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Climate change mitigation potential of biochar from forestry residues under boreal condition

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Abstract

Forest harvest residue is a low-competitive biomass feedstock that is usually left to decay on site after forestry operations. Its removal and pyrolytic conversion to biochar is seen as an opportunity to reduce terrestrial CO_2 emissions and mitigate climate change. The mitigation effect of biochar is, however, ultimately dependent on the availability of the biomass feedstock, thus CO_2 removal of biochar needs to be assessed in relation to the capacity to supply biochar systems with biomass feedstocks over prolonged time scales, relevant for climate mitigation. In the present study we used an assembly of empirical models to forecast the effects of harvest residue removal on soil C storage and the technical capacity of biochar to mitigate national-scale emissions over the century, using Norway as a case study for boreal conditions. We estimate the mitigation potential to vary between 0.41-0.78 Tg CO_2 equivalents yr^{-1}, of which 79% could be attributed to increased soil C stock, and 21% to the coproduction of bioenergy. These values correspond to 9-17% of the emissions of the Norwegian agricultural sector and to 0.8-1.5% of the total national emission. This illustrates that deployment of biochar from forest harvest residues in countries with a large forestry sector, relative to economy and population size, is likely to have a relatively small contribution to national emission reduction targets but may have a large effect on agricultural emission and commitments. Strategies for biochar deployment need to consider that biochar's mitigation effect is limited by the feedstock supply which needs to be critically assessed.

Keywords: biochar; boreal forests; carbon dioxide removal; forest harvest residues; national-scale emission reduction; negative emission technology

1. Introduction

Global CO_2 emissions from fossil-fuel burning and industrial processes have increased by 1.3% each year for the last decade (Friedlingstein et al., 2019), setting Earth on a course of rapid climate change with consequences to global health and safety (IPCC, 2018). Large inconsistencies remain between science-based targets and national commitments, and immediate actions need to be taken to “decarbonize” human activities and curb climate change (Rockström et al., 2017). Reducing emissions of greenhouse gases (GHG) and improving the strength of natural carbon (C) sinks are key strategies to mitigate the increase in atmospheric CO_2 content (Rumpel et al., 2018; Vermeulen et al., 2019).

The cumulative emission of GHG gases must be kept below a maximum upper limit to stabilize the global mean temperature (Hansen et al., 2008; Meinshausen et al., 2009). Consequently, emission reduction alone cannot lower the risk of exceeding a dangerous and irreversible climate change (Solomon et al., 2009), thus technologies that can remove CO_2 from the air must be additionally implemented to achieve long-term climate change mitigation (Anderson and Peters, 2016). The Paris agreement sets the long-term goal of limiting global warming this century to “well-below” 2°C above pre-industrial levels. To avoid rising atmospheric GHG concentrations and to achieve the Intended Nationally Determined Contributions (INDCs) set by the Paris agreement, we are required to deploy negative emission technologies (NETs) that can remove CO_2 from the atmosphere over a regional scale (Anderson and Peters, 2016). The vision of future cost-effective NETs is politically appealing, but their true potential and risks for failure need to be carefully assessed before implementation in national emission reduction plans (Fuss et al., 2014).

Biochar is a recalcitrant C-rich solid product created from pyrolysis of biogenic organic residues (e.g. sludge, wood- and agricultural waste) that is applied to soil to improve soil C storage. Biochar is counted as one of the most viable options among NETs, because of its C sequestration potential and low environmental footprint and cost impacts (Smith, 2016; Tisserant and Cherubini, 2019). The climate benefit of biochar stems mainly from its slower decomposition rate than the raw biomass from which it is generated from (Lehmann et al., 2006). Biochar also provides several co-benefits such as providing renewable energy products (e.g. bio-oil and syngas) that can displace fossil fuels, reduce GHG emissions of N_2O and CH_4 from soil (Blanca Pascual et al., 2020; Borchard et al., 2019), and increase crop yield in degraded agricultural soils by improving soil conditions and nutrient retention (Jeffery et al., 2011).

On a global scale, the use of biochar may increase the terrestrial C sink by 0.6-11.9 Pg CO_2 yr^{-1} (Fuss et al., 2018) and displace a maximum 12% of anthropogenic emissions (Woolf et al., 2010). The mitigation potential of biochar depends on the rate at which feedstocks can be collected and processed (Fuss et al., 2018). However, most biomass feedstock compete with other demands and high economic costs impose constraints on biomass collection and therefore waste feedstocks are needed for an economically viable biochar deployment (Dickinson et al., 2015; Fuss et al., 2018; Vochozka et al., 2016).

Forest harvest residues are tree components with a low market value, which are typically left to decay on site after forestry operations. Since the mid-twentieth century, the European forest stocks have at least doubled in size as a result of maturing age structure and harvesting rates remaining lower than forest growth (Ciais et al., 2008; Nabuurs et al., 2003). Because of this increase in forest biomass and potential supply excess, conversion of forestry residues into biochar and its long term C-storage in soil could be an important element in pursuing national emissions reduction targets and mitigating climate change, particularly in the boreal region where the forestry residues are usually not collected. Furthermore, forest harvest residues present advantages over other organic waste feedstocks, as it can be harvested year-round, which is a benefit in cold climates with short growing seasons. Although combined collection of tree stems and harvest residues has been shown to reduce forest soil organic carbon stocks, the C losses are usually lower under cold climates (e.g. boreal conditions) compared with temperate climates (Achat et al., 2015).

