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Life-cycle assessment to unravel co-benefits and trade-offs of large-scale biochar deployment in Norwegian agriculture

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ABSTRACT

Limiting temperature rise below 2 °C requires large deployment of Negative Emission Technologies (NET) to capture and store atmospheric CO_2. Compared to other types of NETs, biochar has emerged as a mature option to store carbon in soils while providing several co-benefits and limited trade-offs. Existing life-cycle assessment studies of biochar systems mostly focus on climate impacts from greenhouse gasses (GHGs), while other forcing agents, effects on soil emissions, other impact categories, and the implications of a large-scale national deployment are rarely jointly considered. Here, we consider all these aspects and quantify the environmental impacts of application to agricultural soils of biochar from forest residues available in Norway considering different scenarios (including mixing of biochar with synthetic fertilizers and bio-oil sequestration for long-term storage). All the biochar scenarios deliver negative emissions under a life-cycle perspective, ranging from -1.72 ± 0.45 tonnes CO_2-eq. ha^{-1} yr^{-1} to -7.18 ± 0.67 tonnes CO_2-eq. ha^{-1} yr^{-1} (when bio-oil is sequestered). Estimated negative emissions are robust to multiple climate metrics and a large range of uncertainties tested with a Monte-Carlo analysis. Co-benefits exist with crop yields, stratospheric ozone depletion and marine eutrophication, but potential trade-offs occur with tropospheric ozone formation, fine particulate formation, terrestrial acidification and ecotoxicity. At a national level, biochar has the potential to offset between 13% and 40% of the GHG emissions from the Norwegian agricultural sector. Overall, our study shows the importance of integrating emissions from the supply chain with those from agricultural soils to estimate mitigation potentials of biochar in specific regional contexts.

1. Introduction

The achievement of the Paris agreement of limiting global temperature rise to well below 2 °C is likely to require large amount of carbon dioxide removal (CDR) (Rogelj et al., 2018). Depending on temperature pathways, 95% of the estimated cumulative need for CDR falls between 130 and 1600 GtCO2 (Huppmann et al., 2018; Rogelj et al., 2018). Several options have been proposed as negative emission technologies (NET) for CDR: afforestation and reforestation, soil carbon sequestration, biochar, bioenergy with carbon capture and storage (BECSS), direct air capture, enhanced weathering and ocean fertilization, among others (Minx et al., 2018).

Biochar is produced from thermo-chemical conversion of biomass in absence of oxygen and it is considered a NET because it is a stable carbon-based product that can be stored in soils for centuries (Smith, 2016). Depending on the future socioeconomic scenarios and temperature targets considered, biochar can provide from 10 to 35% of the required CDR deployment rate in 2050 (Tisserant and Cherubini, 2019). Biochar production can rely on today's non-used resources, like forest and crop residues, and it has several co-benefits. For example, it produces useful co-products, such as non-condensable gasses and bio-oil (a mixture of organic compounds and water) (Crombie and Mašek, 2015; Woolf et al., 2014). The technology is well known and easy to implement, although large facilities are still lacking (Minx et al., 2018). Bio-oil, which is also rich in biogenic carbon, could be stored in geological deposits to further improve the CDR potential of biochar (Schmidt et al., 2018; Werner et al., 2018). There is also evidence of a series of positive effects of biochar use in agriculture, such as increases in plant yields (Jeffery et al., 2017), reduction of N_2O emissions and nitrogen leaching from soils (Borchard et al., 2019; Liu et al., 2019),

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improved soil water retention (Razzaghi et al., 2020), restored soil fertility, prevention of land degradation (Ali et al., 2017; Saifullah et al., 2018; Yu et al., 2019), and remediation of contaminated sites (Abbas et al., 2018; Yuan et al., 2019; Zama et al., 2018). Biochar is thus attracting increasing attention as one of the most promising options to achieve large-scale CDR deployment and simultaneously co-deliver improvements on multiple sustainability issues (Semida et al., 2019; Smith et al., 2020; Tisserant and Cherubini, 2019).

Assessing the climate change mitigation potential and the environmental sustainability profile of a technology requires a life-cycle perspective that accounts for direct and indirect emissions along its value chain. Life-cycle assessment (LCA) is a useful method to monitor potential co-benefits or trade-offs by tracking several environmental impacts. Many LCA studies of biochar application to agricultural soils have been performed over the years and have been reviewed in two recent literature reviews (Matuštík et al., 2020; Tisserant and Cherubini, 2019). All studies generally converge on the net climate mitigation benefits of biochar, but the magnitude depends on a variety of factors such as type of biomass feedstocks, pyrolysis conditions, biochar treatment, agriculture management and methodological assumptions. Results are thus highly case-specific. Most of the existing studies mainly assessed the climate effects using the Global Warming Potential (GWP) with a time horizon (TH) of 100 years as the default characterization factor (or emission metric), and only consider impacts from greenhouse gasses (GHGs), mainly CO_2, CH_4 and N_2O. This approach has limitations because on the one hand it ignores multiple temporal dimensions of the climate system response to emissions (e.g., either in the short-term or in the long-term), and on the other hand it does not take into account the climate change effects of the so-called near-term climate forcers (NTCFs), such as aerosols (SO_x, black carbon (BC), organic carbon (OC)) and ozone precursors (NO_x, non-methane volatile organic compounds (NMVOC), CO), which cause a strong but time-limited perturbation to the climate (Cherubini et al., 2016; Jolliet et al., 2018; Levasseur et al., 2016a). Further, recent literature reviews noted that analysis of other impact categories besides climate change is limited, and argued future studies should include an assessment of effects in other environmental areas of concerns that are relevant for biochar production and use (Matuštík et al., 2020; Tisserant and Cherubini, 2019). For example, despite its clear importance, only a few LCA studies include biochar's effects on soil emissions (Azzi et al., 2019; Field et al., 2013; Roberts et al., 2010; Thers et al., 2019; Wang et al., 2014). Biochar can potentially affect nitrogen emissions from soils like N_2O, ammonia volatilization, NO_x, and nitrogen leaching (Borchard et al., 2019; Liu et al., 2019; Pourhashem et al., 2017), but the influence of these biochar-induced changes for a range of environmental impact categories has not yet been explored within a life-cycle perspective. These emissions, together with other NTCFs, are important drivers of air quality, eutrophication, or acidification. Similarly, only some LCA studies include positive effects of biochar on yields and nutrients, by either modeling increase in food production or reduction of fertilizer inputs (Field et al., 2013; Mohammadi et al., 2016; Robb and Dargusch, 2018; Sparrevik et al., 2013).

In Norway, increasing soil carbon stock is an important strategy from a climate perspective and for soil health and food production, and biochar has been identified as one of the technologies with the highest potential (Rasse et al., 2019). Norway has large amounts of forest residues that are left unused after extraction of commercial roundwood or from wood industries (Cavalett and Cherubini, 2018), and they are a promising feedstock for biochar production to stimulate a circular economy perspective and reduce pressure on terrestrial ecosystems. In this study, we assess the life-cycle environmental sustainability effects of alternative scenarios of large-scale deployment of biochar production from forest residues and application to agricultural soils in Norway. Biochar production is modelled using a process simulation software to derive emission factors and the mass and energy balance. Different biochar scenarios are investigated, and they differ by the type of biochar used as soil amendment in agriculture (untreated biochar or a biochar-fertilizer mix), and use of biochar co-products (production of heat and power or pumping bio-oil into geological storages to maximize carbon sequestration). The analysis focuses on grain production (barley) and quantifies the environmental impacts from both the life-cycle stages and the changes in soil emissions under Norwegian conditions of biochar use in agriculture. Co-benefits and trade-offs are explored for a range of impact categories: climate change, stratospheric ozone depletion, fine particulate matter formation, tropospheric ozone formation, terrestrial acidification, marine eutrophication and terrestrial ecotoxicity. Multiple climate metrics are used to assess climate change mitigation benefits across different time dimensions, and effects of both GHGs and NTCFs are considered. The overall robustness of the results is evaluated with a Monte-Carlo analysis (10 000 simulations) that considers a variety of uncertainty ranges in key process parameters, modeling assumptions, emission factors, and climate metrics (especially NTCFs). The climate change mitigation potential and other environmental sustainability effects of large-scale biochar deployment in Norway are quantified both per individual process unit (e.g., hectare of land, kg of biochar, or kg of grain) and for a national large-scale deployment (i.e., per year), so to estimate the overall mitigation potentials and side-effects.

2. Methods

The methods section is structured as follows: Section 2.1 presents the system boundaries and an overview of the reference system and the different scenarios; Section 2.2 describes the reference system; sections from 2.3 to 2.6 introduce the modeling of the various aspects of the biochar scenarios (i.e. feedstock collection and transport, pyrolysis, biochar-fertilizer production and application to soil); Section 2.7 presents the effects of biochar on soil; Section 2.8 explains the different climate metrics and impact categories considered for the analysis; Section 2.9 presents the approach to scale up the analysis of the potentials and effects of large-scale biochar application in Norway; Section 2.10 describes the uncertainty analysis.

2.1. System boundaries and biochar scenarios

Fig. 1 shows an overview of the scenarios and system boundaries for the life-cycle assessment of biochar production and application to agricultural soils in Norway. The analysis compares grain production in Norway without or with biochar application.

The reference system includes farming activities (ploughing, fertilization, pesticide application) and inputs (fertilizers, machineries, lime) required for the management of one hectare of land producing barley over the period of one year without addition of biochar to soil.

The reference system is compared to four scenarios where biochar produced from forest residues is spread on land, while the other farming activities remain the same (unless those affected by biochar, such as changes in fertilizer management and soil emissions). The four biochar scenarios are: (i) "biochar", where biochar is directly applied to agricultural soils and biochar co-products are burnt to provide heat for pyrolysis and feedstock drying (no use of the extra heat available); (ii) "biochar-fertilizer", where biochar is grinded and mixed with inorganic fertilizers and pelletized before its application to soils, and biochar co-products are burnt to provide heat for pyrolysis and feedstock drying (no use of the extra heat available); (iii) "biochar-fertilizer with CHP", as in (ii) but co-products are burnt in a CHP unit to meet the electricity and heat demand of the pyrolysis plant, and the excess energy is assumed to displace electricity from the grid and heat from natural gas; (iv) "biochar-fertilizer with bio-oil sequestration", where biochar is treated as in (ii) and all the syngas and part of the bio-oil are combusted to provide heat for pyrolysis, and the remaining of the bio-oil is recovered, transported and pumped into off-shore geological deposits to maximize carbon storage.

