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On the role of forests and the forest sector for climate change mitigation in Sweden

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Abstract
We analyse the short- and long-term consequences for atmospheric greenhouse gas (GHG) concentrations of forest management strategies and forest product uses in Sweden by comparing the modelled consequences of forest resource use vs. increased conservation at different levels of GHG savings from carbon sequestration and product substitution with bioenergy and other forest products. Increased forest set-asides for conservation resulted in larger GHG reductions only in the short term and only when substitution effects were low. In all other cases, forest use was more beneficial. In all scenarios, annual carbon dioxide (CO_2) sequestration rates declined in conservation forests as they mature, eventually approaching a steady state. Forest set-asides are thus associated with increasing opportunity costs corresponding to foregone wood production and associated mitigation losses. Substitution and sequestration rates under all other forest management strategies rise, providing support for sustained harvest and cumulative mitigation gains. The impact of increased fertilization was everywhere beneficial to the climate and surpassed the mitigation potential of the other scenarios. Climate change can have large—positive or negative—influence on outcomes. Despite uncertainties, the results indicate potentially large benefits from forest use for wood production. These benefits, however, are not clearly linked with forestry in UNFCCC *reporting*, and the European Union's Land Use, Land-Use Change and Forestry *carbon accounting*, framework may even prevent their full realization. These reporting and accounting frameworks may further have the consequence of encouraging land set-asides and reduced forest use at the expense of future biomass production. Further, carbon leakage and resulting biodiversity impacts due to increased use of more GHG-intensive products, including imported products associated with deforestation and land degradation, are inadequately assessed. Considerable opportunity to better mobilize the climate change mitigation potential of Swedish forests therefore remains.

KEYWORDS
adaptation, conservation, forest, land set-asides, LULUCF, mitigation, substitution

1 | INTRODUCTION

Forests and the forest sector influence atmospheric carbon dioxide (CO_2) concentrations through removing CO_2 from the atmosphere and storing carbon in forests and forest products (Pilli et al., 2015). Greenhouse gas (GHG) savings arise when forest products substitute fossil fuels and other GHG-intensive products, such as cement and steel (Gustavsson et al., 2017, 2021; Leskinen et al., 2018; Lundmark et al., 2014; Sathre & O'Connor, 2010). Sweden aims to become a fossil-free society with net zero GHG emissions by 2045 and negative emissions thereafter. Biomass is currently the largest energy source in Sweden, and a nation-wide initiative to develop roadmaps towards a fossil-free future (Fossilfritt Sverige, n.d.) highlights that biomass-based solutions are increasingly considered. Most of this biomass is expected to come from forests, which are expected to support mitigation also through enhanced forest carbon storage, wood supply for industries producing construction wood and other bio-based products. The European Union's (EU's) LULUCF (Land Use, Land-Use Change and Forestry) policy, on the other hand, may in effect discourage further increases in forest biomass use (Ellison et al., 2014, 2021; Grassi et al., 2019; Matthews, 2020; Nabuurs et al., 2017).

Potential trade-offs exist between the objectives of storing carbon in the forest, on the one hand, and harvesting wood, on the other. How best to balance forest carbon storage and wood production with respect to the climate has long been a subject of debate and scientific inquiry (Cintas, Berndes, Cowie, et al., 2017; Cintas, Berndes, Hansson, et al., 2017; Cowie et al., 2021; Eriksson & Klapwijk, 2019; Klapwijk et al., 2018). Forest owners tend to favour the harvesting of wood to produce forest products, while environmental groups tend to favour conservation, highlighting short-term carbon storage effects in standing forests and improvements in biodiversity (Eriksson & Klapwijk, 2019). Continued debate surrounds the share of set-aside forests required to ensure the protection of wildlife habitats and associated wild species (Dinerstein et al., 2019; Ellis, 2019; Roberts et al., 2020) with some studies identifying important weaknesses in arguments favouring set-asides (Schulze, 2018; Schulze et al., 2022). Disagreements may also arise due to opposing views concerning short- vs. long-term climate objectives, expectations regarding society's future dependence on carbon-based energy and materials, and whether these needs can be met in climate-friendly ways without using biomass (Berndes & Cowie, 2021; Cowie et al., 2021; Rodrigues et al., 2022).

In principle and with respect to the climate, forestry is considered an acceptable practice because wood supply is ideally and traditionally harvested when annual growth rates slow and mean annual carbon accumulation rates begin to plateau (Eriksson, 1976). When conducted under the conditions of sustainable harvest to growth ratios, forestry allows for harvest without comparable C stock declines. Moreover, because younger forests sequester more carbon and net rates of carbon sequestration decline as forests get older (Gao et al., 2018; Holdaway et al., 2017; Repo et al., 2021), it is preferable from a mitigation perspective to harvest growth and produce forest products that provide mitigation through product substitution and carbon storage in harvested wood products (HWPs). In line with existing regulations, forests are immediately and actively regenerated after harvest in Sweden, and carbon sequestration in young forests increases rapidly turning forests into net carbon sinks in 5–20 years (Misson et al., 2005; Rebane et al., 2019). Growth rates and timber production can further be improved by increased regeneration efforts, the use of improved plant material, fertilization, forest thinning and planting density modifications (Högberg et al., 2017; Nilsson et al., 2011; Kauppi et al., 2022).

Forest set-asides are proposed both as a means for providing biodiversity protection as well as for increasing climate change mitigation potential through carbon sequestration in forests. Immediate biodiversity goals are best achieved with older forests (Gao et al., 2015; Martikainen et al., 2000) and the common practice in Sweden is to set aside remote, little-used forests and older high biodiversity potential forests (much like the primary forests highlighted in the *EU Biodiversity Strategy for 2030*). However, such older forests can weaken the potential short- to medium-term climate change mitigation benefits many anticipate due to the likely trade-offs with biodiversity-driven set-aside goals.

UNFCCC *reporting* and EU LULUCF *accounting* rules favour and even encourage net removals in standing forest and land set-asides. By constraining harvest levels (Forest Reference Level, FRL), disincentivizing benefits for promoting additional forest growth (cap) and not linking mitigation benefits from wood use with the forest sector, both UNFCCC *reporting* and EU-level carbon *accounting* create disincentives to additional climate change mitigation (Ellison et al., 2014, 2020; Nabuurs et al., 2017). UNFCCC *reporting* outcomes, however, are reputational in character and do not weigh heavily on individual Parties. EU *accounting* outcomes, on the other hand, can result in penalties (i.e. debits). Parties (or EU Member states) who fall short of their commitments are expected to purchase surplus carbon credits from other Parties/countries.

By displacing fossil fuels, the use of forest biomass for energy helps countries meet their UNFCCC targets and provides real, positive contributions to mitigation. However, UNFCCC reporting focuses only on the net change in forest carbon pools and does not assess effects beyond the impact on carbon pools. All biomass used for bioenergy and HWPs which substitute other products are considered 'oxidized' and fully accounted as 'harvest'. Since harvest is already accounted as a decline in living biomass (i.e. an emission), to avoid double-counting emissions, the combustion of tree biomass is accounted as zero in the energy sector. Although the biomass use for energy results in 'avoided emissions', these are not attributed to the LULUCF sector.

EU LULUCF carbon accounting similarly does not credit the LULUCF sector for avoided emissions, via substitution of fossil fuels, cement, etc. in other sectors. Furthermore, the EU LULUCF regulation (2018/841) creates a separate LULUCF pillar which limits the role of climate-promoting incentives by making it possible to regulate 'flexibility' (the trading/offsetting of credits/debits) across sectors. By limiting the impact of LULUCF on other sectors of the climate policy framework, setting limits on forest resource use with the FRL, and placing a crediting cap on managed forest land (MFL), the EU policy framework represents one of the most restrictive LULUCF frameworks in the world. For another, debits are imposed for harvesting beyond (i.e. for failing to achieve) the FRL. These strategies explicitly discount and set strict limits on the offsetting potential of the forest and forest resource-based sector. Likewise, not achieving the FRL (no-debit rule) is perceived as a policy failure (Solberg et al., 2019). The EU LULUCF regulation does, on the other hand, promote long-lived HWP-based carbon sequestration. The remaining components of the carbon accounting framework, however, fail to incentivize the climate benefits of forest growth and substitution (Ellison et al., 2013, 2014, 2020; Nabuurs et al., 2017).