Management of boreal forests is an important component for climate change mitigation strategies as boreal forests store 32% of the global forest C stock (Pan et al., 2011). The Nordic boreal forests have been under intensive management during the past century, resulting in an increased harvest yield potential with the growing stand density (Lundmark et al., 2014; Rautiainen et al., 2011). Forest management of Nordic boreal forests is characterized by patch clearcutting which produces large volumes of residues which are usually left to decay at site. Relative to the size of the population and economy, the Nordic forest sector is large, and because of the substantial volume of forest residues produced each year, the Nordic region presents an attractive case location for the analysis of the climate change mitigation potential of biochar supplied by forestry under boreal conditions.

In the present study, we used Norwegian forest inventory data and empirical models of forest growth and logging activity to quantify the technical capacity of biochar made from forestry residues to mitigate national-scale emissions, using Norway as a case study for boreal conditions. We forecasted the supply of forest residues from Norwegian forests over the period

2020-2120 and performed a biomass-C budget analysis to quantify the effects of harvest residue removal and biochar amendments on soil C storage over the combined forest-biochar system. Decomposition dynamics in forest soils was modelled using the Yasso07 decomposition model (Tuomi et al., 2009). The C sink potential of biochar was assessed using emission coefficients sourced from the 2019 refinement to the 2006 IPCC Guidelines for national GHG inventory (IPCC, 2019), and fossil fuel offset was estimated by calculating energy yield from producing biochar from forestry residue. The mitigation potential of biochar, inclusive of the avoided emissions of GHG from the co-production of bioenergy, was evaluated for two biochar deployment scenarios, one represented by economically constrained conditions (scenario 1) and another represented by a maximal forest residue utilisation (scenario 2). The net effect was compared against Norway's Intended Nationally Determined Contribution (INDC) set by the Paris agreement and its national production-based emissions.

2. Methods

2.1 Scenario description

Two biochar deployment scenarios were evaluated with the results compared against a non-biochar reference scenario. For both scenarios we simulated the difference in soil C sink over time (from 2020 to 2120) and assumed that biochar was produced from annual supplies of forest harvest residues (crown, unmarketable stem sections and foliage). Scenario 1 was represented by an economically limited scenario where the harvest residue supply is constrained by the expected costs. Specifically, the extraction costs to road side was constrained to 30 Euro/ton which, according to Bergseng et al. (2013), yield an annual feedstock availability of approximately 0.85 Tg per year. Scenario 2 represents the maximum-intensity deployment of biochar, where 70% of the total residues were assumed to be used for biochar production, representing the maximal yield residue recovery after logging operations (Nurmi, 2007).

2.2 Forecasting harvest residue removal

All the data used for this study were from the Norwegian national forest inventory (NFI) which records forest resources from a 3 x 3 km grid on 22,008 permanent plots (Breidenbach et al., 2020), 58% of which are classified as forest. We used a total of 12,307 plots, representing 12.56 Mha from the last complete measurement of the Norwegian NFI (2013-2017). The records comprised individual tree measurements such as diameter and height, as well as forest characteristics such as species composition, site index and stand age (Breidenbach et al., 2020), as well as information on silvicultural treatments that have been implemented since the last measurement.

From the Norwegian NFI data, forest development was forecasted over 2018-2120 using the sitree simulator R package (Antón-Fernández, 2019). Climate change was included in the simulations through a climate-sensitive site index model (Antón-Fernández et al., 2016), and the climate data followed the IPCC scenario RCP 4.5, downscaled to a 1 x 1 km grid according to SeNorge (Lussana et al., 2019). The total supply of logging residues (tree tops, branches and needles) was estimated using the species-specific tree allometric equations developed by Smith *et al*. (2016, 2014) Marklund (1988), and Petersson and Ståhl (2006).

Logging activity was predicted based on the single tree simulator (sitree) and followed a similar approach to the Forest National Accounting Plan of Norway (Ministry of Climate and Environment, 2020). In short, the total forest area was divided into seven strata according to the dominant tree species, site quality and the expected cost for felling, which were further divided into young and mature forest. Using the last three measurements of the Norwegian NFI (2003-2017), the ratio between the total- and felled area was calculated for each stratum and maturity class and used as a proxy for harvest intensity. For each stratum and maturity class the plots were ranked according to the probability of a harvest model fitted to NFI data as described by Anton-Fernandez & Astrup (2012), until the area defined by the harvest intensity was reached. This harvest model predicts the probability of thinning and final felling based on forest attributes and assumes that harvests are more frequent when profit can be made, thus the probability of final felling increases with site index (site fertility), volume and maturity, and decrease with slope and distance to road (Antón-Fernández and Astrup, 2012).

At the beginning of the simulations (2020) 5% of the total forest area was protected forests, corresponding to the current area protected in Norway, and we assumed that the protected area was increasing by 15000 ha yr^{-1} until 10% of the total forest area was under protection. No form of harvest was allowed on protected areas. We also assumed 15-83% harvest restrictions for forests located within urban areas, mountains, riparian zones and swamps, according to legislation and certification schemes; for further details see Søgaard et al. (2012). When more than one restriction category applied the highest percentage was used. For “mountain forest” we followed the definition described by Stokland *et al*. (2020), while riparian forest plots compromised all the NFI plots with a center <10 m from a mire, stream or water body. Swamp forests corresponded to NFI plots with waterlogged organic soils and vegetation types characteristic of wet woodlands, according to the vegetation classification system of Larsson (2005). Restriction categories were established from meta-data of the NFI and by overlaying maps, maintained by the Norwegian Mapping Authority, the Norwegian Environment Agency and the Norwegian Institute of Bioeconomy Research.

