$ and $N_2O$ emissions and supports climate change adaptation during longer drought periods.

Lehtonen et al. (2023) found that conversion from even-aged forest to uneven-age forest to CCF (ie, selective harvesting) in fertile drained peatlands in Finland reduces annual emissions by 1–1.2 Tg $CO_2$ eq, while the harvesting level remains constant. At the site level, CCF on fertile drained peatlands provides higher economic benefits to landowners (Juutinen et al. 2021), reduces nutrient loading to waterways (Palviainen et al. 2022), and maintains higher 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 (Günther et al. 2020).

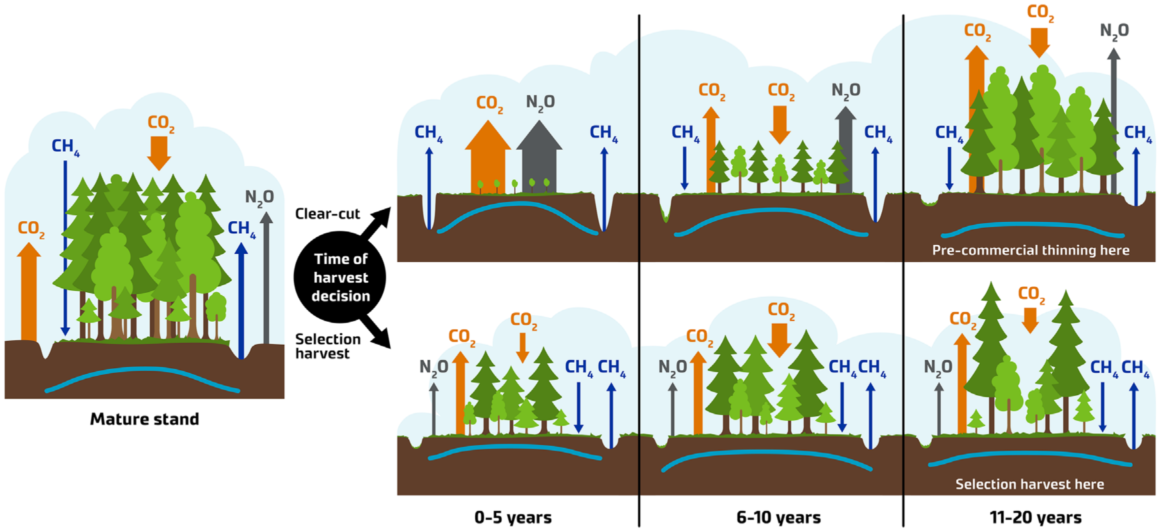


**Figure 9.** Land owners make harvesting decisions and, in the case of mature stands on fertile drained peatlands, owners can either opt for clear-cut (top row) or for selection harvests (bottom row, as a continuous cover forestry measure). With selection harvests, the amount of timber obtained is less compared to clear-cut, but adverse environmental impacts, such as GHG emissions and nutrient loading, are smaller.

In **Northern Europe, Finland** is the country with the highest area of drained peatland forests, covering 4.7 million hectares. Additionally, there are 390,000 hectares of drained fertile peatlands nearing their final felling age, prompting landowners to explore various management options (Lehtonen et al. 2023). In **Sweden**, according to the GHG inventory of Sweden, drained peatland forests cover around one million hectares. Meanwhile, in **Norway**, peatlands span nearly three million hectares, with 650,000 hectares having undergone drainage.

### 2.5.2 Wood ash fertilisation of drained peatlands

Another potential strategy for carbon sequestration in drained peatlands is wood ash fertilisation by improving **soil fertility and stimulating tree growth** (Hytönen 2016). Ash fertilisation significantly influences peat chemistry by increasing soil pH and enhancing the availability of phosphorus (P) and potassium (K) and so is suitable for sites with low pH and fertility (see Jansone et al. 2020). At ecosystem level, ash fertilisation leads to a long-lasting carbon sink in nutrient-poor sites, while in more fertile sites it results either in minimal carbon uptake or even $CO_2$ emissions, suggesting potential soil carbon losses over longer time spans (Ojanen et al. 2019). However, the effects of ash fertilisation on soil GHG emissions remain **less understood as there are few measurements** in ash-fertilised peatlands (for example, 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 **additional increment of 1.2 Tg $CO_2$ $yr^{-1}$** (current soil emissions from drained fertile forested peatlands are 10.6 Tg $CO_2$ eq. in Finland).

In terms of **soil carbon accumulation**, 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 **soil organic carbon mineralisation** (Zimmermann and Frey 2002; Rosenberg et al. 2010), which may offset carbon sequestration in biomass.

Another issue is the accumulation of **heavy metals in wood ash** and **environmental concerns** about its application in natural ecosystems. These concerns, combined with the lack of clear legislation, have so far prevented wider use of wood ash in central Europe. As for biodiversity impacts, the effects of ash fertilisation are not well-known, apart from a few studies reporting a positive impact on biodiversity (for example, Silvan and Hytönen 2016; Zusevica et al. 2022).

According to the IPCC (Hiraishi et al. 2013), key recommendations for wood ash fertilisation in forested peatlands include the use of ash originating from bioenergy plants and avoiding conducting ash fertilisation near waterways to prevent nutrient runoff and leaching. By adhering to these guidelines, we can enhance soil fertility, promote tree growth and contribute to sustainable land management in peatland ecosystems.

### 2.5.3 Peatland restoration

Peatland restoration can significantly reduce soil GHG emissions (Wüst-Galley et al. 2016), and rewetting of drained peat soils under agricultural use is already taking place in central Europe. Due to high emissions per area, restoration of peatlands soils **involves much less land** compared to other mitigation measures, and therefore these practices should be prioritised (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 (for example, ~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 to 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 $ha^{-1}$ $yr^{-1}$ according to Wilson et al. 2016) or in some cases even turn them back into a net GHG sink in the longer-term (Günther et al. 2020).

A side-effect of rewetting is **increased methane emission**, which can offset much of the cooling in the early decades after re-wetting (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 to 11.5 Mg $CO_2$ $ha^{-1}$ $yr^{-1}$, while simultaneously the increase in $CH_4$ emissions varied between 0.7 to 6.0 $CO_2$ 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 carbon 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 soil are rewetted, the reduced tree $CO_2$ uptake can offset the GHG emission savings from the rewetted soil for decades.

However, 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 carbon crediting schemes, the initial response of tree biomass to rewetting needs to be carefully considered.

# 3 Measurement challenges

## 3.1 Quantification and monitoring

As reviewed in Chapter 2, there is scientific evidence from different European biomes and forest types suggesting that management practices can significantly increase carbon sequestration of European forests. Over recent decades, a growing number of studies have addressed this issue and have developed multiple approximations to measure carbon fluxes in forest ecosystems.

Under the UNFCC framework, signatory Kyoto Protocol countries have, for example, used **different approximations to report LULUCF activities** with respect to Forest Reference Levels (FRL) – so, hypothetical business as usual forest carbon dynamics. Yet, implementation of carbon farming practices implies scientific and technical challenges associated with carbon quantification across a variety of EU forest types, including assessment of additionality, permanence and sustainability (as described in Chapter 4). This may require down-scaling current modelling approximations and model validation against observations from different forest pools conducted across spatial and temporal scales (Figure 10).

Figure 10 gives an overview of the different datasets available at local, regional and continental scales. It also shows what form of monitoring (for example, remote sensing, tree ring analysis) generates which data at which spatial level and at which temporal scale. The variance among them is very large.

This chapter **reviews state-of-the-art approaches** for the quantification of carbon sequestration in forests focusing on **carbon farming practices validation**.

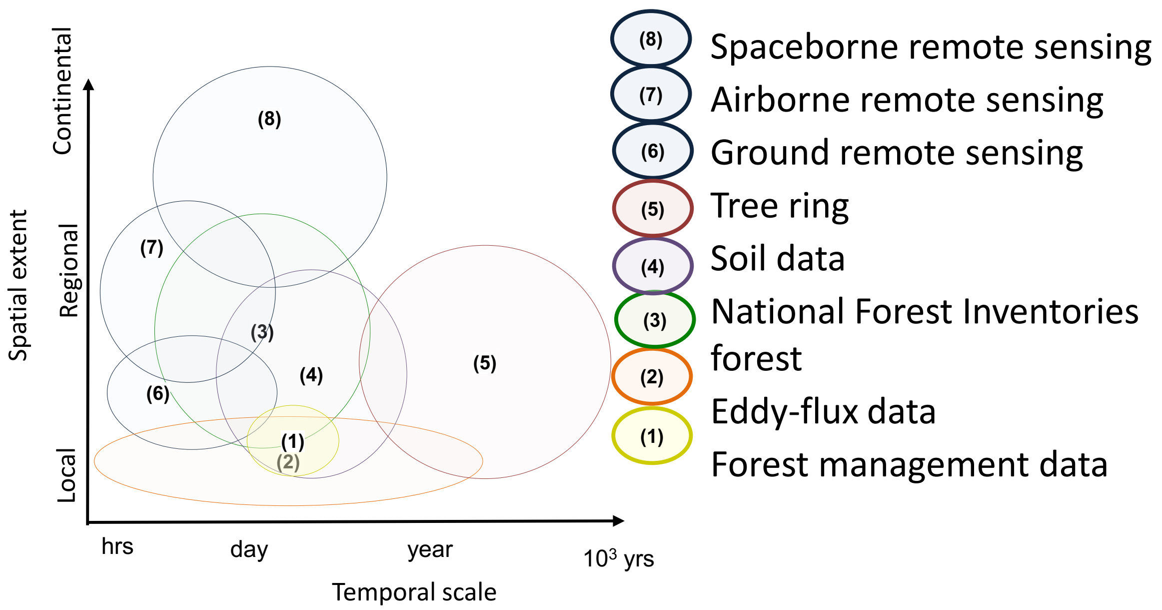


**Figure 10.** Observations available to estimate forest carbon at different spatial and temporal scales (modified from Hartig et al. 2012 and Ruiz Benito et al. 2020) (NFI; National Forest Inventories).

### 3.1.1 Above- and below-ground biomass carbon estimation

**The EU's LULUCF regulation** requires land uses to be tracked in a **spatially explicit way**, meaning that all member states must set up monitoring tools with spatial data. Remote sensing (RS) products are increasingly available at European scale, with variable spatial and temporal coverage dependent on platform and product. While RS-based approximations are rapidly improving, it is important to be aware of the **limitations** of each product in terms of processing costs, uncertainties, scale and resolution.

In general, **hybrid approaches** involving multiple observation and ground data are now considered to be the most promising to obtain robust results, although RS products alone can provide valuable spatialised information in areas lacking field surveys. Open-access **large-scale airborne LiDAR data** and the more recently launched Global Ecosystem Dynamics Investigation Mission (GEDI) provide ample coverage of space-borne LiDAR light detection and ranging for **southern Europe**.

**Tree height and diameter** are common inventory variables that can also be obtained from airborne LiDAR and ground-based remote sensing with higher accuracy than inventory-based calculations (Zolkos et al. 2013). LiDAR can provide sub-metre surface heights (Lee et al. 2010) with an accuracy that depends on canopy height distribution and other factors such as ground slope and sampling intensity (Hopkinson and Chasmer 2009).

**Low-point density data** can be used to calculate stand tree density, basal area and **above-ground biomass** (Lee and Lucas 2007; Simonson et al. 2016). Biomass or wood volume can also be estimated from **space-borne remote sensing** as passive microwave data (Liu et al. 2015), passive optical data (for example, from Landsat: Avitabile et al. 2012), SAR data from L-band (Mitchard et al. 2011) and mixed ground-data estimation and C-band instruments (Rodríguez-Veiga et al. 2017).

Products from **SAR (P-band Synthetic Aperture Radar**) are also increasingly used for forest biomass monitoring. It uses backscatter coefficients related to wood volume scattering mechanisms and/or allometry using height estimates derived through polarimetric interferometry (PolInSAR; Le Toan et al. 2011). Space-borne LiDAR allows biomass quantification at large scales (Simard et al. 2011) resulting in similar results to airborne products (Popescu et al. 2011).

**Terrestrial laser scanning** (TLS), also referred to as terrestrial LiDAR or topographic LiDAR, can provide structural measurements at finer spatial scales and ground validation of carbon estimates from space-borne and airborne methods (White et al. 2016). TLS offers an efficient and accurate alternative for local biomass estimation when the user can afford fieldwork and processing of large data observations (Calders et al. 2015).

**Integrated methods** that combine observations from different sources are a promising approximation to upscaling forest biomass estimation. For example, stand inventory data and TLS can be integrated with airborne LiDAR for upscaling forest biomass (Hancock et al. 2017).

More detailed carbon estimates can be derived from **forest structure** (ie, tree density, basal area, tree biomass, or crown metrics) obtained from forest inventories alone or combined and/or remote sensing data. **National forest inventories** (NFI) measure tree-level diameter/height, allowing a direct estimation of plot level basal area or tree density, which allows the application of models of stand volume, biomass, or carbon estimates through species-specific allometric equations developed from destructive biomass estimations – for example, felling and uprooting trees (Montero et al. 2005, Annighöfer et al. 2016) (see example in Figure 11).