Biochar is assumed to be produced by three large-scale facilities

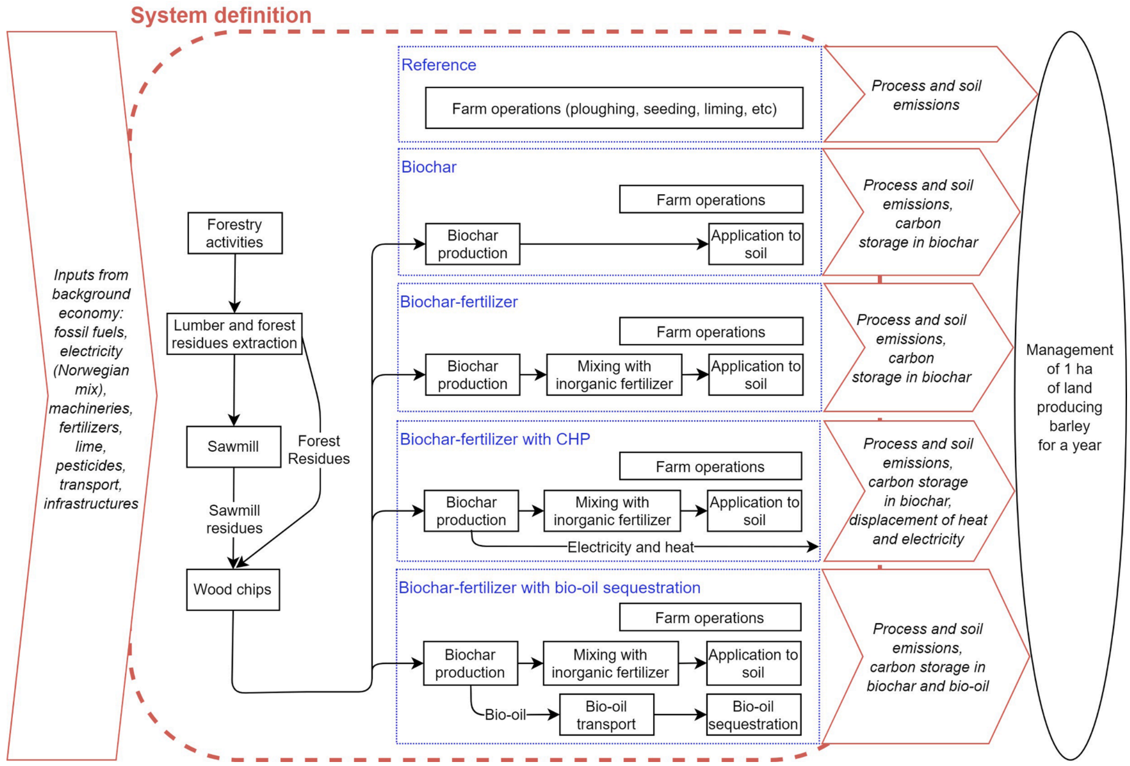

Fig. 1. Overview of the system boundaries and biochar scenarios.

located in Oslo, Stavanger and Trondheim. Biochar supply chain starts with the provision of the feedstock to the plants and includes forestry activities and extraction of forest residues. Residues from the wood industry are also included as potential feedstock. Biochar's effects on soil include changes in N_2O, NH_3 and NO_x emissions, changes in nitrogen leaching, and in the case of biochar-fertilizer application, a positive effect on yield is considered. If not indicated otherwise, Ecoinvent 3.5 (Wernet et al., 2016) was used to gather emission inventories, energy consumption and emission factors associated with the provision of equipment, materials and inputs.

2.2. Reference system

The reference system is the management of one hectare (one complete crop cycle) for one year producing barley, which is the main grain produced in Norway on about 50% of the total grain area (SSB, 2020a). We used reported yields data of barley in Norway from the official national statistics (SSB, 2020b), and estimated an average barley yield of 3756 kg ha^{-1} over the 2009–2018 timespan, with a standard deviation of 495 kg ha^{-1} (here assumed as a proxy of variability in terms of climate and location). Barley production is modeled by adapting the ecoinvent process for barley production in Germany (given on kg barley basis) to Norwegian practices. Field work follows common practices on Norwegian farms and includes ploughing, sowing, harrowing and leveling with stone picking, fertilizing, rolling, pesticide application (typically two applications per year, plus a chemical fallow every three years) and liming (250 kg CaO equivalent per year) (Henriksen and Korsæth, 2013). Fertilizer requirements per year are based on Norwegian average inorganic fertilizer application for barley: 127.5 kg N ha^{-1}, 17.3 kg P ha^{-1}and 63 kg K ha^{-1} (Gundersen and Heldal, 2013; Kolle and Oguz-Alper, 2018). Pesticides application follows typical Norwegian practices for barley (Aarstad and Bjørlo, 2019) and the fields are not irrigated. The inventory is available in Table S1.

2.3. Biomass collection and transport

Feedstock availability and life-cycle inventory for collection, processing and transport follows the model developed in a previous work (Cavalett and Cherubini, 2018). The model is based on county and species-specific production of commercial roundwood removals in Norway over the period 2011–2016. The amount of residues extractable is calculated using age-dependent and species-specific biomass expansion factors to quantify the amount of biomass left in forest after harvest (Lundmark et al., 2014). It is common practice in Norway to leave all forest residues in the forest due to a lack of market for utilizing branches and low-quality wood. In the country, forest residues typically represent a promising feedstock to enhance renewable material supply at no additional pressures from expansion of harvest and to revitalize rural areas through increased circular economy. A residue extraction rate of about 34% is assumed in our analysis, based on sustainable rates of extraction in other Scandinavian countries, where the utilization of forest residues is more common than in Norway (de Jong et al., 2017; Lundmark et al., 2014). A potential of 1.14 Mtonnes year^{-1} of forest residues is estimated, to which we can add an additional 0.56 Mtonnes year^{-1} of by products from the wood industry. Overall, about 82% of forest wood residues are from spruce, 17% from pine and 1% from birch. Life-cycle inventories for feedstock supply include the complete biomass value chain and account for inputs and emissions from harvesting, transport, chipping and processing of forest residues and wood industry residues in Norway. Norwegian-specific data for forestry operations and logistics were used (Cavalett and Cherubini, 2018).

Feedstock transport to the biochar conversion plants is modeled by assigning residues in each county to the nearest biochar conversion plant, after satisfying an equal share of forest residues to the three conversion plants. It is also assumed that the lumber output from forestry is treated within the same county, and the same transport distance is assumed for wood industry residues to the conversion plant. The distance from the county's capital to the conversion plant is used to estimate truck transport distances, or it is assumed to be 40 km if residues are located within the same county of the plant. Distances are weighted by the county's share of feedstock produced and a weighted average transport distance of 190 km from forest to plant is estimated at national level. It is assumed that the feedstock is transported at 40% moisture.

2.4. Pyrolysis

Inventories for biochar production are estimated by modeling the pyrolysis process in Aspen Plus process simulation software. The approach chosen is to model the feedstock biomass, biochar and tar (i.e. organic fraction of the bio-oil, which is a mixture of organic compounds and water) as non-conventional components, while syngas is modeled as a mixture of gas species. For modeling the pyrolysis reaction, a simple approach of converting the feedstock into products using yields is used. The pyrolysis is modeled at 500 °C, and the mass (carbon) yields are 28% (45.7%) to biochar, 56% (42.6%) to bio-oil, and 16% (11.7%) to syngas.

Non-conventional components modeling in Aspen plus requires the proximate analysis (i.e., composition in moisture content, fixed matter, volatile matter and ash content), the ultimate analysis (i.e., content in C, H, O, N, S, Cl) and the sulfate analysis (i.e., content in different forms of sulfur pyritic, sulfate and organic). These data are shown in Table S2 in the supplementary information (SI). The feedstock is modeled as spruce wood, whose composition is taken from the Phyllis2 database (phyllis.nl). Elemental composition and lignin content are taken from the average of the 43 samples in the database for Spruce. Fixed matter, volatile matter and ash contents are also taken from the same database. Biochar yield is determined as function of pyrolysis temperature and feedstock lignin content, and the yield of CH_4, CO, H_2 and C_2H_2 are estimated from regressions based on pyrolysis temperature (Woolf et al., 2014). Tar, CO_2 and water yields are determined from elemental mass balance. N can volatilize as HCN and NH_3 during pyrolysis, S as H_2S and Cl as HCl, CH_3Cl or KCl. Figures S1-S3 in the SI show regression analysis based on literature data of the share of conversion rates of N, S, Cl from the feedstock into different gasses as a function of temperature. These regressions are used to estimate the yield of these gas species for the specific temperature of our pyrolysis system.

For the ultimate analysis of biochar and tar, C, H, and O compositions are estimated from pyrolysis temperature and C, H, O content of the feedstock, using regressions from (Woolf et al., 2014). N content of biochar is assumed to be 0.1% (Morales et al., 2015). S and Cl content in biochar are determined from regressions in Figures S1-S3 in the SI. Tar is used to balance N, S and Cl elements. For the proximate analysis, it is assumed that all feedstock ashes remain in the biochar, which has a fixed matter content of 80% (Weber and Quicker, 2018) and volatile matter is determined to complete the balance. The proximate analysis of the tar (supposed ash-free) is determined using the average value for fixed and volatile matter for bio-oils (given on a dry basis) in the Phyllis2 database: 33.2% for fixed matter, 66.8% for the volatile matter.

The composition of the biomass, biochar and tar and the yields of the different products of pyrolysis are shown in the Tables S2 and S3 in the SI. Description of the Aspen Plus simulations is available in the supplementary text 1 together with Aspen Plus flow charts (Figure S4 and Figure S5) in the SI.

In the case of pyrolysis with combined heat and power (CHP) production, the tar and syngas are burned for recovery of electricity and heat at 28.5% and 71.5% of efficiency, respectively, in line with standard values for steam cycle CHP (Sipilä, 2016).

In the case of biochar production with bio-oil recovery for geological storage, part of the tar (11%) is used for combustion with syngas to produce the required heat for the pyrolysis plant to avoid relying on external fossil fuel. The rest of the bio-oil is transported to Stavanger and transferred to a tanker for transport of 400 nautical miles (one-way) (Gassco, 2017). Infrastructures required for pumping the oil to geological deposit is estimated from Ecoinvent process of offshore petroleum and gas production.

Electricity consumption for drying the feedstock and the pyrolysis reactor are taken from a model of biomass torrefaction (Manouchehrinejad and Mani, 2019), and energy requirements for blowing air for the combustion are given by Aspen Plus. Drying of wood is associated with emissions of NMVOC, estimated at 56 mg/kg biochar produced (Granström, 2009). In the case of the CHP, the energy requirement for producing the biochar-fertilizer is taken by the electricity output from the CHP, and it is thus subtracted from it. Similarly, the heat required for drying the feedstock is subtracted from the heat from the CHP. For the other cases, electricity consumption for producing the biochar or biochar-fertilizer is assumed to be from the Norwegian electricity mix from ecoinvent database (Wernet et al., 2016).

Aspen Plus-derived emissions from the pyrolysis-CHP system are complemented with emission factors measured from a medium scale pyrolyser (Sørmo et al., 2020). They include emission factors for polycyclic aromatic hydrocarbon (PAHs), NMVOC, PM10 and heavy metals associated with particulate matter (As, Cd, Cr, Cu, Pb, Hg, Mo, Ni, Sn). In the case of the pyrolysis with bio-oil recovery, the emission factors are corrected by the amount of tar sent to combustion.

The inventories for the different biochar production scenarios are shown in Table S4, and for the sequestration of bio-oil in Table S5.