2.3 Accounting for fate of harvest residues left in the forest

To forecast forest soil carbon stock changes due to the removal of forest harvest residues, we used the soil carbon and decomposition model Yasso07 (Tuomi et al., 2009). The modelling features of "Yasso07" corresponds to the IPCC Tier 3 method and thus represents the highest standard of analytical complexity. In Yasso07, decomposition is described nonlinearly, and organic matter is divided into five different compound groups, according to their solubility (acid-, water-, ethanol- and non-soluble, in addition to humus), and assumes a mass loss rate for each group (Tuomi et al., 2009), and the resulting development of the soil C stock is projected based on litter chemistry, air temperature and precipitation. Since the net effect on soil C stock and GHG emissions of biochar varies with time, Yasso07 and other tier 3 models are suitable. Yasso07 is run assuming no climate change to avoid systematic error differences between the regions of Norway, see Dalsgaard et al. (2016) for further details.

2.4 Contribution of biochar to soil C stock changes and Monte-Carlo uncertainty analysis

The 2019 IPCC refinement includes the first IPCC methodology for national emissions accounting of biochar (IPCC, 2019). Based on the refined IPCC guidelines, biochar yields and contribution to C stock changes was calculated from the simulated logging residues by assuming a biochar mass yield of 30% from logging residues (Crombie et al., 2013; Woolf et al., 2014; Yan et al., 2011), biochar C content of 77%, and that 80% of the biochar C remains in soil after 100 years (IPCC, 2019). The fraction of biochar C content was based on pyrolysis wood and the fraction of biochar C to remain in soil was based on estimates assuming a medium pyrolysis temperature (450-600°C; IPCC, 2019).

Uncertainty of the residue-to-biochar conversion factors was accounted for by allowing the applied factors to randomly vary with a Gaussian distribution and according to 95% confidence intervals, based on the variances reported by the refined guidelines of IPCC. In short, the conversion factor of biochar C content (0.77) was allowed to vary by a factor of ±0.42 and the fraction of biochar C to remain after 100 years (0.8) was allowed to vary by ±0.11 (IPCC, 2019). For the factor of char yield (0.3) we assumed it to vary by a factor of ±0.17 since char yield from pine wood chips usually varies between 25-35% (Crombie et al., 2013; Yan et al., 2011). Variation in the conversion variables was then used in a Monte-Carlo analysis, where the random variation was assigned to the calculation to quantify the uncertainty that was propagated to the final C balance predictions. Briefly, the calculations were bootstrapped 5000 times with the assigned variation, and the 5th and 95th percentile of the range of the calculations was used to assess the total uncertainty of the predicted biochar C stock changes. Changes in forest- and net soil C stock between the two scenarios was evaluated for statistical significance using Gaussian generalized linear regressions in R version 3.5.2 (R Core Team, 2017).

2.5 Biochar energy yield and potential fossil fuel offsets

Maximal energy yield (Mj) from biochar production was calculated according to Woolf *et al*. (2010), following the formula:

$$E_{max} = m_{dm}LHV_{dm} - m_{bc}HHV_{bc} - m_w(\Delta H_{VAP} + (T_{VAP} - T_A)C_w) \qquad \text{Eqn 1}$$

where,

m_{dm} = mass of feedstock (dry weight of forest harvest residues)

LHV_{dm} = lower heating value of forest harvest residues (19.2 MJ kg^{-1}) (Ringman, 1995)

m_{bc} = mass of biochar (assuming a biochar mass yield of 30%) (IPCC, 2019)

HHV_{bc} = higher heating value of biochar derived from wood (31.2 MJ kg^{-1}) (Phyllis2)

m_w = mass of water generated from the pyrolysis of forest harvest residues

ΔH_{vap} = specific latent heat of evaporation of water (2.26 MJ kg^{-1})

T_{VAP} = evaporation temperature of water (100°C)

T_A = ambient air temperature (taken as 20 °C)

C_W = specific heat capacity of water (0.00418 MJ kg^{-1} K^{-1})

The realized energy yield from pyrolysis (E) was calculated based on the maximum energy yield (E_{max}) and the pyrolysis energy efficiency (η_p), represented by the proportion of energy recovered from the theoretical maximum, according to:

$$E = \eta_p E_{max} \qquad \text{Eqn 2}$$

Energy efficiency was assumed to represent 38%, based on the operation efficiency of 'Best Energies' pyrolysis plant when optimized for biochar production (Gaunt and Lehmann, 2008). The amount of C released in delivering the energy (C emission penalty) must be known to calculate fossil fuel offset. For the calculations, we assumed that coal, oil and natural gas have a C emission penalty of 0.024, 0.019 and 0.014 Mg C GJ^{-1} (Fowles, 2007), respectively.