In the current study, we assess how different forest management strategies in Sweden influence the forest carbon stock and wood harvest over time. Forest management strategies are discussed in relation to their climate change mitigation potential, while possible climate change impacts on forests are also considered. More specifically, the study aims to analyse:

- the role of forests and forestry by comparing how atmospheric CO_2 concentrations are affected over different timescales by carbon storage in forests and HWPs, and by substitution (given a fixed management system)
- forest protection, nature conservation and their long-term impacts on forest-based climate change mitigation
- the potential for increased fertilization to sustainably increase net CO_2 substitution and removals
- the potential benefits and/or increased risks associated with a changing climate on mitigation (we simulate both positive and negative effects on growth due to a rise in global temperatures and potential nutrient deprivation)
- the differences between the *real* effect of forests and forestry on atmospheric CO_2 concentrations and the *reported* and *accounted* climate reporting estimates implied by different accounting frameworks

2 | MATERIALS AND METHODS

2.1 | Scenarios at National Scale: Modelling of biomass, carbon flows and pools

To study the cumulative climate impacts of harvest and stocks in standing forest over time in Sweden, national-level scenarios were generated using the empirical Heureka RegWise decision support system to simulate the future given initial natural resources, biological limitations on growth and assumptions about forest management practices (Wikström et al., 2011). Modelled growth is conditioned by measured site fertility and the initial stand at the beginning of the simulation. These two factors are set by the measured data on sample plots inventoried by the Swedish National Forest Inventory (NFI). The Swedish NFI compiles detailed, robust and constantly updated information about the state of the forest. Multiple types of data are recorded at tree, site and stand level and data quality is checked in several steps after the inventory. The models for basal area growth, mortality and ingrowth (with varying growth equations for young stands, productive forests, unproductive stands [growth less than 1 m^3/ha/year] and natural mortality) are empirical in character and build primarily upon data from NFI permanent plots.

For all scenarios, the initial state is set by adopting the existing measured data on the permanent sample plots of the Swedish National Forest Inventory (NFI) in 2010 (Fridman et al., 2014). The Swedish NFI employs area-based sampling on 30,000 permanent sample plots and each sample plot measures 10 m in radius. All plots together represent the total land and freshwater area of Sweden. The NFI is an annual, systematic cluster-sample inventory organized as a systematic grid of sample clusters. The square-shaped clusters are distributed in a denser pattern in the southern than in the northern part of the country. Each cluster consists of four to eight sampling plots. Each sample plot is occasionally delineated into more than one land-use category. A variety of tree, stand and site variables are registered on the plots. On each plot, all trees with DBH ≥ 4 cm are calipered, height is measured and sample tree damage recorded. Dead wood with diameter ≥ 10 cm is calipered and stumps are measured (Marklund, 1987; Näslund, 1947; Petersson & Ståhl, 2006). Land use is assessed in the field with the help of site and stand variables and the existing vegetation cover.

The NFI data used to simulate these scenarios consist of many parameters. Stem volume and living tree biomass starting in 2010 are estimated with the help of allometric equations (Marklund, 1987; Näslund, 1947; Petersson & Ståhl, 2006; Wikström et al., 2011). The dead wood state is measured on the plots (Lundblad et al., 2021; Sandström et al., 2007). Changes in carbon pools (living biomass, dead wood, stumps, litter, soil and HWP) are estimated using the stock difference method (IPCC, 2006). Inflows to the HWP pool are estimated based on simulated harvest. All stem wood is harvested, and a proportion (equivalent to approximately 10 TWh) of tops and branches are also harvested for bioenergy. Stumps are not extracted. Since substitution factors are uncertain, we model the scenarios with three levels of substitution, where 1 m^3 harvested stem wood is assumed to result in 0.5, 1 and 1.5 tonnes of *avoided* CO_2 *emissions* (Leskinen et al., 2018; Lundmark et al., 2014). We return to the debate on substitution factors in the Discussion. Other emissions (Tables 2 and 3; IPCC 2006) are generally minor under Swedish conditions and were assessed as a constant emission of 0.096 $MtCO_2e$/year for all years and scenarios.

Heureka consists of several underlying models (e.g. stand growth, mortality and decomposition models). Decomposition and changes in pools for dead wood, litter and soils are modelled using the Q-model (Ågren et al., 1996). The Q-model is a process-based model that uses empirical data. The inflow of organic material is assumed to originate from dead organic matter after harvest, natural mortality and non-tree vegetation. Model parameterization settings for the four main Swedish climatic regions (see Figure S1) have been applied. For model initiation, carbon stock estimates from the Swedish Forest Soil Inventory were used as the starting point for a 20-year spin-up period preceding the actual simulation starting point. Inflow/turnover rates were modelled for branches, needles and root fractions, and constants assessed for grasses, herbs, shrubs, mosses and lichens. Inflows from harvest residues were estimated per fraction of needles, branches, stems, tops, stumps, roots and excluded stem wood. In Sweden, roundwood is harvested. But a small share of the stems is left on harvest sites. Natural mortality is empirically modelled. Stumps and harvest residues left in the forest are assumed to decompose at an annual rate of 4.6% (Melin et al., 2009) and 15% (Lundblad et al., 2021) respectively. The Q-model is only applied to mineral soils, and emissions from drained organic soils are estimated using activity data (area) multiplied by emission factors. Different emission factors are used per nutrient status and climate region (Lundblad et al., 2021).

Fahlvik et al. (2014) demonstrate that the growth and mortality modelling in Heureka generates reliable results. The specification of forest management between two consecutive points in time may include, for example, fertilization, harvest type and intensity, regeneration type and areas set aside for nature conservation. An algorithm (based on forest owner behaviour identified on NFI sample plots) was used to select stands for harvest. Given regular harvests and no natural disturbances, we assume that no unknown variable would change the principal findings. As with all empirical models, precision diminishes if the aim is to simulate the development under circumstances that deviate from the prevailing circumstances at data collection. The positive climate effects from assuming the IPCC scenario RCP 4.5, for example, rely on process-based assumptions. The future climate impact on tree vitality and growth, however, is uncertain.

2.2 | Scenarios for future forest management—Scenario model specification

To study the cumulative climate impacts of harvest and standing forest-based stocks over time in Sweden, future developments are simulated using five different scenarios. The total Managed Forest Land (MFL) area was estimated at 27.5 Million hectares (Mha) in 2010. MFL was further subdivided into productive (average growth >1 m^3/ha/year; around 23.4 Mha) and unproductive MFL (average growth <1 m^3/ha/year, around 4.0 M ha). Productive forests consisted of approximately 19.8 Mha of forests used for wood supply and another 3.6 Mha of formally and voluntarily protected forests in which harvest was and is not permitted. Low productive forests were also considered 'protected'. In total, 7.6 Mha have been protected and excluded from harvest, representing approximately 28% of total MFL.

We focus on MFL defined as forest land remaining forest land (e.g. Lundblad et al., 2021), that is, we include land-use conversions to forest land and exclude forest land converted to other land-use categories. Land transitions from and to MFL are simulated based on the average conversion rate over the period 1990–2017 (Lundblad et al., 2021; afforestation rates are approximately 15 kha/year and deforestation rates 11 kha/year). Land actively converted to forested land is first classified as Afforested Land for 20 years and thereafter included under MFL. Land actively converted from MFL is immediately considered and reported as Deforested Land for 20 years and thereafter reported in the land category it was converted to.

In all scenarios, we assume 100% of the growth on productive MFL used for wood supply, minus self-mortality, is harvested. We assume zero harvest in protected forests. We further assume an equilibrium stem volume (biomass)

TABLE 1 Scenarios and objectives

Scenarios	Assumptions
Maximum Potential Harvest	Base scenario
Increased Nature Conservation	Study effects of increasing forest land set-asides (3.7 Mha)
Increased Fertilization	Fertilization (restricted by law)
Negative Climate Effects on Growth	Double mortality
Positive Climate Effects on Growth	Growth based on IPCC RCP 4.5 scenario

will emerge, as well as a steady state on land set-asides after approximately 200 years. Table 1 provides an overview of the five scenarios and their objectives.