2.5. Biochar-fertilizer

In the biochar scenario, biochar is directly applied to the field as a biochar soil amendment. In the biochar-fertilizer scenario, biochar is mixed with fertilizers before application to the soils to form the so-called biochar-based fertilizer (BCF). BCF is produced by grinding biochar into fine particles, then mixing them with a fertilizer and then pelletizing into a final product. Applying biochar in the form of BCFs is found to improve effects on yield and nitrogen use efficiency (Chew et al., 2020; Liu et al., 2020; Shi et al., 2020). Such an expected effect is especially important in Nordic conditions where biochar alone does not necessarily increase yields (O'Toole et al., 2018). Biochar has been shown to substantially reduce N_2O emissions, but this effect is more pronounced the first year after application (Borchard et al., 2019). For this reason, annual applications of biochar mixed with nitrogen fertilizer is expected to maximize the reduction in N_2O emissions (Guenet et al., 2021). Positive interactions between the carbon structure of biochar and nitrogen fertilizer in BCF are also expected to reduce NO_3^- leaching and thereby increase nitrogen use efficiency (Guenet et al., 2021). These positive effects of BCFs on nitrogen use efficiency and yield result from the slow release to the soil of the nitrogen absorbed on the biochar structure (Ibrahim et al., 2020). However, there are physico-chemical limits to how much nitrogen can be absorbed on a biochar structure. Most studies report nitrogen-sorption for biochar below 20 g N per kg biochar (Zhang et al., 2020), but we hypothesized that above-average products would be developed and selected towards a realistic upper value of 50 g nitrogen per kg biochar, which is still lower than several high values reported in the literature (Zhang et al., 2020). Our working hypothesis translates into 50 kg nitrogen per tonne of biochar, which implies that 2552 kg of biochar per hectare need to be applied as BCF to fulfill the nitrogen fertilizing requirements of a barley cropland in Norway. As softwood biochar has 0.51% K_2O available to plants (Ippolito et al., 2015), this reduces the need for potassium by 10.7 kg. The final loading of fertilizers to biochar to fulfill barley's requirements is thus 50 kg N, 6.75 kg P and 20.5 kg K per tonne of biochar.

Energy requirements for grinding and pelletizing the biochar is taken from (Manouchehrinejad and Mani, 2019). Due to lack of data for grinding the fertilizers, the same energy requirement of biochar per unit of (dry) mass is assumed. The total energy requirement is 0.21 kWh per kg biochar-fertilizer, which is assumed to be taken from the Norwegian grid for all scenarios, except for the biochar-fertilizer with CHP scenario where it is taken from the electricity output of the pyrolysis plant. Emissions of particulate matter from the grinding and pelletization of the biochar-fertilizer are taken as proxy from the ecoinvent process of lignite briquetting. Emissions of heavy metals associated with the particulate matter are accounted for assuming that the particles are biochar and using heavy metals concentration in biochar as in (Sørmo et al., 2020).

The inventory for the biochar-fertilizer production is shown in Table S6.

2.6. Biochar application to soil

For estimating transport distances for biochar application to the field, each county is assigned one of the conversion plants based on proximity and equally shared grain land area. Distance from the county's capital and conversion plant is considered as a proxy for transportation distances or assumed to be 40 km if biochar is applied to a field within the same county. Distances are weighted by the county's share of grain land area and an average transport distance of 226 km is estimated.

Biochar application to the field is assumed to be broadcasted and followed by harrowing for incorporation into soil. It is assumed that 74% of the carbon in biochar remains in soil after 100 years based on biochar stability in soils measured under Norwegian conditions (Budai et al., 2016). It is assumed that all the calcium in the feedstock remains in biochar as $CaCO_3$, reducing the need for liming by 145 kg year^{-1}. The inventory is available in Table S7.

2.7. Biochar's effects on soil emissions

Emission factors from soils in the reference system are taken from the Norwegian emissions inventory report (Miljødirektoratet, 2019). Soil N_2O emissions from fertilizers are estimated considering that 1% of the nitrogen applied, 1% of the volatized nitrogen and 0.75% of the leached nitrogen are emitted as N_2O. NO_x emissions are 0.04 kg NO_x per kg nitrogen applied, NH_3 emissions are 5% of the nitrogen applied, and 22% of the nitrogen applied as fertilizer is leached from the soil as nitrates. Table S8 in the SI provides a summary of these factors and the range used in the uncertainty analysis.

Modelled effects of biochar include changes in soil N_2O, NO_x and NH_3 emissions and in nitrogen leaching. Direct biochar application to soil in Norway is not expected to have significant effect on grain yield (O'Toole et al., 2018), as also observed in other Nordic countries (Tammeorg et al., 2014a, 2014b). However, biochar-fertilizer has the potential to improve fertilizer efficiency and can therefore induce a positive effect on yields. A literature survey of 10 studies finds that BCFs based on inorganic fertilizer have an average effect on crop yield of 19%, with a standard deviation of 22% (Chew et al., 2020; González et al., 2015; J. Liao et al., 2020; Magrini-Bair et al., 2009; Puga et al., 2020; Qian et al., 2014; Schmidt et al., 2017; Shi et al., 2020; Wen et al., 2017; Yao et al., 2015). An uncertainty range of −3% to +41% for the effects of BCF on grain yields was therefore considered in our analysis.

Given the high uncertainty of effects on soil emissions, uncertainty ranges are considered in a Monte-Carlo analysis. The reduction potential of biochar on N_2O emissions from soils is considered to be between 22 and 50% (with an average effect of 38%), according to a meta-analysis (Borchard et al., 2019). This range is consistent with results from regression modeling for biochar from wood under Norwegian soil conditions under low application rate (0–10 tonnes per hectare) (Liu et al., 2019), and with observed field measures in Norway (O'Toole et al., 2014). Biochar's effect on ammonia volatilization is modeled using regression modeling for biochar from wood under Norwegian soil conditions and low application rate of 0–10 tonnes biochar per hectare (Liu et al., 2019). According to these data, NH_3 volatilization increases between 0 and 10%, with an assumed average increase of 5%. Biochar's effect on soil NO_x emissions from nitrogen fertilizer is based on a review of literature data (Fan et al., 2020, 2017; X. Liao et al., 2020, p.; Nelissen et al., 2014; Niu et al., 2018; Obia et al., 2015; Wang et al., 2019; Weldon et al., 2019; Xiang et al., 2015; Zhang et al., 2019, 2016). NO_x reductions can be as high as 75–80% for biochar produced at high temperature and at high biochar application rates (Wang et al., 2019; Weldon et al., 2019). However, increased NO_x emissions under biochar amendment can also be observed, but mainly from biochar produced at low temperature (<400 °C) (Weldon et al., 2019). At biochar application rates of 3–3.75 tonnes/ha, NO_x reductions of 5–20% are reported (X. Liao et al., 2020; Niu et al., 2018; Xiang et al., 2015). In our scenarios, biochar is produced at 500 °C and an increase in NO_x emissions is not expected. The lower bound of the uncertainty range is thus set at 0%, the average reduction at 10% and the upper bound at 20%. Biochar's effect on nitrogen leaching is taken from (Liu et al., 2019), and it is expected to be a reduction by 0–16% (average 8%). It is assumed that biochar and biochar-fertilizer have the same effect on soil emissions.

Biochar's effects on soil are considered to be effective only for one year after its application, according to recent evidence (Borchard et al., 2019; Liu et al., 2019). It is assumed that biochar is applied annually and long-term effects of biochar on crop yield and nitrogen leaching are not included in the analysis as they are still unclear and uncertain (Borchard et al., 2019; Jeffery et al., 2017).

2.8. Climate and other environmental impacts

The climate impact analysis includes the effects of both greenhouse gasses (CO_2, N_2O and CH_4) and NTCFs (NO_x, CO, SO_x, non-methane volatile organic compounds (NMVOC), organic carbon (OC), black carbon (BC)). These different climate forcers affect the climate system on different time scales: GHGs have long life-time that allows for uniform atmospheric mixing and affect the climate globally; whereas NTCFs have short life-time, are not well-mixed in the atmosphere, and their climate impacts are highly heterogeneous (Levasseur et al., 2016b; Myhre et al., 2013). A single metric like the GWP100 can never capture the full picture of the climate impacts from forcing agents with such a variety of timescales. To overcome these limitations, the United Nations Environment Programme-Society of Environmental Toxicology and Chemistry Life-Cycle Initiative proposed the combined use of multiple metrics that quantify the effects of different climate forcers on different timescales, for example in terms of the rate of climate change or long-term temperature increase (Cherubini et al., 2016; Jolliet et al., 2018; Levasseur et al., 2016a). These metrics are GWP20 and GWP100 to assess short-term and mid-term impacts, and the global temperature change potential (GTP) with TH of 100, GTP100 (Levasseur et al., 2016b). GTP is a metric that evaluates the contribution of an emission to global average temperature at a specific point in time in the future indicated by the TH. A detailed description of these metrics can be found elsewhere (Joos et al., 2013; Myhre et al., 2013; Shine et al., 2005). Since GWP100 characterization factors are numerically similar to the values of GTP40, GWP100 can be interpreted as a metric assessing temperature changes within approximately 40 years (Allen et al., 2016). GWP20 and GWP100 can thus mostly capture short (GWP20) and medium-term (GWP100) climate change impacts that are relevant for the rate of climate change, and, since they are based on integrated (cumulative) effects, they tend to assign relatively higher importance to short-lived forcers like NTCFs or CH_4 (especially for short TH, as in GWP20). GTP100 represents the instantaneous (i.e., non-integrated) effects on temperature at 100 years. It is therefore a proxy for long-term climate impacts and the temperature stabilization goal stated in the Paris Agreement (Levasseur et al., 2016b; Tanaka et al., 2019). In our analysis, GWP20 and GWP100 include the effect of both NTCFs and GHGs, while GTP100 only quantify contributions from GHGs (the ones from NTCFs are negligible).

Characterization factors for NTCFs for GWP20 and GWP100 are taken from (Levasseur et al., 2016b), and are based on world average estimates available from the latest IPCC Assessment Report (Myhre et al., 2013). Values and uncertainty ranges for all the characterization factors are reported in Table S9 in the SI.

We selected six additional impact categories to investigate potential trade-offs or co-benefits: stratospheric ozone depletion, fine particulate matter formation, tropospheric ozone formation, terrestrial acidification, marine eutrophication and terrestrial ecotoxicity. Different types of emissions contribute to varying impact categories. For example, N_2O emissions contribute to stratospheric ozone depletion (in addition to climate change), NO_x participates in tropospheric ozone formation with implication for human and ecosystem health, ammonia (and NO_x) contributes to terrestrial acidification (with potential impacts on plant diversity) and to fine particulate matter formation (with potential impacts on human health), leaching of nitrogen is associated with marine eutrophication, and emissions of heavy metals are key drivers of terrestrial ecotoxicity impacts. All emissions are characterized using averaged mid-point characterization factors from ReCiPe 2016 v1.1 (Huijbregts et al., 2017).

2.9. Large-scale biochar deployment

The biochar potential from forest residues availability is assumed to be applied annually to the grain producing area in Norway, which on average over the period 2010–2020 is about 0.28 Mha (35% of the cultivated area) (SSB, 2020a). From the amount of forest residues available in the counties and the biochar yields of the pyrolysis process described above, we estimate a national biochar production potential of 0.48 ± 0.03 Mtonnes year^{-1}. Assuming an application rate to agricultural soils of 2.5 tonnes year^{-1}, a total of 0.19 ± 0.01 Mha can be annually treated with biochar (representing about 68% of the grain cultivated area). Changes induced by biochar or biochar-fertilizer to barley yields and soil emissions are estimated by considering the specific average effects (and uncertainty ranges) mentioned above over all the treated area.