Together with the energy obtained from the pyrolysis (E) and the emission penalty, the potential fossil fuel substitution from the production of biochar was calculated according to the equation (Woolf et al., 2010):

$$A = E\ C_E \eta_g / \eta_f \qquad \text{Eqn 3}$$

where,

A = Avoided C emission from producing biochar

E = realized energy yield form pyrolysis of biochar feedstock

C_E = carbon emission penalty

η_g = the fraction of thermal energy that is obtainable from the pyrolysis gas (32%)

η_f = the fraction of thermal energy that is obtainable from fossil fuels burning (40%)

In the calculations we assumed that the total energy production from the displaced non-renewable sources was 0.4% coal, 48.0% oil, and 51.5% gas and based on primary energy production over 2009-2019 (Statistics Norway, 2020). The ratio of η_g/η_f was assumed to be 0.32/0.40. Finally, we assumed carbon costs associated with the energy consumed for feedstock transportation and processing was proportional to export of residues and equivalent to 2.5% of the biochar C storage (Woolf et al., 2010). Thus C-cost for transport was fixed for scenario 1 and variable for scenario 2, according to the level of residue removal.

3. Results

3.1 Changes in forest biomass stock and forestry residue supply

Forest biomass stock was predicted to increase by 82% over 2020-2120 and 9% over 2020-2030 (Fig. 1a). As a result of growing biomass stock, the Norwegian forests were predicted to act as a sink of atmospheric CO_2 and assimilate about 17.2 Tg CO_2-eq yr^{-1} over 2020-2030 and 15.0 Tg CO_2-eq yr^{-1} over 2020-2120 (Fig. 1b). Because of the increase in forest biomass, the feedstock supply of forest harvest residues for biochar production was forecasted to increase from 0.8 to 1.2 Tg over 2020-2120 (Fig. 1c).

3.2 Changes in forest soil carbon stock from forestry residue removal

Over the initial five years, removal of forest harvest residue for biochar production decreased the simulated C sink capacity of forest soils by 0.44 and 0.39 Tg CO_2-eq yr^{-1} for scenario 1 and

2, respectively (Fig. 2a). The decline in forest soil sink capacity varied over time ($P < 0.01$) and was different between the two scenarios ($P = 0.01$). For scenario 1 and 2, the median decline in forest soil sink was 0.02 and 0.12 Tg CO_2-eq yr^{-1}, respectively (Fig. 2a). For scenario 1, forest soil C sink strength was predicted to reach a steady-state around 2040, at which the simulated forest soil C stock was predicted to have decreased by 5.0 Tg CO_2-eq since 2020 (Fig. 3). For scenario 2, forest soil C stock was predicted to continuously decline at an average rate of 0.10 Tg CO_2 yr^{-1} (Fig. 2a). Over the entire simulated period (2020-2120), forest soil C stock was predicted to decrease by a total of 5.5 and 10.4 Tg CO_2-eq for scenario 1 and 2, respectively (Fig. 3), corresponding to an average reduction of 180 and 341 kg C ha^{-1} over the entire productive forest area of Norway (8.3 Mha).

3.3 Climate mitigation potential of biochar

Biochar produced from forestry residues was predicted to increase the agricultural soil C sinks by an average of 0.58 and 0.71 Tg CO_2-eq yr^{-1} and bioenergy obtained from the production of biochar was predicted to offset 0.15 and 0.19 Tg CO_2-eq yr^{-1} of GHG emissions for scenario 1 and 2, respectively (Fig 2b). For scenario 2, the displacement of fossil fuels was predicted to increase (P<0.001) by 0.92 Gg CO_2-eq yr^{-1} to displace a total of 18.8 Tg CO_2-eq by year 2120 (Fig 3b). The overall climate benefit of biochar was estimated to be 79% from the sequestration of biochar-C and 21% from the coproduction of bioenergy (Fig. 3).

On average, the net mitigation effect of biochar was predicted to correspond to a CO_2 removal of 0.66 (±0.013) and 0.78 (±0.022) Tg CO_2-eq $yr^{-1}$for scenario 1 and 2, respectively, to achieve a cumulative fossil fuel offset and soil C stock change corresponding to 68.1 and 80.4 Tg CO_2-eq by 2120 (Fig. 3), and a total net effect of 65.9 and 77.9 Tg CO_2-eq. Over the initial ten years (2020-2030), the net mitigation effect of biochar was 35-45% lower than the average, corresponding to a sink strength of 0.41 Tg CO_2-eq yr^{-1} for both scenarios. Consequently, biochar was predicted to have removed 4.1 Tg CO_2-eq by 2030 (Fig. 3).

4. Discussion

Here we quantified the technical climate change mitigation potential of biochar from forest harvest residues under boreal conditions, using Norway as a case study. From the increase in soil C storage and the displacement of fossil fuels we forecasted that 0.66-0.78 Tg CO_2-eq emission would be mitigated on average each year over 2020-2120 when 0.80-1.20 Tg logging residues are used as a feedstock for producing biochar. Because of a rapid initial decline in forest soil C sink capacity, the climate benefit of biochar was estimated to be 35-45% lower than the average mitigation effect, over the initial ten years (2020-2030). Over this period, the climate benefit of biochar was estimated to be 0.41 Tg CO2-eq yr^{-1}, representing 0.8% of the current annual GHG emissions of Norway (52.5 Tg CO_2-eq yr^{-1}; Statistics Norway, 2020) and 1.6% of the emission reduction target of 50% reduction by 2030, according to the INDC of the Paris agreement. The mitigation effect over 2020-2120 corresponded to 1.3-1.5% (0.66-0.78 Tg CO_2-eq yr^{-1}) of the GHG emissions of Norway. In comparison, forest biomass stock was predicted to increase by 9% over 2020-2030, corresponding to a C removal of 17.2 Tg CO_2-eq yr^{-1} and 66% of the INDC. Furthermore, the mitigation effect of biochar corresponded to 9% of the agricultural emission (4.5 Tg CO_2-eq yr^{-1}) over 2020-2030, and 15-17% over 2020-2120. The commitment of the agricultural sector is to reduce emissions by 5 Tg CO_2-eq over 2020-2030 (Government of Norway, 2019), thus it would about 12 years to reach that target under our biochar deployment scenarios.