In the *maximum potential harvest* scenario, areas of different land-use classes as well as management practices (excluding harvest intensity) are assumed to simulate the conditions specified by the Forest Agency for the period 2000–2009 (Claesson et al., 2015; Forest Agency, 2008, 2015). This scenario is closely modelled on a previous scenario analysis of Swedish Forest Reference Level (FRL) options (Lundblad, 2018) and is described in more detail in a companion document (Petersson et al., 2022). To study the consequences of setting aside additional MFL for nature conservation, we assume an additional 3.7 Mha of mainly productive MFL is set aside for nature conservation in the simulation. This amount is equivalent to approximately 18.5% of currently available, productive MFL, bringing the total protected forest area to 11 Mha. In the *increased nature conservation* scenario, except for the area set aside for nature conservation, all parameters remain the same as in the *maximum potential harvest* scenario. In the increased conservation scenario, we assume comparatively younger forest set-asides.

To study the consequences of increased investments in forestry on net removals in carbon pools and substitution of fossil fuel-based alternatives, we simulate the *increased fertilization* scenario. This model specification represents a moderate fertilization scenario approximating established fertilization practices on a larger area, but within the legal fertilization guidelines. Established fertilization mainly targets, older, middle-aged Scots pine stands after thinning, around 10 years before final felling (Högberg et al., 2014; Jacobson & Pettersson, 2010). The simulated fertilized area is thus about 200 kha per year or approximately 1% of productive MFL, roughly seven times more fertilization than assumed in the other scenarios. The simulated fertilization thus considers the effect of a one-time addition of 150 kg N/ha (ammonium nitrate). Apart from fertilization, all other parameter settings are identical with the *maximum potential harvest* scenario.

To study the potential risks of negative climate effects on growth, net removals in carbon pools and assumed substitution of fossil fuel-based alternatives, we modify the *maximum potential harvest* scenario by assuming a doubling in natural mortality. For the *negative climate effects on growth* scenario, all other parameter settings remained identical (currently mortality is estimated at around 11% of the growth in Sweden; Forest Statistics, 2021). To estimate the potential consequences of positive climate effects on tree growth, we use the corresponding IPCC RCP 4.5 pathway (IPCC, 2013) to simulate the *positive climate effects on growth* scenario. Using the process-based model BIOMASS (McMurtrie et al., 1990), the IPCC RCP 4.5 pathway has been calibrated for Swedish conditions (Bergh et al., 1998, 2003). The principal components for the process-based growth adjustment comprise age, basal area, site index, vegetation index and temperature sum. In both the negative and positive climate effects scenarios, all other parameter settings remain identical.

3 | RESULTS

In the *maximum potential harvest* scenario (Figure 1), after around 200 years both stocks (storage) and growth become linear. This equilibrium finding occurs because we assume 100% harvest of the net growth on MFL and 0% harvest in protected forests. After peaking, the constant annual sustainable harvest is estimated at 99 Mm^3/year. After 200 years, approximately two forest rotation periods, the cumulative harvest is 4.3 times greater than stocks (we assume mortality is emitted to the atmosphere through decomposition). Total gross growth is estimated at 119 Mm^3/year, similar to current gross growth in Sweden (Forest Statistics, 2021). Assuming forests remain viable over the very long term, this relationship is expected to continue in a linear fashion over time.

In the *increased nature conservation* scenario (Figure 1), on the other hand, after peaking, the constant annual sustainable harvest (growth) is estimated at 85 Mm^3/year. The long-term loss in forest growth from setting aside an additional 3.7 Mha of productive forest land for nature conservation compared to the *maximum potential harvest* scenario is 14 Mm^3/year, from peak to perpetuity (a loss of 3.8 m^3/ha/year of additional forest growth per year over the entire scenario period). Since the two scenarios generate similar total amounts of forest growth, after 200 years, estimated stocks + cumulative mortality + cumulative harvests were not significantly different. An important share of the growth in the *increased nature conservation* scenario, however, is lost to cumulative mortality and eventually becomes an emission. In the *maximum potential harvest* scenario, on the other hand, less growth is

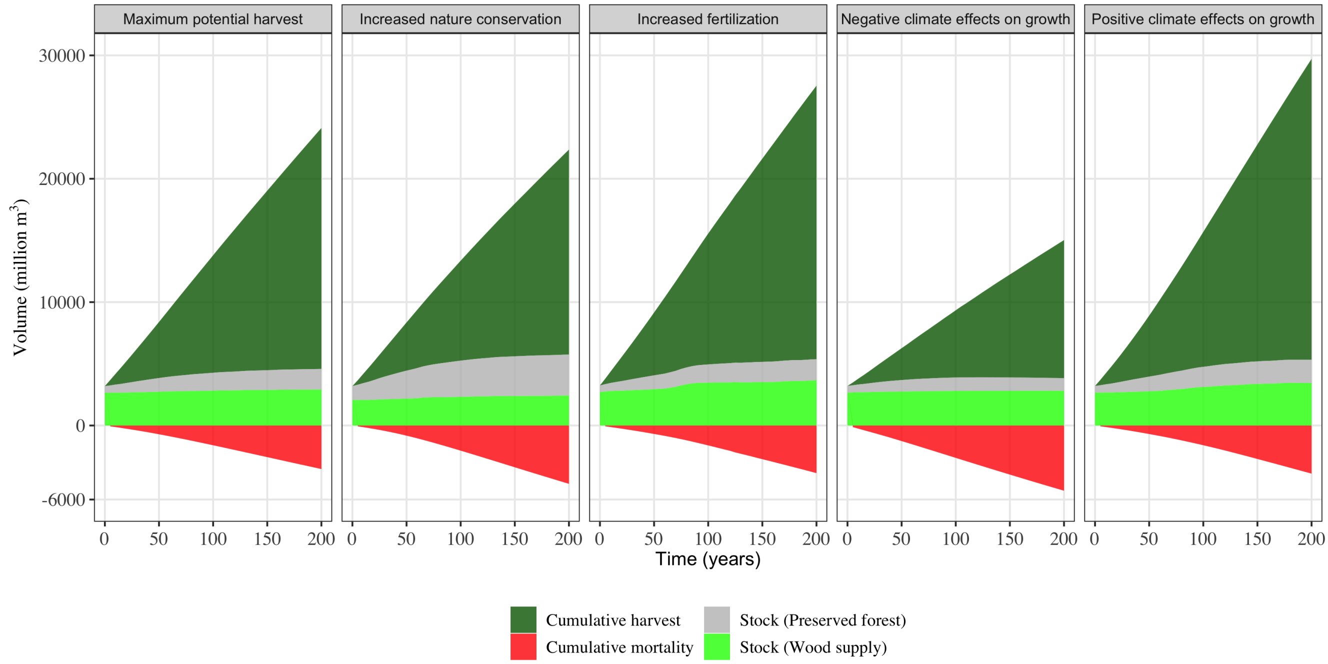

FIGURE 1 The simulated cumulative stem volume [Mm^3] stored in MFL forests, cumulative harvest and (decomposed) cumulative mortality over a period of 200 years, assuming that 100% harvest of net growth in MFL is used for wood supply and no harvest in preserved forests. In all scenarios, after 200 years, cumulative harvest is greater than storage. From a pure climate perspective, if harvest is used for substitution or storage (where no decomposition occurs), harvesting mature trees and using *maximum potential harvest* provides greater climate benefits than storing carbon in standing forests. This is explained by the higher mortality in the *increased nature conservation* scenario (compared with the *maximum potential harvest* scenario). The *increased fertilization* scenario simulates the possibility that intensive forest management may increase the climate benefits associated with forest use. Finally, the *negative* and *positive climate effects on growth* scenarios simulate outcomes depending on whether climate change is negative or positive for tree growth. Since 100% of the net growth is harvested, after peaking at 200 years, the yearly additional growth for harvest remains linearly constant

lost and mortality is substantially lower. In the *increased nature conservation* scenario, more volume is also stored in the forest and harvest is smaller than in the *maximum potential harvest* scenario.