2.10. Uncertainty analysis

In addition to the uncertainty ranges presented in the previous sections (mostly about soil emissions), our uncertainty analysis considers variability in a range of key parameters that are relevant in the biochar value chain. Uncertainty factors are used for biochar yields, carbon content in biochar and its long-term stability, carbon content in bio-oil, heat required by pyrolysis, transport distances (± 20%) of feedstocks or biochar, climate metrics, biochar's effect on crop yield and soil emissions. Biomass composition, such as moisture or ash content, can influence both yield and fixed carbon content of biochar (Peters et al., 2015; Woolf et al., 2014). Variability in biochar yield, carbon content and stability in the uncertainty analysis is performed to capture these variations. Among the uncertainty factors, a key role is played by biochar yields, because it affects emission factors for pyrolysis, the amount of feedstock per kg of biochar to be extracted and transported, and ultimately the total amount of land that can be treated. Further, BC and OC emissions are not included in the emission inventory database, and they are estimated by multiplying PM10 emissions with factors representing the shares of BC and OC emissions from both stationary and mobile sources (Bond et al., 2004). The uncertainty analysis is performed with a comprehensive Monte-Carlo analysis, where 10,000 runs produce results by randomly selecting one value within each of the uncertainty ranges per each run. LCA usually relies on lognormal distribution for uncertainty analysis of parameters, because of qualitative appraisal of knowledge strength using a pedigree matrix approach (Ciroth et al., 2016; Funtowizc and Raveitz, 1990). In our study, we gathered, when available, quantitative literature data on various parameters and establishing a normal distribution was not always possible due to limited sample size. A triangular distribution was thus selected, as recommended by the principle of maximum entropy (Mishra and Datta-Gupta, 2018; van der Spek et al., 2020). The minimum, maximum and mode of each parameters define the triangular distribution. The uncertainty factors and ranges of values is available in Tables S8, S9 and S10 in the SI.

3. Results and discussion

3.1. Climate change impacts

Fig. 2 shows the results (GWP100) for the reference case and the four biochar scenarios considered in our analysis. These results include the effects of both GHGs and NTCFs and show contributions by life-cycle stage (Fig. 2a) or climate forcing agent (Fig. 2b).

In the reference system, managing one hectare of land for barley production without biochar causes about 2.8 ± 0.2 tonnes CO_2eq. ha^{-1} year^{-1}. A key step is fertilizer production (1.13 tonnes CO_2eq. ha^{-1} year^{-1}) followed by farming operation (0.76 tonnes CO_2eq. ha^{-1} year^{-1}). Soil emissions account for 0.67 tonnes CO_2eq. ha^{-1} year^{-1}. There is a similar share of impact from CO_2 and N_2O with 1.23 and 1.42 tonnes CO_2eq. ha^{-1} year^{-1}, respectively. About half of the N_2O emissions in the reference system are due to soil emissions, while the other half comes from nitric acid production for ammonium nitrate supply.

Producing barley in one hectare of land with biochar has a net climate impact of −1.72 ± 0.45 tonnes CO_2eq. ha^{-1} year^{-1}. Farm operations remains the second main contributor to warming emissions, which are higher than those in the reference system (about 85 kg CO_2eq. ha^{-1} year^{-1}) because of additional emissions from biochar application (spreading and harrowing). On the other hand, the reduction in liming use due to biochar reduces emissions by about 76 kg CO_2eq. ha^{-1} year^{-1}. Transportation activities (including both the transport of the feedstock from the forest to the biochar plant and that of biochar from the plant to the field) cause about 0.62 tonnes CO_2eq. ha^{-1} year^{-1}. Pyrolysis does not significantly contribute to direct warming emissions, as power consumption comes from the low-carbon Norwegian electricity grid, which mostly consists of hydropower. Pyrolysis emissions contribute to slightly cooling effects from emissions of NO_x and SO_x. Soil emissions are reduced by about 0.22 tonnes CO_2eq. ha^{-1} year^{-1} compared to the reference case (from 0.67 to 0.45 tonnes CO_2eq. ha^{-1} year^{-1}). Biochar causes both a cooling effect by reducing soil N_2O emissions and a warming effect by reducing soil NO_x emissions (which is a cooling agent), but, because the former is larger than the latter and N_2O has a stronger climate effect than NO_x with GWP100, the net effect is a reduction in characterized emissions. The application of 2.5 tonnes of biochar per hectare also allows the sequestration of 5.35 ± 0.33 tonnes CO_2eq. ha^{-1} year^{-1} in agricultural soils. This amount of negative emissions is larger than the warming effects from emissions along the biochar's value chain and from the farm, so the system has net negative emissions also under a life-cycle perspective. Warming contributions from black carbon and cooling contributions from NO_x and SO_x are increased compared to the reference case, due to the added fuel consumption during the feedstock collection and transportation processes in the biochar supply chain.

Results from the biochar-fertilizer scenario are similar to the biochar scenario. The fertilization stage accounts for the production of the biochar-fertilizer (e.g. grinding and pelletization) and emissions associated with fertilizers production. Power consumption for production of the biochar-fertilizer and higher transport needs due to the increased weight of the biochar loaded with fertilizers are among the key factors for the lower net climate impacts compared to biochar (−1.65 ± 0.48 vs. −1.72 ± 45 tonnes CO_2eq. ha^{-1} year^{-1}).

The biochar-fertilizer with CHP scenario has a climate effect of −4.59 ± 0.74 tonnes CO_2eq. ha^{-1} year^{-1}. Results follow the same pattern of the biochar-fertilizer scenario, but with additional climate benefits from substituting electricity generation and heat production (assumed from natural gas). Avoided emissions are mostly from reducing burning natural gas (96% of the benefits), given the low carbon intensity of the Norwegian electricity mix. The small cooling effect of CH_4 is due to avoided methane losses in the supply chain of natural gas for heat production.

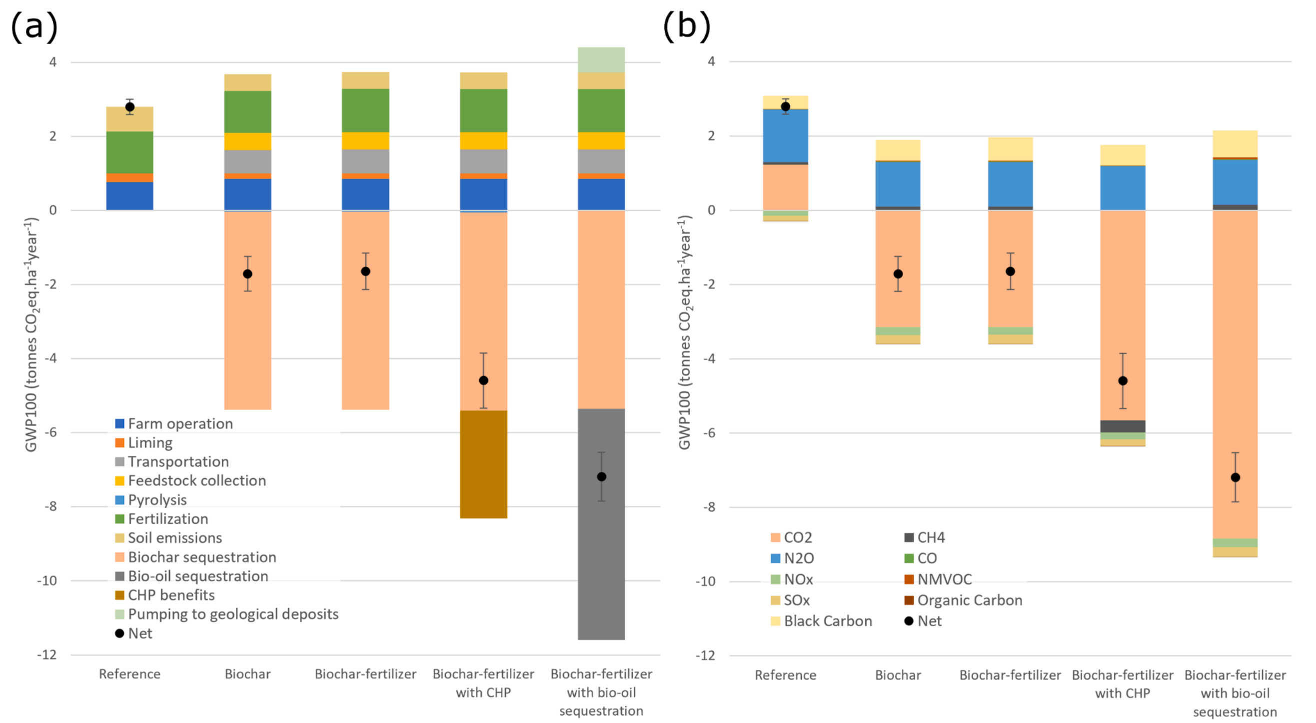

Fig. 2. Climate change effects of the biochar scenarios against a reference system. Results are based on the use of GWP100 to characterize climate impacts and include contributions from both near-term climate forcers (NTCFs) and greenhouse gasses. Both contributions by life-cycle stages (a) and climate forcing agents (b) are shown. Transportation accounts for both feedstock and biochar. Black dots represent the net climate impact and the whiskers show uncertainty ranges from the Monte-Carlo analysis (± one standard deviation).

The biochar-fertilizer with bio-oil sequestration scenario can achieve the largest negative emissions, at -7.19 ± 0.66 tonnes CO_2eq. ha^{-1} $year^{-1}$. Results follow the same pattern as the biochar-fertilizer scenario, but with an additional carbon sequestration from bio-oil of 6.23 ± 0.49 tonnes CO_2eq. ha^{-1} $year^{-1}$. Transport and sequestration of the bio-oil to off-shore geological deposits add 0.69 tonnes CO_2eq. ha^{-1} $year^{-1}$. This means that using excess bio-oil for long-term storage provides larger climate change mitigation benefits than using it to supply heat and power. These results are clearly sensitive to the background energy system, and may vary in other locations where, for example, coal is a primary source for heat or the electricity supply is more dependent on fossil energy sources than Norway.

Figures S6–8 in the SI show the results according to alternative functional units, namely, kg barley, kg biochar and kg feedstock. In terms of impacts per kg barley, the difference between the biochar and biochar-fertilizer scenarios is larger. The climate mitigation is slightly smaller for the latter because BCF increases barley yields, but not biochar production. This implies that the climate mitigation of biochar-fertilizer is spread over a larger grain production and the net benefits are divided by a larger number (as yields are higher), so lowering climate mitigation potential per kg barley as compared to the biochar scenario.

3.2. Sensitivity of results to climate metrics

Figs. 3a and 3b show the sensitivity to the use of alternative climate metrics representative of different types of impacts and time perspectives. GWP20, GWP100 and GTP100 are climate metrics that measure the climate system response within a short, mid and long-term period, respectively (see Section 2.8). GWP20 is a metric that focuses on the very short-term and attributes relatively higher importance to NTCFs. It can be interpreted as an indicator to the impact to the rate of climate change. GTP100 is a long-term metric that addresses the temperature stabilization as stated by the Paris Agreements, and it gives comparably little importance to NTCFs and short-lived GHGs (like CH_4). GWP100 lies in between, and it can be interpreted as a metric assessing temperature impacts within about four decades after emissions.