Among the Nordic countries, Finland, Sweden and Norway are the largest producers of forest products and silviculture is characterized by patch clearcutting of *Picea abies* and *Pinus sylvestris* forest stands, which yields vast quantities of residues that remain on clearcut areas. In Finland and Sweden, wood harvest yields are respectively 6 times (56.8 Mm^3) and 9 times (83.9 M m^3) greater than the harvest volume of Norway (9.6 Mm^3), based on NFI data (Natural Resources Institute Finland, 2021; Norwegian Agriculture Agency, 2021; Swedish national forest inventory, 2021). Assuming that the supply of harvest residue is proportional to harvest yield, and a maximal residue recovery of 70%, the climate benefit of using the entire Nordic supply of logging residues over 2020-2030 would be 6.0 Tg CO_2-eq yr^{-1}, representing 11% of the annual GHG emissions of Norway. Furthermore, wood harvest yield in Russia (176 Mm^3), Canada (157 Mm^3) and Alaska (<1 Mm^3) is about twice the volume of the Nordic countries combined (150 Mm^3) (Canadian council of forest ministers (CCFM), 2020; Food and Agriculture Organization (FAO), 2012; Marcille et al., 2017). Thus, by using the logging residue supply from the majority of the boreal forest biome, the climate benefit from producing biochar from that feedstock would mitigate 20.2 Tg CO_2-eq yr^{-1}, about 39% of the national emissions of Norway. By 2120, we estimated that biochar produced from residues from the Norwegian forest sector has the potential to mitigate a total of 65.9 and 77.9 Tg CO_2-eq. Scaled up over the entire boreal region and based on wood harvest yields, biochar from forestry residues has the potential to mitigate about 3300-3900 Tg CO_2-eq over the next hundred years. In comparison, Woolf et al. (2010) estimated that biochar produced from the global supply of forestry residues to have the capacity of mitigating about 4800 Tg CO_2-eq over the century.

A key challenge of using forest residues for biochar or bioenergy purposes is the removal of nutrients, which can impair forest growth over the long term and thus it C sink potential (Helmisaari et al., 2011). Under boreal conditions forest growth is mainly limited by the availability of nitrogen (N), as most of the N is assimilated in biomass, litter and humus (Högberg et al., 2017). Assuming that logging residues from Norway spruce and Scots pine have an average N content of 0.48% (Helmisaari et al., 2011), about 3900-5800 metric ton N would be removed each year under the two study-scenarios. This amount of N corresponds to 260-425 km^2 of conventional N application (150 kg N ha^{-1}; Pettersson & Högbom, 2004), representing about 0.4-0.6% of the total productive forest area of Norway. In comparison, about 50-100 km^2 forest area is N fertilized annually in Norway (Norwegian Environment Agency, 2014), thus the current fertilization regime would need to increase by a factor of 3-8 to compensate for the N removed with logging residues, which would impose additional energy costs to produce fertilizer and cause leaching of N to water sources (Skowrońska and Filipek, 2014). To limit nutrient removal with residues the recommended practice is to leave the residues in the field for one year before collection. About 50% of the N in residues are stored in needles (Ukonmaanaho et al., 2008), thus the corresponding N removal can be reduced by half by recovering the residues when the majority of the foliage has shed off the branches. Storage of logging residues on the logging area decreases residue's dry matter by 27% over 6 months and 47% over 18 months (Thörnqvist, 1985). Under our biochar deployment scenarios, this would reduce the maximal residue recovery from 70% to about 51-37%, resulting in a 27-47% lower biochar mitigation potential.

While biochar application alone is not sufficient to satisfy nutrient demands for tree growth, biochar can indirectly affect growth by modifying forest soil properties (Li et al., 2018), thus negating some of the negative effects of harvest residue removal. In boreal forests, application of biochar may enhance stand growth by increasing soil N mineralization rates and NH_4 availability, in addition to reducing nutrient losses (Gundale et al., 2016). Furthermore, biochar-based fertilizer products, with the aim of increasing plant growth and N use efficiency (Shi et al., 2020), could directly contribute towards solving the problem of returning N to forests. However, forest growth responses to combined biochar and nutrient application are yet uncertain together with the long-term effects on soil properties and GHG emissions (Li et al., 2018).