In the *increased fertilization* scenario (Figure 1), after 200 years, the long-term harvest increases to approximately 112 Mm^3/year (Figure 1), or approximately 13 Mm^3/year more than in the *maximum potential harvest* scenario (and about 27 Mm^3/year greater than in the *increased nature conservation* scenario). As noted above, however, since significant restrictions on the use of fertilization apply, fertilization is only simulated on approximately 1% of available MFL.

The *negative climate effects on growth* (Figure 1) and the *positive climate effects on growth* (Figure 1) scenarios both suggest powerful impacts on forest-based mitigation potential. After 200 years, total sustainable harvest growth was significantly lower under the *negative climate effects on growth* scenario (57 Mm^3/year) and emissions from mortality were higher. On the other hand, the *positive climate effects on growth* scenario, primarily because this scenario affects all MFL equally, resulted in the highest sustainable harvest levels and the largest cumulative impact (137 Mm^3/year).

Table 2 provides more detailed information on the direct climate impact/benefit of change across all carbon pools in the *increased nature conservation* and *maximum potential harvest* scenarios over the same 200-year period, expressed in terms of their climate impact in carbon equivalents ($MtCO_2e$/year). In the short term (i.e. by 2025), the climate benefits are similar in both scenarios. However, in the long term, i.e., after the carbon pools peak within a period of approximately 200 years, significant differences arise between the two scenarios. In this case, the climate benefit in the *maximum potential harvest* scenario is 16% greater per year (−99.5 $MtCO_2e$/year) than in the *increased nature conservation* scenario (−85.6 $MtCO_2e$/year). The climate benefit from the *maximum potential harvest* scenario is −99.5 $MtCO_2e$/year from peak to perpetuity, an amount greater than the benefit from *increased nature conservation* by −14 $MtCO_2e$/year. Setting aside an additional 3.7 Mha MFL for nature conservation thus reduces the growth/harvest cycle in the circular bioeconomy, thereby impacting future mitigation opportunities.

The results in Table 2 are further sensitive to the assumed substitution effect (here 1 m^3 to 1 tonne CO_2e). Depending on the assumed rate of substitution, projected outcomes for the total annual net forest-related impact on

TABLE 2 Total climate benefit across carbon pools for the increased nature conservation and maximum potential harvest scenarios, given a '1 to 1' substitution effect (the shaded year corresponds to the 2021–2025 reporting period under EU/2018/841)

Year	Increased nature conservation (maximum potential harvest) [M tonne CO_2/year] Living biomass	Soil litter	Other emissions	(stumps) Dead wood	(Lying/Standing) Dead wood	Long-lived HWP	Short-lived HWP	'1 m^3 to 1 tonne CO_2' substitution harvest	Sum
10	−26.8 (−11.6)	−5.41 (−4.47)	0.10 (0.10)	−1.14 (−3.23)	−2.97 (−2.47)	−2.79 (−4.65)	0.05 (−0.52)	−77.9 (−89.0)	−117 (−116)
15	−33.3 (−16.9)	−5.32 (−5.44)	0.10 (0.10)	−0.02 (−2.25)	−2.63 (−2.07)	−1.96 (−3.88)	0.27 (−0.25)	−75.3 (−88.1)	−118 (−119)
30	−31.3 (−17.1)	−3.40 (−1.87)	0.10 (0.10)	−1.50 (−2.24)	−2.47 (−1.52)	−2.29 (−3.30)	−0.43 (−0.44)	−80.3 (−92.3)	−122 (−119)
50	−22.9 (−13.8)	−3.74 (−0.97)	0.10 (0.10)	−1.47 (−1.97)	−2.28 (−1.21)	−0.83 (−1.45)	−0.54 (−0.56)	−83.5 (−96.5)	−115 (−116)
70	−18.3 (−9.38)	−2.34 (0.20)	0.10 (0.10)	−1.27 (−1.33)	−2.07 (−1.06)	−1.11 (−1.29)	−0.37 (−0.17)	−84.4 (−99.7)	−110 (−113)
90	−14.2 (−7.78)	−0.87 (0.86)	0.10 (0.10)	−0.49 (−0.92)	−1.36 (−0.73)	−0.66 (−1.26)	−0.04 (−0.14)	−84.9 (−101)	−102 (−111)
110	−10.5 (−5.58)	−0.81 (0.39)	0.10 (0.10)	−0.40 (−0.48)	−0.81 (−0.51)	−0.64 (−1.17)	0.03 (−0.07)	−86.0 (−101)	−99.0 (−109)
130	−7.33 (−2.79)	−1.00 (0.69)	0.10 (0.10)	−0.29 (−0.30)	−0.48 (−0.25)	−0.19 (−0.44)	−0.01 (−0.03)	−85.8 (−101)	−95.0 (−104)
150	−3.36 (−3.46)	−0.38 (0.35)	0.10 (0.10)	0.26 (0.22)	−0.10 (−0.06)	0.02 (−0.14)	0.07 (0.07)	−84.7 (−99.6)	−88.1 (−103)
170	−0.26 (−0.97)	0.36 (0.76)	0.10 (0.10)	−0.40 (−0.23)	0.13 (0.06)	−0.20 (−0.29)	−0.12 (−0.07)	−86.5 (−101)	−86.8 (−101)
190	−3.34 (−1.62)	−0.02 (0.86)	0.10 (0.10)	0.49 (0.26)	0.24 (0.15)	0.19 (−0.19)	0.16 (0.10)	−83.4 (−99.1)	−85.6 (−99.5)

climate change mitigation vary dramatically (see Figure 2). In most cases, however, the short-term impact of setting aside an additional 3.7 Mha of land for nature conservation is relatively minor compared to the long-term impact of forest use, carbon sequestration in long-lived products and substitution. Only in the most conservative case (1 to 0.5) is the additional carbon sequestration in standing forests simulated by the *increased nature conservation* scenario greater in the short term than the sequestration/substitution impact of *maximum potential harvest*. The difference in impact is measured as the space between the *increased nature conservation* impact (blue line) and the *maximum potential harvest* impact (green line). As the estimated sequestration/substitution impact increases in size, however, the respective substitution benefits of *maximum potential harvest* increase relative to the *increased nature conservation* scenario. In the Discussion section, we further elaborate the logic behind different estimated substitution impacts.

The *positive climate effects on growth* and *increased fertilization* scenarios likewise have very large, continuous impacts on the total net annual sequestration/substitution potential. While the *positive climate effects on growth* are larger, due to legal restrictions in Sweden, we assume fertilization only on a total of 1% of the available MFL in the *increased fertilization* scenario. The *positive climate effects on growth* scenario, however, is not similarly restricted in extent. We cannot really say, however, what might happen if fertilization were permitted on an additional 10% or more of the Swedish MFL (see, however, the example of Gustavsson et al., 2021).

4 | *REPORTING* AND *ACCOUNTING* RULES IMPACT ON MITIGATION INCENTIVES

Table 3 highlights the UNFCCC and EU level *reporting* and *accounting* consequences of the respective LULUCF frameworks based on each of the five simulated scenarios for the period 2021–2025. Ideally, the optimal choice is the scenario that both sequesters the most carbon over both the short and the long term and has the greatest potential climate impact. Based on the scenario results, the short-term benefits of increased nature conservation are marginal, while the potential long-term gains from the maximum potential harvest scenario are significantly greater. However, as noted in the Introduction both the UNFCCC *reporting* and the EU *accounting* frameworks ignore the LULUCF role in the climate effects that arise from the avoided emissions associated with HWP carbon sequestration, product substitution and bioenergy use. Considering this factor, it may make more sense to pursue long-term strategies.

