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Biochar and Its Potential to Deliver Negative Emissions and Better Soil Quality in Europe

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Key Points:

- Biochar from biomass residues in Europe has a negative emission potential of 70–290 $MtCO_2\ yr^{-1}$
- Biochar application to cropland generally increases soil carbon, water holding capacity, crop production, and reduces soil nitrogen losses
- Some trade-offs can occur in few locations, mainly for lower yields and air pollution

Supporting Information:

Supporting Information may be found in the online version of this article.

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Abstract Negative emissions are essential to limit global warming, but their large-scale deployment rises sustainability concerns. At the same time, agricultural soils are under increasing threat of degradation, as measured by losses in soil organic matter, water holding capacity, and nutrient retention, with increasing negative effects on plant productivity. Biochar from biomass residues is a technically mature option that does not compete for land and can typically restore key functions of degraded soils while delivering negative emissions. However, quantitative estimates of its potentials in Europe and a detailed spatially-explicit analysis of the co-benefits and trade-offs for agricultural land are unclear. Here, we estimate an annual negative emission potential of biochar from forest and crop residues available in Europe from 1.7% to 3.9% of 2021s European greenhouse gas emissions (15.2%–35% of the agricultural emissions), depending on residue potentials and biochar scenarios. At the same time, biochar application to cropland increases water holding capacity (+6.5%-9%), crop production (+7.1%-8.4%), NH_3 volatilization (+21.7%-24.2%), and reduces soil N_2O emissions (−13.7%–34.7%) and nitrogen leaching (−17.5%–22.7%). There are spatially heterogeneous trade-offs for some soil effects (ammonia volatilization and yields) and air pollution (mainly due to emissions from biochar systems). Biochar offers synergistic solutions that co-deliver across different sustainability challenges, but its optimal deployment requires strategies tailored to local conditions.

Plain Language Summary Limiting global warming requires a large-scale deployment of negative emission technologies (NETs) to capture and store atmospheric carbon dioxide. However, many NETs typically require land use and compete with food security or nature conservation. Biochar, a stable charcoal-like material, can be produced from biomass residues without competing for land and, when applied to agricultural soils, it delivers negative emissions while providing agronomic benefits. An analysis of the environmental and agronomic effects of biochar production and use in Europe is key to identify sustainable pathways. Our results offer high-resolution estimates of negative emissions and maps of biochar-induced co-benefits and trade-offs with crop production, soil quality and soil emissions. When correctly implemented, biochar can co-deliver for multiple global challenges (climate change, food security, contrast land degradation) and help a cross-sectoral sustainability transition in Europe.

1. Introduction

Limiting global warming to well below 2°C requires large and cost-effective deployment of negative emissions technologies (NETs) (IPCC, 2018). Several NETs have been suggested, but they are usually not technologically or economically mature (e.g., bioenergy with carbon capture or storage, BECCS; direct air capture), require land use (BECCS, afforestation) and/or may compete with other sustainable goals or planetary boundaries when deployed at large scale (Heck et al., 2018; Minx et al., 2018). Among NETs, increasing soil carbon with biochar is recognized as one of the most effective measures (Smith, 2016; Smith, Davis, et al., 2016). Biochar, a stable form of carbon, is the solid remainder of biomass decomposition at high temperature in the absence of oxygen, and can be produced from a variety of feedstocks (Lehmann et al., 2006; Tisserant & Cherubini, 2019). Biochar is an attractive NET as it is technologically mature and capable to co-deliver several agricultural co-benefits (Minx et al., 2018; Schmidt et al., 2021). Co-products of biochar production (i.e., bio-oil and syngas) can also be used to

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produce energy and avoid use of fossil fuels (Azzi et al., 2019; Yang et al., 2017, 2021). Sequestering the liquid bio-oil for long-term storage has been explored as an option to further increase the negative emission potential (Schmidt et al., 2018; Werner et al., 2018).

Biochar can be theoretically applied to all managed land without changing its current use, and unlike other NETs it does not compete for land. Agricultural soils are under increasing threat of degradation (IPBES, 2018; Prăvălie et al., 2021; Smith, House, et al., 2016). Loss in soil organic matter and soil erosion lead to loss in soil structure, water holding capacity, and nutrient retention, causing direct loss in plant productivity (Prăvălie et al., 2021; Stolte et al., 2015). About 35% of European cropland is exposed to aridity (Prăvălie et al., 2021), which is expected to become more severe under a warming climate (IPCC, 2021). More than 6% of European soils have severe erosion rates above 11 t ha^{-1} $year^{-1}$, and about 25% have unsustainable erosion rates above 2 t ha^{-1} $year^{-1}$ (Panagos et al., 2020). These erosion rates lead to degradation in soil quality with progressive losses of soil carbon, nutrients, and water retention capacity (Borrelli et al., 2017; Mokma & Sietz, 1992; Stolte et al., 2015), requiring some form of mitigation or rehabilitation measures (Prăvălie et al., 2021).

Application of biochar to agricultural soils increases below-ground carbon storage and generally helps restoring degraded soils by inducing potential co-benefits such as improved soil structure (Islam et al., 2021), nutrient retention (Gao et al., 2019; Q. Liu et al., 2018; Y. Liu et al., 2018), crop yields (Jeffery et al., 2017; Ye et al., 2020), soil carbon accumulation (Blanco-Canqui et al., 2020; Ding et al., 2018), and water retention capacity (Fischer et al., 2019; Razzaghi et al., 2020). Biochar also affects soil nitrogen emissions from fertilizer use, typically reducing N_2O and NO_x emissions and nitrogen leaching, and with variable effects on NH_3 volatilization (Borchard et al., 2019; J. Liao et al., 2020; Pourhashem et al., 2017; Weldon et al., 2019).

The extent of the biochar-induced agronomic effects depends on the type of biomass feedstock, soil characteristics, local climate and biochar application rate (Chen et al., 2018; Gao et al., 2020; Jeffery et al., 2017; Liu et al., 2019). Existing studies on large-scale potentials show a large mitigation potential of biochar at regional or global levels (Woolf et al., 2010; Yang et al., 2021), but they usually have a poor representation of local biophysical constraints by taking a top-down approach (Griscom et al., 2017; Pratt & Moran, 2010) or a coarse resolution in the quantification of resource availability (Woolf et al., 2010), and do not quantify the associated soil effects nor the life-cycle emissions of the supply chain (Griscom et al., 2017; Powell & Lenton, 2012; Pratt & Moran, 2010). Environmental analysis of individual biochar systems are more common and typically rely on life-cycle assessment (LCA) (Azzi et al., 2021; Matuštík et al., 2020; Tisserant & Cherubini, 2019; Yang et al., 2021). LCA studies of biochar systems generally converge on the importance that specific factors have in shaping the climate benefits, but they usually disregard effects on soil emissions and do not estimate mitigation potentials of large-scale biochar deployment. The latter depends on the volume of biomass resources that are available without exacerbating adverse side-effects on natural ecosystems, food production, or competitive biomass uses. In Europe, agricultural and forestry activities generate large amounts of residues that are currently under-utilized (Scarlat et al., 2019; Verkerk et al., 2019). Estimates of residue potentials are available for different sustainability constraints, socio-economic factors, and spatial resolutions (Fulvio et al., 2016; García-Condado et al., 2019; Scarlat et al., 2019; Verkerk et al., 2019). There is a growing political interest toward their mobilization to support a growing bioeconomy, favor a circular economy perspective, and revitalize rural areas (European Commission, 2020). The negative emission potentials of a large-scale deployment of biochar produced from available residues in Europe, and the concomitant effects on key soil quality indicators, are still unclear.

Here, we address this issue from a bottom-up perspective by integrating spatially-explicit estimates of residue availability in Europe (Scarlat et al., 2019; Verkerk et al., 2019) with biochar production systems and applications to cropland. Depending on various sustainability constraints, two estimates of agricultural and forest residue availability representative of a low and high potential are considered (see Methods). A life-cycle approach is used to model supply chain emissions, including emissions of both greenhouse gases (GHGs) and near-term climate forcers (NTCFs), and logistic models are used for optimal distribution of biochar conversion plants and transport distances. The analysis considers alternative treatment of co-products: no valorization, used for co-production of heat and power, or long-term storage of bio-oil. Biochar's effects on soil emissions, crop yields, and soil water retention are quantified using empirically-derived and spatially-explicit data sets parameterized for biochar types (wood or straw), application rates, soil properties and other local environmental factors (Kroeger et al., 2021; Liu et al., 2019). As the soil response varies with the amount of biochar applied, two biochar application rates are considered (at either 5 t ha^{-1} re-applicable after 10 years or as a single application at 30 t ha^{-1}), and cropland with the highest rates of soil erosion is prioritized for biochar treatment (as it is the one with more urgent remediation needs). The robustness of the results is tested using complementary climate metrics and a Monte-Carlo analysis based on 10'000 runs with variations in key process parameters and biochar-induced soil effects. A set of environmental indicators are included in the analysis of co-benefits and trade-offs: terrestrial acidification and ecotoxicity, air quality via ozone and fine particulate matter formation, stratospheric ozone depletion, and marine eutrophication.

2. Materials and Methods

2.1. Feedstock Availability

Estimates of biomass residues available in Europe considers both crop residues and forest residues. Gridded data of crop residues availability represent the collection potential of residues from the main crops cultivated in Europe (i.e., wheat, rye, barley, oats, maize, rice, rapeseed, and sunflower) (Scarlat et al., 2019). Different potentials are available according to varying constraints: theoretical potential (i.e., all residues available), technical potential (i.e., recoverable with current harvesting machines, about 60% of the theoretical potential), environmental potential (i.e., maximum harvest level that does not negatively impact soil organic carbon (SOC) stocks over 15 years) and sustainable potential (i.e., minimum value between the technical and environmental potential). Gridded forest residues availability were taken from a study that quantified the potentials under different set of constraints connected to harvest intensities and residue extraction rates: a base scenario representing current practices, a high potential scenario with increased mechanization and more flexible guidelines for harvest, and a technical scenario for the maximum harvestable biomass level (Verkerk et al., 2019).