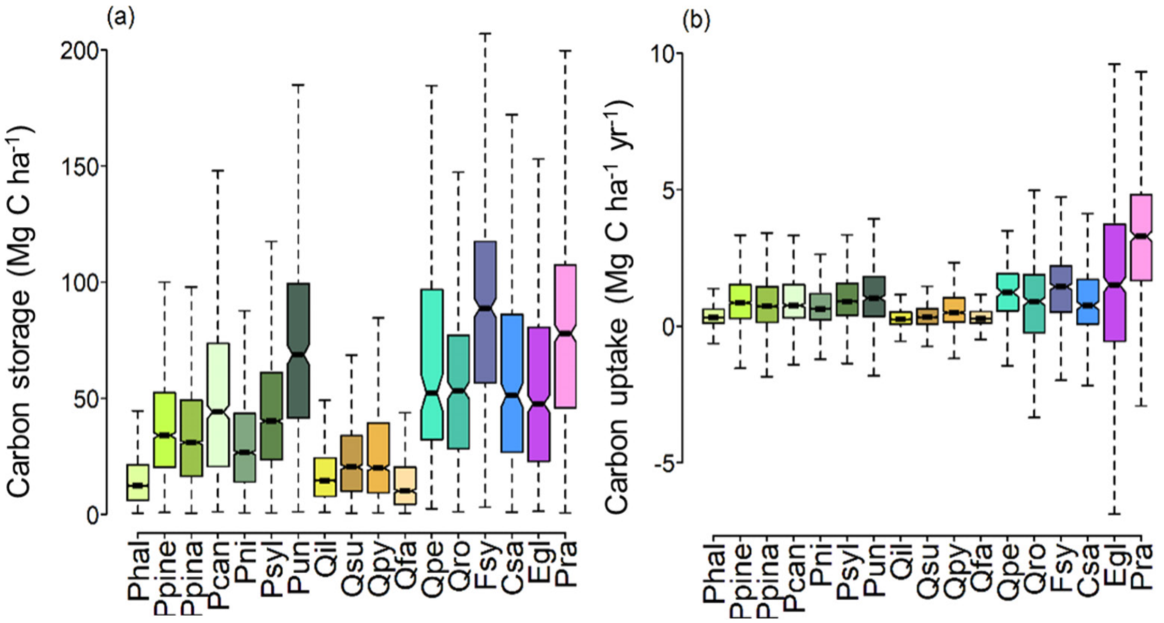


**Figure 11.** Above-ground (a) carbon storage (ie, total carbon of living trees, in Mg C ha$^{-1}$), and (b) carbon uptake (ie, changes in stand carbon storage based on carbon gains due to tree growth and ingrowth and losses from natural mortality and harvesting, in Mg C ha$^{-1}$ yr$^{-1}$). Estimates are based on repeated forest inventory surveys (Second and Third Spanish National Forest Inventory) in combination with allometries from Montero et al (2005). Boxplots include the median and 25–5 interquartile ranges, with lines showing the min-max excluding outliers. Forest types are labelled according to species relative abundance (species basal area is greater than 50% of the total stand basal area). Phal = *Pinus halepensis*, Ppine = *P. pinea*, Ppina = *P. pinaster*, Pcan = *P. canariensis*, Pni = *P. nigra*, Psyl = *P. sylvestris*, Pun = *P. uncinata*, Qil = *Q. ilex*, Qsu = *Q. suber*, Qpy = Q. *pyrenaica*, Qfa = Q. faginea, Qpe = *Q. petraea*, Qro = Q. *robur*, Fsy = *Fagus sylvatica*, Csa = *Castanea sativa*, Egl = *Eucayptus globulus*, Pra = *P. radiata*. Stand carbon storage averages 43.35 Mg C ha$^{-1}$ and stand carbon uptake 1.02 Mg C ha$^{-1}$ yr$^{-1}$ . Modified from González-Díaz et al. 2019.

**Harmonisation initiatives** are resulting in the availability of NFI data at the European level, such as species occurrence (Mauri et al. 2017) or forest structure (Moreno et al. 2017). **Integrated NFI and remote sensing products** are also increasingly used to monitor changes in carbon uptake with LIDAR data (see Falkowski et al. 2009) and with optical images (for example, Haakana et al. 2023). NFI permanent plots allow us to estimate tree growth, mortality, recruitment and harvesting at the individual and stand levels at regular intervals (often each around 5–10 years) (Kunstler et al. 2021). Demographic changes can be upscaled to provide biomass and forest carbon estimates. These observations can be mapped and combined with remote sensing products to infer spatio-temporal variability of carbon stocks.

**Carbon fluxes** of individual ecosystem components are very dynamic in time. Ecosystem-level approximations of carbon fluxes have the advantage of providing a whole ecosystem time-continuous carbon balance. **Eddy covariance measurements** turn out to be critical for quantifying the spatial differences and temporal dynamics in $CO_2$, $N_2O$ and $CH_4$ across large abiotic and biotic gradients and can be used to estimate gross primary productivity (GPP) (for example, Wohlfahrt and Galvagno 2017).

Eddy covariance towers are consolidated research infrastructures with standardised data for more than 15 years across Europe and worldwide (Aubinet et al. 2012; Papale et al. 2012; Franz et al. 2018).

### 3.1.2 Soils

Optimal **soil organic carbon monitoring methods** depend on soil type and, usually, soil carbon stock changes are determined with repeated soil carbon inventories, while the soil carbon change in peatlands is mostly based on soil $CO_2$ flux measurements.

When quantifying soil carbon stock changes, it is essential that, in addition to soil carbon content, **bulk density, gravel content and stones and boulders are also quantified** to have unbiased estimates. To estimate bulk density, one needs to have volume precise sampling of different soil layers.

**Carbon in forest soils** typically shows **high spatial variability** within short distances and so it is important to have a spatial sampling design within a plot, ensuring that plot level estimate for soil properties is unbiased (for example, Häkkinen et al. 2011). Such sampling designs easily **require 100+ sample points** to detect changes in soil organic carbon at a local scale (for example, for 7-8 ha forest area, as shown in Schrumpf et al. 2011).

While the measurement of carbon concentration itself is inexpensive, sampling the soil from several depth layers, determining bulk density, rock content, etc is **labour-intensive**. Therefore, determining carbon changes in the soil in the field is almost always much more costly than measuring carbon changes in the above-ground biomass.

The required repetition of sampling points depends not only on the variability of the background carbon stock in the soil but also on the desired interval for re-measurement (to calculate the change over time) and on the impact of the carbon farming method in terms of changing carbon input to the soil (Smith 2004). If the impact of carbon farming practice is low and the interval for repeated measurements is short (<10 years), an unreasonably high number of sampling points would be required to detect any significant changes in soil carbon. For more details for appropriate soil sampling, see Cools and de Vos (2013).

There are several **soil inventories in Europe**, such as the *International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests* (ICP forests) (Cools and de Vos 2013) and *Land Use and Coverage Area frame Survey* (LUCAS soil inventory) (Orgiazzi et al. 2018). Sampling for **ICP forests** is more comprehensive as it covers several soil layers, considers spatial variability, and includes guidance for the sampling of various nutrients and elements. ICP forest inventory also measures trees and quantifies tree biomass and its change on those same plots, while LUCAS soil inventory excludes tree measurements. The LUCAS soil inventory covers the whole of Europe and focuses on a few key variables in the top 20 cm layer. Unfortunately, the earlier LUCAS soil inventory for forests does not differentiate organic from mineral soil layers. As a result, generated soil estimates have systematic differences against national soil inventories (see Ziche et al. 2022).

*The European Soils Data Centre* (ESDC) and *International Soil Reference and Information Centre* (ISRIC) *World Soil Information* provide a wealth of soil science information, and FAO provides a global soil organic carbon map, which is mostly open access and directly downloadable at 1 $km^2$. In addition to soil property and quality datasets, the ESDC hosts information on different soil functions and threats to soil functioning. Soil water content, temperature and snowpack has been estimated from 1979 to 2010 in the ERA-INTERIM/Land at a resolution of 0.125° (Balsamo et al. 2015) and soil organic carbon is mapped at 1 $km^2$ resolution in the Global Soil Organic Carbon Map.

## 3.2 Modelling tools

Forest ecosystems are complex and dynamic systems, influenced by various ecological, biophysical and socio-economic factors. Modelling is needed to understand how different factors drive carbon balance and to design scenarios exploring tentative system trajectories under a different assumption.

Despite the complexity of forest dynamics, there has been a steady advance in modelling techniques to understand and predict changes in forest dynamics and, to some extent, carbon stocks and sequestration. Available **forest models range from empirical to process-based approaches**, covering from local to global scales (Bugmann and Seidl 2022). The use of empirical data at different scales combined with modelling is a powerful tool to quantify carbon stocks and forest dynamics and evaluate synergies and trade-offs with biodiversity and other ecosystem services (Urban et al. 2016; Franklin et al. 2016).

In this section we review the diversity of existing models and discuss their application in the context of carbon farming implementation.

Forestry has a long **tradition of empirical models at the stand level,** such as yield tables. These are **empirical models** describing above-ground stand productivity based on correlational relationships between tree and stand-level variables (for example, quadratic diameter, etc) (Tesch et al. 1980). For example, ***yield tables*** describe forest volume as a function of tree or stand variables and management characteristics (such as rotation period) (Pretzsch et al. 2008). These models are **calibrated for specific site conditions,** which can be evaluated through different indicators such as maximum tree height. Estimates of stand volume can be used to derive carbon stocks through standard allometric relationships or combined with LiDAR estimates.

A more flexible type of model are **size-structured models** that describe cohort dynamics as a function of tree size and competition and can include climatic dependency to project climate-based scenarios. These models can be easily parameterised from forest management data, have an analytical expression, and can be used to contrast forest biomass or carbon stock dynamics under several climatic conditions and management regimes (for example, to explore changes in the rotation time and harvesting intensity) (Zavala et al. 2024).

**Soil carbon stocks** can be also estimated through empirical approaches. It has been shown that empirical models and machine learning linking vegetation information, climate and topographic information, are able to estimate soil carbon content with a precision that is suitable for large-scale soil mapping, but not appropriate for point locations due to high spatial variation (Baltensweiler et al. 2021).

Confronting the multiple changes of forest carbon dynamics often implies a level of complexity that requires the **inclusion** of both **empirical and causal components in the model** (see Mäkelä et al. 2000; Landsberg et al. 2003). For example, in addition to climatic dependency, forest productivity is driven by factors such as atmospheric $CO_2$ concentration or atmospheric deposition that are not incorporated into empirical demographic-based models (Cramer et al. 2001; de Vries et al. 2014).

**Eco-physiological process-based models** incorporate some mechanistic components of carbon uptake such as photosynthesis and losses through ecosystem respiration that underlie carbon allocation to growth of leaves, stems and roots (for example, GOTILWA+, Gracia et al. 2003). Model calibration and validation methods that take into account the hybrid character of models, have been successfully implemented to describe tree growth and stand productivity and can be readjusted to predict forest carbon dynamics at several scales from the stand to region (Fontes et al. 2010; Medlyn et al. 2011).

Soil carbon stocks and changes can also be projected with process-based modelling. These models have been constructed by conceptualising different soil carbon pools, typically according to their decomposability. In addition to different pools, in these models soil organic matter decomposes according to its quality and environmental drivers (such as temperature and soil moisture).

Besides climate forcing, properties such as **soil texture** (for example, clay content) may affect soil organic matter aggregation and have an impact on soil organic matter decomposition.

First order soil carbon models have been widely used with forest planning tools and with GHG inventories. With these models (such as CENTURY, Yasso07, RothC, Q and Romul), soil carbon changes are driven by litter inputs and trends (Palosuo et al. 2012).

Process models are under development and additional processes have been implemented and tested with those models, such as the role of soil microbial biomass and activity (Abramoff et al. 2022). More recently, in addition to microbial biomass, **microbial biodiversity and its impact on soil organic matter decomposition** has also been evaluated (Khurana et al. 2023).

Modelling soil carbon changes nonetheless prerequisites a thorough **initial assessment** of field soil carbon stocks and the distribution of different soil carbon pools for model initialisation. Based on the model complexity, several further (microbial) parameters need to be measured initially or consecutively.

**Ecosystem process-based models** of productivity can be also parameterised from remote sensed products and can project productivity estimates at different spatial scales (Running et al. 2004). Eddy covariance approximations also provide data from the main components of the forest carbon cycle, for example, heterotrophic and autotrophic components of ecosystem respiration which is key to developing process-based models of carbon balance (see Brændholt et al. 2018; Darenova et al. 2024).

For example, Virkkala et al (2021) compiled eddy covariance and chamber measurements of annual and growing season ecosystem $CO_2$ fluxes of gross primary productivity (GPP), ecosystem respiration (ER), and net ecosystem exchange (NEE) during 1990–2015 from 148 terrestrial high-latitude (tundra and boreal) sites to analyse the spatial patterns and drivers of ecosystem $CO_2$ fluxes and test the accuracy and uncertainty of different statistical models.

**Upscaling forest carbon dynamics** often requires accounting for demographic processes, forest management and disturbances.

- **Individual based models** (IBM) scale up from predicted growth and survival over time for each tree in small patches of forest land (Shugart, 1984; Pacala et al. 1993; Bugmann and Seidl 2022).

- **Spatial heterogeneity** arising from biotic interactions and landscape heterogeneity is critical for a realistic projection of forest productivity (Coates et al. 2003) but also to assess changes in biodiversity and other ecosystem services over extents ranging from stand to region (Ameztegui et al. 2017).

- More **complex landscape models** can simulate tree growth through physiologically driven ecosystem processes, including spatial biogeochemical fluxes across the landscape to project carbon fluxes incorporating site conditions heterogeneity, neighbourhood interactions and responses to changes in the environment such as climate, $CO_2$, fires or management (Kimmins et al. 1999).

- ***Dynamic Global Vegetation Models*** (DGVM) couple physiological ecosystem processes (such as carbon and water exchange) with demographic changes, including competition between functional types and feedback with soil processes using models such as CENTURY (Cramer et al. 2001; Purves and Pacala 2008). These models can downscale to quantify the differential impact of multiple factors on forest mitigation potential.