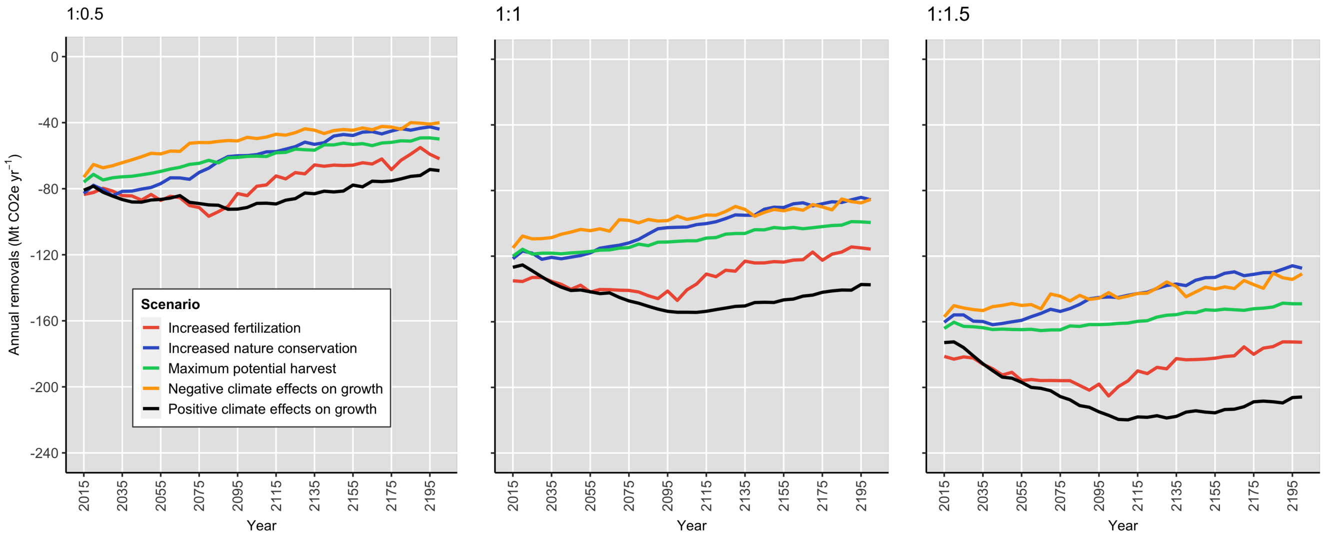

FIGURE 2 Total annual estimated net carbon sequestration and substitution, selected scenarios (2015–2195). The scenarios include changes in all carbon pools (see Table 2) and substitution for three different assumed substitution effects (0.5, 1 and 1.5 tonne CO_2e per m^3 stem wood)

Both UNFCCC reporting and EU accounting, however, encourage short-term impacts. As highlighted in Table 3, both for the UNFCCC *reporting* framework and the EU *accounting* framework, the scenario yielding the largest benefits is *increased nature conservation*: this strategy provides −42.9 MtCO_2e/year in UNFCCC reporting benefits and −1.1 MtCO_2e/year in EU accounting benefits. The principal difference between the UNFCCC and EU outcomes derives from the decision to harvest 100% of the annual net increment. Since the EU accounting framework penalizes harvesting above the FRL, this framework yields an emission in all the scenarios except increased nature conservation. Harvesting less to fulfil the FRL would improve accounted removals in all cases but would not alter the linear relationships between the different scenarios and would not change the substitution-related benefits from harvesting the full amount. Moreover, this would only serve to further raise lost net potential harvest to a point further below the increased nature conservation scenario over the longer term (since this amount is greater than the set-aside amount of forest). Harvesting less (this is the direct FRL impact) is essentially 'equivalent' to increasing the relative share of protected forest and would yield mitigation outcomes comparable to those predicted by this scenario, with a comparable reduction in the substitution effect. Apart from the *negative climate effects on growth* scenario, both reporting and accounting frameworks (UNFCCC and EU) encourage LULUCF sector strategies that will, in the long term, provide smaller total climate benefits relative to the alternatives.

Depending on future substitution effects, the long-term mitigation loss from adopting an *increased nature conservation* scenario may become significant. Given constant climate conditions, compared to the *maximum potential harvest* scenario and based on 1 to 1 substitution, the *increased nature conservation* scenario provides climate benefits of approximately −85.6 MtCO_2e/year, while the *maximum potential harvest* scenario provides −99.5 MtCO_2e/year from peak to perpetuity. If we consider the potential climate effects across variation in the substitution effect, the opportunity costs of failing to choose either the *maximum potential harvest* or the *increased fertilization* scenarios may be substantially greater.

5 | DISCUSSION

Forest management in Sweden involves harvesting (final felling) about 1% of total MFL per year and legally requires immediate, active regeneration after harvest (Forest Agency, 2020a, 2020b). Tree species composition has not been significantly altered over the course of the 20th century (Forest Statistics, 2021), but there is concern about a gradual decline in old growth forests (Jonsson et al., 2019). Furthermore, the targeting of biologically young stands for harvest may limit the delivery of several ecosystem services, resulting in less multifunctional forest (Jonsson et al., 2020). In parallel with increasing wood harvesting levels, forest management has resulted in significant increases in forest carbon stocks. Following a historic period of declining forest resources, forests in Sweden have continuously accumulated carbon since the early 1920s, resulting in more than a doubling of carbon stocks over the past century (Forest Statistics, 2021; see also Kauppi et al., 2022). The focus on forest policies and management strategies has, over time, integrated

TABLE 3 UNFCCC *reported* and EU *accounted* LULUCF impacts relative to their pure climate change mitigation effects, 2021–2025

UNFCCC Reporting	NET change in pools [M tonne CO_2/year]						REPORTED
Sweden: MFL 2021–2025							
Scenario	Living biomass	Soil litter	Other emissions	Dead wood	Long-lived HWP	Short-lived HWP	Total
Maximum Potential Harvest	−16.9	−5.4	0.1	−4.3	−3.9	−0.2	−30.7
Increased Nature Conservation	−33.3	−5.3	0.1	−2.7	−2.0	0.3	−42.9
Increased Fertilization	−9.3	−4.7	0.1	−7.7	−6.1	−1.2	−28.9
Negative Climate Effects on Growth	−10.6	−5.7	0.1	−5.6	−3.3	−0.1	−25.2
Positive Climate Effects on Growth	−19.1	−5.8	0.1	−5.6	−4.4	−0.5	−35.3
EU Accounting	**NET change in pools relative to the required Reference Level [M tonne CO_2/year]**						**ACCOUNTED**
Sweden: MFL 2021–2025							
Scenario	**Living biomass**	**Soil litter**	**Other emissions**	**Dead wood**	**Long-lived HWP**	**Short-lived HWP**	**Total**
Maximum Potential Harvest	13.4	−4.0	0.0	−1.6	−0.6	0.8	8.1
Increased Nature Conservation	−3.1	−3.8	0.0	0.1	1.3	1.4	−1.1
Increased Fertilization	20.9	−3.2	0.0	−5.0	−2.8	−0.1	9.9
Negative Climate Effects on Growth	19.6	−4.2	0.0	−2.8	0.0	1.0	13.5
Positive Climate Effects on Growth	11.1	−4.3	0.0	−2.9	−1.1	0.6	3.4
Reference Levels (effective caps)	−30.2	−1.5	0.1	−2.7	−3.3	−1.1	−38.7
	(cap)	(cap)	(cap)	(no cap)	(no cap)	(cap)	(Total FRL)

1 m^3 to 0.5 tonne CO_2		1 m^3 to 1.0 tonne CO_2		1 m^3 to 1.5 tonne CO_2	
Harvest	**Total climate effect**	**Harvest**	**Total climate effect**	**Harvest**	**Total climate effect**
−44.0	−75	−88.1	−119	−132.1	−163
−37.6	−81	−75.3	−118	−112.9	−156
−50.8	−80	−101.7	−131	−152.5	−181
−42.1	−67	−84.2	−109	−126.4	−152
−46.8	−82	−93.6	−129	−140.4	−176

Changes in carbon pools are reported to the UNFCCC. For MFL, changes in carbon pools, living biomass, soil + litter, other emissions and short-lived HWP are accounted with a cap compared to the FRL, while dead wood and long-lived HWP are accounted without a cap compared to the FMRL (under CP2). For Sweden, the cap, which limits credits from MFL, is −2.5 M tonnes CO_2e/year. The total climate effect is calculated as the reported net change in carbon pools, plus the substitution effect. Three alternative substitution effects have been used.

securing wood supply for the forest industry with other objectives, such as climate change mitigation and adaptation, biodiversity conservation, social aspects and water resource management (Eriksson et al., 2018).