Manure is also a feedstock that can be used for biochar production, but it is not considered in this work because it is a well-established feedstock for anaerobic digestion and biogas production. Being more wet than lignocellulosic residues, it cannot be blended in the same pyrolysis process as it requires energy-intensive pre-drying (Rajabi Hamedani et al., 2019) or, when hydrothermal carbonization is used, it results in a so-called hydrochar that contains less carbon and is more unstable than biochar (Bamminger et al., 2014; IPCC, 2019).

Table S1 in Supporting Information S1 provides an overview of the possible combinations of the different residue potentials, with the corresponding biochar production capacity, cropland areas that can be treated at 5 or 30 t ha^{-1}, and the average number of year to treat all cropland areas in Europe (160 Mha). The table also includes an estimate of the forest and crop residues that are already in use for other applications (e.g., fodder, bioenergy, etc.), that is, 32 Mt year^{-1} (Camia et al., 2021) and 29 Mt year^{-1} (Thorenz et al., 2018). Throughout the article, tables and figures, the unit abbreviated as "*t*" refers to metric tonnes (or Mg). Out of the possible options, two scenarios representative of a low and high residue supply were selected for our analysis. The low residue supply is given by the sum of the sustainable supply potential for crop residues (149 Mt) and the base supply potential (38 Mt) for forest residues, to which the residues that are already used for other applications are subtracted. This scenario is conservative and intended to reflect a sustainable removal rate of residues. The high residues supply combines the technical potential for crop residues (212 Mt) and the high potential (71 Mt) for forest residues, without the subtraction of the residues used for other purposes. The technical potential of forest residues supply is not considered feasible, as well as the use of tree stumps or roots, for both ecological and economic reasons and they are excluded from the analysis. Figure S1 in Supporting Information S1 shows the resulting maps at 1 km resolution of the residues supply scenarios used in our study.

2.2. Pyrolysis, Biochar Production, and Residues Collection

The pyrolysis system is modeled with a biomass throughput of 80 t hr^{-1} and 7,000 operating hours a year, leading to a consumption of about 560 kt of dry feedstock per year (Dickinson et al., 2015; Haarlemmer et al., 2012; Tews & Elliott, 2014). Typically, costs of bioenergy production plants decrease with increasing plant capacity (Haarlemmer et al., 2012; Tews & Elliott, 2014), and similar considerations apply to biochar production systems (Dickinson et al., 2015; Shackley et al., 2011; Yang et al., 2021). Table S2 in Supporting Information S1 provides an overview of the scale of deployment of pyrolysis for biochar production under different residue supply scenario. The pyrolysis process is modeled in the Aspen process simulation software, where biochar and the tars (i.e., the organic content of the bio-oil) are modeled as non-conventional compounds, to calculate specific emission factors and energy requirements of the plant. A complete description of the modeling can be found in a previous publication (Tisserant et al., 2022). A pyrolysis process temperature of 500°C was chosen to balance carbon sequestration, as biochar yields decrease with higher pyrolysis temperature, but biochar stability in soil increases with increasing pyrolysis temperature (IPCC, 2019). Biochar yields from forest and crop residues are 26% and 24% dry ash-free mass basis, respectively. Biomass composition (i.e., content in cellulose, hemicellulose, lignin and C, H, O, N, S, Cl) was retrieved from the Phyllis2 database as averages of wood and straw. Yields and composition of co-products from pyrolysis (i.e., biochar, bio-oil, and syngas, mainly made of CO, H_2, CH_4, CO_2) are modeled using equations from Woolf et al. (2014) and complemented with fate factors for N, S, and Cl (Tisserant et al., 2022). The carbon content of the biochar produced at 500°C is 82% and the carbon content of the tars is 61% for straw and 64% for wood.

Three pyrolysis systems with alternative uses of co-products are modeled. In Py, biochar is recovered, and the co-products are burnt to provide heat for the pyrolysis without using the extra heat generated. The case PyCHP is similar to Py, but the extra heat supplies a combined heat and power cycle (CHP) at 71.5% and 28.5% of efficiency, respectively (Sipilä, 2016). The heat is assumed to replace heat from natural gas, and electricity the EU average electricity mix. In the case PyCS, syngas and about 11% of the bio-oil are combusted to meet the energy needs of the pyrolysis plant, while the remaining of bio-oil is condensed and recovered for storage. Aspen plus simulations also provide emissions to air of the major chemical compounds, including NO_x, N_2O, and SO_x. Emissions of polycyclic aromatic hydrocarbon (PAHs), NMVOC, PM10 and heavy metals associated with particulate matter (As, Cd, Cr, Cu, Pb, Hg, Mo, Ni, Sn) are taken from the literature (Sørmo et al., 2020). In the case of pyrolysis with bio-oil recovery, the emission factors are corrected by the amount of tar sent to combustion. Inventories for the different biochar production cases can be found in Table S3 in Supporting Information S1.

A linear programming logistic model is developed to equally distribute the available residues to the biochar plants while aiming at minimizing transport distances. A more detailed description of the logistic model can be found in the Text S1 in Supporting Information S1 and an overview of the computed transport distances is shown in Table S4 in Supporting Information S1. As this approach has inherent uncertainties and alternative models could be considered, the influence on the final results of different transport distances is explored in a sensitivity analysis (Monte Carlo uncertainty analysis, see below).

2.3. Biochar Application to Cropland Soils

This study considers a single application of 30 t ha^{-1}, or the application of 5 t ha^{-1} that can be reapplied on the same field every 10 years. The analysis takes into account the difference in soil response, the scaling of the effect to area treated (i.e., six times more land treated each year at 5 t ha^{-1} compared to 30 t ha^{-1}), and consequences for life-cycle emissions (i.e., transportation requirements).

Soil erosion leads to losses in soil structure, soil carbon, nutrients, and water retention, increasing the risks of sustaining long-term yields (Stolte et al., 2015). Biochar can help restoring some functionalities of soils threatened by erosion, as it usually increases carbon storage (Blanco-Canqui et al., 2020; Ding et al., 2018), improves soil structure and soil aggregate (Islam et al., 2021), retains nutrients (Liu et al., 2019), increases soil water holding capacity (WHC) (Edeh et al., 2020), and improves root traits, which contribute to alleviate nutrient or water deficiency of eroded land (Xiang et al., 2017). It can also have direct positive effects against soil losses (Li et al., 2019; Seitz et al., 2020), but some mixed effects have been observed (Blanco-Canqui, 2021). Soil erosion rate of European cropland is thus chosen as an aggregated indicator of degradation processes to identify land areas that could benefit the most of the possible direct or indirect restoration effects induced by biochar, as soils under medium to high degradation rates urgently require some form of treatment (Borrelli et al., 2017; Prăvălie et al., 2021). Soil erosion rates in European cropland are shown in Figure S2 in Supporting Information S1. The map is produced by processing a resampled raster at 25 km resolution (Borrelli et al., 2017) and selecting cropland areas from the Global Land Cover 2000 database (GLC2000) (Bartholomé & Belward, 2005), as this is the data set used to simulate biochar-induced changes in soil emissions (see section below). Soil erosion represents erosion by water expressed in mass of soil loss per unit area and time and is estimated with the RUSLE model (Borrelli et al., 2017).

Locations of biochar application each year (for the application rates considered, 5 or 30 t ha^{-1}) were determined using the amount of arable land in each pixel, prioritized according to the highest soil erosion rate and proximity of the biochar plant. The amount of land treated each year is removed from the optimization to identify the next

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area to be treated. In the case of re-application (5 t ha^{-1}), the land is reintroduced in the optimization routine after 10 years. A complete description of the biochar application model can be found in Text S2 in Supporting Information S1.

2.4. Soil Effects

Multiple studies investigated the soil response after biochar application for a variety of soil emissions and agronomic indicators. The available meta-analyses highlight the need to consider the type of biochar, its production conditions, application rate, local soil and climate conditions to understand and model soil responses to biochar. Here we use the only available spatially explicit and empirically-derived model that estimates biochar-induced effects to agricultural soils (Liu et al., 2019). This is used to quantify grid-specific biochar's effects on crop yield, soil N_2O and NH_3 emissions and nitrogen leaching. The model is based on a global meta-analysis of biochar's effects on soils that is used to feed a multivariate Random Forest algorithm. It considered a variety of variables, such as biochar's feedstock types (including straw and wood), biochar's application rates (from 0 to 10 up to >120 t ha^{-1}), soil properties (pH, cation exchange capacity, SOC content, soil texture) and climatic zones. This approach can estimate biochar's effects on soil by simultaneously considering these multiple drivers, and it is an improvement relative to the use of default factors from traditional meta-analysis methods that typically disaggregate results using one or two variables only (Liu et al., 2019).

Biochar's effect on soil NO_x emissions was not included in the model used, and it is in general less studied. In our analysis, we use a single factor (with associated uncertainty ranges) for each specific biochar's application rate, and it is assumed to be the same for wood or straw biochar. At an application rate of 5 t ha^{-1}, an average effect of 10% reduction was chosen (range 0%–20%), in line with previous studies (J. Liao et al., 2020; Niu et al., 2018; Xiang et al., 2015). At 30 t ha^{-1}, the effect ranges from 0% to 67% reduction (maximum measured reduction observed at 20 t ha^{-1}) (Nelissen et al., 2014), with an average factor of 34%. These factors are based on a review of literature data (Fan et al., 2017, 2020, X. Liao et al., 2020; 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., 2016, 2019). In general, soil NO_x emissions are mainly reduced when biochar is produced at high temperature and under high application rates (Wang et al., 2019; Weldon et al., 2019). Increased NO_x emissions can occur for biochars produced at temperature below 400°C (Weldon et al., 2019).