  For example, Gregor et al (2024) uses LPJ-GUESS to investigate how the relative weight of different factors – forest age and type, climate change, disturbances, harvest intensities, wood usage patterns and carbon-intensity of substituted products – influences the carbon mitigation potential of Bavarian forests.

  Currently, there is an active investigation to validate these models at the forest level so potential users (such as private forest owners considering a carbon farming contract) can estimate additional carbon stored based on local context (tree species, climate, soils etc).

## 3.3 Future projections and scenarios

Forest management is expected to attend to multiple objectives; from biodiversity preservation to ecosystem services provision, including climate regulation services (mitigation) and promoting resilience (adaptation).

**Models of forest dynamics and DGVM** allow us to explore how different policies and management regimes influence biodiversity and carbon removal so they can be used to assess sustainability trade-offs and additionality. However, these models can be **difficult to parameterise** and **are not yet widely available** for policymakers. Alternatively, **scenario building can facilitate decision-making** when implementing carbon farming practices.

Next, we discuss the **main factors** to consider when **evaluating trade-offs and synergies in emerging carbon farming practices** and we develop likely biodiversity-carbon removal scenarios.

### Box 4. Woody encroachment

Woody encroachment consisting of shrub colonisation of abandoned agricultural lands when silvopastoral activities are abandoned is a widespread phenomenon in Europe (22 million hectares during the period from 1992 to 2020 with a net forest area increment of 0.4 million $ha^{-1}yr^{-1}$, period 2010–2015; FAO 2015).

Woody encroachment can sequester carbon by increasing biomass production and litter inputs, although it can also lead to increased decomposition rates of soil carbon, potentially offsetting the initial carbon gains, leading to soil carbon losses, particularly in grasslands (Fino et al. 2020).

Lack of range and forest management, coupled with the increased risk of wildfires, poses significant challenges for ecosystem resilience, carbon sequestration and biodiversity conservation in these novel ecosystems. Effective land management strategies, including active forest management and wildfire prevention measures, are essential for mitigating these impacts and promoting sustainable land use practices in the region.

Forest management can mainly operate on two forest attributes: **stand composition**, which is linked to biodiversity, and **stand structure**, which is associated with several ecosystem services, including climatic resilience.

There is ample evidence of the relationship between biodiversity and forest multifunctionality as a driver of ecosystem services provision (Brockerhoff et al. 2017; Mori et al. 2017), including climate regulation services such as mitigation (Díaz et al. 2009) and climatic resilience (Brockerhoff et al. 2017).

At the **stand level** there is **evidence of positive (α-) diversity and productivity relationships** (see Ruiz-Benito et al. 2013.

- Complementary resource use by functionally different plant species results in higher productivity of mixed forests in relation to monospecific forests (Morin et al. 2011).

- Biodiversity is also linked to forest stability (Allan et al. 2011): biodiversity increases the probability of finding life strategies that can withstand a given impact and maintain a given ecosystem function, suggesting a link between biodiversity and climatic resilience (Barrere et al. 2023).

Although many of these relationships are context-specific, these two pieces of evidence suggest that forest diversification can provide win-win strategies between biodiversity, mitigation and adaptation.

In addition to forest composition, defining optimal **stand structure** to attend multiple objectives is a central issue in current forest management.

As shown in Chapter 2, different forest management **practices impact forest carbon pools in different ways** and silvicultural treatments can be readjusted to reach mitigation objectives. But, in addition to carbon sequestration, forest management can also impact biodiversity, provision of other ecosystem services and climatic resilience.

Many studies have shown that **stand structural diversity** is linked to biodiversity. For example, vertical forest stratification and size distribution, and spatial heterogeneity; a mosaic of dense and open canopies contribute to habitat variability and biodiversity (Hämäläinen et al. 2024). Forest structure also influences several ecosystem services such as water provision, according to a trade-off between forest stock and net precipitation (Jackson et al. 2005). Forest structure also plays a pivotal role in forest resilience to climatic impacts.

In **water-limited environments** increased standing biomass correlates to higher tree mortality during drier periods, thus supporting the aggravating role of crowding in drought-induced mortality, suggesting that forest densification is a key predisposing factor to mortality (Vilà-Cabrera et al. 2023).

Experimental and modelling studies suggest that moderate thinning reduces tree mortality by alleviating water stress locally. A reduction in tree density improves the tree growth of remnant trees, but also improves stand-level drought resilience during a limited number of years (Sohn et al. 2016).

### 3.3.1 Win-win scenarios

**Biodiversity enhances productivity** (Feng et al. 2022b), **multifunctionality** (Van der Plas et al. 2016a) and **climatic resilience** (Jactel et al. 2017). Hence, promoting regeneration and coexistence of mixed forests and a heterogenous forest structure is a “win-win” option in most cases (Messier et al. 2021). Biodiversity is also very important for maintaining multifunctionality at larger spatial scales. Both high local (α-) diversity and a high turnover in species composition between locations (high β-diversity) are important drivers of ecosystem multifunctionality (Van der Plas et al. 2016b).

**Large-scale emphasis** on high functional levels (such as intensive carbon farming) can lead to biotic homogenisation and a decrease in other key ecosystem services at the landscape level.

⇨ **Sustainable carbon farming** will likely be better achieved by forest management options at the landscape levels and over a design that integrates pure land sparing (intensive plantations and forest reserve) and land-sharing options in a variable proportion (Muys et al. 2022; Figure 12b).

Landscape planning is more challenging in landscapes with a large proportion of **private ownership** (Muys et al. 2022; Bollmann and Braunisch 2013) yet carbon farming can be an important incentive for driving this landscape transformation and promoting a shift in the proportion of land sparing and land sharing options, ie maintaining cultural forest systems or restoring multifunctional ecosystems.

### 3.3.2 Win-lose scenarios

A focus on biodiversity and conservation with overprotection policies can also have detrimental long-term effects on several ecosystem services and, paradoxically, on biodiversity (Varela et al. 2020; Figure 12).

High biodiversity in some European forest ecosystems is linked to **cultural** landscapes with high biodiversity and multifunctionality levels. Because of legacy effects, abandoned forests can exhibit higher climatic risks than managed forests – for example, highly densified plantations, abandoned coppices or unmanaged newly formed forests are more vulnerable to drought and have higher risks of wildfires (Vilà-Cabrera et al. 2023).

### 3.3.3 Lose-win/lose-lose scenarios

Some specific ecosystems can present **high species richness** levels of some taxa, yet they can exhibit **low carbon storage** and net primary productivity.

- For instance, certain agricultural landscapes can form human-made slopes that support a diverse range of bird species, but carbon storage would be lower relative to agricultural/woodland transitional stages.
- Similarly, some Mediterranean woodlands with moderate to high fire frequency can exhibit autosuccesional dynamics following disturbances and can maintain high diversity values, yet carbon is periodically lost by wildfires (Figure 12d).

In abandoned agricultural lands the **outcome of succession is context specific**. In many cases, biodiversity and key ecosystem services such as soil and above- ground carbon accumulation and infiltration improve along secondary succession. Shrub roots can help stabilise soils and reduce erosion, although in others an increase in erosion is also possible (Puttock et al. 2014).

However, the increase of biomass increases the **risk of intense and widespread wildfires** during dry and hot periods, resulting in a loss of soil organic matter through combustion and increased rates of decomposition, soil erosion after fire and even desertification on steep slopes (Box 4, Figure 12c).

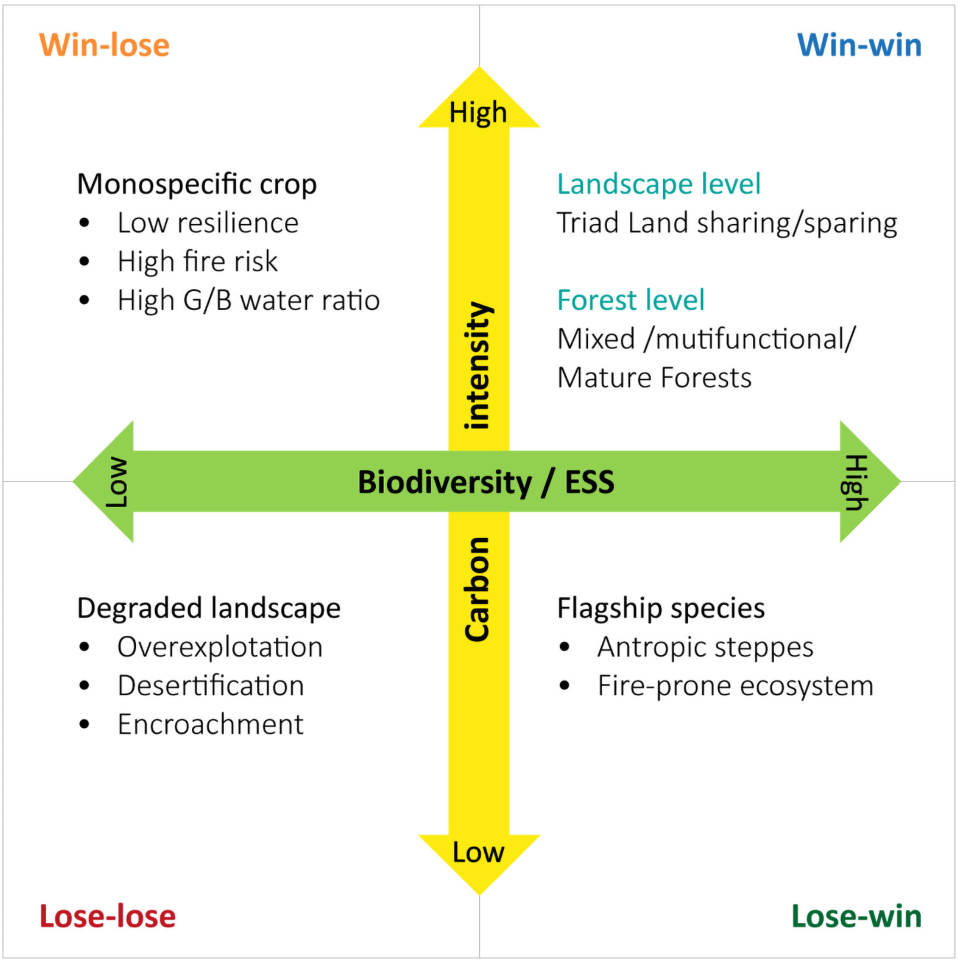


**Figure 12.** Likely scenarios emerging for carbon farming trade-offs with biodiversity, ecosystem services (ESS) and adaptation.

# 4 Policy and economic aspects

This chapter outlines key criteria for the successful implementation of carbon farming instruments. We also draw on experiences from outside Europe and, more generally, on the implementation of payments for environmental services. Four examples of instruments that have been implemented in the European forestry sector and one example from the US are used as illustrations.

## 4.1 Market-based approaches

**Carbon farming** has been identified as a mechanism to help **internalise the positive externalities of land-use changes or practices** that **enhance carbon sequestration in vegetation and soils** as well as a new business model that can contribute to the income of land users. Paying for carbon sequestration through **carbon credit markets** can be a tool to provide land managers with incentives to manage their land according to carbon reduction targets.

As previously discussed, carbon farming depends on **complex** biophysical processes, has long time horizons and many risk factors (Tang et al. 2016). These are all factors that challenge the development of a transparent and well-functioning carbon market in the land sector (Raina et al. 2024), particularly so in the forestry sector (Haya et al. 2023).

For **some forest types,** forest management targets operate **over more than a century** and may be affected by multiple risks (such as fire, insect attacks, windthrow, etc). A **diverse forest ownership structure** with many small forest owners makes the development of well-functioning markets even more difficult as the costs of concluding and enforcing contracts will be high relative to the amount of carbon emission reduction obtained (Lee et al. 2018). Moreover, as **timber is traded on a global market**, it may also be important to consider non-local indirect effects of changes in forest management. Section 3.3 discusses some of these challenges in more detail.

**Carbon credits** can be traded in the so-called **compliance market** or in the **voluntary carbon market**. The compliance market focuses on mandatory emission reductions through legal mechanisms, while the voluntary carbon market facilitates voluntary offsets to support broader environmental goals and commitments.

- Existing carbon farming mechanisms in Europe concern mainly the **voluntary carbon market**. In the voluntary market, carbon reductions are typically purchased by **companies or individuals** seeking to meet corporate or personal sustainability goals.
- Several **voluntary carbon emission reduction registries and standards** have emerged (Cevallos et al. 2019). These standards typically require companies to try to reduce emissions in their primary production before buying reductions on the voluntary market.
- **Purchased carbon reductions** cannot be used as offsets in regulated sectors but contribute to national reduction targets for the land sector.

While the focus of carbon farming is on carbon credits traded on voluntary markets, payments can also come from public funds, such as agri-environmental schemes under the Common Agricultural Policy (see section 4.4).

There has been repeated **criticism of the methods** applied for generating certificates for voluntary carbon markets, including improved forest management and forest protection projects (Haya et al. 2023; West et al. 2023). But the use of carbon credits from carbon farming for compensation claims can also be seen as a threat to effective mitigation as it leads to the permanent GHG emissions that could have been avoided.