As highlighted in Figure 3, forest stands are traditionally harvested when annual growth rates decline and mean annual carbon accumulation rates begin to plateau (Eriksson, 1976). For a given site in Sweden, the optimal harvest should occur after the year annual growth culminates. The optimal rotation period is the period which maximizes growth (or carbon uptake) in trees. On average, this occurs after approximately 100 years of growth and carbon sequestration (Figure 3), later in the North and earlier in the South. After this peak, each additional year of forest growth sequesters less additional carbon. Harvesting at later than optimal time points is thus assumed to yield declining amounts of additional biomass and slowing rates of carbon uptake. In the long term and moving from an approximately even-aged stand distribution to the landscape scale, carbon uptake will eventually become equal to decomposition rates, yielding a steady state, net zero rate of carbon sequestration. Over time, older forests therefore provide no significant additional mitigation benefit via carbon sequestration.

From a climate perspective, the impact of forestry is typically considered acceptable because, given an even-aged stand distribution, constant fertility over the forest landscape and maximum harvest at the optimal rotation period for all stands, net marginal tree growth will eventually stop increasing (saturation point). This point defines a steady-state equilibrium where tree growth and harvest removals are essentially balanced and equal at the landscape scale. The *maximum potential harvest* scenario (Figure 1) illustrates that after around 200 years both stocks (storage) and growth become linear, reflecting this equilibrium prediction. Harvesting after the saturation point ultimately means the loss of additional net growth potential in replanted forest. Although equilibria can be affected by 'natural disturbances' such as insect attacks, wildfires and storms (Forzieri et al., 2021; Senf et al., 2020; Senf & Seidl, 2021a, 2021b), the concept of a more or less stable, long-term equilibrium potentially remains relevant (Eliasson et al., 2013). However, as discussed below, there are many uncertainties.

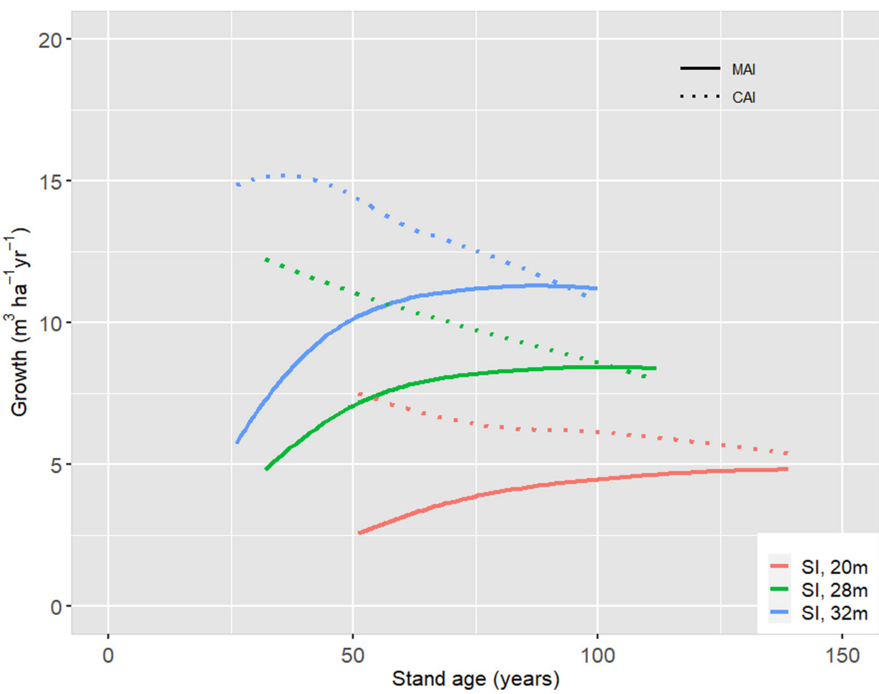

FIGURE 3 Measured development of mean annual and current annual increment in three common Norway spruce stands. SI = "site index" and refers to tree height at 100 years (considering total tree biomass, 1 m^3 stem wood roughly corresponds to 1.4 tonnes CO_2e). *Notes*: After stand establishment, a tree stand first grows slowly, then more rapidly, then peaks, and after this point the growth rate begins to decline. The optimal rotation period which maximizes growth occurs when the MAI line crosses the CAI line. Growth will be lost if harvest occurs before or after this optimization point. Current annual increment (CAI) is the total annual growth in any given year. Mean annual increment (MAI) is the average annual growth a stand exhibits at a given age and is calculated as the cumulative growth divided by the stand age. The figure is based on measured data and supports the theory of growth in tree stands (Eriksson, 1976)

5.1 | Growth in older forests

One possible objection to this analysis arises from the literature on old growth forests. Suggestions that old-growth forests arrive at a steady state with stable C stocks have been challenged with evidence that old-growth forests continue to act as C sinks (Hadden & Grelle, 2016; Luyssaert et al., 2008; Seedre et al., 2015). Measurements over entire landscapes or long-term stand measurements may, however, provide more robust determinations about old growth forests (Gundersen et al., 2021; Luyssaert et al., 2021). Based on data from 874 forest plots, Holdaway et al. (2017) estimated biomass growth and identified drivers for biomass change in old-growth forests and secondary managed forests in New Zealand. Over the period 2002–2014, a significant biomass increase was detected in the secondary forest, whereas no significant change was detected in the old-growth forest. The drivers for biomass growth in the secondary forest were growing stock (biomass) and past disturbance, while growth in old-growth forests was determined by recent disturbance (mortality among large trees). This pattern is confirmed by Derderian et al. (2016) who resampled a 700-year chronosequence in Colorado (US) and discovered that C stocks, due to high spruce tree mortality caused by a bark beetle attack, had declined compared to 30 years earlier, despite the fact that net C uptake was recorded at both sampling occasions.

A recent study conducted by a team of 25 scientists appointed by the International Boreal Forest Research Association (IBFRA) (Högberg et al., 2021) found substantially increased C stocks in living tree biomass from 1990 onwards in the more intensively managed, higher harvest Nordic countries, as compared to the boreal forests in Canada, Russia and Alaska, where C stocks have remained stable or even declined (Alaska). A partial explanation was C losses due to forest fires in the less intensively managed boreal forests, together with fire protection measures in the Nordic countries, which maintained annual burned area well below natural conditions. Sharma et al. (2013) demonstrate that national parks in Canada with large shares of old-growth forests have large C stocks and low annual CO_2 uptake, whereas parks with large shares of younger forests, due to natural disturbances, have reduced but still large C stocks and comparatively high annual CO_2 uptake. Thus, while C stocks can be preserved in national parks, large C stocks have limited mitigation potential.

In another recent study, based on data from the Norwegian NFI, the rapid drop in current annual increment (CAI) at a certain age (cf. Figure 3) was challenged (Stokland, 2021a). The methodology behind that study has been questioned (Brunner, 2021), but Stokland defends his findings and suggests more studies looking into the fate of CAI after the point where CAI crosses MAI are needed (Stokland, 2021b). A more stable CAI over several decades would, from a climate mitigation perspective, speak for extended rotation periods in managed forests. However, with the exception of very poor sites that illustrate flatter CAI development with age, another recent study based on the Finnish NFI confirms the rapid drop in CAI and hence in MAI (Repo et al., 2021).

One might consider the old forest sink issue a 'red herring' in the sense that the question whether forests eventually reach a steady state or continue to sequester more carbon each year distracts from the more relevant questions: (1) How much mitigation per hectare can old forest landscapes generate?; and (2) is the mitigation per hectare large or small compared to the mitigation from an average hectare of forest managed to maintain net forest growth at high levels under sustained harvest practices?

5.2 | Can increased fertilization provide additional climate benefits?

Although the *increased fertilization* scenario increased growth from about 99 (*maximum potential harvest* scenario) to about 112 Mt CO_2e/year, fertilization was only applied on 1% of the forest area per year. This raises interesting questions about the possible outcome of greatly increasing MFL fertilization rates. Due to legal restrictions in Sweden, we have only simulated small changes in older forests fertilized about 10 years before final felling. Gustavsson et al. (2021), however, have investigated a more intensive fertilization scenario in which growth increased by 40% after 100 years. Presumably there is great potential to increase growth with the help of fertilization. In the same study, Gustavsson et al. (2021) ran a scenario where as much as 50% of the forest land area was protected. Fertilization may thus provide an opportunity to preserve larger areas for biodiversity while simultaneously managing other forest areas more intensively, thereby maintaining total growth.