Biochar's effect on soil water content at field capacity (water holding capacity, WHC) is estimated using a regression model (Kroeger et al., 2021). The regression model considers a variety of parameters such as sand content in the soil, the biochar application rate (5 or 30 t ha^{-1} in our analysis), biochar particle size (1 mm), pyrolysis temperature (mid-range, 500°C) and biochar feedstock (either wood or straw). The main limitation of this model is that it is based on experimental data biased toward sandy soils, as about 70% of the data points used to train the model have a sand content above 45%. For this reason, in our analysis we only estimate the potential changes in WHC for European cropland soils that have a sand content over 45%. This is consistent with the findings of another meta-analysis that found significant biochar's effect on WHC in soils with sand content over 50%, and no effect in those with less than 50% (Edeh et al., 2020). This type of soils is usually referred as coarse (sand, loamy sand and sandy loam).

Still many uncertainties remain regarding the soil effects of biochar, in particular regarding the persistence of the effect and whether field effects will be of similar intensity than laboratory scale studies, which are typically used in meta-analyses. To account for these uncertainties, we take a conservative approach and only account for biochar effects on soil emissions for the year following its application to the field. We also perform an uncertainty analysis on biochar's effects on soils via a Monte-Carlo analysis, where 10'000 maps of biochar's soil effects were generated using the uncertainty ranges of each individual soil effect (see section Uncertainty analysis). For each year, the effect factors are read at the locations where biochar is applied and averaged. Biochar's effect on WHC averages is weighted by the cropland area treated with biochar in each pixels. Biochar's effect on crop yield averages is weighted by the amount of crop production in each pixel (i.e., product of cropland area treated with biochar and crop yield). Biochar's effect on N_2O and NH_3 emissions and nitrogen leaching are weighted by the amount of nitrogen applied in each pixel (i.e., product of cropland area treated with biochar and nitrogen application rate). The combined averages of effects from wood and straw biochars are further weighted by the relative amount of wood or straw biochars applied in each pixels. With this procedure, we generate 10'000 averaged effect of biochar over the locations that were treated with biochar for each given year. The standard deviation of these

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10'000 average effects is calculated and represent the uncertainty of the effect, while the mean of the 10'000 represent the mean biochar's effect. To derive the average and uncertainty of biochar's effect over the 30 years span of the analysis, the average of the annual means is taken and the standard deviation, s_{pooled}, is calculated by pooling the annual standard deviation, s_{year}, using the following equation.

$$s_{pooled} = \sqrt{\frac{\sum_{year=1}^{30} s_{year}^{2}}{30}} \tag{1}$$

In the Results section, black whiskers represent the uncertainty of the mean effect (± standard deviation) from a Monte-Carlo analysis based on the uncertainty in the response of each individual grid, while the blue whiskers represent the mean ± standard deviation of the spatial variability of the effect. The uncertainty analysis of the soil effects is then integrated with that of the life-cycle analysis (see below).

2.5. Life-Cycle Assessment (LCA)

The LCA considers the main stages of the biochar systems, such as harvesting and collection of the residues, their transport to the biochar conversion plants, pyrolysis, biochar transport to the fields and its application (spreading and harrowing in soils). Agriculture residues collection is modeled using the EcoInvent process Straw {RoW}| wheat production | APOS, U and forest residues as Wood chips, dry, measured as dry mass {RER}| market for | APOS, U. Instead of defining a generic inventory for crop production (which can differ from country to country, farmer to farmer) and because of a lack of robust data on how biochar would affect other farming practices, we only accounted for direct and indirect emissions associated with the value chain of biochar. We thus consider that the only difference between cultivation with and without biochar is due to the biochar system itself (fertilization level, soil work, pesticides etc., remain the same). Since we only show the difference in the results relative to the present state where no biochar is used, this is not affecting the results because the unchanged inventories cancel out each other. Bio-oil for long-term storage is assumed to be transported by truck to either Karsto (Norway) or Teeside (UK), depending on the biochar conversion plant location, and then to the geological deposits by oil pipeline (Norsk Petroleum, 2022). Figure S3 in Supporting Information S1 shows the system boundaries considered in our analysis. EcoInvent 3.5 is used to gather indirect emissions inventories from energy consumption and emission factors associated with the provision of equipment, materials and inputs (Wernet et al., 2016). Tables S5 and S6 in Supporting Information S1 show a summary of the inventories for the different LCA stages.

Climate impacts are evaluated using the global warming potential at a time horizon of 100 years (GWP100), which is the most commonly used climate metric. Emissions of both GHGs (mainly CO_2, CH_4, N_2O) and NTFCs (NO_x, NH_3, CO, SO_x, non-methane volatile organic compounds (NMVOC), organic carbon (OC) and black carbon (BC)) are considered. These two categories of climate forcers affect the climate differently: GHGs are long-lived and are well mixed in the atmosphere, and affect the climate globally; NTFCs are short lived and are not well mixed, meaning that the resulting climate impacts are sensitive to the emission location and are thus spatially and temporally heterogeneous (Aamaas et al., 2016; Levasseur et al., 2016). As no single climate metric can capture the impacts of the climate forcers with such different lifetimes, the sensitivity of the climate impacts were assessed using short-and long-term metrics (Cherubini et al., 2016; Jolliet et al., 2018; Levasseur et al., 2016). For the short- and long-term climate impacts the global temperature potential (GTP) at time horizon 20 and 100 years are chosen. GTP measures the instantaneous contribution of an emission to future global temperature increase at its given time horizon (Joos et al., 2013; Myhre et al., 2013; Shine et al., 2005). GWP100 can be interpreted as a mid-term climate indicator as its values are numerically close to those of GTP40 (Allen et al., 2016). Characterization factors for NTCFs are taken from a multi-model intercomparison study (Aamaas et al., 2016), and are averaged between summer and winter for Europe. Uncertainty ranges are chosen as the largest range between winter and summer. Characterization factors with uncertainty ranges can be found in Table S7 in Supporting Information S1.

For the other impact categories, emissions are characterized using averaged mid-point characterization factors from ReCiPe 2016 v1.1 (Huijbregts et al., 2017). Biochar can affect these impacts by its effect on soil N_2O emissions (stratospheric ozone formation), NO_x (tropospheric ozone formation, fine particulate matter formation and terrestrial acidification), NH_3 (terrestrial acidification, fine particulate matter formation), and nitrogen leaching

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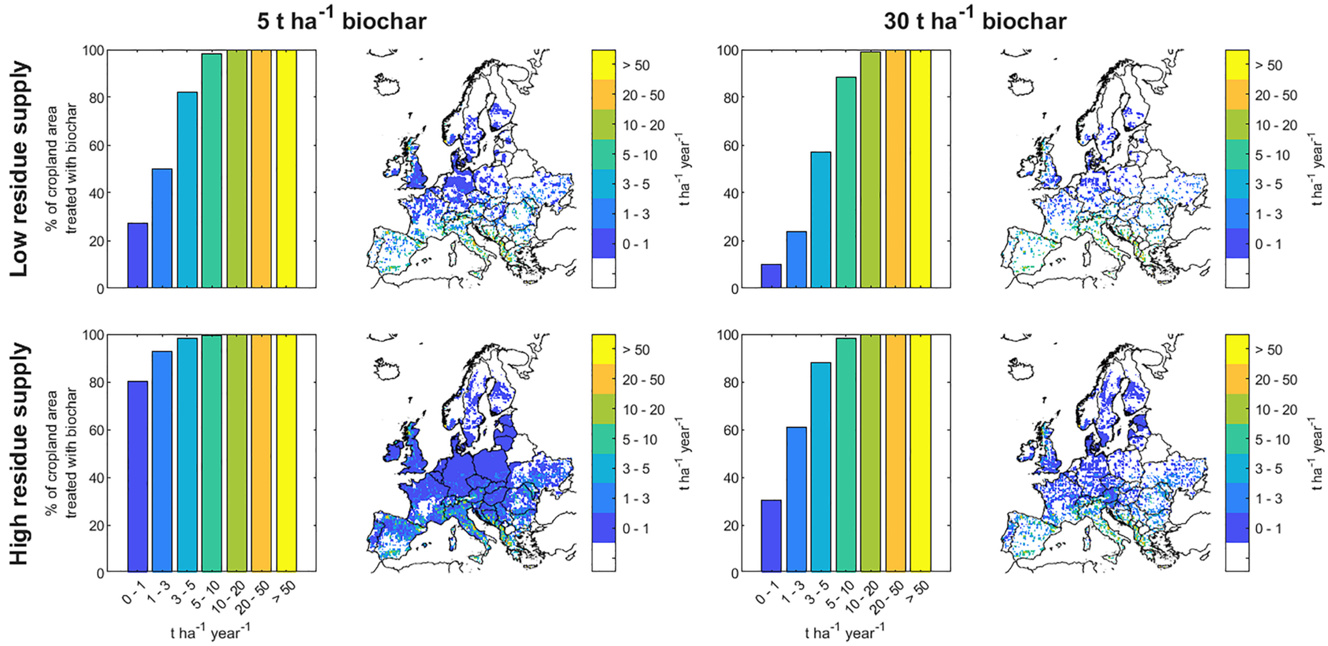

Figure 1. Fraction and spatial distribution of cropland under different soil erosion rates treated with biochar at different residue supply potentials and biochar application rates. The application of biochar to agricultural soils is prioritized according to a soil erosion gradient: cropland threatened by high levels of soil erosion is treated first. For the biochar application rate at 5 t ha^{-1}, all cropland is treated three times. At application rate of 30 t ha^{-1}, all cropland is treated only once.

(marine eutrophication). Emissions of heavy metals and particular matter are key drivers of terrestrial ecotoxicity impacts.

2.6. Uncertainty Analysis

To test the robustness of the LCA results, a Monte Carlo simulation is performed with 10'000 runs. The distribution is assumed to be triangular, as we have an estimate of the mean of the parameters and their range of variability. In addition to the uncertainty factors for biochar's effects on soil emissions and crop yields discussed in the previous section, the uncertainty analysis considers variability in the following key factors: emission factors from the supply-chain processes, biochar yield and its carbon content and stability in soils, carbon content of the bio-oil, transport distances, climate metrics and. Further, OC and BC emissions are not currently represented in the EcoInvent database and were estimated from PM10 emissions and emission factors from Bond et al. (2004). Values and uncertainty ranges for the different parameters are summarized in Table S8 in Supporting Information S1.