In general, the net climate effects tend to decrease with the longer time perspective of the climate metric (GWP20 – GWP100 – GTP100). This is mainly due to the smaller effect from NTCFs, especially BC, NO_x and CH_4, when a longer TH is considered. For the reference scenario, it means reduced warming, while for the biochar scenarios it means increased cooling. In all the cases, the contributions of the life-cycle stages remain similar across the climate metrics. For the biochar-fertilizer with CHP scenario, the net climate impact remains the same for all climate metrics considered. This occurs essentially because changes in cooling effects are nearly entirely compensated by changes in warming effects.

Warming contributions from soil emissions increase as time perspective increases, because cooling effects of NO_x emissions become less important relative to warming from N_2O (which remains approximately constant) at longer TH. Emissions associated with pyrolysis have larger cooling effects with GWP20 compared to the other metrics due to the higher cooling of NO_x and SO_x at shorter TH, and the impact decreases over longer time scales.

Finally, uncertainty in the climate response decreases as time perspective increases. Uncertainty ranges for GWP20 and GWP100 are dominated by intrinsic uncertainties in characterization factors for NTCFs. These uncertainties are particularly relevant for biochar and biochar-fertilizer scenarios under GWP20, where the ranges are large and the net climate effects can either be of strong cooling or nearly climate neutral (if not slightly positive). For example, characterization factors for BC can range from 270 to 6200 kg CO_2eq. kg^{-1} (mean: 3200 kg CO_2eq. kg^{-1}), or for NO_x from -53 to -27 kg CO_2eq. kg^{-1} (mean: -40 kg CO_2eq. kg^{-1}) (see Table S8 in the SI).

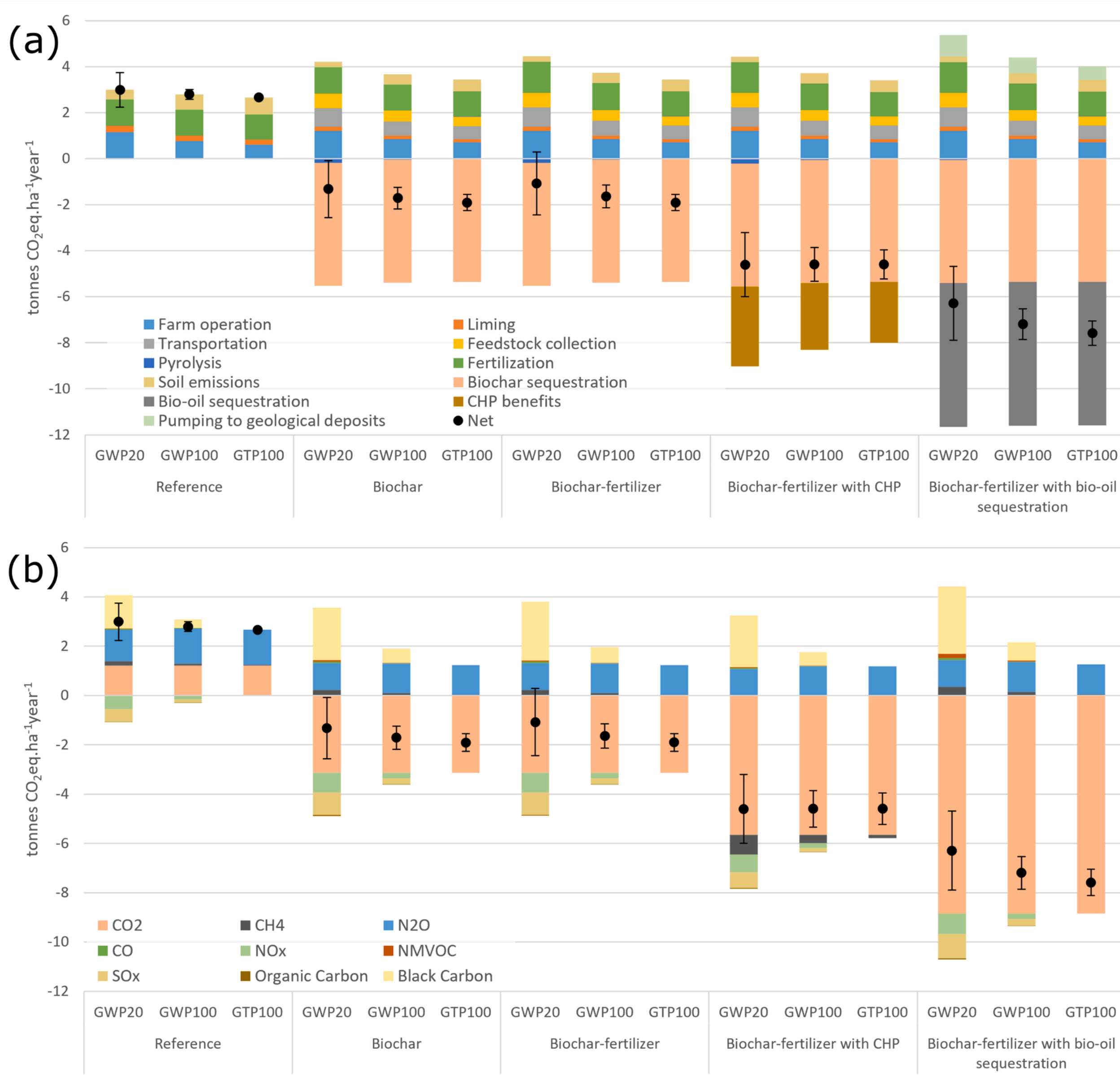

Fig. 3. Climate change effects using different metrics for characterization of impacts: global warming potential at 20 years' time horizon (GWP20), global warming potential at 100 years' time horizon (GWP100) and global temperature potential at 100 years' time horizon (GTP100). Results are presented by life-cycle stage (a) and by contributions of the climate forcing agents (b). Black dots represent the net impact and the whiskers show uncertainty range from our Monte-Carlo analysis (± one standard deviation).

3.3. Net climate mitigation of biochar scenarios

Fig. 4 shows the net mitigation potential of the different biochar scenarios by taking the difference between the climate impact of each given biochar scenario and that of the reference system. In all the cases and irrespective of the climate metric, a net climate mitigation is achieved. Considering each metric and the corresponding uncertainty range, negative emissions can range from −3.7 tonnes CO_2eq. ha^{-1} $year^{-1}$ (GWP20, higher end) to −4.9 tonnes CO_2eq. ha^{-1} $year^{-1}$ (GTP100, lower end) for the simplest biochar scenario, from −3.3 tonnes CO_2eq. ha^{-1} $year^{-1}$ (GWP20, higher end) to −4.9 tonnes CO_2eq. ha^{-1} $year^{-1}$ (GTP100, lower end) for the biochar-fertilizer system, from −6.7 tonnes CO_2eq. ha^{-1} $year^{-1}$ (GTP100, higher end) to −8.5 tonnes CO_2eq. ha^{-1} $year^{-1}$ (GWP20, lower end) for the biochar-fertilizer with CHP and −8.3 tonnes CO_2eq. ha^{-1} $year^{-1}$ (GWP20, higher end) to −10.8 tonnes CO_2eq. ha^{-1} $year^{-1}$ (GTP100, lower end) for the biochar-fertilizer with bio-oil sequestration. Overall, the net mitigation is relatively insensitive to the climate metric used, as all results of each scenario are within the respective uncertainty ranges. In particular, biochar and biochar-fertilizer scenarios have similar net mitigation. If coproducts of the pyrolysis are used to generate heat and electricity, about 65% more climate mitigation is achieved, compared to only producing biochar. Sequestration of the bio-oil into geological deposits can potentially more than double the net climate benefits of biochar alone (+ 120%).

3.4. Other environmental impact categories

Fig. 5 shows an overview of the results for other environmental impact categories of the reference case and the different biochar scenarios. Results are normalized relative to the impact from the reference case in each category. Absolute results are presented in Figures S9-S14 in the SI.

Biochar application to agricultural soils can provide co-benefits in

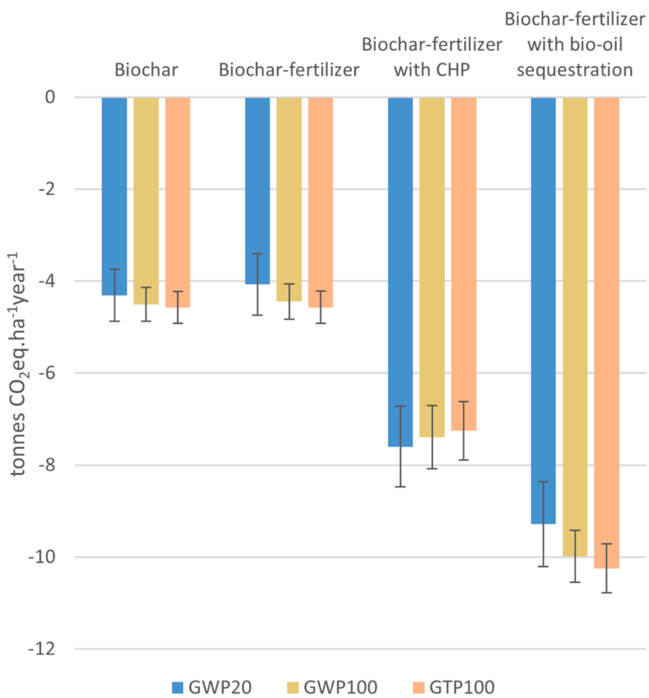

Fig. 4. Net climate change mitigation per biochar scenario and climate metric. Net mitigation is defined as the climate impacts of the given scenario minus the climate impacts of the reference system. Black whiskers show uncertainty ranges from the Monte-Carlo analysis (± one standard deviation).

terms of stratospheric ozone depletion and marine eutrophication, although for the latter the uncertainty range prevent drawing robust conclusions. The magnitude of these co-benefits is relatively insensitive to the type of biochar scenario. On the other hand, tropospheric ozone formation (which affects human health), fine particulate matter formation, terrestrial acidification and terrestrial ecotoxicity have higher impacts for the biochar scenarios than the reference case.

In general, co-benefits occur for those impact categories where biochar's value chain (e.g. transportation, feedstock collection, pyrolysis) does not contribute with relevant emissions. Stratospheric ozone depletion impacts are mainly due to N_2O emissions from nitrogen fertilizers production and soil emissions. Reduction in soil N_2O emissions by biochar explains the lower impacts in stratospheric ozone depletion. Marine eutrophication is mostly driven by soil leaching of nitrogen from the fertilizers, and the biochar's mitigation potential for nitrogen leaching explains the reduced impacts.

In the reference system, contributions to tropospheric ozone formation are mostly due to NO_x and NMVOC emissions from combustion of fuels during land management and soil NO_x emissions from nitrogen fertilizer use. In the different biochar scenarios, there is a reduction in NO_x emissions from soils by 10%, but it is outweighed by higher emissions of NO_x (and to less extent NMVOC) from the combustion of fuels during transportation, feedstock collection and pyrolysis. In the biochar-fertilizer with CHP, avoided production of heat from natural gas prevents some NO_x emissions, which is the reason for the overall lower impacts compared to the other biochar scenarios. In the case of biochar-fertilizer with bio-oil sequestration, the pyrolysis stage has almost no impacts because there are much less NO_x emissions (most of the bio-oil is recovered rather than burnt), but the additional emissions from transportation and sequestration of the bio-oil more than offsets this reduction, and make this scenario the one with the highest impact in tropospheric ozone formation.