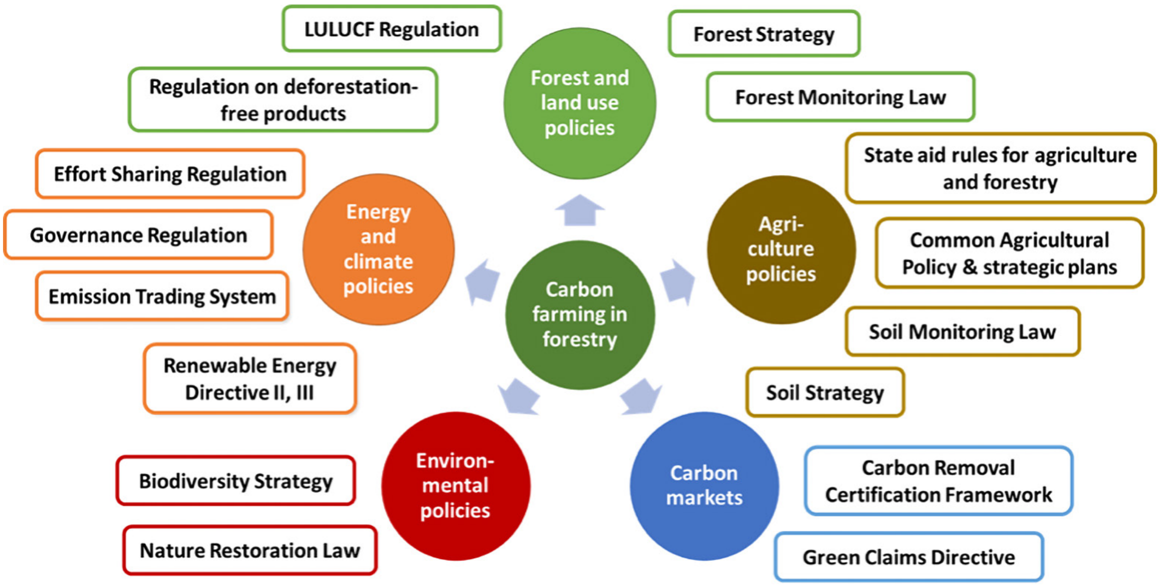


**Figure 13.** Overview of legislative context for carbon farming in forestry in EU.

## 4.2 Carbon farming in the EU legislation

Figure 13 summarises the policy environment for carbon farming in forestry in the EU. Its main policy instruments are described below.

For the European Union to achieve a net GHG emission decrease of at least 55% by 2030, there are several legal instruments in place. These include the *Regulation of the Governance of the Energy Union and Climate* 2018/1999, which sets out rules for planning, reporting and monitoring, the EU *Emissions Trading System* (ETS), and the *Effort Sharing Regulation* (ESR) both designed to reduce GHG emissions from energy production, industrial processes, the agriculture sector and waste management.

Moreover, the EU Climate Law explicitly integrates emissions and removals that are occurring in the LULUCF sector. The EU Climate Law sets a maximum contribution of natural removals from LULUCF to 225 Mt $CO_2$ to avoid mitigation deterrence, ie that the necessary emission reductions in other sectors are being delayed. At the same time the EU sets a minimum target for net removals to be achieved with its LULUCF Regulation (Regulation (EU) 2018/841[3]) that was amended in 2023 (Regulation (EU) 2023/839[4]) – the expected contribution of the land sector to achieving the EU's climate targets.

In **2030,** the **EU land sector** is committed to generating **removals of -310 Mt $CO_2$eq**. The LULUCF Regulation also sets out binding national targets for 2030 for each Member State, covering all emissions and removals in the LULUCF sector reported in GHG inventories. From 2026 to 2029, Member States must respect a LULUCF 'budget', defined as the sum of all net removals that are required.

The basis for the LULUCF Regulation is data on emissions and removals annually reported by Member States to UNFCCC in their **National Inventory Reports** (NIR) and reviewed and aggregated by the European Commission.

3 https://eur-lex.europa.eu/eli/reg/2018/841/oj
4 https://eur-lex.europa.eu/eli/reg/2023/839/oj

The **reporting** involves differentiating GHGs, classifying the land into reporting categories, and documenting the impact of land management on emission factors or carbon pools. **Methodologies** for the reporting are provided by the IPCC *Good Practice Guidance* (IPCC 2019). In general, the common system enables Member States to report GHG emissions and removals from the land sector in a transparent, accurate, complete, consistent and comparable way.

The **reporting system documents land-use changes** and the associated GHG emissions and removals. It also reflects changes in management practices within a land reporting category that occur in these areas. Visibility of activities in GHG inventories of the land sector is important for effective climate policy. The carbon credits generated by activities that affect emissions and removals in Member States can only be counted towards achieving the NDC of the country where the activity takes place.

The EU's ***Carbon Farming Initiative*** was launched in 2021 as part of the *Communication on Sustainable Carbon Cycles* (COM (2021) 800[5]) and can be considered as the instrument to involve the private sector in mitigation in the land sector (see Box 5). The initiative outlines the need to **improve monitoring, reporting and verification methodologies** for managing carbon in the land sector and the need to provide financial incentives. It sets out two targets: by 2028, every landowner or manager should have access to verified emission and removal data, and carbon farming should support the achievement of the 2030 net removal target of the LULUCF Regulation.

## Box 5. The EU Carbon Removals and Carbon Farming (CRCF) Regulation

On 10 April 2024, the EU adopted a Regulation establishing a Union certification framework for permanent carbon removals, carbon farming and carbon storage in products. Its aim is to provide a basis for certifying high-quality carbon removals in the EU so that such carbon removals have a minimum level of quality and comparability across the EU. Removals include those generated from carbon farming activities, as well as industrial technologies or processes storing atmospheric carbon in geological pools (such as Bioenergy with CCS) and long-lasting products (for example, wood materials in construction).

The framework establishes four quality criteria (quantification, additionality, long-term storage, sustainability) and rules for third party verification and certification. Under this framework, the European Commission will develop methodologies to operationalise the quality criteria, considering the specific characteristics of different carbon removal activities.

The CRCF can be used by Member States to generate contributions to achieving the LULUCF targets by the private sector that invests in carbon farming and carbon products.

Before these above-mentioned climate policy-related instruments were agreed, the EU land sector was already regulated by other legislation.

The **Common Agricultural Policy** (CAP) has been one of the most important European policy instruments since the unification of Europe. Since 1962, the CAP regulates funding streams between EU Member States and farmers to improve agricultural productivity, ensure food production and support the rural economy in seven-year rural development programmes. The CAP 2023-27 entered into force on 1 January 2023.

5 https://climate.ec.europa.eu/system/files/2021-12/com_2021_800_en_0.pdf

The CAP sets specific objectives for agricultural practices and provides funding for actions with positive effects on carbon sinks. It closely relates to carbon farming because it also promotes the establishment of landscape features and soil protection, which are essential for carbon storage. Twenty-four percent of CAP direct payments are allocated to eco-schemes that incentivise climate-friendly farming practices, such as organic farming, agroforestry, or soil conservation. Member States outline in their CAP Strategic Plans how they will allocate funding to different interventions to achieve the specific objectives of the CAP.

EU countries can choose to fund **forestry interventions** through their CAP Strategic Plans. These interventions might aim at protecting the forest, making it more resilient to climate change and safeguarding its multiple functions. Specific activities that can be targeted are:

- **afforestation** or creation of woodland, both their establishment as investments and their maintenance
- **creation and maintenance** of new **agroforestry** systems, regeneration or renovation of existing agroforestry systems (where trees and agricultural crops or pastures occupy the same land)
- **prevention of forest damage** caused by fires, natural disasters or catastrophic events, and restoring damaged forests
- **investment in improving climate resilience** and environmental value of forest ecosystems
- investment in forest technologies, mobilising, processing and marketing of forest products
- **land management contracts** for forest-environment-climate services and **forest conservation**
- conservation and promotion of forest genetic resources.

Under the ***Carbon Removals and Carbon Farming (CRCF) Regulation***, to receive **certification**, the carbon removals will need to be correctly quantified, deliver additional climate benefits, strive to store carbon for a long time (at least five years for temporary credits and for “several centuries” for permanent credits), prevent carbon leakage, and contribute to sustainability (also referred to as **QU.A.L.ITY criteria**: **QUantification, Additionality, Long-term storage and sustainabilITY**).

The CRCF builds on emerging national initiatives on carbon farming certification such as *Label Bas Carbone* in France, *MoorFutures* in Germany or *Registro Huella de Carbono* in Spain (further details provided in Table 2).

Regarding **carbon farming, eligible activities** include farming and forestry practices such as afforestation, reforestation and activities within sustainable forest management, agroforestry and other forms of mixed farming, use of catch crops, cover crops, conservation tillage and increasing landscape features, conversion of cropland to fallow or set aside areas to permanent grassland, and restoration of peatlands and wetlands.

The CRCF is considered an important **funding instrument for Member States** to incentivise carbon removals on their territories to contribute to national and EU net GHG targets. Moreover, the CRCF is expected to help make other funding instruments, such as the CAP, more effective for climate change mitigation.

It sets the frame for installing **voluntary or compliance markets at national level**s that address private sector actors. **Carbon credits** certified under the CRCF can be considered **additional** because they occur above defined baselines. Because there are currently no compliance targets for the participants in carbon farming projects (forest and land owners), there is also no risk of double counting.

**Table 2.** Examples of carbon farming mechanisms in forestry

| Mechanism | Label Bas Carbonne (LBC) | UK Woodland Carbon Code (WCC) | Klimaskovfunden (KSF) | California's forest offset program US Forest Projects (USFP) | Wald-Klimastandard (WKS) |
|---|---|---|---|---|---|
| **Country** | France | UK | Denmark | US | Germany |
| **Year (since)** | 2019 | 2011 | 2020 | 2012 | 2023 |
| **Volume by 2023** | 388 afforestation projects certified (but not all funded) | 2037 afforestation projects | 94 afforestation projects | 127 US forest protocol projects (end 2022) | 24 projects |
| **Markets** | Voluntary carbon markets | Voluntary carbon markets | Voluntary carbon markets | Compliance market (California's cap-and-trade system) and voluntary carbon markets | Voluntary carbon markets |
| **Payment timing** | At project start | Pending Issuance Units upfront; Woodland Carbon Units once verified | At project start | When credit generated | |
| **How buyers meet sellers** | No central market, but platforms with labelled projects searching funding | No central market, but platforms projects which can be bought, Government is buyer through reverse auction | Reverse auction organised by KSF | California's Cap-and-Trade Program | No governance |
| **Use of certificates** | Contribution claims | Offsetting and contribution claims | Contribution claims | Offsetting and contribution claims | Offsetting and contribution claims |
| **Activities related to forests** | Afforestation<br>Forest restoration<br>Converting coppice forest to high forest | Afforestation | Afforestation | Improved forest management<br>Increasing rotation age<br>Increasing productivity through selective thinning and management of brush and short-lived forest species<br>Increasing stocking on under-stocked areas | Forest restoration after disturbances |
| **Carbon pools** | Forest biomass, soil, litter, products, substitution | Forest biomass, soil, litter | Forest biomass, soil, litter | Forest biomass and forest products | Forest biomass |
| **Quantification** | Production tables, provide conversion models | Provide a simulation model for carbon assessment | Site class assessment, provide simulation model | Provide simulation models – repeated field measurement of stock and harvest volume | Project scenario tool including site and tree species-specific data derived from NFI data. |

**Table 2.** continued.

| Mechanism | Label Bas Carbonne (LBC) | UK Woodland Carbon Code (WCC) | Klimaskovfunden (KSF) | California's forest offset program US Forest Projects (USFP) | Wald-Klimastandard (WKS) |
|---|---|---|---|---|---|
| **Monitoring** | Third party audit year 5 | Review at year 5 and 15. Depending on project size if review each 10 years | Third party audit year 2.5-3.5 and thereafter every 10 years | Measurement every 6 years | Five years after the start of project activities, and then monitored every three to five years |
| **Additionality** | Afforestation: not forest the last 10 years, financial test (on agricultural land relative to agriculture, on nature land relative to natural regeneration of forest) | Legal test; not used for compensation planting; financial tests | Legal test; financial test, no other public support and not be common practice | Legal tests; financial tests; higher stock in project than in neighbouring forest properties | Financial additionality; policy goal not fulfilled |
| **Permanence** | 30 years, after 30 years no restrictions | Permanent, as under forest legislation (if forest converted new forest should be planted) | Permanent (forest cannot be converted back to agriculture as new projects on the forest act) | 100-year time horizon for projects. If project conditions violated owners should buy back issued carbon credits | Exclude forest project prone to high (fire) risk. |
| **Risk** | 10% discount in estimated reductions (>10% in regions with high wildfire risks) | 20 % of reductions enter collective buffer. Risk reducing measures are mandatory | 15 % of reductions enter collective buffer | 17-19 % of reductions enter collective buffer | 15 % of reductions enter collective buffer. Baseline scenario account for different risk factors |
| **Leakage** | No account | An assessment of leakage (outside project area but inside UK) and if >5% then accounted for. | No accounting for international leakage as only addressing national targets. If landowner increases harvest on other areas this will be subtracted from the project effect. No consideration of market effects | Account for activity shifting and market-shifting leakage: (20% of reduced harvest is considered leakage) | Consider leakage not being important |
| **Sustainability** | indicators of side-effects: socio-economic, soil preservation, biodiversity and water | Online reporting on co-benefits: biodiversity, water, community, economy | Several criteria on afforestation practices to ensure biodiversity, water protection, recreation objectives | | Project should follow general guidelines for climate resilient forest management and be compatible with FSC or PEFC certification |

The **voluntary carbon market suffers from different standards** and a lack of comparability of the quality of different certificates. The CRCF intends to provide a common EU framework for removal certification for landowners and land managers in EU Member States that will increase their opportunities for participation in carbon markets.

**Buyers of carbon credits**, on the other hand, will benefit from more transparent rules and more robust methodologies. To further implement the QU.A.L.ITY criteria, the European Commission is currently reviewing existing certification methodologies as well as relevant legislation to collect experiences and good practice, including key issues of certification, quantification, monitoring and reporting, additionality, durability, environmental integrity and transparency.