For biochar's effects on soils, low and high uncertainty bounds are taken from Figure 1 in Liu et al. (2019) and in Edeh et al. (2020), and represent the 95% confidence interval. Since the standard deviation of biochar's effects is not available at the grid level, we assume that the standard deviation is constant and the 95% range of uncertainty is the same for all grid cells, but centered around the grid cell estimate. Uncertainty ranges for biochar's effects on soils are shown in Table S9 in Supporting Information S1.

3. Results

3.1. Biochar Potentials and Cropland Area Treated

We identify a low and high biomass residue potential in Europe of 127 and 287 Mt year^{-1}, respectively (see Figure S1 in Supporting Information S1). Crop residues are 94% and 74% of the supply, respectively, and are primarily concentrated in the major agricultural districts. Forest residues from harvesting and thinning (stumps excluded) are more evenly widespread in the continent. These biomass potentials can generate 30.6 ± 2.4 Mt (low) and 70.4 ± 4.5 Mt (high) of biochar. This biochar can be applied to a large extent of European cropland (Figure 1). At a biochar application rate of 5 t ha^{-1} (re-applied every 10 years), 6.1 and 14.1 Mha of cropland can be treated each year for the high and low residue potentials (or respectively 3.6% and 8.3% of European cropland). At 30 t ha^{-1}, the cropland treated are 1 and 2.3 Mha per year potentials (or respectively 0.6% and 1.4% of European cropland). Over an assessment

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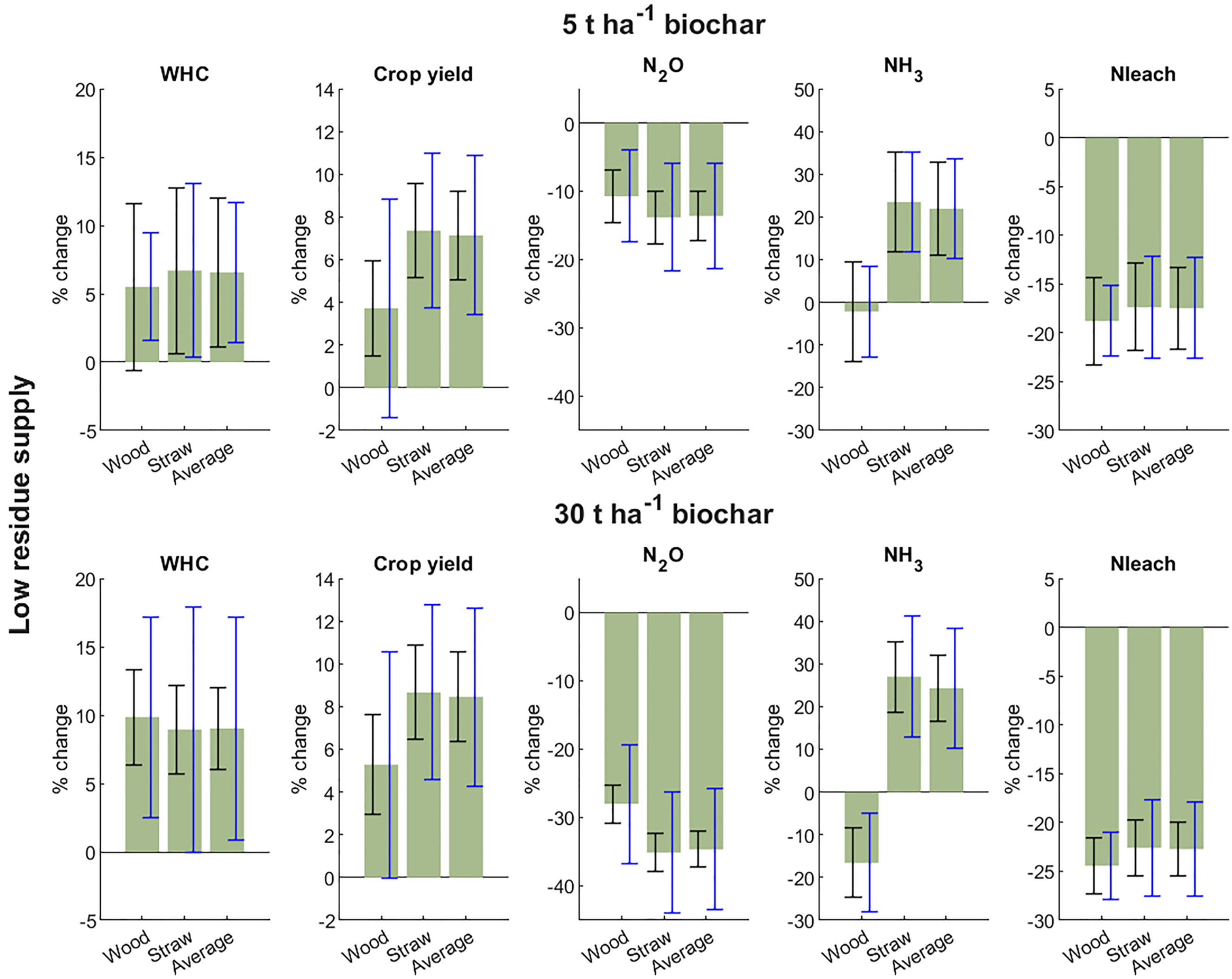

Figure 2. Effects of biochar from the low residue supply potential on key soil quality-related indicators: soil water holding capacity (WHC), crop yield, N_2O and NH_3 emissions, nitrogen leaching. Results are shown for wood biochar, straw biochar, and the average effect of the two biochar types. Black whiskers represent the uncertainty of the mean effect (± standard deviation) averaged over 30 years of biochar application for all European cropland as derived from a Monte-Carlo analysis based on the uncertainty in the response of each individual grid. The blue whiskers represent the mean ± standard deviation of the spatial variability of the effect. The same results for the high residue supply potential are shown in Figure S5 in Supporting Information S1.

period of 30 years, the total cropland area treated at 5 t ha^{-1} ranges from 37% (62.7 Mha treated three times over 30 years) at low residue supply to 84% (141.9 Mha treated three times over 30 years) at high residue supply. For a single biochar application at 30 t ha^{-1}, the area treated is 18% (30.4 Mha) and 42% (71.0 Mha) of European cropland.

Nearly all cropland in Europe can be treated with the combination of high resource potential and low biochar application rate. At an application rate of 5 t ha^{-1} with the low residues supply, nearly all cropland with erosion rates above 5 t ha^{-1} yr^{-1} receive three biochar applications. For the high residues supply, this occurs to all cropland with erosion rates above 3 t ha^{-1} yr^{-1}. Most of the cropland with high soil erosion rates is located in South and Southeast Europe.

3.2. Biochar's Effects on Soil

Biochar application to European cropland is expected to increase water holding capacity (WHC), crop yield and ammonia volatilization, and reduce N_2O emissions and nitrogen leaching (Figure 2). There is a large variability in the soil response (see Figure S4 in Supporting Information S1 for the spatial distribution), which depends on biochar's feedstock type, application rate and local conditions. Since there are limited studies on the long-term effects of biochar on soil emissions and crop productivity, biochar-induced soil effects are only considered for the year following application. In general, the relative changes of soil responses are rather similar between the low (Figure 2) and high residue supply scenarios (Figure S5 in Supporting Information S1). The main difference is in the averaged effect between wood and straw biochar, as the share of wood biochar from forest residues is larger in the high supply case.

The average increase in soil WHC is about 6.5 ± 5% at 5 t ha^{-1} and about 9.0 ± 8.1% at 30 t ha^{-1} (mean ± standard deviation of the spatial variability), and WHC increases in all locations. Increased plant available water can help to reduce irrigation needs and secure yields, as eroded land tend to increase water stress leading to yield losses (Arriaga & Lowery, 2003; Mokma & Sietz, 1992). The effects of biochar on crop yields are heterogeneous. On average, biochar enhances crop yields by 7.1 ± 3.7% at 5 t ha^{-1} and 8.5 ± 4.2% at 30 t ha^{-1}, but there are caveats. Higher biochar application rates and straw biochar are associated with increases in crop yield (Figure S4 in Supporting Information S1), while wood-based biochar has uncertainty ranges in the negative domain (i.e., yield losses). Decreases in yields mainly occur in Scandinavia for both biochar types and application rates.

Reductions in soil emissions of N_2O (a powerful GHG and a depleting agent of stratospheric ozone) are almost three times larger at a biochar application rate of 30 t ha^{-1} than 5 t ha^{-1}. Mitigation of soil N_2O emissions is generally stronger at higher application rates. In several locations around the Mediterranean Sea, effects on soil N_2O emissions change from positive (an increase in emission) to negative (a decrease in emission) when going from 5 t ha^{-1} to 30 t ha^{-1}. Changes in volatilization of NH_3 (an aerosol precursor affecting air quality, terrestrial acidification, and with a cooling effect on climate) depend on both the type of biochar applied and its application rate. Straw biochar mostly increases NH_3 volatilization at both application rates, while the response to wood biochar is highly scattered at 5 t ha^{-1}, resulting in an almost neutral continental average change. At 30 t ha^{-1}, a reduction in NH_3 volatilization is predicted in almost all cropland areas treated with wood biochar. However, since the fraction of crop residues is larger than that of forest residues, the net effect is an average increase in NH_3 volatilization. In the high residue supply case, the fraction of wood-based biochar is larger, and the average increase in NH_3 volatilization is reduced (Figure S5 in Supporting Information S1). Biochar has a clear effect in reducing nitrogen leaching in agricultural soils in Europe, and the more the biochar applied to the field the stronger is the reduction. This has positive effects in terms of both increased nitrogen retention in soils and reduced eutrophication of downstream water bodies.

3.3. Effects on Crop Production and Environmental Impacts of Soil Emissions

Biochar is found to simultaneously increase crop production in Europe and decrease impacts of soil emissions for climate change (CC), ozone depletion potential (ODP), marine eutrophication potential (MEP), and ozone formation potential (HOFP), but increase terrestrial acidification potential (TAP) and particulate matter formation potential (PMFP) (Figure 3).