Fine particulate matter is mostly formed by emissions of particulate matter (PM2.5) and aerosol precursors like NO_x, NH_3 and SO_x, and it is a potential threat to human health. In the reference system, nearly half of the impact comes from farm operations, and the other half from soil emissions. In terms of individual drivers, the most relevant are SO_x emissions from fertilizers production and emissions of PM2.5 and NO_x from fertilizer production and use of fossil fuels in machineries. Emissions of NH_3 and NO_x from soils lead to the remaining impact for the reference system. Under the biochar scenarios, the combined effect of increase in NH_3 and decrease in NO_x emissions from soils due to biochar application leads to a slight increase in impact from soil emissions of about 2%. This is due to the fact NH_3 is more than twice more impactful compared to NO_x (Huijbregts et al., 2017). The additional emissions of NO_x, SO_x and PM2.5 from combustion processes during transportation, feedstock collection, pyrolysis and biochar-fertilizer production leads to larger impact for all the biochar scenarios compared to the reference case. For the biochar-fertilizer with CHP scenario, there are emissions of NO_x avoided by the displacement of heat from natural gas, leading to the lowest impact score for fine particulate matter formation. For the biochar-fertilizer with bio-oil sequestration scenario, less material is burned in the pyrolysis process with lower emissions of NO_x and SO_x. However, also in this case higher emissions of these compounds during transportation and sequestration of the bio-oil lead to the highest impact in this category.

Deposition of NO_x, SO_x, and NH_3 on terrestrial ecosystems lead to terrestrial acidification, which is a threat to ecosystem health and functioning. Soil emissions are the main contributors to the acidification potential in the reference case and all biochar scenarios and are due to the emissions of NH_3 and NO_x from fertilizer use. About a third of the impact comes from farm operations and mainly from SO_x, NH_3 and NO_x emissions during fertilizer production (due to ammonia and sulfuric acid production) and NO_x emissions from machinery use. In the biochar scenarios, the combined effect of increase in NH_3 and decrease in NO_x emissions from soils due to biochar application lead to a slight increase in impact from soil emissions of about 3% (the acidification potential of ammonia is 5.4 times larger than NO_x (Huijbregts et al., 2017)). Emissions of NO_x and SO_x during pyrolysis contribute to 7% of the impact, while transport and feedstock collection account for 12% together. For the biochar-fertilizer with CHP scenario, avoided use of heat from natural gas saves emissions of NO_x, leading to the lowest impact score for terrestrial acidification among the different biochar scenarios. For the biochar-fertilizer with bio-oil sequestration scenario, lower impact is observed for the pyrolysis process due to less material burnt (and less emissions of NO_x and SO_x), but these savings are more than compensated by higher emissions of these compounds from transportation and sequestration of the bio-oil.

Terrestrial ecotoxicity impacts in the reference system are from emissions of heavy metals during fertilizer production (about 63%, 45% from ammonium nitrate production only), and the remaining are mostly from heavy metals emissions from combustion of fossil fuels in agricultural machinery during farming operation. Contributions of pesticide in soils are below 0.5% of the total impact. The higher needs for transportation of materials and the emissions of pollutants from the pyrolysis stage make the effects on terrestrial ecotoxicity from the biochar scenarios from 3.5 to 4.5 larger than those from the reference system. In the biochar scenarios, transport becomes the main contributor to terrestrial ecotoxicity, with emissions of heavy metals from fossil fuels combustion, mostly copper (92%) and zinc (5%). Biochar-fertilizer production's impacts are largely due to emissions of heavy metals from grinding and pelletization. Impacts from pyrolysis come from emissions of heavy metals (mostly copper 78% and nickel 11%) during combustion of the biochar's co-products. Contribution of PAH emissions during pyrolysis are negligible (lower than 0.0001% of the pyrolysis process's impact). Pyrolysis impacts are lower in the case of bio-oil recovery and sequestration, because it is assumed that most of the heavy metals are recovered with the bio-oil. However, these lower impacts are partly offset by emissions during transport and sequestration processes of the bio-oil.

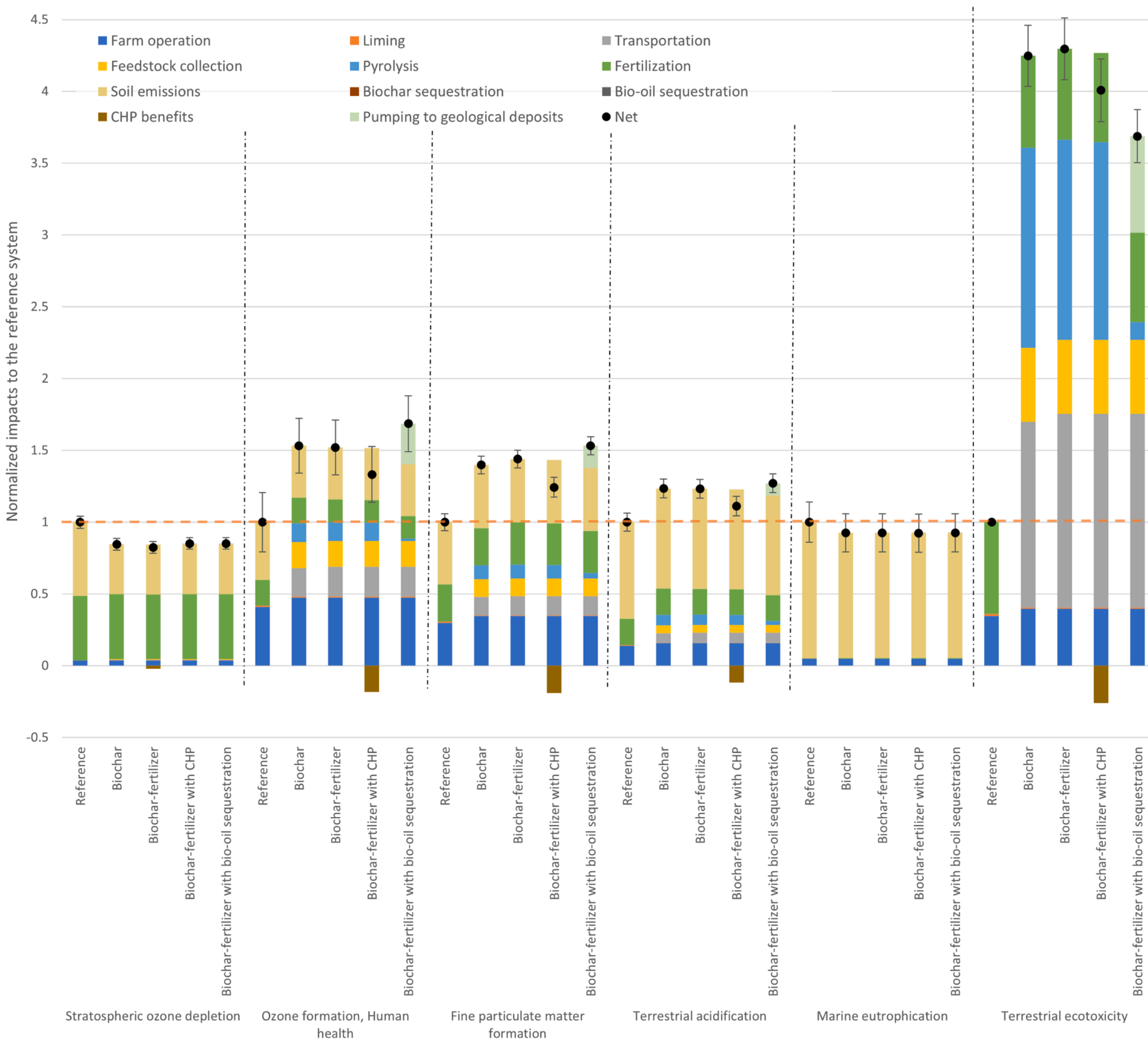

Fig. 5. Life-cycle impacts from the reference system and the four biochar scenarios for 6 impact categories: stratospheric ozone depletion, ozone formation (human health), fine particulate matter formation, terrestrial acidification and terrestrial ecotoxicity. Results are presented by life-cycle stages and are normalized to the impact of the reference system per each category. Transportation accounts for transportation of both feedstock and biochar. Black dots represent the net impact and the whiskers show uncertainty ranges from the Monte-Carlo analysis (± one standard deviation).

3.5. Effects at a national level

Fig. 6 offers an overview of the potential of carbon sequestration (or negative emissions) from a large-scale deployment in Norway of the biochar scenarios analyzed in this study, either with or without a life-cycle perspective. Deployment scenarios are calculated by scaling up the biochar production potential to the total feedstock available, with the associated logistics described in the methods. From the estimated 1.7 Mtonnes of forest residues available per year, about 0.48 ± 0.03 Mtonnes year^{-1}of biochar are produced. Assuming an average application rate to agricultural soils of 2.5 tonnes biochar per ha, about 68% of the 0.28 Mha of grain producing land in Norway can be annually treated with biochar.

Accounting only for the carbon sequestered without a life-cycle perspective, the mitigation potential is 1.01 ± 0.1 Mtonnes CO_2eq. year^{-1}, and it can be about twice as much (2.19 ± 0.1 Mtonnes CO_2eq. year^{-1}) when bio-oil is also captured and stored. Under a life-cycle perspective that accounts for emissions along the whole supply chain, the mitigation potential in the biochar and biochar-fertilizer scenarios is reduced by 15–24%. Adding the generation of electricity and heat adds 36–42% to the climate mitigation of the simple biochar scenario. The consideration of life-cycle emissions in the case of bio-oil sequestration reduces the climate change mitigation potential by 12–20% relative to the case where only the carbon in biochar and bio-oil is taken into account. With the exception of the scenario of biochar-fertilizer with CHP, the life-cycle based yearly mitigation potentials tend to increase when extending the temporal perspective of the climate metric.