## 4.3 Carbon farming – challenges for environmental integrity

The **carbon farming approach** promotes **on-site carbon storage**. This helps mitigate uncertainties in broader carbon-focused *payments for ecosystem services* (PES) schemes, where complexities arise from accounting for substitution and product storage effects. These uncertainties mainly revolve around reporting inaccuracies such as double counting or inaccurate biomass to carbon conversion factors (Sato and Nojiri 2019; Jasinevičius et al. 2017).

Nevertheless, there remain **risks** that carbon farming approaches may not deliver on their objectives, thereby compromising their environmental integrity, ie their ability to contribute to effective climate change mitigation without compromising other environmental and social objectives (Haya et al. 2023).

In the following, we draw on a framework for assessing PES schemes initially put forth by Pagiola (2005) and further developed by Engel et al (2008). This framework is equally applicable to carbon farming approaches (for further elaboration, refer to Thamo and Panell 2016).

Ensuring the effectiveness of carbon farming schemes entails employing various approaches with varying levels of rigour, and the selection of an approach significantly impacts the shape, results and overall success of carbon farming initiatives. We illustrate the five key elements of environmental integrity that are to be addressed, relating also to the **QU.A.L.ITY criteria: (1) ensuring additionality, (2) permanence, (3) avoiding leakage, (4) robust quantification and (5) reducing trade-offs and realising synergies with other sustainability goals (Table 3).**

**Table 3.** Summary of potential forest management practices for carbon farming with the QUALITY criteria.

| PRACTICE | Additionality | Permanence | Robust leakage prevention | Level of quantification | Trade-off and synergies |
|---|---|---|---|---|---|
| **Afforestation** | **High**<br>Increased biomass and soil carbon | **High**<br>If the planted forest is sucessfully maintained | **High**<br>If on abandoned farmland | **High**<br>Data availability high<br>Easy to quantify | Biodiversity benefits on cropland<br>Potential soil carbon and biodiversity decline on grassland |
| **Silvicultural practices** | | | | | |
| ***Species selection*** | **Medium**<br>Many forests are already stocked with productive species | **High**<br>Increased soil carbon (long term) | **Medium**<br>If industry demands specific species then it is sourcing them from elsewhere if supply is reduced | **Medium**<br>Only quantifiable on long time horizons | Can incentivise the promotion of productive species that are not adapted to the climate change<br>Greater stability and reslience with climate adapted species<br>Improved biodiversity |
| ***Less harvest/ Lengtheened rotation period*** | **Medium**<br>Can be economically beneficial in some cases<br>**High**<br>For climate additionality | **Medium**<br>If disturbance does not increase, can be quickly reverted when management changes | **Less harvest: Low**<br>High chance that reduced harvest amounts are produced elsewhere<br>**Lengthened rotation: Medium**<br>If harvest reductions are only moderate | **Low**<br>Defining the baseline is challenging | Increased biodiversity with increased tree size and habitat structures |
| ***Reduced thinning intensity*** | **Medium** | **Medium** | **Medium**<br>Reduced supply for pulp and paper | **Medium**<br>Easier for trees but difficult for soils | May increase the risk of various disturbances |
| ***Diversification of forest structure (vertical structure, species and age composition)*** | **Medium** | **High**<br>More resilient and resistant first stands under climate change | **High**<br>If quantitative and qualitative changes of timber are low | **Medium**<br>Requires a long time horizon | Increased biodiversity, stability and resilience |
| ***No harvesting*** | **Medium**<br>If harvest not economically attractive<br>**High**<br>Climate additionality | **Medium**<br>Depends on the disturbance regime | **Low**<br>Wood demand is likely being met from elsewhere | **Medium** | Increases biodiversity |

**Table 3.** continued.

| PRACTICE | Additionality | Permanence | Robust leakage prevention | Level of quantification | Trade-off and synergies |
|---|---|---|---|---|---|
| ***Site fertilisation*** | **Low**<br>Economic additionality as it increases economic value<br>**Medium**<br>Climate additionality due to fossil and N emissions | **Medium**<br>Impact of N fertilisation in boreal conditions is short lasting | **High** | **Medium**<br>Quantification of impact on soils needs to be improved | Increases both tree biomass and soil C sinks and compenstes energy costs for N fertiliser production, N losses into ground and surface waters |
| **Fire management** | **High**<br>High costs for fire management (in the south) | **Medium**<br>Depends on effectiveness of fire management | **High** | **High**<br>With the help of modelling | Saving lives, biodiversity, reducing air pollution |
| **Agroforestry** | **High** | **Medium-high**<br>Systems need to be maintained | **High** | **Low**<br>Mostly little or no carbon inventory data available | Loss of a certain amount of agricultural land, increased diversity of habitats, improved biodiversity, soil protection, recreation value |
| **Peatland management** | | | | | |
| **Peatland restoration** | **High**<br>Loss of revenue and high GHG emission reductions | **High**<br>High GHG emission reductions | **Medium**<br>Wood production is reduced and likely produced elsewhere but with lower GHG emissions (if shifted to mineral soil forests) | **Medium**<br>More research needed for soil GHG emission quantification | Biodiversity benefits, improved water management |
| **CCF on drained peatlands** | **Medium** | **High**<br>High GHG emission reductions, low incentive to change back to rotation forestry | **Medium to high**<br>If harvest amounts are reduced, more harvest may be targeted to upland soils | **Medium**<br>Better soil GHG models are needed | Improves water quality by reduced nutrient loading. Improved water management |

### 4.3.1 QU.A.L.ITY criteria 1: How can we ensure additionality?

In the CRCF, additionality is addressed by requiring carbon farming projects to demonstrate that carbon reductions or removals are directly caused by the respective carbon farming project and would not have occurred without it. This is to be validated through rigorous baseline setting, continuous monitoring and transparent reporting mechanisms.

Three key types of additionality must be ensured.

1. **Legal additionality:** the project goes beyond meeting the minimum legal requirements in a given region. It should implement practices that are not mandated by current laws.

2. **Economic additionality:** the environmental benefits of the project, such as reduced emissions or increased carbon storage, would not happen without the specific funding it receives.

3. **Climate additionality:** the project results in genuine, measurable increases in carbon sequestration or reductions in emissions compared to what would have happened in its absence.

The evaluation of additionality usually involves comparing the actual carbon additional changes driven by the project against a **baseline** or reference scenario. The assessment is typically ex ante where the baseline is established before the project starts and may include simulations of expected development without the project and calculations of land opportunity costs (Wunder 2005; Ferraro 2011; Blanco et al. 2021). Alternatively, ex-post assessments measure carbon additionality over the life of the project, where the baseline is typically represented by similar sites but without a project (see Box 6).

#### Box 6. Additionality examples

In the emerging European initiatives, for example French *Label Bas Carbon* (LBC) or the *UK Woodland Carbon Code* (UKWCC), additionality is typically ensured by conducting an investment analysis justifying that the baseline scenario is economically preferable for the project holder (economic additionality) and that that there is no legal obligation to carry out the project (legal additionality).

In the **French *Label Bas Carbon*** (LBC), climate additionality is documented through a simulation of carbon removals or reduced emissions, comparing baseline and anticipated project scenarios.

The ***Woodland Carbon Code*** (UKWCC), on the other hand, relies on regular on-site measurements rather than simulations.

The **German *Wald-Klimastandard*** (WKS) assumes that, despite the fact that forest restoration by forest owners after calamities is legally required, there is not sufficient funding for establishing forests with higher resilience towards climate change impacts.

**Challenges:** forest ecosystems are complex and dynamic systems, influenced by various biophysical and socio-economic factors. The accurate establishment of reliable baselines for carbon farming projects is therefore highly challenging, and full information on a project scenario and its baseline will never be achievable. Five relevant factors causing uncertainties are:

1. **Climate change:** global warming and associated extreme weather events or forest calamities increase the uncertainty of any forest dynamics projection

2. **Diverse site conditions:** potential project sites differ significantly in their historical background, environmental conditions, regulatory frameworks, and human impact, which complicates the creation of standardised methodologies.

3. **Data availability:** for smaller, privately owned forests, there may be a lack of available current or historical forest inventory data.

4. **External influences:** market trends and policy interventions can significantly affect forest dynamics and are challenging to predict accurately.

5. **Decision-making:** the choices of landowners and managers are influenced by a mix of objective and subjective factors, including forest management plans, silvicultural regulations but also personal values and alternative land-use options, which are often not fully apparent to project developers.

**Recent criticism:** forest-based carbon-related PES schemes, such as Reducing Emissions from Deforestation and Degradation (REDD) and Improved Forest Management (IFM) programmes, have repeatedly been accused of lacking additionality. In a significant proportion of recently evaluated REDD projects, deforestation would not have occurred even if the project had not been established (West et al. 2020; West et al. 2023). Similar uncertainties surround the established baseline scenarios for IFM projects (Haya et al. 2023).

**Suggested measures:**

1. **Use ex-ante testing with ex-post testing:** establishing reliable baselines for carbon farming projects is subject to large uncertainties. Ex ante testing of additionality by merely modelling hypothetical reference scenarios against expected sequestration and storage trajectories therefore runs the risk of failing to reflect reality. Instead, ex-post testing with on-site measurements, or at least spot checks, are essential for the reliable validation of carbon farming projects.

2. **Monitoring through on-site measurements and remote sensing**: The on-site measurements could be accompanied by stringent remote sensing monitoring (see Chapter 3). These monitoring efforts should include validating baseline scenarios of active projects by comparing project sites with similar sites (regarding forest type, structure, productivity, management etc) not included in a carbon farming scheme. For example, applying matching methods (Andam et al. 2008; Haya et al. 2023).

3. This approach allows the expected baseline to be assessed against actual developments in the real world, and potentially the amount of carbon certificates generated to be adjusted throughout the project process (dynamic baselines).

4. **regular updated on baselines**: Moreover, it can enhance the accuracy of both baselines and project scenarios for future projects. Baselines, whether static or dynamic, need to be updated regularly, such as every five years, to account for potential changes in policies and market conditions that were not anticipated at project start.

5. **baseline consistency**: Moreover, consistency of baselines across different projects needs to be achieved in such a way that baselines accurately reflect developments at the national level. This means that the sum of baselines in a country or region must not exceed the national level baseline as, for example, reflected in the country's NDC.

### 4.3.2 QU.A.L.ITY criteria 2: How can permanence be guaranteed?

Carbon farming activities are characterised by the fact that emission reductions or carbon removal may only be temporary, and the sequestered carbon may be released back into the atmosphere at a later point in time (Balmford et al. 2023 – see Box 7). The CRCF therefore differentiates between carbon farming activities, as temporary removals, and permanent removals that store carbon in geological formations for centuries. Reversal of sequestered carbon or reduced emissions can be unintentional (due to natural processes or project failure) or intentional (due to changes in management). Non-permanence refers to the risk of such reversals occurring within or after the conclusion of funding. If carbon reversals occur during the monitoring period, operators are liable under the CRCF.

#### Box 7. Permanence examples

One approach involves regulatory measures and specific contract conditions, as seen in France's *Label bas Carbone* initiative.

This method integrates site-specific risk factors into each contract, accounting for the potential failure of projects due to natural and economic uncertainties when generating carbon credits.

Another method, used in the *Danish* KSF, the German WKS, and the UKWCC, involves creating carbon buffers. A certain portion of carbon credits is held back and not sold to buyers. If a project fails, these reserved credits are used to refill the buffer.

**Challenges**: uncertainties caused by climate change and the associated increase in frequency and intensity of extreme weather events and forest calamities complicate predictions of the permanence of carbon farming programmes. Uncertainty about future management, for example induced by changes in timber demand, add to the risk of not ensuring permanence. Risk factors must therefore be adaptable to a changing, yet unknown, risk environment.

Furthermore, it is important to assess the time profile of the change in carbon – for example, is the rate of removal constant or does it change with the age of the trees, and for how long will there be a net removal of carbon?

**Suggested measures:**

The **risk of non-permanence** should be addressed with:

(1) **Temporal carbon credits:** they are specifically designed to account for the time-sensitive nature of carbon removal. Unlike traditional carbon credits, which assume permanent removals quantified ex ante, temporal credits recognise that forest carbon storage is dynamic and may not be permanent.
    These credits are periodically verified and adjusted according to ongoing assessments of carbon storage, offering a more accurate representation of the true carbon benefits achieved by carbon farming projects. Once these temporal certificates are expired, they are no longer valid and must be replaced.

(2) **Dynamic carbon buffers:** dynamic carbon buffers allow for an adaptation to changes in actual emission reductions and storage performance. Unlike static buffers, which rely on ex-ante hypothetical assumptions about risks, dynamic buffers are more flexible, and their size can be adjusted based on real-time empirical data (for example, data on fire risk).

(3) **Tonne-year-accounting:** implementing tonne-year accounting can help to quantify and track the climate benefits of temporary carbon storage. This method integrates the amount of carbon stored with the duration it remains sequestered, providing a metric proportional to the climate impact of avoided warming (Matthews et al. 2023).

Tonne-year accounting calculates the climate impact of carbon storage by considering both the quantity of carbon and the length of time it is stored. This approach values temporary storage by expressing it in terms of its equivalent impact if the carbon were permanently stored.

For example, storing one tonne of carbon for one year could be considered equivalent to permanently storing a fraction of that tonne, based on the climate benefits over a specified period. This method allows for more nuanced crediting that reflects the actual climate mitigation achieved through temporary storage. This approach would also allow for shorter contract durations, thus lowering the entrance barrier of carbon farming projects.