Potentially, nutrient-enriched biochar may increase crop yield and N use efficiency under boreal and temperate conditions, where pure biochar applied in large quantities at a single application has shown to not increase crop growth (Jeffery et al., 2017; O'Toole et al., 2018;

Soinne et al., 2020). In addition to increasing soil C storage, biochar often reduces soil N_2O emissions by an average of 32% and decreases soil N leaching by 26% via sorption of nitrate (Borchard et al., 2019; Liu et al., 2018). As the ability of biochar to reduce N_2O emissions decreases with time (Borchard et al., 2019), it is likely that repeated application of biochar amended with nutrients may contribute to reduction of soil N_2O emissions while enhancing crop yield (Guenet et al., 2021). In Norway, N_2O emissions from agricultural soils account for about 1.7 Tg CO_2-eq yr^{-1}, which is 38% of agricultural emissions (Norwegian Environment Agency, 2018). Assuming such biochar products will be used on 25% of N applied to soil and that N_2O emissions are reduced by 32%, the corresponding mitigation effect would be an additional 0.13 Tg CO_2-eq yr^{-1}, representing 2.9% of the agricultural GHG emissions (4.5 Tg CO_2-eq yr^{-1}). While the yield of benefits from biochar application are less under boreal conditions, (Soinne et al., 2020), biochar-based fertilizers may reduce fertilization requirements which may displace emissions from fertilization production and make biochar more economically viable for farmers (Field et al., 2013; Sohi et al., 2010).

In the present study, the estimated mitigation potential of biochar was calculated from factors sourced from the refined IPCC guidelines and uncertainty in the factors may contribute to our error in our prediction. Pyrolysis temperature is the main factor determining the stability of biochar (Crombie et al., 2013), and when biochar is produced at medium pyrolysis temperature (450-600 °C the persistence of biochar is estimated to vary within 95% CI limits from 0.71 to 0.89 (IPCC, 2019). In the present study we assumed 80% of biochar C would remain in soil after 100 years, but it is possible that boreal conditions extend the persistence of biochar because of the cold climate. Assuming 71% and 89% of biochar C stability, the total mitigation effect of biochar over 2020-2030 would correspond to 0.34-0.47 Tg CO_2-eq for both scenarios, opposed to 0.41 Tg CO_2-eq when assuming 80% stability over 100 years. For comparison, assuming 100% persistence over 100 years would increase biochar's C sink capacity to 0.55 Tg CO_2-eq. Thus, either a faster or slower biochar decomposition rate would only have a minor contribution to the estimated mitigation effect. Similarly, extending our approach to an IPCC Tier 3 approach to include biochar decomposition dynamics would not affect the estimated mitigation potential to a major extent in terms of C storage.

Another uncertainty is the projected forest growth. Our harvest model is based on the assumption that logging activity is related to forest development and the standing stock volume. In the present study, the total Norwegian forest biomass was predicted to increase by 9% over the decade and by 83% over the century, assuming RCP 4.5 climate scenario. With a changing climate the trajectory of future forest growth is uncertain, but increase in growth is usually reported for European- and north-eastern US forests, at least under mild-moderate climate warming (Gustafson et al., 2017; Härkönen et al., 2019; Wang et al., 2017). However, future increases in forest stocks may be displaced by increased frequencies of natural disturbances (Nabuurs et al., 2003), as well as altered forest management regimes (Härkönen et al., 2019). Still, our results are in line with projections of future forest growth of European forests, predicting that the growing stock volume will increase by about 13% over the decade and 73% by 2100 (Härkönen et al., 2019). Thus, we consider the predicted biomass increase to be conservative, at least over the next decade

A third uncertainty is the extent to which biochar may affect mineralization of native soil organic matter. Cycles of C and N are tightly to soil microbial microbes and the biogeochemical effects of biochar are likely dependent the availability of resources to the microorganisms (Lehmann et al., 2011). Biochar may interact with microbial processes and thus affect the mineralization of soil organic matter (Cross and Sohi, 2011). On average, this effect results in a reduced mineralization of the indigenous soil organic matter (SOM) and a further increase in C sequestration (Wang et al., 2016), and biochar is now largely considered a method to increase the stability of non-pyrogenic organic matter in soils (Lehmann et al., 2006). However, the effect is variable, and accelerated SOM decomposition has also been observed, especially in poor sandy soils (Wang et al., 2016). The biochar effect on SOM mineralization rate also depends on pyrolysis temperature and feedstock type (Purakayastha et al., 2016), and there is currently a lack of long-term studies to fully evaluate these effects.

5. Conclusion

To limit global warming below 2°C by 2100, and possibly 1.5°C, according to the Paris agreement, drastic reductions of GHG emissions are required but not sufficient. Therefore, we depend on technologies that remove CO_2 from the atmosphere. Here we show that biochar produced from forest harvest residues in Norway has a maximal capacity to remove 0.41 Tg CO_2-eq yr^{-}1 over 2020-2120 and 0.78 Tg CO_2-eq yr^{-1} over 2020-2120, corresponding to 0.8% and 1.5%, respectively, of the current production-based GHG emissions of Norway. These values also correspond to 9-17% of the total GHG emission from the Norwegian agricultural sector and are nearly equal to the entire sector's emission reduction target set for 2030. This illustrates deployment of biochar produced from logging residues in countries with a large forestry sector, relative to economy and population size, may have only a small contribution to INDCs but may have a relatively large effect on agricultural GHG emission and commitments. Strategies for biochar implementation need to consider that the mitigation potential of biochar is limited by the supply of the feedstock which needs to be critically assessed to quantify the C removal of biochar. The potential positive and negative effects of biochar on agricultural-forest systems need to be carefully assessed before using biochar as a national-scale GHG emission mitigation measure. While biochar may contribute to increasing soil C storage in cold-temperate and boreal conditions, reduced GHG emissions and other strategies for CO_2 removal must be additionally implemented to reach national emission reduction goals.