Relative to the Norwegian territorial GHG emissions in 2019 (SSB, 2020c), the carbon storage from the biochar without and with bio-oil sequestration can mitigate 2.0% ± 0.2% and 4.3% ± 0.2% of the national emissions, respectively. Taking life-cycle emissions into considerations for the different metrics and uncertainty ranges, the mitigation potential is between 1.3% (biochar-fertilizer, GWP20) and 1.9% (biochar, GTP100) and between 3.1% (biochar-fertilizer with bio-oil sequestration, GWP20) and 4.0% (biochar-fertilizer with bio-oil sequestration, GTP100) respectively. Compared to emissions from the Norwegian agricultural sector only, the climate change mitigation potential of the carbon sequestration in biochar and in biochar and bio-oil is 20.6% ± 1.7% and 44.5% ± 2.1% respectively. Under a life-cycle perspective for the different metrics and uncertainty ranges, these

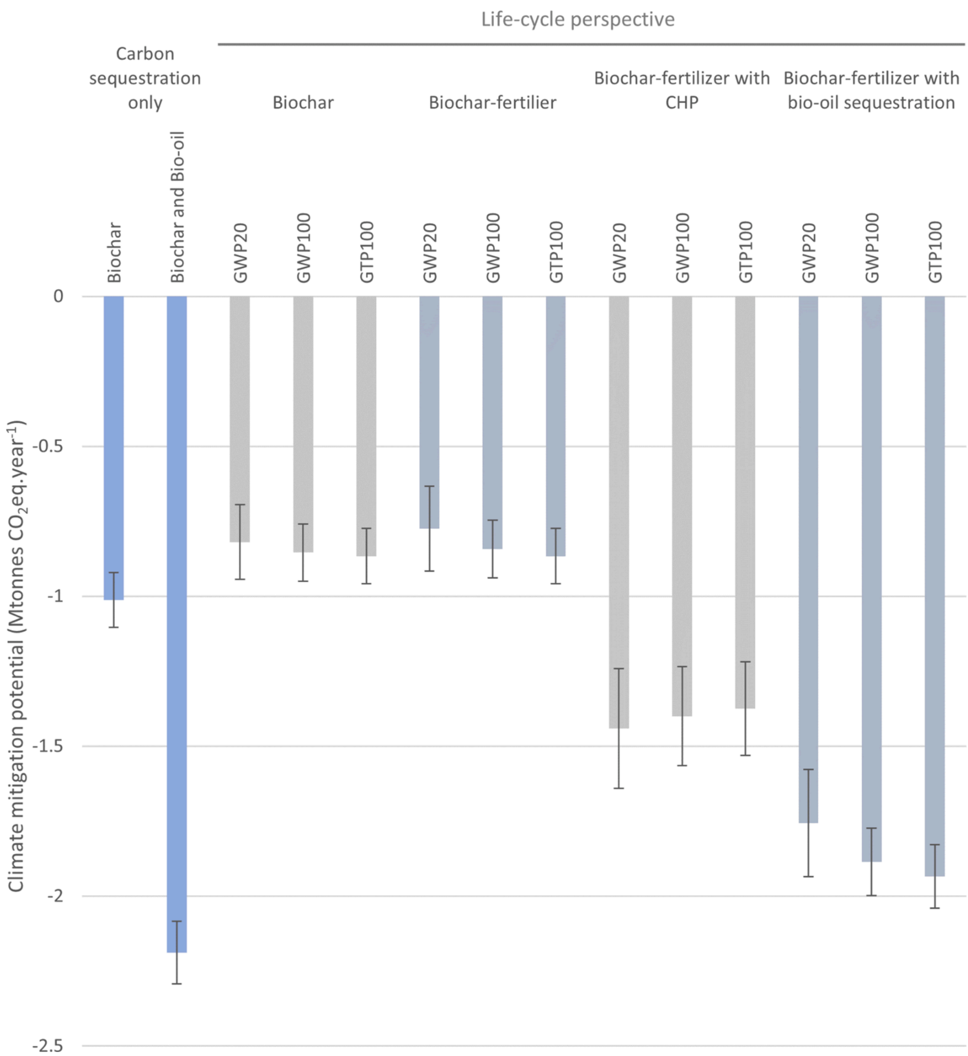

Fig. 6. Comparison of the climate change mitigation potential of a large-scale deployment of biochar in Norway considering only the carbon contained in biochar and bio-oil or taking a life-cycle perspective. Black whiskers show uncertainty ranges from the Monte-Carlo analysis (± one standard deviation).

figures are between 12.9% (biochar-fertilizer, GWP20) and 19.4% (biochar, GTP100) and between 32% (biochar-fertilizer with bio-oil sequestration, GWP20) and 41.4% (biochar-fertilizer with bio-oil sequestration, GTP100).

Fig. 7 shows how the large-scale deployment of biochar in Norwegian agriculture affects yields of barley and soil emissions. Based on national availability of forest residues, biochar can be annually applied to about 0.19 ± 0.1 Mha of grain production area, resulting in a yield increase of about 0.14 ± 0.06 Mtonnes per year (+12%) (under the assumption that all the land is dedicated to barley production). The mitigation of N_2O emissions is 21% ± 4% compared to baseline emissions where land is not treated with biochar. This mitigation is due to a reduction of direct emissions of N_2O from fertilizer application (25% ± 4%), a decrease of indirect N_2O emissions due to a decrease of nitrogen leaching from soils (5% ± 2%), and an increase of about 3% ± 1% of indirect N_2O emissions from the overall increase of ammonia volatilization. Compared to the national statistics for 2019, the reductions of N_2O emissions correspond to 1.8% of the national N_2O emissions and 2.4% of the agricultural N_2O emissions (SSB, 2020d).

The application of biochar causes additional ammonia volatilization by around 3% ± 1%, corresponding to an increase of 0.26% and 0.27% of the national and agricultural total ammonia emissions, respectively (Miljødirektoratet, 2019; SSB, 2020e). The low contributions to both national and agricultural emissions is due to the comparatively high emissions of ammonia from handling of manures from livestock systems.

Soil emissions of NO_x decrease by about 7% ± 3%, corresponding to 1.6% of NO_x emissions from the agricultural sector in Norway. At the total national level, this reduction becomes negligible because agricultural NO_x emissions only represent 5% of the Norwegian emissions (which are dominated by oil and gas extraction and transportation) (SSB, 2020e). The somewhat larger uncertainty range for NO_x emissions from soils comes from the large uncertainty of NO_x emission factor from fertilizer, which can range from 0.005 to 0.104 kg NO_x kg^{-1} N applied (12.5 to 260% of the average emission factor of 0.04 kg NO_x kg^{-1} N applied) (Miljødirektoratet, 2019).

Biochar can reduce nitrogen leaching in agricultural soils by about 5% ± 2%, corresponding to 0.4% of the total anthropogenic nitrogen input to Norwegian coastline or 1.5% of the agricultural nitrogen losses compared to 2018 emissions (Selvik and Sample, 2018). However, uncertainty ranges are large and overlapping.

The potential energy recovery from pyrolysis can produce additional electricity and heat. The electricity potential is 880 ± 180 GWh $year^{-1}$, and heat potential is 1800 ± 370 GWh $year^{-1}$. This electricity generation represents about 0.6% of the electricity production in Norway in 2020 (SSB, 2020f), but heat production from pyrolysis has a larger potential contribution to the national energy system, as it can deliver about 30% of the current district heating production (SSB, 2020 g).

In general, the main co-benefits with climate change mitigation are

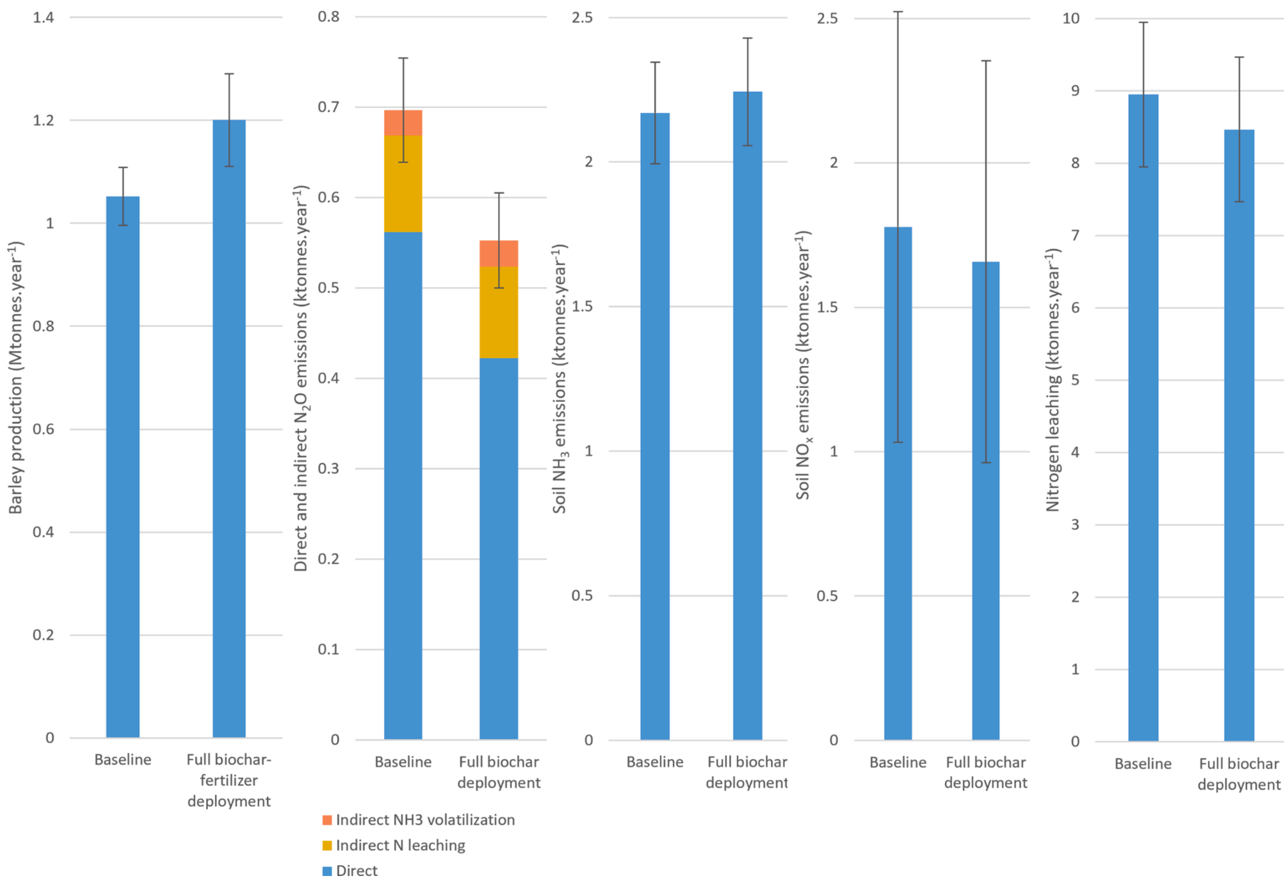

Fig. 7. Effects of a large-scale deployment of biochar application to Norwegian agricultural soils on grain yields, soil emissions (N_2O, NH_3, NO_x) and nitrogen leaching. Black whiskers show uncertainty range from the Monte-Carlo analysis (± one standard deviation). Results only refer to biochar effects in soils, and do not consider life-cycle emissions.

related to increased yields and reduction of N_2O emissions, while the decrease in soil NO_x emissions and nitrogen leaching has less confidence because of the large overlap of the uncertainty ranges. The increase in soil ammonia emissions is also not significant.

4. Uncertainties and limitations

Our results are subject to a series of uncertainties and limitations. In particular, soil conditions, climate and land management affect the level of soil emissions and how biochar alters them. Biochar's production and supply chain are also subject to uncertainties, such as variability in feedstock composition, which can affect yield and biochar's carbon content, or the distances between forest, biochar production plants and fields. The climate response to emissions of NTCFs is also dependent on a variety of factors. Variability on these parameters have been included in the Monte-Carlo analysis to investigate how they influence our results. The results are generally robust to these uncertainty factors, especially climate change effects based on GWP100 and most of the other impact categories (except marine eutrophication). Net negative climate effects from biochar with GWP20 are uncertain in two scenarios, for which the negative emissions of the biochar scenarios can be questioned, but overall a net mitigation relative to the reference system is achieved.