These proposals to address non-permanence suggest that payments for carbon credits should be made recurrently based on ongoing monitoring and verification, rather than as a single upfront payment.

### 4.3.3 QU.A.L.ITY criteria 3: How can leakage be avoided?

Leakage refers to the unintended consequence of activities being displaced to areas outside the supported region, which can cause increases in emissions or decreases in removals, leading to an inflated estimation of a carbon farming project's net carbon effects (see Box 8).

Empirical studies of leakage in US forest carbon sequestration programmes range from under 10% to more than 90% (Murray et al. 2004). This effect is frequently reported as the *greatest hurdle* in the design and implementation of carbon crediting projects (Alix-Garcia and Wolff 2014; Jack et al. 2008; Plantinga and Richards 2009).

Leakage occurs in **two main forms.**

1. **Activity shifting leakage:** this happens when landowners transfer planned production activities from land within a carbon farming project to another land.

2. **Market leakage:** this occurs when land withdrawn from production affects the general market equilibrium, leading to higher prices for certain commodities and increased pressure on land not covered by the carbon farming initiative (Wu 2000, 2005; Wu et al. 2001; Roberts and Bucholz 2005; 2006; Jack et al. 2008; Alix-Garcia and Wolff 2014).

While activity-shifting leakage can often be reduced through well-defined contract terms and diligent monitoring, market leakage presents an intrinsic and difficult-to-solve dilemma in large-scale forest carbon sequestration scheme implementation (Rose and Sohngen 2011). In the CRCF, leakage is to be addressed through strict monitoring and reporting requirements and through the liability of the operator.

**Market leakage** manifests in

- **Quantitative leakage:** this occurs when the quantity of products produced by the forest changes, potentially leading to increased production elsewhere to meet demand.

- **Qualitative leakage:** this refers to changes in the type of end products supplied by a carbon farming-supported forest and their associated carbon storage performance. Such changes may be caused, for instance, by a modification to the species composition of the supported forest. This is due to differences in the suitability of different tree species for different end uses. Promoting broadleaved or mixed forests may, for example, result in a shift from using coniferous species for timber to broadleaved species for energy wood and thereby reduce the carbon stored in forest products.

### Box 8. Leakage examples

Leakage is not considered in the **LBC**, while addressed in both the UK *Woodland Carbon Code* and the KSF at a national level. Both schemes emphasise national targets, and due to strict legal restrictions on forest conversion to other land uses, afforestation is argued to not contribute to national deforestation.

The ***German Wald-Klimastandard*** assumes that leakage does not occur through projects covered by the methodology, because the amount of timber is not altered. However, the scheme supports changes in species composition, which may lead to qualitative leakage effects due to changes in end products and their carbon storage performance.

**Challenges**: the risk of market leakage is particularly high for projects that negatively affect wood supply (reducing harvesting intensity) or delay wood supply (extending rotation), with estimates ranging up to 89% replacement (see Chapter 4) (Jonsson et al. 2012; Hu et al. 2014; Filewood and McCarney 2023).

Market leakage effects are notably more severe in efficient timber markets, as these markets tend to expand the geographical scope of market transactions and, consequently, the area vulnerable to leakage effects (Murray et al. 2004). This implies that establishing novel, large-scale carbon farming schemes in the EU carries a substantial risk of market leakage effects, with harvesting activities possibly shifting to areas not covered by the scheme, including non-EU countries, significantly altering the scheme's net carbon mitigation performance.

Additionally, an increased demand for carbon certificates could make afforestation on agricultural land more profitable, potentially decreasing the availability of agricultural products. This reduction could then exert pressure on agricultural or potential agricultural land in regions not covered by carbon farming schemes, possibly leading to higher deforestation rates and thus increased carbon emissions.

**Suggested measures** to tackle leakage could involve:

(1) **Expanding geospatial monitoring:** enhance monitoring of carbon stocks and sequestration at European level, including sites not covered by any carbon farming initiative to inform decisions about scheme adaption and expansion.

(2) **Targeting funding:** direct carbon farming funding towards activities that do not significantly reduce or delay harvesting activities, for example restoring degraded forests, minimising the impact on the overall timber supply.

(3) **Harmonising regulations:** align carbon farming regulations with broader EU-level regulations to avoid conflicting incentives, such as those that might arise with the Renewable Energy Directive (RED). For example, the demand for biomass for energy by the RED may increase competition for land and not support cascade use of wood if the use for energy is not constrained, for example due to residues.

(4) **Accounting for leakage:** despite any precautions, leakage will take place in many carbon farming activities. In these cases, leakage effects must be accounted for in the quantification of net climate benefits and the amount of carbon credits generated must be adjusted accordingly.

### 4.3.4 QU.A.L.ITY criteria 4: How can we guarantee robust quantification?

Forest-based carbon farming financed through carbon markets requires **reliable quantification of forest carbon** to calculate emission reductions and removals as a basis for accounting. This includes requirements for the consideration of all relevant pools and gases and the treatment of uncertainty, both of which determine the number of credits being generated (see Box 9).

**Robust accounting** also relies on the comparison of quantified carbon benefits with credible baselines and deductions for leakage. The quantification should also address transparency, legitimacy and accountability, and synergies and trade-offs.

#### Box 9. Quantification examples

Quantification methods among existing standards differ according to the activities covered. They often refer to the IPCC Guidelines that include a description of the scope of emissions and removals to be considered and respective equations, procedures, list of pools and gases and default values to be used in the case of missing data.

The French LBC's method for afforestation considers carbon removals and emissions from changes in above- and below-ground tree biomass, soil carbon and litter as well as emission reductions related to wood products substitution effects. The latter two are optional.

The German *Wald-Klimastandard* only considers above- and below-ground biomass.

**Challenges**: quantification is not only difficult with respect to identifying relevant baselines, which form the basis for ensuring additionality. The high spatial and temporal variations make a precise measuring of, for instance, SOC difficult and costly.

Furthermore, it is important that quantification considers the time profile of carbon removals, so average annual carbon removals should be associated with information about the time horizon for this removal rate. Quantification should also assess the degree to which the removals are permanent (see 4.3.2)

Transparency of instruments (for example for measurement and monitoring) is crucial for accountability and legitimacy and, thus, for long-term success (Delacote et al. 2024).

Assessment of synergies and trade-offs also contributes to the legitimacy of carbon-farming approaches. In a forest context, forest changes in management will not only influence SOC and timber production but many other ecosystem services. However, transparent quantification and rigorous monitoring are costly and often not economically viable for project developers and buyers (Paul et al. 2023). Cost-effective governance is also hampered by a lack of regulatory requirements or common standards.

**Suggested measures**: baseline scenarios form the foundation of any carbon quantification.

1. **Baseline setting and conservative approaches**: Appropriate baseline setting and conservative approaches are therefore essential for any carbon farming initiative to ensure that overestimates are avoided. A conservative approach means adopting methodologies that systematically underestimate the emission reductions or carbon removals of activities. Excluding carbon pools or gases is often justified with conservativeness but proofs must be provided. In particular, accounting for the temporal aspects of carbon farming removals is crucial for accurate quantification (see 4.3.2).

2. **legal framework**: Development of a legal framework and widely applied standards may be a way forward to reduce transaction costs and facilitate a market for credible certificates. Thus, CRCE certification can potentially lead to a boost in the adoption of carbon farming initiatives.

### 4.3.5 QU.A.L.ITY criteria 5: How to reduce trade-offs and realise synergies with other sustainability goals

Carbon farming activities can have multiple impacts on forest ecosystems, including effects on water cycles, soil fertility, biodiversity and habitat protection, and provision of recreational services (Lin et al. 2013; Asbeck et al. 2021). These effects can be positive or negative (see Box 10).

When dealing with carbon farming activities, a **common risk** is not considering both positive and negative side effects. To address this, the CRCF framework uses a "*Do No Significant Harm*" principle and adds that carbon farming activities should, as a minimum, generate co-benefits for the objective of protecting and restoring biodiversity and ecosystems, including soil health and preventing land degradation (Council of the European Union Commission 2024).

#### Box 10. Co-benefits examples

Most European programmes have co-benefit evaluation and reporting requirements. It may also be required to include socio-economic indicators (French LBC and UK *Woodland Carbon Code*) that, for example, describe the potential impacts on the local economy and employment of a given project.

Consideration of co-benefits also implies restrictions on carbon farming projects – for example, restrictions on the tree species that can be used for afforestation and forest restoration. It may also be required that the forest will be certified by FSC or PEFC (in the case of the German WKS).

In the French LBC projects, the co-benefits are reported using a system of indicators for which a project can obtain scores between one and five based on a qualitative assessment of the project.

**Challenges:** consideration of carbon mechanisms independent of co-benefits such as biodiversity conservation would not be optimal (Tedersoo et al. 2024). For example, carbon farming may favour exotic, fast-growing tree species that have a negative impact on biodiversity. However, due to the multiple effects of changes in land use or forest management, quantification can be difficult and costly.

Additionally, the dynamic nature of ecosystems and the variability of responses to different management practices pose significant challenges for consistent assessment (Smith et al. 2019; Thompson et al. 2020). At the same time, the acceptance of carbon finance by credit buyers and the general public will depend on the absence of negative side effects from carbon projects.

**Suggested measures:**

a. **Cost-effective methods development**: identifying synergies with other sustainability objectives will require the **development of cost-effective methods** to assess the co-benefits of carbon farming, including new technologies, high-throughput DNA sequencing, analysis of environmental samples to measure biodiversity, or the establishment of best practice standards.

b. **Integrated ecosystem assessment framework**: Using integrated ecosystem assessment frameworks can help in evaluating the multiple benefits provided by forest-based carbon projects (Egoh et al. 2007; Nelson et al. 2009). Engaging local communities in monitoring and reporting can also enhance the assessment of socio-economic co-benefits and increase project acceptance (Danielsen et al. 2010; Chhatre and Agrawal 2009).

In addition, creating standardised metrics for biodiversity and ecosystem services will facilitate the comparison and aggregation of co-benefit outcomes across different projects and regions (Mace et al. 2018). Ensuring transparency in the assessment of co-benefits will also support the credibility of carbon credits.

## 4.4 How can carbon credits be used?

Carbon **removal units or certificates** can, in principle, be **used for various purposes** (see Box 11). Possible uses include:

1. **Compliance use of** units under EU climate regulations for achieving national or EU climate targets, thereby offsetting emissions covered by these regulations (EU NDC, EU ETS, LULUCF Regulation, Effort Sharing Regulation).

2. **Compliance use of units by other non-national compliance systems**, such as the Carbon Offsetting and Reduction Scheme for International Aviation (CORSIA), offsetting international aviation emissions.

3. **Compliance use of units under other** EU, national or sub-national **policies**, such as the EU Fuel Quality Directive,

4. **Voluntary use of units for offsetting** by companies, institutions, jurisdictions or individuals.

5. **Voluntary use of units for purposes** other than offsetting, including contribution claims, or using certificates for labelling purposes for meeting legal requirements.

### Box 11. Carbon removal credits examples

Carbon removal credits are mostly used to 'offset' or balance out GHG emissions associated with a buyer's activities on the voluntary carbon market (LBC, CWC, WKS).

This option is attractive to buyers as it provides them with a concrete product (carbon credits) that can easily be integrated in a company's environmental reporting system and can be related to production units, such as per unit of product.

However, offsetting contributes to overall mitigation only if the removals can be considered permanent, ie they remove carbon from the atmosphere for several centuries.

**Challenges:** the offsetting uses make removals and emission reductions from carbon farming activities and fossil fuel emissions equivalent, despite their inherent differences. This bears considerable risk for environmental integrity.

As discussed above, unlike carbon stored in coal, gas or oil in the ground, **storage of carbon** in biomass and soil organic matter is **only temporary**. The equivalence can also be questioned regarding the above discussed issues concerning additionality and complexities in measuring carbon.

The **issue of equivalence** is also illustrated by a massive discrepancy between prices for carbon farming offsetting certificates and prices of $CO_2$ at the European ETS market or costs estimated for the damage caused by the emission of one tonne of $CO_2$ (200 EUR/t $CO_2$).

**In conclusion,** it is of utmost importance that fossil fuels and biogenic emissions are reduced as quickly as possible to avoid dangerous climate change impacts. It has been questioned to what extent the use of credits from carbon farming for offsetting will be globally effective (Paul et al. 2023). This is due to their **temporary storage** character that is not suitable for permanent offsetting of emissions. An alternative approach makes use of the certificates to promote global climate protection via private financial contributions. A key difference is that the participating companies cannot count the removals towards their own targets of becoming climate neutral, but declare it as a contribution to finance climate protection. So far, however, this approach has not been widely implemented.

# 5 Conclusions and recommendations

Forests occupy almost 40% of the European Union's land area and constitute the main terrestrial carbon sink. The European Union strongly relies on forests to achieve its target of climate neutrality by 2050 and has put in place a framework to implement carbon farming initiatives and develop a regulatory framework for the accounting and certification of carbon removals from the atmosphere. However, whether EU forests can help meet this ambitious goal in such a short time is questionable and there are significant challenges.

First of all, **forests in Europe are aging** and are, despite some afforestation activities, already showing a clear decline in their carbon sink potential, which is also a result of climate change. 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.

**Forest policies** 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. Moreover, **forest management practices** must take into consideration a holistic approach with **several forest functions**, and so there are also environmental, economic and social perspectives.

We have analysed the current scientific literature and identified the main socioeconomic challenges and problems for the implementation of carbon farming practices in the forestry sector. We have also reviewed the scientific evidence of management practices potentially suitable for carbon farming projects, and we have remarked on current methodological problems and drawbacks. In the following sections we provide recommendations based on our findings.