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Figures and figure text

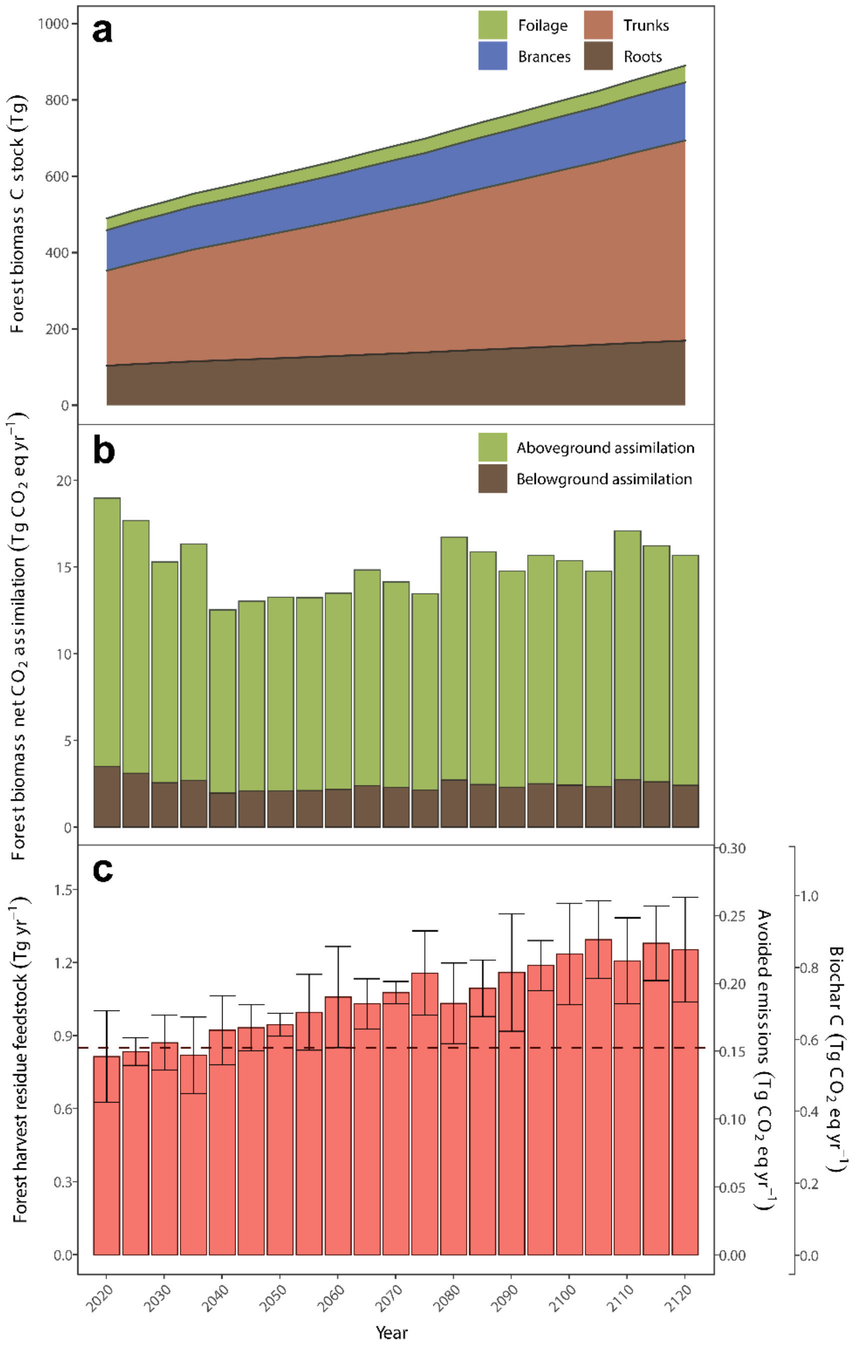

Figure 1 Forecasted increase in forest biomass C stock in Norway (a), assimilation rate of atmospheric CO_2 in forest biomass (b), and (c) development of Norwegian forest harvest residue feedstock (left y-axis), fossil fuel offset from biochar bioenergy (innermost righthand y-axis), and stock of biochar C in soil (outermost righthand y-axis). Dashed line in (c) represents conditions of scenario 1 when a fixed amount (0.85 Tg) of forestry residues are used as feedstock to produce biochar. Conversely, red bars in (c) represent conditions of scenario 2 when 70% of the national supply of forest harvest residues in Norway are used to produce biochar. Error bars in (c) represent ± one standard deviation.

