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## EDITED BY

Leonard Rusinamhodzi,  
International Institute of Tropical  
Agriculture (IITA), Nigeria

## REVIEWED BY

Ou Sheng,  
Guangdong Academy of Agricultural  
Sciences (GDAAS), China  
Ben Faber,  
University of California, Berkeley,  
United States

## \*CORRESPONDENCE

Liisa Kulmala  
liisa.kulmala@fmi.fi

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# Effects of biochar and ligneous soil amendments on greenhouse gas exchange during extremely dry growing season in a Finnish cropland

Liisa Kulmala<sup>1,2\*</sup>, Kenneth Peltokangas<sup>1,3</sup>, Jussi Heinonsalo<sup>2,4</sup>,  
Mari Pihlatie<sup>3,5</sup>, Tuomas Laurila<sup>1</sup>, Jari Liski<sup>1</sup> and  
Annalea Lohila<sup>1,6</sup>

<sup>1</sup>Finnish Meteorological Institute, Helsinki, Finland, <sup>2</sup>Department of Forest Sciences, Institute for Atmospheric and Earth System Research (INAR), University of Helsinki, Helsinki, Finland, <sup>3</sup>Department of Agricultural Sciences, University of Helsinki, Helsinki, Finland, <sup>4</sup>Department of Microbiology, University of Helsinki, Helsinki, Finland, <sup>5</sup>Department of Agricultural Sciences, Viikki Plant Science Center (VIPS), University of Helsinki, Helsinki, Finland, <sup>6</sup>Department of Physics, Institute for Atmospheric and Earth System Research (INAR), University of Helsinki, Helsinki, Finland

Organic soil amendments such as manure, biochar and compost are among the most efficient and widely used methods to increase soil carbon sequestration in agricultural soils. Even though their benefits are well known, many wood-derived materials are not yet utilized in Nordic agriculture due to a lack of incentives and knowledge of their effects in the local climate. We studied greenhouse gas exchange, plant growth and soil properties of a clay soil cultivated with oat in southern Finland in an extremely dry year. Two years earlier, the field was treated with three ligneous soil amendments—lime-stabilized fiber from the pulp industry, willow biochar and spruce biochar—which we compared against fertilized and non-fertilized controls. We found that the soil amendments increased porosity and the mean soil water holding capacity, which was most noticeable in plots amended with spruce biochar. There was a trend indicating that the mean yield and overall biomass production were larger in plots with soil amendments; however, the difference to unamended control was seldom significant due to the high variance among replicates. Manual chamber measurements revealed that carbon dioxide and methane exchange rates were reduced most probably by the exceptionally hot and dry weather conditions, but no differences could be found between the amended and unamended treatments. The nitrous oxide emissions were significantly smaller from the vegetated soil amended with willow biochar compared with the unamended control. Emissions from non-vegetated soil, representing heterotrophic respiration, were similar but without significant differences between treatments. Overall, the studied soil amendments indicated positive climatic impact two years after their application, but further research is needed to conclusively characterize the specific effects of organic soil amendments on processes affecting greenhouse gas exchange and plant growth.

## KEYWORDS

biochar, crop yield, primary production, soil moisture, climate-smart agriculture, drought

## Introduction

The targets of the Paris Agreement cannot be reached without effective ways to remove carbon dioxide (CO<sub>2</sub>) from the atmosphere (Millar et al., 2017; Minasny et al., 2017). Therefore, intensification of land CO<sub>2</sub> sinks has been suggested as a solution, for example, by managing agricultural soils, increasing afforestation, or restoring degraded soils and ecosystems (Griscom et al., 2017; Roe et al., 2019). Climate-smart management in agriculture often advocates the use of carbon-rich materials as soil amendments or fertilizers to sequester carbon (C) into long lasting forms. This has the potential to mitigate climate change and to support soil fertility (Rosenstock et al., 2016) by increasing soil organic carbon (SOC) content, which promotes soil structure, soil water retention (Rawls et al., 2003; Bronick and Lal, 2005), as well as supporting nutrient cycling and microbial activity (Biederman and Harpole, 2013; Wang et al., 2016).