## 5.1 Forest management practices

**A challenge for enhancing forest carbon is the long and variable time-scale inherent in forestry activities**, while natural disturbances can lead to rapid carbon losses.

Some carbon farming practices can lead to increased carbon sequestration immediately (such as *afforestation or reduced harvesting*), while others require longer lead times (10 years or more as for example, *tree species change or changes in forest structure*). Some carbon farming practices can sequester carbon for centuries and have a high degree of permanence, such as *rewetting of peatlands*, while others can sequester additional carbon for a period until a new steady state is reached (such as *longer rotation period*).

Management must therefore balance the potential benefits (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.

All this must be considered when selecting a carbon farming practice. Some important aspects related to the application of carbon farming practices are summarised below.

- **Afforestation and forest conservation** are essential to counteract deforestation and to maintain the current forest carbon sink.
  *Caution* needs to be taken in the area selected to avoid negative impacts on other ecosystem services. The use of productive pastures and arable land for afforestation purposes can lead to adverse effects on food production, biodiversity, disturbance regimes and ecosystem services trading off with carbon uptake such as water provision.

- Practices such as **reduced harvests** (final and thinning) and/or **longer rotations** increase carbon stocks in tree biomass and soils, thereby sequestering additional carbon dioxide from the atmosphere.

  Carbon sequestration is permanent if natural disturbances such as windthrow, fires, diseases or insect outbreaks do not result in the release of sequestered carbon back into the atmosphere. As the susceptibility of many tree species to disturbance increases with tree height and age, the risk of carbon loss is generally higher in older and taller forests than in younger forests, particularly in unsheltered locations. Tree species composition, age structure, local disturbance agents and their interaction with climate change need to be carefully considered.

- Promoting **tree species selection** for increasing carbon sequestration, particularly on degraded forest areas, whereas in managed forests the potential for species change to maximise carbon sequestration is limited. Tree species diversification and mixing are among the most effective strategies for climate change adaptation and are already implemented in many management plans.

  Adaptive changes in tree species therefore represent current management baselines, *making it difficult to argue for additionality* in carbon sequestration. In any case, carbon farming is not intended to provide an incentive to stick with fast-growing but climate-sensitive species instead of switching to climate-adapted and more resilient tree species, even though these latter species may have lower carbon sequestration potential in the short term.

- For some **forest types**, fostering the shift from clear-cutting towards closer to nature forestry (uneven-aged stands; continuous cover forestry). In particular, the positive effects on carbon sequestration in soils may make closer to nature forestry superior to clear-cutting.

  However, *conversion to closer to nature forestry* is a long-term process and is often associated with carbon losses in the short term, while in the long term, the benefits for carbon sequestration should outweigh.

- Despite the fact that forest **carbon uptake** can be tangible in the long run, given that European forests are suffering a densification process, forest management is required to enhance forest resilience with respect to some hazards (such as drought). Selective cutting is a potential measure to increase natural forest productivity and preserve existing forest carbon stocks while improving forest resilience.

- Under certain **conditions**, promoting the *conversion of coppices* into high forests, which leads to an increase in carbon sequestration in forest biomass in the long term and reduces forest vulnerability to drought and other hazards. The impact of conversion on soil carbon is still unclear.

- **Forest fertilisation** (for example, with nitrogen fertiliser) can increase tree growth and thus carbon sequestration in biomass for a certain period and also reduce the decomposition of organic matter in soils, thereby increasing soil carbon sequestration.

  However, *negative environmental effects* such as nitrogen pollution of surface and groundwater or loss of biodiversity limit its use in many European regions. Liming or wood ash application is less problematic in terms of nitrogen losses, but the effects on tree growth and especially soil carbon are poorly quantified so far.

- **Afforestation and restoration** require systematic planning to reconcile mitigation objectives with biodiversity conservation and multifunctionality at local and regional scale. Both local (stand to forest) and landscape to regional scales must be considered in the analyses to generate sustainable territorial practices.

An overview of potential carbon farming measures in forests and their pros and cons is provided in Table 3.

## 5.2 Measurement challenges

Establishing reliable systems for monitoring and reporting carbon sequestration is critical to verify the impact of carbon farming. A robust monitoring, reporting and verification (MRV) system can help ensure transparency and accountability, and provide data to support policy and market development. Important methodological challenges need to be addressed in order to make scientific advances available for policymakers and forestry practitioners.

- **Models** can contribute to **increasing the reliability** of carbon estimations. There are solid estimation and modelling methodologies, covering a wide range of spatial and temporal scale of analysis, which are accessible to foresters and applied scientists. Upscaling forest inventory observations and LiDAR biomass estimations through empirical modelling approximations that combine information from both sources is recommended.

- **Measuring** carbon sequestration in **above-ground forest biomass is less demanding and cost intensive** than measuring/modelling carbon sequestration in forest soils. However, there are cost-efficient empirical approximations that can be incorporated in forest inventory surveys.

- **Hybrid approaches** involving Remote Sensing multiple observation and ground data are one of the most promising approaches to obtain spatialised information on land use changes and forest carbon dynamics. Yet RS products alone can provide valuable spatialised information in areas lacking field surveys.

- **Dynamic modelling approximations** are also available and provide a more integrated perspective on carbon farming by considering other aspects such as durability and sustainability. However, these large-scale approximations **require local validation** to account for specific land use, management and site characteristics of different European forests. A long-term perspective is key in forest mitigation. Carbon stocks must be evaluated at periods in agreement with biological processes operating in forests (ie, secondary succession) as well as forest management planning, which can imply rotation periods longer than a century. This implies the use of modelling that includes disturbances such as wild or prescribed fires or catastrophic storms, which imply processes of several decades to stabilise forest carbon.

## 5.3 Socioeconomic remarks and recommendations

Forest carbon sequestration in European forests can be achieved through three main pathways:

(i) by **protecting** current forests to avoid carbon emissions to the atmosphere (for example, fire, insects, windthrow)

(ii) by **promoting** forest practices that enhance biomass and soil carbon stocking

(iii) by **increasing** the forest area and generating forests that are more resilient to climate change hazards.

However, setting baselines and verifying carbon removal and gains resulting from afforestation and reforestation, forest protection and silviculture presents important challenges, as previously discussed. As a principle, under any form of carbon farming, **additionality requires "proof" of a lower carbon sequestration** in the absence of the adopted carbon farming practice. Carbon farming needs to deliver multiple environmental co-benefits, such as biodiversity conservation, water regulation, or soil health improvement among other ecosystem services. Practices that enhance these co-benefits should be prioritised.

Some other important aspects are summarised below.

- There is an **urgent need to establish standardised methodologies with transparent guidelines for baseline development** (for example, a business as usual scenario). It is central that methods for estimating baselines align with national-level reporting standards and policy objectives. Furthermore, they need to be regularly adjusted to reflect changes in policy, market dynamics and the climate environment. In this regard, dynamic baselines are preferred to static baselines.

- **Market leakage** is a key issue that can challenge net carbon benefits, considering that measures aimed at reducing harvest intensity offer the greatest potential for credit generation but also carry the highest risk of market leakage. Therefore, new regulations must prioritise market leakage prevention, along with implementing rigorous and transparent accounting practices for any residual leakage effects.

- Addressing non-permanence in carbon farming activities requires adaptive strategies that account for the temporary nature of forest carbon storage. Given the risk of carbon reversals due to natural events or management changes, the adoption of dynamic measures is essential.

- Recommended approaches are the use of temporal carbon credits, which are periodically verified and adjusted, and dynamic carbon buffers that can be resized based on real-time data. Additionally, implementing tonne-year-accounting, which integrates the amount of carbon stored with the duration it remains sequestered, offers a way to measure the climate impact of temporary carbon storage more accurately by expressing it as an equivalent to permanent storage. These methods ensure a more flexible and realistic representation of carbon benefits, encouraging continuous monitoring and recurrent payments for carbon credits, rather than single upfront payments.

- Carbon farming **methodologies need to be prescriptive in defining the exact scope** for removal projects. Excluding carbon pools and emissions sources from quantification can be conservative, after demonstrating that the pool is not representing a source. This should, however, be determined by the methodology itself and not left to the discretion of individual projects.

- **Conflicting policies (directive/regulations) can undermine the results**. For instance, assessing voluntary carbon market methodologies and national and EU level policies reveals that there are potentially conflicting goals. For example, the *Renewable Energy Directive* supports the use of biomass for bioenergy, while national schemes for payments for ecosystem services provide funding for lowering wood harvest. Such conflicts need to be resolved and policy needs to provide clear priorities for the development of strategies.

- **Activities funded by voluntary carbon markets** need to be **visible in the country's national GHG inventory**. GHG inventories are the main tool for Member States to steer the country towards achieving the national targets and assess compliance.

- **Carbon farming methodologies** must include provisions that removals should be determined **conservatively**, rather than using the most accurate estimate. The degree of conservativeness should be based on the magnitude of uncertainty associated with estimating removals (so, in cases of high uncertainty, approaches should be more conservative).

- An **alternative use of credits** should be foreseen. A major factor for the attractiveness of credits from carbon farming in forestry of existing standards is the question of whether credits can be used for offsetting GHG emissions. Striving for GHG neutrality enterprises or products is a major driver for the expansion of the voluntary carbon market. However, market-based approaches with an emphasis on carbon credits bear the risk of overestimating net effects on the atmosphere due to several incentives to do so.

Credits do not necessarily have to be used for offsetting. Instead, alternative uses could be pursued, including compliance use of units for contribution claims, or for getting access to subsidy schemes. Such options lower the risk of overestimating the contribution of carbon farming to climate mitigation but need to be made mandatory by policy to increase demand.

## 5.4 Conclusions

Carbon farming in forests is suggested as a way towards increasing land $CO_2$ uptake in EU Member States and some companies have already seized the opportunity to provide carbon certification schemes for forest owners in the so-far poorly regulated private carbon market. EU legislation is in the process of formalising the definition of regional baselines and reporting procedures. However, this is very challenging and the outcome is still uncertain.

- The number of easily applicable management practices suitable for carbon farming in forests is limited for several reasons.

- Many measures require **long time spans or lead times** (10 years or more) before they provide a carbon benefit (tree species change, etc), making them difficult to implement in certification schemes that operate on decadal or even shorter time scales.

- Other potential carbon farming measures **suffer from methodological** (quantification) **problems** – in particular, changes in soil carbon are difficult to measure and quantify, and there is a great need for method harmonisation and improvement.

- Nevertheless, measures such as **reduced harvesting or afforestation of arable land** are already being implemented in certification schemes, as carbon sequestration in forest biomass is readily measurable.

The success of such measures may critically depend on regional forest risk exposure (fire, wind, pests, etc) 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), whereas adaptation rather than mitigation policies should be prioritised in regions where forests are already on the brink.

# Annex 1

## Literature review

The literature review was conducted using a common review protocol by searching relevant articles from the scientific, peer-reviewed literature from Scopus and Google Scholar using mainly the following keywords: (a) forest management, (b) carbon sequestration, (c) soil organic carbon, (d) carbon farming, (e) mitigation, (f) European, (g) biodiversity, (h) greenhouse gas emissions.

- The search was mainly limited to publications from the last 10 years (2013-2023), including reviews and reports.

- Paper abstracts were screened to identify studies reporting on the carbon sequestration potential in biomass and/or soils of different forest management practices.

- The geographical scope is limited to the EU-27 and Norway to represent boreal, temperate and Mediterranean forests.

- Data on forest biomass and/or soil organic carbon were extracted from all selected studies and stored in a database, where the data was harmonised to facilitate comparison.

- For each study, the carbon sequestration rate of selected management practices was taken directly from the study or estimated by calculating the amount of emission reduction and/or carbon removal attributed to a specific forest management activity compared to a business as usual scenario (baseline).

- A total of 34 studies were found in the boreal zone, 49 in the temperate zone and 35 in the Mediterranean zone. These publications included a total of 349 case studies reporting changes in above-ground (83) and/or soil organic carbon (72). See Table 4 for all the studies included.