5.3 | Consequences of a changing climate

Despite the perceived benefits of climate change for forest growth, considerable uncertainty surrounds the possible positive and negative climate effects on future growth. For the Nordic boreal forests, the prevailing assumption is that gradual climate change will be positive for growth due to higher temperature, precipitation, atmospheric nitrogen deposition and extended growing seasons (Appiah Mensah et al., 2021; Etzold et al., 2020; Henttonen et al., 2017; Kauppi et al., 2014; Keenan, 2015; Kellomäki et al., 2008; Koca et al., 2006; Tamm, 1991). Compared to current levels in Sweden, climate effects suggest positive future productivity increases of about +300% and +100% in the northern and southern regions, respectively, resulting in shorter rotation periods (Bergh et al., 2005). Forested northern regions are expected to expand further northward and into higher elevations, yielding larger areas of forest cover and increasing forest density (Claesson et al., 2015). However, we have not modelled these expanding growth patterns with the Heureka model.

Climate-related factors can, however, challenge the positive effects of CO_2 fertilization and longer growing seasons (Hanewinkel et al., 2013; Reyer et al., 2017). Both negative and positive growth responses have been measured across the Canadian forest landscape (Girardin et al., 2014; Taylor et al., 2017). Changes in water availability provided one possible explanation for these divergent responses. Water availability could further level out or reverse positive effects on growth in boreal tree species (Reich et al., 2018). Likewise, nutrient availability has also been singled out as a potential explanation for not benefitting from these increased concentrations (Hyvönen et al., 2007; Norby et al., 2010; Sigurdsson et al., 2013).

Extreme and frequent changes in abiotic conditions could have damaging effects on trees, thereby affecting growth capacity in succeeding years (Keenan, 2015). Tree growth rates in Sweden, for example, were found to be about 20% lower than expected in 2018 due to summertime hot and dry conditions (personal communication, Swedish NFI). While temperature effects on tree growth can be severe, precipitation effects may be minimal during the growing season due to the recharge of the ground water table from melted winter snow (Bergh et al., 2005). Increases in evapotranspiration may result in more persistent drought during the growing season, potentially counteracting growth (Koca et al., 2006). Higher groundwater levels and shorter winter soil frost seasons may increase the risk of storm damages and soil damages from off-road timber transport (Oni et al., 2017). On the other hand, events such as storms, frosts and droughts can trigger wildfires, pest and disease outbreaks (e.g. root rot and bark beetles) that may reduce forest growth and productivity (Björkman et al., 2011; Blennow & Olofsson, 2008; Subramanian et al., 2015). For example, Pinto et al. (2020) found that both climate and vegetation correlate with fire size, whereas human-related landscape features shape ignition patterns.

Increasing disturbances from wind, bark beetle and wildfires at European level (Seidl et al., 2014), may become greater concerns in Sweden. From a Swedish perspective bark beetle attacks by the spruce bark beetle (*Ips typographus* [L]) are an identified threat—not least in the climate change context (Eidmann, 1992; Jönsson et al., 2009; Kärvemo et al., 2016). Incident rates and magnitude of forest damage by spruce bark beetle are higher in older stands (Martikainen et al., 1999). Under specific climatic conditions, forest growth could exhibit varied risks depending on stand age, development stage and management practice (Blennow, 2012).

Hence, both the boreal forest growth response and carbon cycle feedback to climate change remain uncertain. Adaptive forest management practices could be essential for mitigating negative effects, while maximizing forest growth and production (e.g. Bolte et al., 2009; Keenan, 2015). For instance, reducing the intensity of forest thinnings and rotation lengths has been suggested as the best practice to enhance stem volume production and the profitability of Norway spruce in southern Sweden due to reduced storm risk, root and butt rot (Subramanian et al., 2015).

5.4 | Substitution effects

The relative magnitude of substitution effects is key to the understanding of the mitigation effects of forest use and different forest products. At 'low' levels of substitution (1 m^3 to 0.5 tonne CO_2e), the *increased nature conservation* scenario performs better than the *maximum potential harvest* scenario up until approximately 2085. However, in later years and especially with larger substitution effects, the *maximum potential harvest* scenario quickly becomes the better short- *and* long-term scenario. Moreover, even under the more conservative low substitution estimate, the long-term differences are substantial. Vis-à-vis the climate, only the *positive climate effects* and the *increased fertilization* scenarios perform better than the maximum forest use scenario. The magnitude of the substitution effect thus strongly impacts outcomes.

Hudiburg et al. (2019) suggests that, at least in places like the United States, large shares of HWP simply end up in landfills and are never used for substitution. While such outcomes indicate important 'missed opportunities', the substitution that occurs earlier in the HWP life cycle still remains. HWPs can substitute for a range of carbon-intensive products (cement, steel, plastics, glass). And long-lived HWPs sequester carbon over extended periods, while newly planted forests simultaneously sequester additional carbon. The circular bioeconomy clearly falters when end-of-life-cycle wood resources are squandered. Prior substitution effects, however, are not thereby eliminated: only opportunities for additional substitution lost. Moreover, even if HWP resources end up in landfills, this is by no means a justification for reducing forest use. Such shortcomings instead signal failed policy intervention and inefficient resource use, requiring corrections of a different kind.

Several factors related to circular economy principles can influence the magnitude of substitution effects. The *quality* of the circular bioeconomy can be measured in the relative efficiency of wood resource use, the number of times forest products are used and reused for different purposes, and the way recycling multiplies the substitution effect (Lundmark et al., 2014; Stegmann et al., 2020; Ubando et al., 2020). The relative share of wood ending up in long-lived vs. short-lived HWPs further influences the mitigation effect. The carbon in short-lived HWPs can, however, remain in the HWP pool via material recycling. When substitution arises several times, this increases GHG savings per unit of wood harvested. When wood waste is used as a bioenergy feedstock, high conversion efficiency is also an important objective. Deployment of carbon capture, utilization and storage (CCUS) can further enhance the circular bioeconomy by recycling biogenic carbon into new products or by storing carbon in geological reservoirs (Shahbaz et al., 2021; Tcvetkov et al., 2019).

The substitution effect is thus a compound component of the following elements: the HWP energy content, the HWP carbon pool content and the substitution content, which in turn is a compound effect made up of multiple substitutions along the entire product pathway. Improving the efficiency and effectiveness of the circular bioeconomy presumably requires public policy intervention, i.e., policy frameworks that promote or require circular behaviours and material flows (e.g. fines on disposing wood resources in landfills, legal requirements on paper and used wood resource recycling, value added and other related tax reductions/benefits to encourage HWP use instead of GHG-intensive products). Policy interventions can further support research on the quality of the circular bioeconomy.

The calculation of multiple substitution rates begs the question of which substitution rates are most appropriate? There is, however, no easy answer to this question. In an analysis of the Swedish marketplace, others estimated a substitution potential of 0.47–0.75 tonnes CO_2e/m^3 stem volume (Lundmark et al., 2014), in line with our lower conservative estimate. Since Lundmark et al.'s study was conducted, the resource and energy efficiency of the forest industry has improved, and the product portfolio expanded, suggesting the substitution factor is now likely higher. Leskinen et al. (2018) provide a review of some 51 studies which provide estimates of different substitution factors ranging from −0.7 to 5.1 kg C/kg C (approximately −0.53 to 3.83 tonnes $CO_2e/1\ m^3$), with an average of 0.9 tonnes $CO_2e/1\ m^3$ and 90% of estimates on the positive side of this range. To the extent substitution effects can be compounded by shifting to longer lived HWPs, increasing the efficiency of wood resource use and reuse via material recycling and ensuring that incineration plants effectively use the energy content of wood waste, the relative substitution effect will be larger.

Additional concerns regard declining substitution potential due to the decreasing GHG intensity of economies (Brunet-Navarro et al., 2021; Harmon, 2019). However, carbon-based fuels and materials are expected to remain important because they are essential in energy, transport and industrial infrastructures which change slowly. In a scenario where variable electricity generation based on solar and wind increases, biomass will remain an important alternative to fossil fuels providing the balancing power needed to maintain power stability and quality (Tafarte et al., 2020). Similarly, electrification of the transport sector is a relatively slow process due to long turnover times of the vehicle stock. Biofuels will therefore remain an alternative to petrol and diesel in the coming decades (Bacovsky, 2020). Moreover, biofuels will remain an alternative to fossil fuels for longer times in sectors where the substitution of carbon-based fuels is difficult, such as long-distance aviation and marine transportation (Skea et al., 2022).