The increase in annual crop production is between 0.06 ± 0.01% (low residues supply potential, at 30 t ha^{-1}) and 0.57 ± 0.16% (high residues supply potential, at 5 t ha^{-1}). The increase in crop production is about five times more important at 5 than 30 t ha^{-1}, as lower biochar application allows to treat more cropland each year (six times more cropland is treated each year at 5 compared to 30 t ha^{-1}).

The climate effects of changes in soil emissions after biochar treatment show an overall mitigation that is mostly driven by reductions in direct N_2O emissions. Additional cooling contributions are due to increased NH_3 volatilization, a cooling agent, and to a reduction in NO_x emissions and nitrogen leaching, which are partly converted to N_2O. Results are similar under different climate metrics representing alternative types of climate impacts or time horizons (Figure S6 in Supporting Information S1).

N_2O is the dominant ozone-depleting substance since the ban of chlorofluorocarbons (Ravishankara et al., 2009). A reduction in soil N_2O emissions thus results in a mitigation of ODP, which is about twice larger at 5 t ha^{-1} than at 30 t ha^{-1}. About three times more reduction in N_2O emissions is achieved at 30 t ha^{-1} biochar compared to 5 t ha^{-1} (see Figure 2), but six times more cropland is treated at 30 t ha^{-1}. Reduction in nitrogen leaching due to biochar lead to a net mitigation of marine eutrophication. The extent of the mitigation follows the similar trend of ODP. A similar pattern is observed for impacts on HOFP, which causes damages to human health and ecosystems.

The increased NH_3 volatilization due to biochar application drives higher impacts in TAP and PMFP. These increases are mainly due to straw-based biochar. Reduction in soil NO_x emissions can to some extent mitigate the increased impacts in these two categories, but the contributions from ammonia are clearly dominating, as the specific impact of NH_3 is 5 and 2 times higher than that of NO_x for TEP and PMFP (Huijbregts et al., 2017).

Climate impacts decrease almost everywhere in Europe (Figure 4), although at 5 t ha^{-1} there is a local warming in some areas in the South. These contributions are largely overwhelmed by the increase in soil carbon

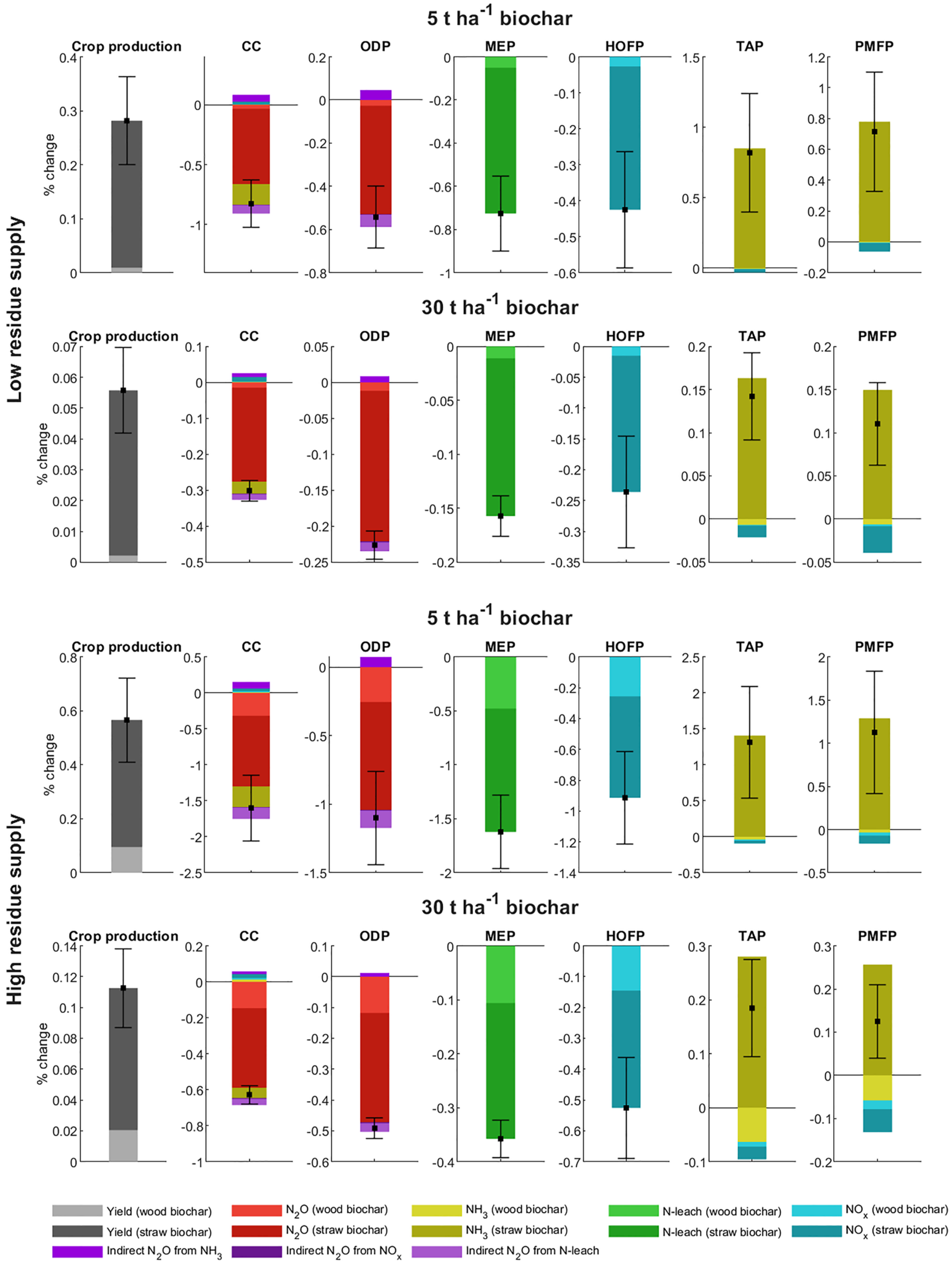

Figure 3. Annual average changes in crop production and soil emission impacts due to the application of biochar in European cropland, relative to a base case where no biochar is applied. Black whiskers represent the mean ± standard deviation of the effect at a European continental scale taking into account the uncertainty of biochar's effects on soil emissions (and uncertainty of GWP100 characterization factors of NH_3 and NO_x). CC: climate change (GWP100), ODP: stratospheric ozone depletion potential, MEP: marine eutrophication potential, HOFP: tropospheric ozone formation potential, TAP: terrestrial acidification potential, PMFP: particulate matter formation potential.

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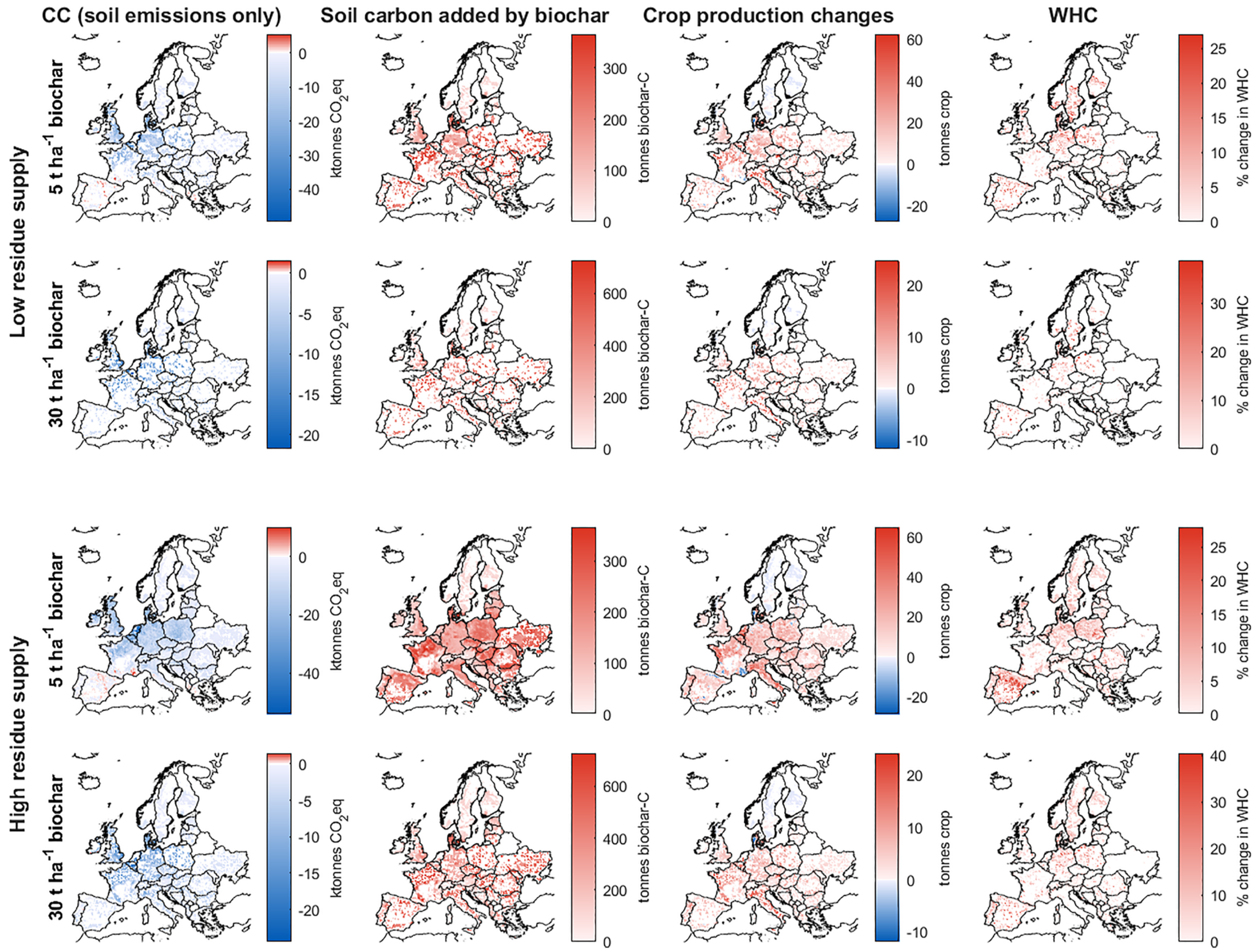

Figure 4. Spatial variability in the contributions to changes in climate change impacts (GWP100) from soil emissions, biochar carbon added to soils, crop production and changes in WHC. Results are shown for biochar produced from two residue supply cases (low and high) and for two different biochar application rates (5 and 30 t ha^{-1}) to agricultural soils in Europe. The maps have a resolution of 20 × 20 km. CC: climate change; WHC: soil water holding capacity.

that results from the addition of biochar. These increases are mostly located on the same cropland from which residues were taken, thereby reducing the risk of carbon depletion from residues harvesting. Crop production increases in most of the locations, but a (weak) reduction in crop yield is also reported, especially in Scandinavia. WHC increases in all the treated soils, but due to a data gap, changes in WHC are only estimated for coarse soils with sand content over 45% (see Materials and Methods) (Edeh et al., 2020). Spain, Portugal, Greece and regions around the black sea that currently face arid conditions (Prăvălie et al., 2021) can thus benefit the most from the biochar-induced increase in WHC. The spatial distribution of the impacts from biochar-induced soil emission changes on ODP, MEP, HOFP, TAP, and PMFP are shown in Figure S7 in Supporting Information S1.