**Table 4.** Summary of the studies included in this report for the different biomes in Europe: boreal, temperate and Mediterranean

| Management | Country | Species | Age | Native | Type | AGB rate | SOC rate | Reference |
|---|---|---|---|---|---|---|---|---|
| **BOREAL** | | | | | | Mg $CO_2$ ha$^{-1}$yr$^{-1}$ | Mg $CO_2$ ha$^{-1}$yr$^{-1}$ | |
| **Afforestation cropland** | Finland | *Picea abies* | 10 | | Conifer | 2.53 | -0.46 | Tupek et al. 2021 |
| | Finland | *Picea abies* | 50 | | Conifer | 5.93 | 0.73 | Tupek et al. 2021 |
| | Finland | *Betula spp.* | 10 | | Broadleaves | 0.40 | -0.37 | Tupek et al. 2021 |
| | Finland | *Betula spp* | 50 | | Broadleaves | 4.68 | 1.32 | Tupek et al. 2021 |
| | Sweden | *Betula spp.* | 9 | | Broadleaves | 3.04 | -2.53 | Rytter & Rytter 2020 |
| | Sweden | *Picea abies* | 9 | | Conifer | 2.82 | -3.51 | Rytter & Rytter 2020 |
| | Sweden | *Populus spp.* | 9 | | Broadleaves | 4.21 | -2.12 | Rytter & Rytter 2020 |
| | Denmark | *Picea abies* | 29 | | Conifer | 15.32 | - | Vesterdal et al. 2002 |
| | Denmark | *Oak spp* | 29 | | Broadleaves | 9.89 | - | Vesterdal et al. 2002 |
| ***Afforestation grassland*** | Finland | *Picea abies* | 10 | | Conifer | 2.38 | 0.40 | Tupek et al. 2021 |
| | Finland | *Picea abies* | 50 | | Conifer | 4.61 | 0.48 | Tupek et al. 2021 |
| | Finland | *Betula spp.* | 10 | | Broadleaves | 0.48 | 0.44 | Tupek et al. 2021 |
| | Finland | *Betula spp.* | 50 | | Broadleaves | 5.49 | 0.81 | Tupek et al. 2021 |
| | Norway | *Picea abies* | 50 | | Conifer | - | 0.07 | Strand et al. 2021 |
| ***Peatland management*** | Global | | | | Both | - | 3.40 | Mander et al. 2024 |
| | Finland | *Picea abies* | | | Conifer | - | 0 | Peltoniemi et al. 2023 |
| | Finland | *Picea abies* | | | Conifer | - | - | Rissanen et al. 2023 |
| | Finland | *Picea abies* | | | Conifer | 0.15 | 0.11 | Lehtonen et al. 2023 |
| | Finland | *Picea abies* | | | Conifer | 10.65 | - | Aro et al. 2020 |
| | Scandinavia | *Pinus sylvestris* | | | Conifer | - | 0.37 | Wilson et al. 2016 |
| | Scandinavia | *Pinus sylvestris* | | | Conifer | - | 0.26 | Wilson et al. 2016 |
| | Scandinavia | *Pinus sylvestris* | | | Conifer | - | 5.38 | Wilson et al. 2016 |
| ***Ash fertilisation*** | Finland | *Picea abies/Pinus sylvestris* | | | Conifer | 4.76 | -0.48 | Ojanen et al. 2019 |
| | Sweden | *Picea abies* | | | Conifer | 11.45 | - | Van Sundert et al. 2021 |
| | Finland | *Pinus sylvestris* | | | Conifer | 3.44 | - | Moilanen et al. 2015 |
| | Finland | *Pinus sylvestris* | | | Conifer | 5.49 | - | Moilanen et al. 2013 |
| | Norway | *Picea abies* | | | Conifer | 3.70 | - | Hanssen et al. 2020 |
| | Finland | *Pinus sylvestris* | | | Conifer | 3.88 | - | Hytonen 2016 |

**Table 4.** continued.

| Management | Country | Species | Age | Native | Type | AGB rate | SOC rate | Reference |
|---|---|---|---|---|---|---|---|---|
| ***Longer rotation period*** | Sweden | *Pinus sylvestris/Picea abies* | | | Conifer | 4.21 | - | Peichl et al. 2023 |
| | Finland | *Pinus sylvestris* | | | Conifer | 0.66 | - | Akujärvi et al. 2019 |
| | Norway | *Picea abies* | | | Conifer | 4.58 | - | Stokland et al. 2021 |
| | Finland | *Pinus sylvestris/Picea abies* | | | Conifer | 2.34 | - | Trivino et al. 2017 |
| **TEMPERATE** | | | | | | | | |
| **Afforestation cropland** | Latvia | *Picea abies/Pinus sylvestris* | 15 | | Conifer | 6.41 | - 0.84 | Petaja et al. 2023 |
| | Latvia | *Betula* spp. | 15 | | Broadleaves | 6.22 | - 1.83 | Petaja et al. 2023 |
| | Czech Republic | *Fagus/Quercus/Acer/Tilia* | 14 | | Broadleaves | 16.03 | 6.22 | Cukor et al. 2022 |
| | Czech Republic | *Picea abies* | 14 | | Conifer | 18.67 | 5.49 | Cukor et al. 2022 |
| | Czech Republic | *Populus/Alnus/Acer* | 52 | | Broadleaves | 9.15 | - | Vacek et al. 2022 |
| | Czech Republic | *Spruce/Larix* | 52 | | Conifer | 8.60 | - | Vacek et al. 2022 |
| | Czech Republic | *Spruce/Larix* | | | Conifer | 12.27 | - | Zeidler et al. 2022 |
| | Poland | *Pinus sylvestris* | 10-50 | | Conifer | - | 1.24 | Smal et al. 2019 |
| **Afforestation grassland** | Italy/Germany | *Picea abies* | 93-112 | | Conifer | 10.06 | 1.06 | Thuille and Schulze 2006 |
| | Germany | *Pinus sylvestris* | 120 | | Conifer | - | 0.91 | Heinsdorf 1994 |
| | Ireland | *Fraxinus /Alder/* | 4 | | Broadleaves | 2.30 | 0.91 | Peichl et al. 2010 |
| | Ireland | *Fraxinus* ssp. | 12 | | Broadleaves | 9.95 | -1.02 | Wellock et al. 2014 |
| | Ireland | *Fraxinus* ssp | 20 | | Broadleaves | 8.60 | -3.04 | Wellock et al. 2014 |
| | Ireland | *Fraxinus* ssp | 27 | | Broadleaves | 5.85 | -3.18 | Wellock et al. 2014 |
| | Ireland | *Fraxinus* ssp | 40 | | Broadleaves | 7.39 | -1.97 | Wellock et al. 2014 |
| | Switzwerland | *Norway spruce* | 40 | | Conifer | - | 0.37 | Speckert et al. 2023 |
| | Switzwerland | *Picea* ssp. | 25-120 | | Conifer | 12.63 | 0 | Hiltbrunner et al. 2013 |
| **Fertilisation** | Germany | *Fagus sylvatica* | 0-145 | | Broadleaves | - | -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* | | | Conifer | - | -1.68 | Hommeltenberg et al. 2014 |
| | Germany | *Alnus glutinosa* | | | Broadleaves | 47.43 | - | Schweiger et al. 2021 |

**Table 4.** continued.

| Management | Country | Species | Age | Native | Type | AGB rate | SOC rate | Reference |
|---|---|---|---|---|---|---|---|---|
| **MEDITERRANEAN** | | | | | | | | |
| ***Afforestation cropland*** | Spain | *Pinus sylvestris* | 52 | Native | Conifer | 13.58 | - | Ruiz Peinado et al. 2016 |
| | Spain | *Pinus pinaster* | 59 | Native | Conifer | 8.23 | - | Ruiz Peinado et al. 2013 |
| | Italy | *Pseudotsuga menziesii* | 11 | Plantation | Conifer | 10.58 | - | Coletta et al. 2016 |
| | Spain | *Ceratonia siliqua* | 26 | Native | Broadleaves | - | 2.60 | Palacios Rodríguez et al. 2022 |
| | Spain | *Quercus suber/ilex* | 11 | Native | Broadleaves | - | 0.66 | Renna et al. 2024 |
| | Spain | *Pinus halepensis* | 18 | Native | Conifer | - | 2.01 | Segura et al. 2016 |
| | Spain | *Populus* ssp. | 10 | Plantation | Broadleaves | - | 6.73 | García Campos 2022 |
| | Spain | *Populus* ssp. | 5-10 | Plantation | Broadleaves | - | 7.10 | Sierra et al. 2013 |
| | Spain | *Populus* ssp. | 20-30 | Plantation | Broadleaves | - | 3.30 | Sierra et al. 2013 |
| | Spain | *Populus* ssp. | 50-100 | Plantation | Broadleaves | - | 1.98 | Sierra et al. 2013 |
| | Italy | *Quercus/fraxinus/salix/populus* | 16 | Native | Broadleaves | 9.52 | - | Magnani et al. 2005 |
| | Italy | *Fraxinus/prunus/quercus* | 9 | Native | Broadleaves | 12.44 | - | Alberti et al 2006 |
| | Italy | *Juglans regia* | 34 | Native | Broadleaves | 3.73 | - | Certini et al. 2023 |
| | Italy | *Quercus robur/Alnus* | 27 | Native | Broadleaves | - | 1.68 | Chiti et al. 2011 |
| | Italy | *Eucaliptus* ssp. | | Plantation | Broadleaves | - | 1.79 | Novara et al. 2012 |
| ***Afforestation Grassland*** | Spain | *Eucaliptus* globulus | 10 | Plantation | Broadleaves | 28.29 | -0.87 | Perez-Cruzado et al 2012 |
| | Spain | *Eucaliptus* globulus | 15 | Plantation | Broadleaves | 36.12 | 0.62 | Perez-Cruzado et al 2012 |
| | Spain | *Eucaliptus* globulus | 20 | Plantation | Broadleaves | 40.00 | 0.66 | Perez-Cruzado et al 2012 |
| | Spain | *Eucaliptus nitens* | 10 | Plantation | Broadleaves | 37.29 | -2.05 | Perez-Cruzado et al 2012 |
| | Spain | *Eucaliptus nitens* | 15 | Plantation | Broadleaves | 44.07 | -0.18 | Perez-Cruzado et al 2012 |
| | Spain | *Eucaliptus nitens* | 20 | Plantation | Broadleaves | 47.58 | 0.18 | Perez-Cruzado et al 2012 |
| | Spain | *Pinus radiata* | 30 | Plantation | Conifer | 35.17 | 1.02 | Perez-Cruzado et al 2012 |
| | Spain | *Pinus radiata* | 35 | Plantation | Conifer | 36.20 | 0.99 | Perez-Cruzado et al 2012 |
| | Italy | *Pseudotsuga menziesii* | 11 | Plantation | Conifer | 10.58 | - | Coletta et al. 2016 |
| | Spain | *Pinus radiata* | 10 | Plantation | Conifer | 28.80 | - | Fernández-Núñez et al. 2010 |
| | Spain | *Betula* ssp. | 10 | Plantation | Broadleaves | 5.60 | - | Fernández-Núñez et al. 2010 |
| | Spain | *Pinus sylvetsris* | 49 | Native | Conifer | - | -0.58 | Nadal Romero et al. 2016 |
| | Spain | *Pinus nigra* | 49 | Native | Conifer | - | 0.10 | Nadal Romero et al. 2016 |

**Table 4.** continued.

| Management | Country | Species | Age | Native | Type | AGB rate | SOC rate | Reference |
|---|---|---|---|---|---|---|---|---|
| | Spain | *Pinus sylvetsris* | 55 | Native | Conifer | - | -0.55 | Campo et al. 2019 |
| | Spain | *Pinus nigra* | 55 | Native | Conifer | - | 1.57 | Campo et al. 2019 |
| | Italy | *Pinus nigra* | 35 | Native | Conifer | 13.36 | - | Iovino et al. 2021 |
| | Italy | *Castanea sativa/Quercus* spp. | 10-25 | Native | Broadleaves | 8.97 | - | Iovino et al. 2021 |
| ***Agroforestry*** | Italy/Spain | *Malus/Pyearus/Prunus* | | Native | Fruit | 19.40 | - | Kay et al. 2019 |
| | Spain | *Prunus ssp.* | | Native | Fruit | 4.98 | - | Lopez Bellido et al (2016) |
| | Italy | *Olea europaea* | | Native | Fruit | 8.16 | - | Proietti et al. 2012 |
| | Spain | *Quercus ilex* | | Native | Native | 0.47 | - | Kay et al. 2018 |
| | Spain | *Quercus suber* | | Native | Native | 3.00 | - | Kay et al. 2019 |
| | Spain | *Quercus suber* | | Native | Native | 3.00 | - | Palma et al. 2014 |
| | Med | *Paulownia* ssp | | Plantation | Plantation | 12.44 | - | Kay et al. 2019 |
| | Italia | *Quercus suber* | | Native | Native | - | 1.65 | Francaviglia et al. 2012 |
| | France | *Juglans regia* | | Native | Native | - | 1.06 | Cardinael et al. 2017 |
| ***Longer rotation*** | Spain | *Pinus sylvestris* | | Native | Conifer | 10.87 | - | Moreno Fernández et al. 2015 |
| | Spain | *Pinus sylvestris* | | Native | Conifer | 13.73 | - | Moreno Fernández et al. 2015 |
| | Spain | *Pinus sylvestris* | | Native | Conifer | 6.88 | - | Pérez Cruzado et al. 2012 |
| | Spain | *Eucaliptus/Pinus radiata* | | Plantation | Conifer | 36.75 | 0.07 | Pérez Cruzado et al. 2012 |
| ***Harvest Intensity*** | Spain | *Pinus sylvestris* | | Native | Control | 6.04 | 4.24 | Bravo-Oviedo et al. 2015 |
| | Spain | *Pinus sylvestris* | | Native | Light | 5.78 | 4.06 | Bravo-Oviedo et al. 2015 |
| | Spain | *Pinus sylvestris* | | Native | Moderate | 5.74 | 4.61 | Bravo-Oviedo et al. 2015 |
| | Spain | *Pinus sylvestris* | | Native | Control | 13.58 | 6.44 | Bravo-Oviedo et al. 2015 |
| | Spain | *Pinus sylvestris* | | Native | Moderate | 10.43 | 6.37 | Bravo-Oviedo et al. 2015 |
| | Spain | *Pinus sylvestris* | | Native | Heavy | 9.04 | 6.40 | Bravo-Oviedo et al. 2015 |
| | Spain | *Pinus pinaster* | | Native | Control | 8.24 | 7.21 | Ruiz-Peinado et al. 2013 |
| | Spain | *Pinus pinaster* | | Native | Moderate | 6.11 | 7.03 | Ruiz-Peinado et al. 2013 |
| | Spain | *Pinus pinaster* | | Native | Heavy | 10.58 | 6.44 | Ruiz-Peinado et al. 2013 |
| | Italy | *Pseudotsuga menziesii* | | Plantation | Light | 16.25 | - | Coletta et al. 2016 |
| | Italy | *Pseudotsuga menziesii* | | Plantation | Moderate | 16.72 | - | Coletta et al. 2016 |
| | Italy | *Pseudotsuga menziesii* | | Plantation | Heavy | 17.31 | - | Coletta et al. 2016 |

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