Carbon dioxide removal (CDR) from the atmosphere will likely be required to meet the Paris Agreement goal of keeping the increase in the global average temperature well below +2°C above pre-industrial levels (Rogelj et al., 2019). Carbon sequestration and storage via forests and forest product-based net removals, including construction wood in buildings and CCUS, are among the options for providing CDR (Burns & Nicholson, 2017; Churkina et al., 2020). Thus, while both future substitution effects of forest products and the relative importance of forest/biomass-based CDR are difficult to project, the scientific literature suggests they will be important in the longer term for reaching climate goals (IPCC, 2019).

For another, as long as forests are sustainably managed and the net annual exchange of biomass use and net annual biomass growth is zero (i.e. harvest does not exceed gross growth), HWPs should remain a core component of a carbon neutral circular bioeconomy, even as GHG savings from substitution become less relevant. By the same token, squandering biomass resources and forest residues by failing to use them represents an absolute loss to the system. A key circular economy issue is how best to source 'renewable' resources and *avoid*, or *reduce*, the use of scarce, *non-renewable* resources. The availability of renewable, replenishable resources is of great significance, especially under the more general conditions of limited resources, peak resource production and declining resource availability. Precisely because wood resources can continue to meet demand due to their *circular* benefits, more emphasis should ideally be placed on better understanding the limits of sustainable forest resource use, in conjunction with the relevant biodiversity requirements.

5.5 | LULUCF in the EU climate policy framework

The FRL strategy potentially discourages some Member states from fully using available forest resources for wood production (Ellison et al., 2014, 2020, 2021; Nabuurs et al., 2017). Although the FRL is theoretically set to protect annual net removals (sinks) and limit the increasing intensity of forest use (Grassi et al., 2021; Matthews, 2020), it ignores the substitution-related benefits of the HWP and bioenergy-based *avoided emissions* accounted outside the LULUCF sector. Since gains (*avoided emissions*) are not weighed directly against harvest-related emissions, the FRL approach may conflict with the bioeconomy interests of many Member states. Thus, to the extent that substitution represents an efficient and effective mitigation strategy, existing policy frameworks fail to represent this appropriately and can thus discourage effective forest use.

Clearly, part of the answer to this question lies both in the magnitude of the real substitution effect, as well as in the shape of public policy intervention. Although the FRL strategy may limit forest use intensity, there is little evidence this will either promote additional forest protections and conservation, or that it will promote climate change mitigation at comparable rates. In fact, our results suggest the opposite. To the extent the FRL has the potential effect of reducing forest use, it is likely to increase mortality and thus reduce the production of usable forest biomass and its related substitution effects. We have highlighted the significant losses in terms of the future biomass resource.

The FRL strategy further sets limits on countries that have regularly been harvesting comparatively low shares of the available net increment across the 2000–2009 period. The EU LULUCF ruling essentially locks in behaviour and suggests countries should continue to harvest *at the same rate*: that is, 'harvest intensity should not increase'. The Netherlands, for example, harvested approximately 55% of the annual net increment over the period 2003–2013 (Arets & Schelhaas, 2019), while Sweden has historically harvested a significantly large share of its overall net increment (on average, approximately 82% over the period 2000–2009, excluding commercial thinnings) (Swedish Ministry of the Environment, 2019). This approach thus discourages countries that use smaller shares of their forest resource from increasing production and promoting increased substitution.

Equally important but frequently neglected (see e.g., Gustavsson et al., 2022; Skytt et al., 2021) are the leakage effects potentially driven by the FRL and/or increasing forest set-asides, thereby reducing the amount of European forest available for harvest. Leakage can take on at least two different forms. On the one hand, it can result in the increased use of carbon-intensive materials like cement and steel (Churkina et al., 2020; Elhacham et al., 2020; Holmgren, 2021). On the other, leakage can lead to increased forest use in other locations around the world (Grassi et al., 2018; Kallio et al., 2018; Solberg et al., 2019). Increased consumer demand pressures on international trade may drive carbon and biodiversity loss in parts of the world that still host the principal share of global primary forests and some of the richest carbon stores (Santoro et al., 2021).

By excluding a share of the forest resource from harvest, the FRL further has the effect of increasing uncertainty regarding future forest resource use, thereby weakening investment incentives. While the potential to gain carbon credits from afforesting unmanaged forest lands may make up for this in some cases, the EU LULUCF regulation requires afforested lands outside forest management be integrated into MFL after a period of 20 years (EU, 2018/841). Because new forest growth can only be accounted for the first 20 years but then presumably becomes subject to the MFL-based FRL and cap strategies, both public and private sector investment incentives may be diminished. Investors interested only in the long-term set-aside effects on biodiversity, for example, may lose interest due to the threat of eventual privatization, while profit-seeking investment is weakened through the FRL and the cap.

On the other hand, to the extent land set-asides and increased forest protections do not affect the practice of forestry, they will likely influence overall mitigation potential only to the extent they involve the regeneration of degraded forest lands. In Sweden, for example, newly proposed land set-asides do not include intensively managed lands. Similarly, the plan to set aside some of the remaining primary forests in Europe does not address degraded lands and is not likely to significantly affect mitigation potential. This could, however, have important impacts on European biodiversity (Sabatini et al., 2018, 2020).

To optimize the mitigation effects of forestry, it is preferable to consider substitution effects and remove inflexibilities in trade across sectors (Ellison et al., 2021). Understanding what is best for the climate requires studying all land and atmosphere fluxes over extended periods of time. Current consideration of the next version of the EU LULUCF policy framework (COM(2021) 554 final) provides opportunities to address these concerns.

6 | CONCLUSIONS

Although storing carbon in standing forests can clearly contribute to climate change mitigation, this strategy has definable limits and potentially unrecognized opportunity costs. To achieve long-term reductions of atmospheric CO_2, it may be better to view the forestry enterprise—the circular forest-based bioeconomy—as the principal mechanism by which climate change mitigation can be progressively maximized, that is, by simultaneously increasing the magnitude of total annual forest growth alongside carbon sequestration and substitution effects. As our scenarios suggest, the reduction in atmospheric GHG concentration is maximized when forest growth and the potential annual substitution effect are maximized.

Given constant climate conditions and compared to the *maximum potential harvest* scenario, the net effect of increasing forest set-asides on a relatively modest share of productive forest land (18.5%) is estimated at −13.9 $MtCO_2e$/year from peak to perpetuity. The total net carbon sequestration impact of Swedish LULUCF during the second commitment period is approximately −49 $MtCO_2e$/year. Based on this total, future mitigation losses from increased forest set-asides amount to approximately 28% per year from peak to perpetuity. If we consider the potential magnitude of substitution effects, as well as the benefits of improved policy intervention, this amount could be greater.

Pursuing forest management as a strategy for maintaining and strengthening the forest role as a 'regulator' of atmospheric GHG concentrations thus makes good sense. Policy interventions that could meaningfully mobilize the climate benefits of forest use are, however, currently hamstrung by a preference for carbon sequestration and storage in forests and a perceived need to harness forest use intensity. As the above scenarios suggest, more refined policy interventions could go a long way toward better mobilizing forest use in favour of the climate. The key policy and research innovations that could further help mobilize forests in favour of the climate are as follows: support for technology and market development in circular bioeconomy solutions; achieving greater flexibility in the trading of carbon credits across the multiple sectors (pillars) of the climate policy framework; eliminating the 'no-debit rule', the FRL and the cap; and improving the accounting of, knowledge about, and policies surrounding substitution effects.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

AUTHOR CONTRIBUTIONS

HP conceptualized and designed the study; DE designed the study within the FORCLIMIT project. DE, HP, GB, GE, JS, ML, TL and AAM drafted and revised the manuscript. PW, JS, AL and AAM were involved in data acquisition, analysis and interpretation.

DATA AVAILABILITY STATEMENT

The article data has been made available at: https://zenodo.org/record/6390892.

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