LCA studies are subject to different assumptions regarding system boundaries and other methodological aspects that make results specific to the individual case and comparison across studies challenging (Matuštík et al., 2020; Tisserant and Cherubini, 2019). Our work is specific to the Norwegian context in terms of resource supply, conversion processes, agricultural operations, and soil emissions. This country specific approach offers an estimate of the national potentials, co-benefits and trade-offs associated with alternative biochar utilization scenarios in Norway. Different local conditions and assumptions will evidently lead to different outcomes. For example, countries with a larger fraction of fossil-based electricity in their power mix will show higher avoided emissions in the CHP scenario, or larger application rates of biochar would cause larger effect on soil emissions. Compared to other LCA studies, our results are broadly consistent despite the large variability found in the literature. A recent review that summarized LCA studies of biochar systems with production and application to agricultural fields estimated total climate change impacts of -0.9 ± 0.3 tonnes CO2eq. tonnes^{-1} feedstock (median ± one quartile), but with 5th and 95th percentile of -0.1 and -1.5 tonnes CO2eq. tonnes^{-1} feedstock, respectively (Tisserant and Cherubini, 2019). For comparison, net climate change effects in our study range from -0.25 tonnes CO2eq. tonnes^{-1} feedstock (biochar and biochar-fertilizer scenarios, GWP100) to -0.8 tonnes CO2eq. tonnes^{-1} feedstock (biochar with bio-oil sequestration, GWP100) (Figure S8 in the SI). In line with our analysis, the review found that climate impacts from the biochar value chain can be up to $+0.5$ tonnes CO2eq. tonnes^{-1} feedstock, and those of biochar sequestration in agricultural soil can contribute -0.25 to -0.75 tonnes CO2eq. tonnes^{-1} feedstock (5th and 95th percentile) (we estimated about -0.6 tonnes CO2eq. tonnes^{-1} feedstock in our study).

Quantification of the effects of different deployment scenarios is based around national average soil conditions, fertilization and crop yields. This is a simplification as agricultural production is very heterogeneous and depends on local climate and practices. To take this variability into consideration, Norway's average barley yield over 10 years are used to partially even out the regional differences in management and fluctuations in weather conditions.

The effect of biochar on soil N_2O emissions is also uncertain. Both carbonization degree of the biochar and soil type appear as key factors controlling effects on N_2O emissions. For example a study reports that well carbonized biochar products, i.e., produced at higher temperature, consistently reduced N_2O emissions from two contrasted soil types (Weldon et al., 2019). Less carbonized biochars suppressed N_2O emissions only in a mineral soil but induced the opposite effect in a peat soil. The duration of the effect on N_2O emissions is also a source of variability. A recent biochar review reports an average reduction of N_2O emissions by 38%, but also indicates that reductions tend to be negligible after one year (Borchard et al., 2019). However, these emission reductions can be sustained over time by annual applications of well-carbonized biochar or BCF. In our Monte Carlo analysis, a range of 22 to 50% reduction is considered to take the variability of this effect into account. Although variable, it is important to consider this positive effect of biochar, especially in light of alternate solutions. On average, agronomic practices aiming at increasing carbon sequestration in soil lead to a slight increase in N_2O emissions, while biochar leads to a reduction (Guenet et al., 2021).

In addition to the uncertainties above, there are a range of processes and considerations that have not been investigated in our study. The economic dimension was not explored but it is a necessary component for a successful large-scale deployment of biochar systems. In general, a pyrolysis system with biochar production has a positive net present value at a feeding rate above 9 tonnes per hour (about 45 MW capacity or larger) at pyrolysis temperature above 450 °C, or above 6 tonnes/hour (about 30 MW capacity or larger) at pyrolysis temperature above 550 °C (Yang et al., 2021). An integrated strategy of producing both biochar and bioenergy is found to have higher net present value than simple bioenergy systems, in particular if there are positive effect of biochar on yields (Woolf et al., 2016). At about 290 MW and a pyrolysis temperature of 500 °C, our modelled scenarios for biochar production are thus within these economic viability criteria based on carbon market, bioenergy and biochar prices.

The possible effect of biochar on the degradation rate of native soil carbon stocks, an effect referred to as priming, has not been included in the analysis. On average, the addition of biochar amendments into soil has been reported to decrease the decomposition rate of the native soil organic matter and thereby further increase carbon sequestration (Ding et al., 2018; Wang et al., 2016). However, there is a large variability in these results (Ding et al., 2018; Wang et al., 2016). Low temperature chars are still rich in labile compounds and can increase mineralization, while higher temperature chars reduce the decomposition of organic matter (Chen et al., 2021). In addition, the priming effect of biochar on soil organic matter is often transient (Budai et al., 2016), and the long-term effects, if any, are uncertain.

Reduction of surface albedo due to darkening of soils after biochar application has been suggested to cause a warming feedback that can reduce the climate mitigation potential of biochar (Bozzi et al., 2015; Genesio et al., 2012; Meyer et al., 2012; Verheijen et al., 2013). For example, a study estimates that changes in albedo could reduce climate mitigation of biochar by 13–22% (Meyer et al., 2012). However, this effect is expected to be limited in Norway because of snow cover and low insulation during winter months. At high latitudes, the exposure of a darker soil would be limited to a few weeks in spring between snow melt and crop growth and in the autumn between harvest and snow fall. Further, the second year following a biochar application of 30–60 tonnes.ha^{-1} a decrease in the effect of biochar on soil albedo was observed due to further soil mixing under subsequent tillage operations, thereby reducing the potential changes in surface albedo in cases of one-off applications (Genesio et al., 2012). Albedo changes after biochar application could also be managed by maintaining a canopy cover in between cropping cycle using cover crops, with potential additional benefits in terms of soil carbon accumulation (Jian et al., 2020).

Our analysis does not include the alternative oxidation rate of forest residues left in the forest to decompose (Guest et al., 2013; Ortiz et al., 2014). Part of the residues will become CO_2 in a few years and a smaller fraction will return to soil and litter. The potential inclusion of these fluxes would alter the profile of our results especially in the short term, as residues would represent a sort of temporary short-term carbon storage, but in the long term the residues will largely oxidize to CO_2 in any case. Their collection and use as biochar will move the temporary storage from the forest to the agricultural soils, as part of the carbon in the feedstock goes to biochar. Several reviews and meta-analysis investigate the consequences of removing forest residues after tree harvest on forest productivity, soil nutrient content, soil carbon stock and soil properties with some contrasting results (Achat et al., 2015a, 2015b; Clarke et al., 2021; Hume et al., 2018; Ranius et al., 2018; Wan et al., 2018), in particular for forest soil carbon stocks (Achat et al., 2015b; Hume et al., 2018; Ranius et al., 2018; Wan et al., 2018) and in cold climates (Achat et al., 2015b; Clarke et al., 2021). In general, the level of residues harvested is an important determinant of the effects on forest ecosystems. In our analysis, a conservative extraction rate of 35% is used, which is below the 50% limit recommended for sustainability criteria in nearby Scandinavian countries (de Jong et al., 2017).

Dust emissions from biochar handling, especially during its application, have raised concerns for implications on human health and climate (Gelardi et al., 2019; Genesio et al., 2016). However, these emissions are hard to measure and robust estimates are not readily available in the literature. A BC emission in the range of 0.3–6.7 kg ha^{-1} (0.01–0.26% of 2.5 tonnes ha^{-1} application rate) of biochar dust can result in no net negative emissions for the biochar and biochar-fertilizer scenarios, when short-term climate change impacts are assessed using the uncertainty ranges of the characterization factors for BC with GWP20 (270–6200 kg CO_2eq. kg^{-1}). Options to limit the potential emissions of dust exist, for example by applying the biochar wet and under low wind conditions, or use biochar pellets (as it is in our biochar-fertilizer scenarios) (Gelardi et al., 2019).

Biochar has been shown to reduce availability of heavy metals in soils and limit their uptake by crops (Chen et al., 2018; Hilber et al., 2017), as well as affecting pesticides' fate in soils (Liu et al., 2018). Biochar's effect on heavy metals was not included in our analysis due to a lack of wide-spread data on concentration and availability of heavy metals in Norwegian agricultural soils. Limited data are also available for the effects of biochar on pesticides, and contrasting findings are sometimes reported, with usually lower availability under biochar amendment but mixed effect on their degradation (Liu et al., 2018). Both effects can be potential co-benefits of biochar application to agricultural soils for human health and terrestrial ecotoxicity, but are expected to have little overall influence on our results. It is unlikely that reduction in availability of heavy metals in soils can offset the effect on terrestrial ecotoxicity, which is primarily linked to the emissions of heavy metals in the supply chain of biochar, while pesticides had a negligeable contribution to terrestrial ecotoxicity impacts.

Storage of the bio-oil in geological deposits can have technical challenges and limitations that are to be overcome. Bio-oils are known to have higher viscosity than heavy oil, and the corrosivity can make transport and pumping difficult. However, a review study argues that bio-oils and fossil crude oils have similar properties in terms of pumping and transportation (Schmidt et al., 2018). Bio-oils are also slightly corrosive due to low pH and should be carefully stored, particularly as they contain toxic compounds (Cordella et al., 2012). If geological sequestration of bio-oils turns out unfeasible or uneconomical, bio-oil can be used in a variety of products to replace fossils (for fuels or chemicals) (Pinheiro Pires et al., 2019). Incorporation of bio-oils in asphalt paving would correspond to an alternative form of carbon sequestration.

5. Conclusions

Biochar production is a mature process and one of the most cost-efficient NETs, and can be a strategic option to be developed in the near-term before other technologies emerge. Our analysis shows that negative emissions can be achieved for all scenarios when accounting for a wide range of emissions (both GHGs and NTCFs) along the entire life-cycle. The exclusion of life-cycle emissions leads to an overestimate of the mitigation potential of 10–20% when benefits of co-products are excluded. Including a variety of biochar-induced soil effects in the analysis allowed to quantify potential co-benefits or trade-offs regarding other environmental impact categories: increased food production, reduced stratospheric ozone depletion and, though uncertain, marine eutrophication, while impacts on tropospheric ozone formation, terrestrial acidification, fine particulate matter formation and terrestrial ecotoxicity are increased. Biochar could significantly reduce emissions of N_2O at the Norwegian level, while application of biochar-fertilizer could represent a benefit in terms of increased grain production. However, the effect is more uncertain in terms of reduction of NO_x emission and leaching of nitrogen and increased NH_3 emissions. Integrating emissions from both the supply chain and soils is important to prevent spill-over effects, as we found that some co-benefits in terms of soil emission reduction can be outweighed by emissions happening in the supply chain. Greener future transportation systems and stricter emission control measures at the pyrolysis facilities can mitigate these adverse effects, with additional benefits for tropospheric ozone formation and fine particulate matter formation. These results show the need of taking a holistic approach in terms of accounting emissions along the biochar supply chain and assessing environmental impacts using multiple assessment methods. Better knowledge regarding soil effects can help to guide an optimal management of biochar and agricultural land based on local conditions.

CRediT authorship contribution statement

Alexandre Tisserant: Investigation, Formal analysis, Methodology, Writing – original draft. **Marjorie Morales:** Methodology, Writing – review & editing. **Otavio Cavalett:** Methodology, Writing – review & editing. **Adam O'Toole:** Investigation, Writing – review & editing. **Simon Weldon:** Investigation, Writing – review & editing. **Daniel P. Rasse:** Project administration, Funding acquisition, Writing – review & editing. **Francesco Cherubini:** Conceptualization, Project administration, Funding acquisition, Supervision, Writing – review & editing.

Declaration of Competing Interest

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

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Supplementary materials

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