3.4. Life-Cycle Assessment

From a life-cycle perspective that integrates emissions from the residue and biochar supply chains with soil effects, a significant mitigation of climate change, ODP and MEP, is found for the three different biochar systems considered (Py, biochar only with no energy recovery from co-products; PyCHP, biochar with combined heat and power (CHP) from combustion of co-products; PyCS, biochar and bio-oil sequestration). There are trade-offs for other impact categories for Py and PyCS, whereas there is still a net mitigation for PyCHP, except for TETP (Figure 5). These results are annual average impacts for low and high residue potentials at a biochar application rate of 5 t ha^{-1}. Results for the application rate at 30 t ha^{-1} are generally similar because soil emis-

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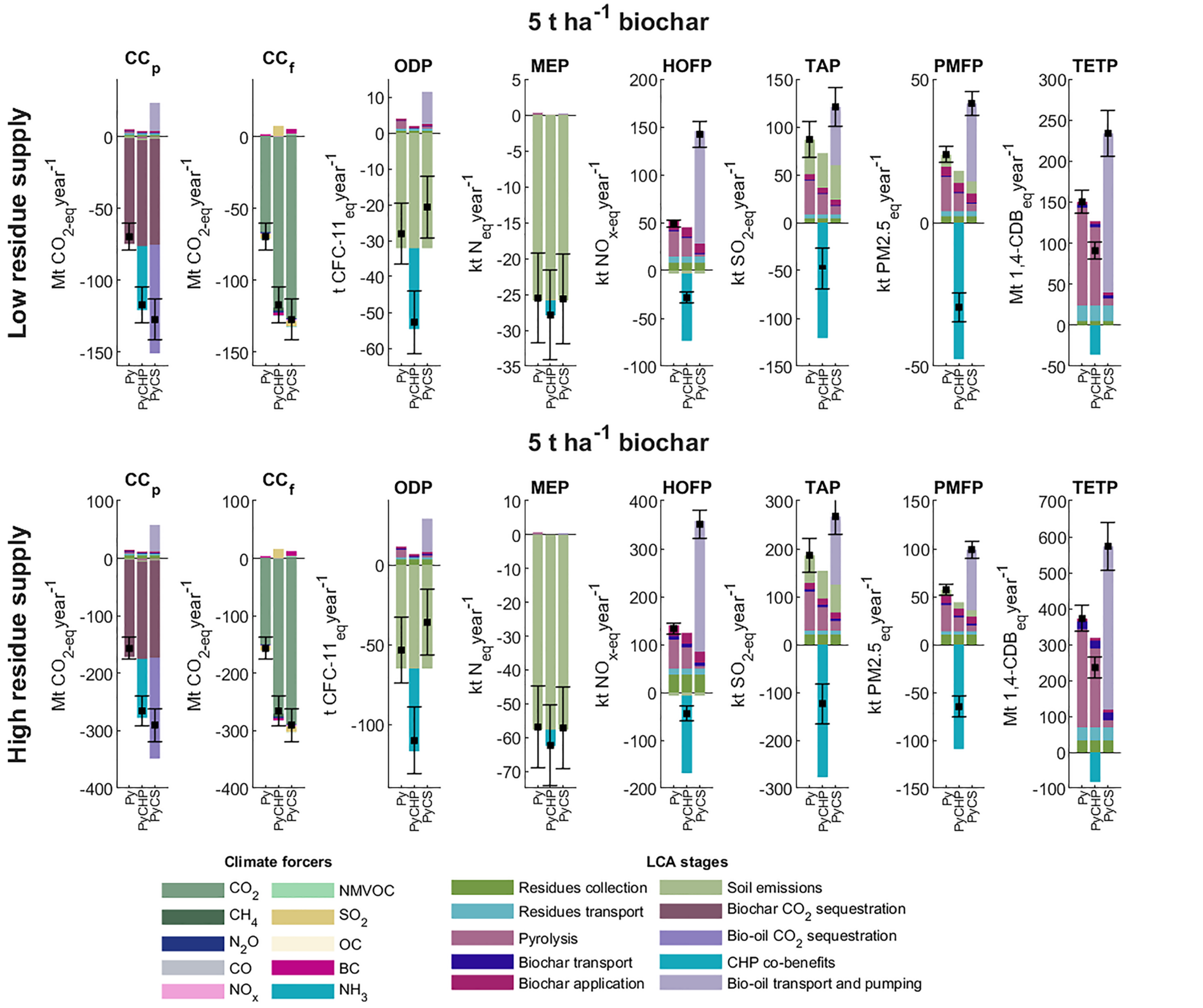

Figure 5. Annual average total impacts from a life-cycle perspective that integrates direct and indirect emissions from the biochar systems with biochar-induced soil emissions. Results are shown for two residue supply potentials (low and high) and a biochar application rate to agricultural soils of 5 t ha^{-1} (see Figure S8 in Supporting Information S1 for those at 30 t ha^{-1}). The three biochar technologies considered are: Py, biochar only, with no external benefits from pyrolysis co-products; PyCHP, with the co-products of the pyrolysis used in a CHP system to generate electricity (replacing European electricity mix) and heat (replacing heat produced from natural gas); PyCS, with the bio-oil produced during pyrolysis recovered, transported by trucks and ships and pumped to geological deposits for storage. Impacts refer to climate change per process (CCp) and climate forcers (CCf), terrestrial acidification (TAP), marine eutrophication (MEP), tropospheric ozone formation (HOFP), stratospheric ozone depletion (ODP), fine particulate formation (PMFP) and terrestrial ecotoxicity (TETP). Black points represent the mean net effect and whiskers show one standard deviation. Note: different axis scale between low and high residue supply.

sions are usually small contributors to the total impacts compared to process emissions from the supply chain (Figure S8 in Supporting Information S1). Table S10 provides results for Figure 5 and Figure S8 in Supporting Information S1.

At 5 t ha^{-1}, the net climate benefits are 69.8 ± 9.3 $MtCO_2$eq $year^{-1}$, 117.2 ± 12.4 $MtCO_2$eq $year^{-1}$ and 127.4 ± 14.0 $MtCO_2$eq $year^{-1}$ for Py, PyCHP and PyCS, respectively, for the low residue potential. The mitigation is two times larger for the high residue potential, which produces about two times more biochar. Despite the larger biochar volumes, normalized transport distances and average soil effects are very similar (Figure 2; Figure S5 in Supporting Information S1), so the mitigation potential scales by a factor of two.

Along the life-cycle, warming contributions are associated with fuel consumption during residues harvesting and biomass/biochar transport (3.7 $MtCO_2eq$ $year^{-1}$ and 12.3 $MtCO_2eq$ $year^{-1}$ in the low and high supply case). Pyrolysis contributes to warming emissions (1.3 and 2.9 $MtCO_2eq$ $year^{-1}$), especially for Py and PyCS that require electricity inputs for plant operations. PyCHP uses internally produced energy, and its climate impacts are only due to direct emissions that result in net cooling (cooling emissions of NO_x and SO_x have a stronger effect than the warming from black carbon and CO). The mitigation potential of biochar-induced changes in soil emissions is 1.0 and 2.0 $MtCO_2eq$ $year^{-1}$, which is very small compared to the mitigation potential secured by biochar CO_2 sequestration (73.7 ± 9.2 $MtCO_2eq$ $year^{-1}$ or 169.8 ± 19.5 $MtCO_2eq$ $year^{-1}$, depending on feedstock availability scenario). Relevant benefits are also achieved when CHP from the biochar plant replaces average EU-electricity and heat produced from natural gas (44.7 ± 5.8 $MtCO_2eq$ $year^{-1}$ or 102.7 ± 12.4 $MtCO_2eq$ $year^{-1}$, and about 46.1% from electricity and 53.9% from heat generation). The geological storage of bio-oil can bring even higher benefits (75.6 ± 8.4 $MtCO_2eq$ $year^{-1}$ or 176.0 ± 18.2 $MtCO_2eq$ $year^{-1}$). Climate change mitigation results are robust across different climate metrics, showing no trade-offs between short and long-term perspectives (Figure S9 in Supporting Information S1).

Soil emissions are the main contributors to MEP and ODP, for which the mitigation potential is much lower at a biochar rate of 30 t ha^{-1} than 5 t ha^{-1} (Figure 5; Figure S8 in Supporting Information S1). The supply chain is responsible for a negligeable fraction of MEP, as soil emission changes are the main drivers. Regarding ODP, N_2O is also emitted from combustion processes. Avoided emissions in the PyCHP system lead to higher ODP mitigation, while for PyCS the large transport need for bio-oil (1,300 km on average, Table S3 in Supporting Information S1) leads to significant N_2O emissions that offset a large share of the benefit of reduction in soil N_2O emissions.