Biochar is currently considered the most promising option for long-term C sequestration (Lehmann et al., 2006; Woolf et al., 2010; Griscom et al., 2017; Bai et al., 2019; Smith et al., 2020) due to its resistance to decomposition (Wang et al., 2016). However, its effectiveness may be limited in areas of high natural SOC content because of possible C saturation or short-term priming effects, even though the long-term positive effects are often considered to outweigh the negative (Smith, 2005; Zimmerman et al., 2011; Wang et al., 2016). In addition, many of the additional benefits to soil fertility have been observed mainly with coarse textured and highly weathered tropical soils (Jeffery et al., 2017). Moreover, agricultural practices that provide C sequestration may have other climatic impacts, which can be either adverse or beneficial. For example, soil amendments are likely to change soil hydrological and thermal properties (Gupta et al., 1977; Abu-Hamdeh and Reeder, 2000; Rawls et al., 2003; Usovich et al., 2016), while introducing external inputs of organic C into the soil system that may thereby reinforce unwanted microbial activity, possibly enhancing the production of nitrous oxide (N<sub>2</sub>O) or methane (CH<sub>4</sub>). To ensure that the overall effect of these management practices is favorable also in fine textured boreal soils, it is important to quantify the net climatic effect of different soil amendments in local conditions.

Most mineral soils cultivated in southern Finland are lacustrine sediments consisting of fine-textured deposits (Mokma et al., 2000; Ylihalla and Mokma, 2001; Matschullat et al., 2018) with organic matter content of 3–12% (Lemola et al., 2018). These clay soils are often reliant on intensive soil management and annual soil frost to loosen the soil. Moreover, the annual precipitation mostly exceeds the annual evapotranspiration, which together with naturally high water tables make these cultivated soils dependent

on artificial drainage to prevent waterlogging. As a result, the intensive soil management has greatly influenced the soil formation of these young agricultural soils (Yli-Halla et al., 2009), and together with a warming climate, it has contributed to the current decline of SOC observed throughout Finland (Heikkinen et al., 2013). Consequently, implementing organic amendments into the cultivation regime may provide an opportunity to improve soil structure and to convert agricultural soils from current sources of C to sinks (Paustian et al., 2016; Bai et al., 2019; Smith et al., 2020).

In Finland, wood derived side streams i.e., ligneous biomasses are produced in large quantities. For example, pulp and paper industry produces almost as much biomass as wastewater treatment (Tampio et al., 2018) making it promising future source for soil amendments in Finland. Several studies have reported that pulp residuals increase soil C content and subsequently soil water holding capacity, one year after their application (Foley and Cooperband, 2002; Chow et al., 2003). However, Beyer et al. (1997) reported that the effects of pulp sludge on soil organic matter content disappeared after one year. Zibilske et al. (2000) and Price and Voroney (2007) suggested that the positive effects could be maintained through repeated applications. On the other hand, Rasa et al. (2021) found that a single application of pulp sludge had only minor effect on soil C content, but reduced soil erodability, even after four years. The inconsistencies concerning the persistence of pulp derived C and the longevity of its effects on soil properties call for further studies.

Our aim was to assess the climatic impact of ligneous soil amendments produced from economically significant biomasses that have the potential for wider application in Finnish agricultural soils. We had three specific research questions:

- Do the soil amendments affect soil physico-chemical properties in a way that would benefit soil fertility?
- Do the soil amendments affect net primary production (NPP), i.e., growth of biomass?
- Do the soil amendments affect greenhouse gas (GHG) exchange between agricultural soil and atmosphere?

For this purpose, we measured soil properties, biomass growth and the GHG exchange of a clay soil growing oat in southern Finland two years after the application of three different soil amendments and compared the effects of the amendments with fertilized and non-fertilized controls. We also monitored soil and weather conditions, which turned out to be important factors affecting plant growth and soil activity during the measuring campaign.

## Materials and methods

### Site and weather station

The study took place in 2018 at Qvidja farm (60° 17' 44" N 22° 23' 35" E), located in south-western Finland in the larger geographical area called Varsinais-Suomi. The field site was chosen for its rather flat terrain with homogeneous management history and soil properties. The soil texture was classified as clay, with 54% clay, 34% silt, and 12% sand. The World Reference Base classification was Vertic Endogleyic Stagnic Cambisol (clayic). The field site has a long cultivation history, possibly dating back centuries. During recent years, it has been cultivated for wheat (five years), caraway (three years), sugar beet with oilseed rape (two years), and grass (five years). Because of the general homogeneity and young age of cultivated soils in southern Finland (Yli-Halla et al., 2009), the site may be considered representative of a typical cropland for the geographical area.

The mean annual temperature and precipitation, measured at a nearby weather station (Yltöinen, 13 km from Qvidja), equalled 5.4°C and 679 mm, respectively, between 1981 and 2010 (Pirinen et al., 2012). In 2018, the annual mean temperature and precipitation were 6.6°C and 486 mm, respectively. In 2018, there was a shortfall of precipitation in June and July compared with the long-term mean, while temperatures much higher than average were observed especially in May and July (Figure 1).