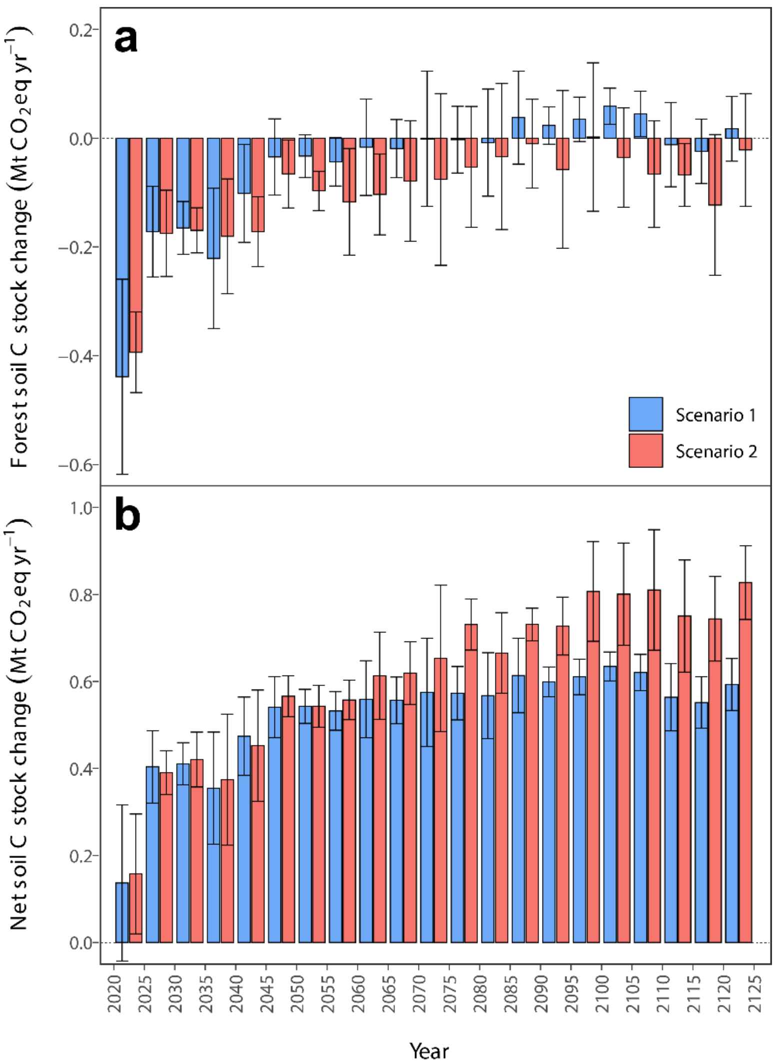

Figure 2 Forecasted forest- (a) and net changes (b) in soil carbon (C) stock under two biochar deployment scenarios when either a fixed amount (0.85 Tg) (scenario 1; blue bars) or 70% (scenario 2; red bars) of the national supply of forest harvest residues in Norway are used as a feedstock for biochar production. All differences in (a) and (b) are related to a business-as-usual scenario when harvest residues are left to decay at site. Error bars represent ± one standard deviation.

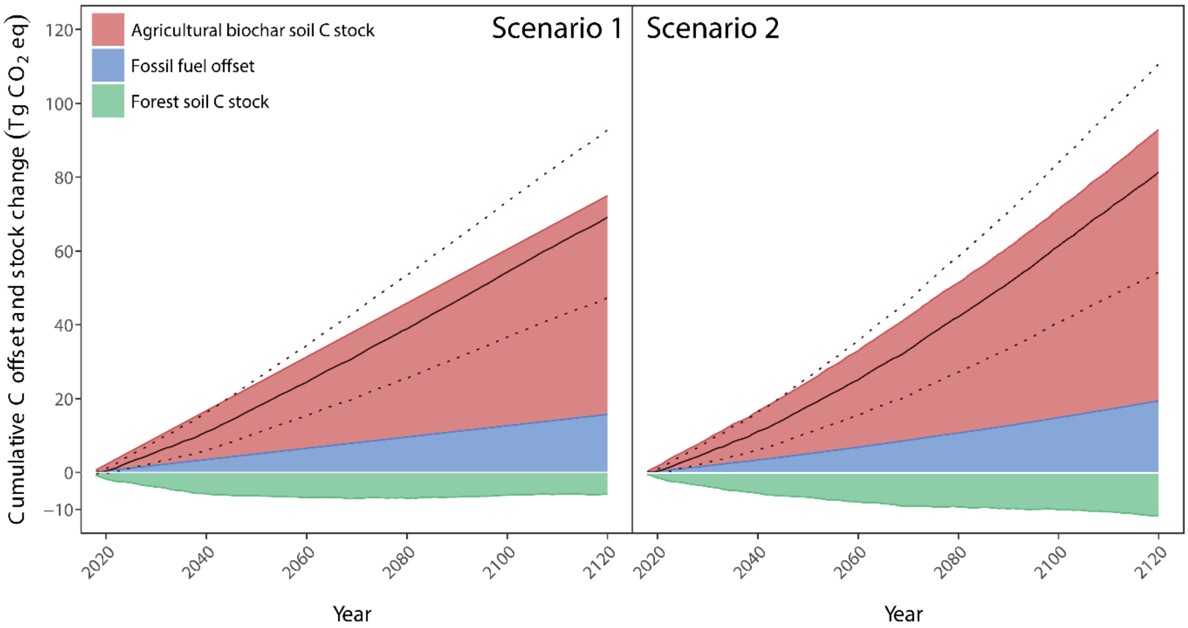

Figure 3 Cumulative carbon (C) offset and soil C stock changes in Norway under two different biochar deployment scenarios. Scenario 1 assume that a fixed amount (0.85 Tg yr^{-1}; Scenario 1) of the forest harvest residues are used as a feedstock for biochar and added to agricultural soils, whereas scenario 2 assume that 70% of the forest harvest residues are annually used as a feedstock for biochar. Solid lines indicate the cumulative net effect from increased biochar C storage (red), avoided emissions of greenhouse gases (GHG) from the coproduction of bioenergy (blue) and the decrease in forest soil C stocks (green). Distance between the two dotted lines corresponds to the range between the 5th and 95th percentiles of the data.

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