NO_x, CO and NMVOC emissions from combustion processes are the main contributors to the formation of tropospheric ozone, which causes air pollution with impacts for human health and ecosystems. The biochar-induced reduction in soil NO_x emissions is almost negligeable compared to the impact from emissions from residue harvesting, transport, pyrolysis and biochar spreading. The contribution of the pyrolysis step is higher for Py than PyCHP, because in Py ozone precursors are emitted during the pyrolysis and indirectly via grid electricity, whereas for PyCHP the latter are avoided as internal electricity is used. Further, in PyCS only the syngas and about 11% of the bio-oil are combusted, thereby leading to lower emissions than Py or PyCHP.

Emissions of NO_x, NH_3 and SO_x contribute to impacts on TAP as it redeposits on natural environment. For the same reasons as for HOFP, pyrolysis contributions to TAP decreases from Py to PyCS. Changes in soil emissions due to biochar are a net positive contributor with an increase in TAP. The contributions of soil emissions at a biochar application rate of 30 t ha^{-1} is smaller (Figure S8 in Supporting Information S1), and impacts are mostly driven by emissions in the supply chain. In the PyCHP case, avoiding NO_x and SO_x emissions from grid-electricity and heat generation allows to more than offset TAP impacts from the value chains and biochar-induced soil emission changes.

PMFP characterizes emissions of NO_x, NH_3, SO_x and PM2.5 that contribute to the formation of particulate matter. Impact patterns across the different technologies, residues supply cases, and biochar's application rates resemble those of TAP. Terrestrial ecotoxicity is the only impact categories with net positive impacts for all biochar systems. For Py and PyCHP, the pyrolysis process is the most impacting stage, mostly due to emissions of heavy metals and particulate matter during the combustion stage.

3.5. Negative Emission Potentials

The application to agricultural soils of biochar from European biomass residues has the potential to achieve large-scale negative emissions (Figure 6). By only accounting for the carbon sequestered in the soils by biochar, the annual average negative emission potential is (mean ± one standard deviation) 73.8 ± 9.4 Mt CO_2eq per year and 169 ± 19.4 Mt CO_2eq per year for the low and high residue supply scenarios, respectively. This corresponds to about 1.7 ± 0.2 and 3.9 ± 0.5% of European GHG emissions in 2021 (WRI, 2023), or to 15.2 ± 1.9 and 35.0 ± 4.0% of the emissions from the agricultural sector only (WRI, 2023). Considering the carbon sequestration from both biochar and bio-oil storage, can 3.5 ± 0.4% and 8.0 ± 0.9% of the European GHG emissions (or 30.8 ± 3.7% and 71.3 ± 7.7% of EU agricultural emissions) can be mitigated.

Taking a LCA perspective, for each residue supply scenario the negative emission potentials are more affected by the biochar system and treatment of co-products than the rate at which biochar is applied to the soils. In the Py system, results are similar to those mentioned above, but large differences occur with the other two systems. In the PyCHP system, the production of heat and electricity can increase negative emissions to 6.1 ± 0.6% of Europe 2021 emissions in the high residue supply case. The associated energy production from the biochar

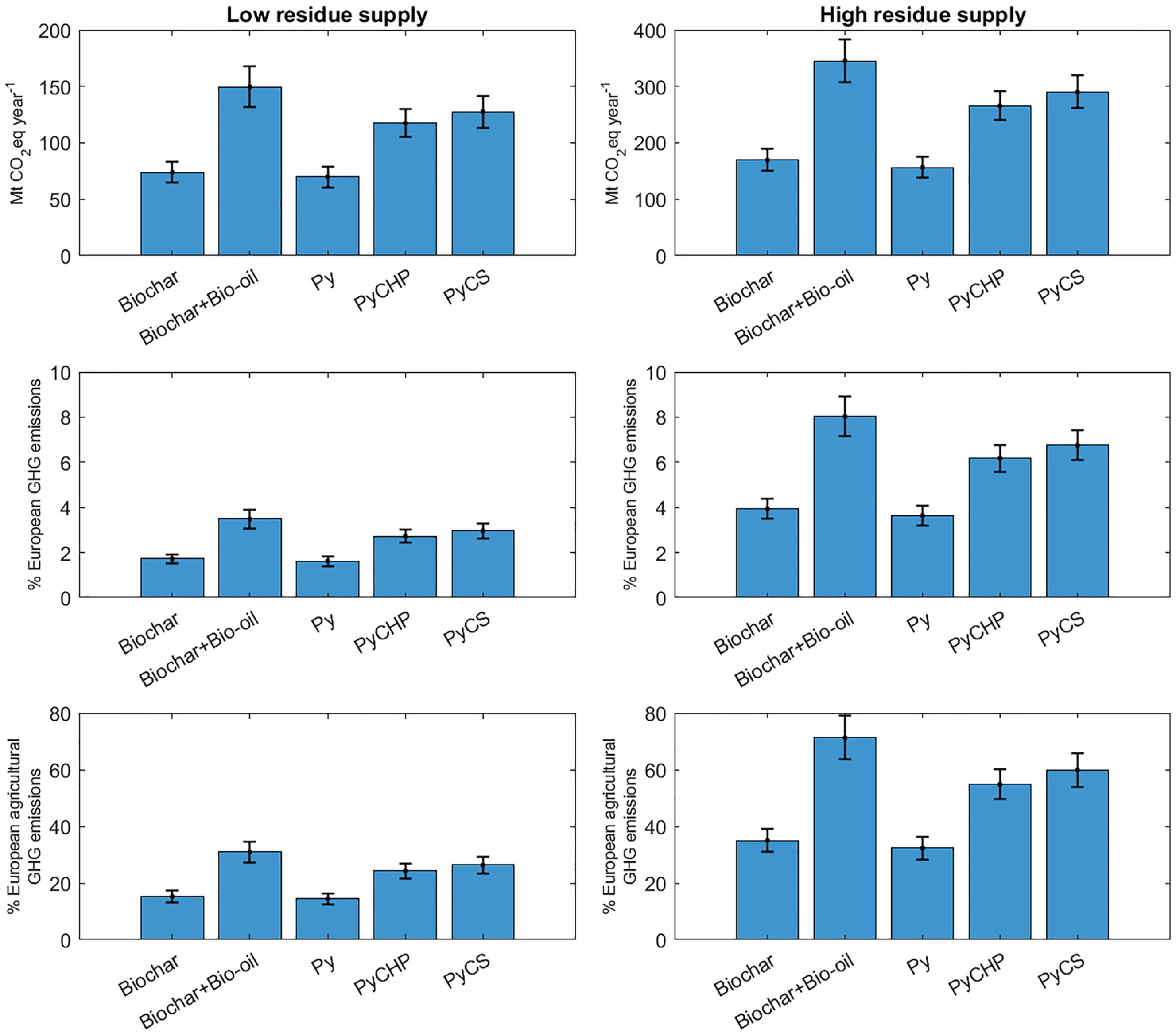

Figure 6. Annual average negative emission potentials of a large-scale deployment of biochar from biomass residues to agricultural soils in Europe. Results are shown considering only the carbon stored in biochar and eventually bio-oil or by taking an life-cycle assessment (LCA) perspective (GWP100) where direct and indirect emissions from the supply chains are integrated with the changes in soil emissions. The LCA perspective is only applied when different biochar systems (Py, PyCHP, and PyCS) are considered. The three biochar technologies considered are: Py, biochar only, with no external benefits from pyrolysis co-products; PyCHP, with the co-products of the pyrolysis used in a CHP system to generate electricity (replacing European electricity mix) and heat (replacing heat produced from natural gas); PyCS, with the bio-oil produced during pyrolysis recovered, transported by trucks and ships and pumped to geological deposits for storage. The negative emission potentials are compared to 2021 European total GHG emissions and European's agricultural GHG emissions. Results are shown for the low and high residues supply potential and a biochar application rate of 5 t ha^{-1} (results are very similar between the two biochar application rates).

co-products corresponds to 3.9% of Europe's 2021 electricity consumption and 34.4% of the heat (65.5 ± 5 TWh vs. 151.1 ± 9.6 TWh and 482.7 ± 37.4 PJ vs. 1122.6 ± 71.3 PJ) (IAE, 2022). Sequestrating both biochar and bio-oil could offset up to 3.0 ± 0.3 and 6.7 ± 0.7% of Europe's 2021 emissions (case PyCS).

4. Discussion

Our analysis shows the potential for large-scale negative emissions from a deployment of biochar to agricultural soils in Europe. Co-benefits are identified for crop yields, water holding capacity of soils, and reductions in soil emissions (with positive effects for impacts to tropospheric ozone formation, stratospheric ozone depletion, and marine eutrophication). At the same time, trade-offs occur with terrestrial acidification, ecotoxicity and fine particulate matter formation, mainly because of emissions happening in the supply chain of the biochar systems.

In general, mitigation potential and effects scale with the amount of residues considered. However, the biochar application rate and the local/climatic conditions of the agricultural soils affect the results. Low biochar application rates have larger effects at a continental scale (larger co-benefits or trade-offs) as they allow to annually treat larger areas compared to an application rate of 30 t ha^{-1}. Higher biochar application rates provide higher increase in WHC, mitigation of soil N_2O emissions and nitrogen leaching and are better at mitigating NH_3 volatilization in some regions. In some areas, benefits for crop yields are minimal, if not detrimental, and this might hinder farmer's acceptance. While a continental-scale analysis is useful to benchmark expected potentials for negative emissions and map co-benefits and trade-offs across space, the diversity in soil responses calls for an implementation of biochar systems tailored to local environmental conditions.