The weather station at the field site was established on 8 May 2018. Air temperature and humidity were measured with a Humicap HMP155 (Vaisala, Vantaa, Finland). Soil moisture was measured with an ML3 ThetaProbe sensor (Delta-T Devices, Cambridge, UK). The soil temperature was recorded from depths of 0.05, 0.10, and 0.30 m by an IKES Pt100 (Nokeval, Nokia, Finland). Photosynthetically active radiation (PAR) was measured with a PQS PAR sensor (Kipp and Zonen, Delft, the Netherlands). Precipitation was recorded with an OTT Pluvio2 (OTT HydroMet, Kempten, Germany). All measurements were logged by a Vaisala QML201C data collection board (Vaisala, Vantaa, Finland) and averaged to a 30-min resolution.

### Experimental set-up and management

The experiment was launched in autumn 2016 with three replicate blocks (Figure 2) each of size 135 m × 20 m. Each of the blocks contained one randomly chosen plot (9 m × 20 m) for each of the treatments (i.e.,  $N = 3$ ).

This study included five treatments (Figure 2; Table 1): (1) C0 was a control without any soil amendment or fertilization. (2) C80N was a control without any soil amendment, but with yearly NPK fertilization. (3) LimeS (lime-stabilized pulp sludge) was amended with lime-stabilized ligneous (i.e., wood-derived) fiber sludge with yearly NPK fertilization. The fiber

was applied as semi-dry mass and is a commercial product of the Soil Food Oy. For a more comprehensive description of the material, see Rasa et al. (2021). (4) SprB (spruce biochar) was amended with biochar produced through pyrolysis at 450°C from spruce (*Picea abies*) chips with yearly NPK fertilization. (5) WilB (willow biochar) was amended with biochar produced through pyrolysis at 450°C from willow (*Salix* spp.) chips with yearly NPK fertilization. Although, all the soil amendments were produced from ligneous feedstock, they each had distinct physical characteristics in semi-dry form, which made their equal use on dry weight basis difficult. Therefore, the application rates used in this study (Table 1) varied to maintain regulations concerning organic fertilizers and to agree with the rates recommended by the producers of the amendments.

After spreading the soil amendments manually in 2016, all plots were tilled with a cultivator (Horsch Terrano 3.5 FX) down to 0.10 m. In the following years, the soil management was kept within the top 0.10 m to contain the effect of the soil amendments within a limited soil layer.

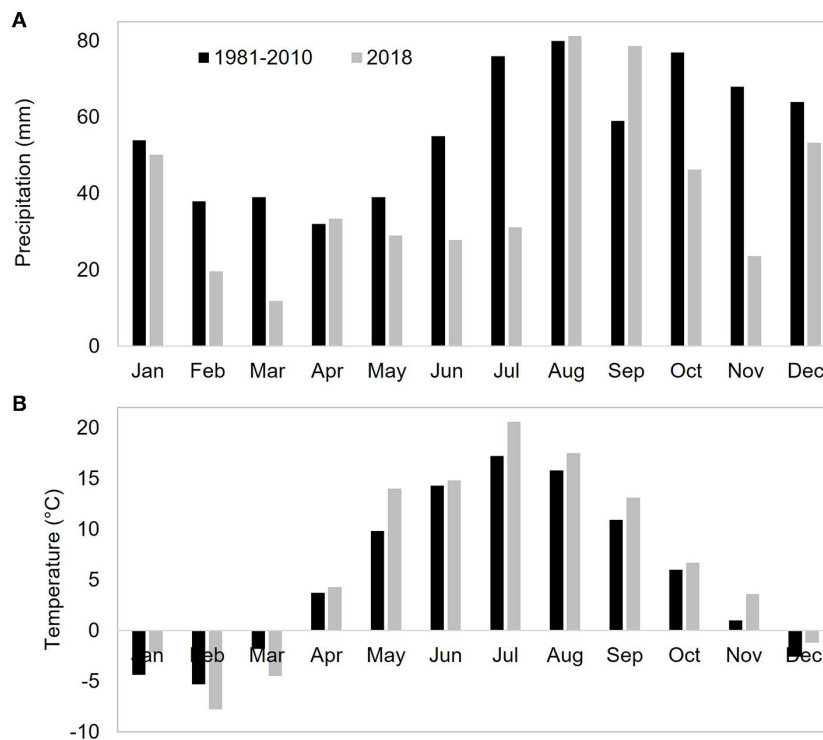
In spring 2017, the field was sown with wheat and all plots except C0 were fertilized using NPK fertilizer (Yara Mila 3, 23-3-8) with 80 kg N ha<sup>-1</sup>. All other practices were equal among the plots in 2017. In May 2018, the field was first prepared down to 0.05 m, and on 15 May it was sown with oat (variety Matty) using a seeder (Överum Tive CD1830), resulting in, on average, 437 germinated seeds m<sup>-2</sup>. At the same time, all plots except for C0 were fertilized using NPK fertilizer (Yara Mila 3, 23-3-8) with 80 kg N ha<sup>-1</sup>. Weeds were manually removed on 13 June. Due to extremely dry weather (Figure 1), the field was irrigated with ~40–50 mm of water during a period of 18 h on 30 June by using 11 rotary sprinklers per replicate block. Crop samples were collected with a 1.5 m wide field plot harvester (Wintersteiger, Ried, Austria) from each plot on 22 August after which the whole field was harvested and straws collected.