We used currently available spatially explicit models to estimate biochar's potential effects in European soils. However, these models still have limitations (see original manuscripts for more details). For example, they include short-term laboratory experiments that might differ from real soil effects (e.g., biochar-induced changes in soil N_2O emissions are suspected to be only short-term or may not be observable on field scale experiments (Borchard et al., 2019; Guo et al., 2023; Verhoeven et al., 2017)). Biochar's effect on soil NO_x emissions is less studied than N_2O, and the long-term effects are unclear. As soil NO_x emissions are usually correlated to N_2O, their effect may be also transient (Vinken et al., 2014). Biochar can increase NH_3 volatilization due to its alkalinity, moving chemical equilibrium toward NH_3, while its surface chemistry can help retain NH_4^+ and help reduce its volatilization as NH_3 (Q. Liu et al., 2018; Y. Liu et al., 2018). For these reasons, the increase in NH_3 can be expected to be transient, as oxidation of its surface will help to better retain NH_4^+ and its alkalinity will decline (Q. Liu et al., 2018; Y. Liu et al., 2018). However, as biochar can reduce soil bulk density and compaction, increased soil aeration can sustain NH_3 volatilization over time, if diffusion is the limiting factor to NH_3 volatilization. It is worth noting that crop residues left on soil would also lead to NH_3 emissions (Xia et al., 2018), and acid treatment of biochar before application could reduce the risk of increasing NH_3 emissions (Asada et al., 2006; Doydora et al., 2011; Puga et al., 2019). Regarding the temporal effects of biochar-induced retention of nitrates in soils, it is unclear how it will develop once the mixture of soil and biochar reaches the new maximal capacity, and it will likely depend on site-specific nitrogen cycling, uptake by plants and inputs of nitrogen fertilizers.

To deal with these uncertainties, we only considered the effects for 1 year following application of biochar and performed a Monte-Carlo uncertainty analysis to explore how variability in soil emission factors affects the results. In addition, when biochar soil effects are integrated with impacts from emissions from the biochar supply chain, they represent a small fraction of the total impacts (except for ODP and MEP). So, using alternative factors for the biochar-induced soil effects is not expected to significantly change the overall conclusions of our study and the estimates of the net mitigation potential of biochar.

In general, our findings are robust to a large range of uncertainty parameters tested in a Monte-Carlo analysis, but various limitations and assumptions exist (see Text S3 in Supporting Information S1). For example, we did not consider other positive effects that biochar can induce to agricultural soils, such as a reduction in the bio-availability of heavy metals and the resulting limited uptake by crops (with reduced threats to human health) (Chen et al., 2018), the sorption of pesticides and their reduced leaching (Y. Liu et al., 2018), or the biochar capacity to reduce degradation of native soil organic carbon (a process called negative priming) (Ding et al., 2018). Biochar can also affect methane cycle of soils (Cong et al., 2018; Ji et al., 2018), in particular at high application rates and for the Mediterranean region (Ribas et al., 2019). However, the effects are mixed and highly interactive, leading to either a net release or uptake of methane (Cong et al., 2018; Ji et al., 2018), and a robust data sets supporting a large-scale analysis across different soil types and climatic conditions is missing. Given the low contribution of N_2O (a stronger climate forcer than methane) to the LCA climate impacts, it is unlikely that inclusion of changes in soil methane emissions will dramatically affect our results. Biochar can also induce changes in biophysical factors that are relevant for the local climate, such as reduced albedo due to darkening of the soil or modifications of land cover properties influencing radiation and moisture fluxes (Fischer et al., 2019; Genesio et al., 2012; Verheijen et al., 2013). These aspects are highly site-specific and uncertain, and more knowledge on their interactions with biochar is needed before they can be robustly included in mitigation studies.

The locations of the biochar conversion plants were estimated by identifying regions of equal residues collection potential and then minimizing the transport distances. This is only one of the many possible ways that can be used to identify plant locations and transport distances. Refined approaches can consider cost-efficient solutions where, for example, pyrolysis heat can be used by nearby industries or for district heating, or existing industrial sites are prioritized to take advantage of retrofitting possibilities and system integration. To consider the inherent uncertainty in simulating transport distances, an uncertainty factor of ±20% was included in the Monte-Carlo analysis. In addition, we explicitly test the specific sensitivity of the results to varying transport distances by using either the mean, the shortest or the longest distances (from Table S4 in Supporting Information S1) for transportation of residues, biochar and bio-oil, keeping everything else equal. The consideration of maximum transport distances can reduce the climate change mitigation potential relative to the mean case by about 7%–20%, 4%–11%, and 21%–28% depending on the residues supply scenario, biochar application rate, and climate metric for Py, PyCHP, and PyCS, respectively (Figure S10 in Supporting Information S1). The higher ranges (above 20%) are only for the PyCS scenario and when a very-short term climate metric (GTP20) is used, owing to the longer transport distances involved and larger impact of near-term climate forcers emitted from diesel trucks. These ranges are slightly higher than those resulting from the full uncertainty analysis (as shown in Figure 6), which are around 9%–14%. Biochar is frequently acknowledged as a small-scale solution for decentralized climate mitigation, where local or farm-scale biochar production and use require short transport distances. In our case, the consideration of the shortest transport distances would increase the climate change mitigation potentials of the biochar systems of typically less than 2% for Py and PyCHP, and around 16% for PyCS (due to the remaining long travel distances for bio-oil storage). Overall, our results show that only marginal gains in climate mitigation efficiency could be obtained if all transport distances are taken as minimal.

Our study considered soil erosion as a general criterion to prioritize biochar application to cropland, as biochar treatment can ameliorate some of the local negative effects of erosion. However, there are some risks that under certain conditions biochar can induce lower soil strength and higher erodibility (Blanco-Canqui, 2021). Other approaches can consider a mix of different threats more locally distributed for optimal applications. For example, soil acidity, low cation exchange capacity or contamination by heavy metals are the soil properties that can be most significantly improved by biochar. The analysis of the large-scale potential performed in this study shows that the majority of cropland can be treated in all scenarios, especially with high residue potential supply and low biochar application rate (when nearly all European cropland can be treated). This means that alternative choices of prioritization are mostly relevant for smaller-scale studies addressing specific regional contexts.

In contrast to BECCS, biochar is technically mature and can provide negative emissions with various co-benefits, although political support, public acceptance and new economic models to pay for sequestered carbon are necessary for its implementation. The multiple environmental co-benefits of biochar can be a key to stimulate its deployment across a range of actors. Relative to other NETs, biochar has the social advantages that it does not need to be implemented at a large industrial scale, but it is suited to be deployed in a way to distribute benefits across different actors or at a subsistence farming level. The question of acceptance and economic viability can particularly arise in regions with annual mean temperatures below 10°C, where yield increases from biochar are not achieved. Future progresses on biochar properties and management can change this picture. For example, pre-mixing of biochar with synthetic fertilizers or manure can improve nutrient uptake efficiencies of the crops (Rasse et al., 2022). Further, the meta-analysis used for crop yield effects is based on experimental trials that were not necessarily designed for yield benefits, but rather testing responses of other agroecological variables or of different types of biochar and application rates. The identification of biochar production processes and optimal management practices in specific locations can improve the positive effects on yields and partially (if not completely) mitigate the negative effects. The crop yield increase is, however, not the only benefit of biochar that can attract farmers. For example, the increase in water holding capacities and decrease in nutrient leaching from arable soils influences management intensities by reducing needs for irrigation and fertilization, and, by favoring more resilient soils, it likely reduces the risks of yield losses due to extreme weather events.

Realization of the high biochar potentials estimated in our analysis requires a massive development of a biochar industry in Europe, with proper supply networks, storage facilities and logistics. Larger plant capacity can secure larger markets for biomass residues, and the use of the existing long-distance transport infrastructure for coal can reduce transportation costs. Making use of the co-generated electricity or heat can also increase profitability, as it will be counted as renewable energy according to the European directive (European Commission, 2018). Smaller scale applications can be designed around district-level network systems that rely on local resources and address specific issues. For example, in Europe there is cropland with high soil erosion rates requiring some actions to prevent irreversible losses of soil functions, or contaminated soils that need remediation, or soils under other forms of environmental stress (e.g., acidity) that can directly benefit from biochar treatment. These local contexts can be used as a test case to start implementing biochar systems where biochar is locally produced from resources available and applied to the field to contrast land degradation, restore soil organic matter and deliver negative emissions. This system can then be gradually expanded, taking advantage of the gains from such practical knowledge. Independently of the scale, integrated environmental, social and economic analysis are essential to identify successful policies and strategies to the production and use of biochar.

5. Conclusions

Biochar deployment across Europe has the potential to achieve large-scale net negative emissions when produced from the available agricultural and forest residues. From a life-cycle perspective, the main factors influencing the results are the type of biomass feedstock, the treatment of biomass co-products, the type of soil receiving the biochar, and the transport distances. Most of the climate benefits come from the long-term storage of carbon in soils, and GHG emissions from the supply-chain are more than one order of magnitude larger than the declines in biochar-induced soil emissions. While delivering negative emissions, biochar from existing residues is sufficient to treat almost all European cropland threatened by soil erosion, and it helps to mitigate its negative consequences by generally increasing crop yield, water holding capacity, and soil nitrogen retention. Reductions in soil emissions of N_2O and nitrogen can also mitigate impacts in ozone layer depletion and marine eutrophication. Trade-offs also occur. Emissions along the supply chain can cause a net increase in all impact categories other than climate change, ODP and MEP, some areas can experience negative effects on yields, and increased soil ammonia volatilization can impact air quality and terrestrial acidification.

This work is a first attempt to quantify the bottom-up and spatially explicit climate mitigation potential of biochar in Europe while considering potential co-benefits or trade-offs, and identify regions where they can occur. Options exist to better optimize biochar properties and induced soil effects, so that different biochar systems can be deployed on local scales to limit (or prevent) trade-offs, maximize co-benefits, and exploit co-products. Regional feasibility studies, policies, incentives and information campaigns are still required to overcome the existing barriers in a sustainable large-scale production and use of biochar.

Data Availability Statement

Gridded data of forest residues availability were obtained from Dr. Hans Verkerk (Verkerk et al., 2019), and those of crop residues from Scarlat et al. (2019). Soil erosion rates were gathered from Borrelli et al. (2017). Modeled biochar effects on crop yields and soils emissions were obtained from Liu et al. (2019), biochar's effect on soil water retention were estimated using the semi-empirical data set from Kroeger et al. (2021). Emission and process inventories for the different life-cycle stages were taken from Tisserant et al. (2022). Gridded results and code for the logistic transport model are available at an open data repository (https://doi.org/10.5281/zenodo.8214764) (Tisserant et al., 2023).

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