### Soil moisture and temperature measurements

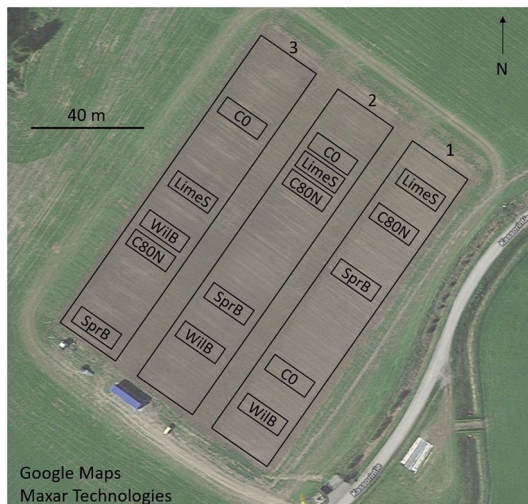
The soil temperature was measured using HOBO Pendant Temperature Loggers (UA-001-64, Onset, Bourne, MA, USA), which were installed in each plot at a depth of 0.05 m. The soil surface moisture was measured weekly from ~0.05 m with a ML3 ThetaProbe (Delta-T Devices) by manually inserting the probe into six different locations in each plot. We used the average of these measurements in the further analysis. The daily soil moisture,  $M$ , was linearly interpolated.

### Soil properties

The soil water holding properties were determined from 90 undisturbed soil cores ( $d = 73$  mm,  $h = 48$  mm,  $V = 0.20$  L).



**FIGURE 1** Monthly average precipitation (A) and temperature (B) recorded at a nearby weather station, Yltöinen, in 1980–2010 (black bars, Pirinen et al., 2012) and in 2018 (gray bars).



**FIGURE 2** Experimental set-up with three replicate blocks (1–3) each having five plots, one for each treatment: CO, non-fertilized control; C80N, fertilized control; LimeS, lime-stabilized pulp sludge; SprB, spruce biochar; WiIB, willow biochar.

Six soil cores were taken from each of the 15 sampled plots: three from the soil surface (0–0.05 m) and three below the managed soil layer (0.20–0.25 m). After sampling, the cores were sealed and transported to the University of Helsinki soil science laboratory, where they were stored at 4°C before determining their soil water retention curves (SWRCs) starting in January 2019.

Each SWRC was plotted using measurements of gravimetric water content (m/m) at 0, –0.3, –6.0, –250 and –1500 kPa (Dane and Hopmans, 2002). The pressure range from 0 to –6.0 kPa was determined using the kaolin sandbox method (Eijkelkamp Agrisearch Equipment, the Netherlands) and thereafter to –1500 kPa using pressure plate extractors (Soilmoisture Equipment Corp., Santa Barbara, CA, USA) connected to a compressor (Kaeser Kompressoren, Coburg, Germany) via a pressure manifold (Soilmoisture Equipment Corp.). Plant available water (PAW) was defined as the water holding capacity at a pressure range of between –6.0 and –1500 kPa. Finally, the samples were dried at 105°C to determine their dry bulk density ( $\rho_{bd}$ ).

Total soil C ( $C_{tot}$ ) was determined from soil samples gathered alongside with the soil cores. The soil was first air-

TABLE 1 Experimental treatments including their application rates in 2016.

Treatment Abbreviation	Non-fertilized control C0	Fertilized control C80N	Lime-stabilized pulp sludge LimeS	Spruce biochar SprB	Willow biochar WilB
Input	–	–	24,200	20,600	33,400
C input	–	–	3,250	18,999	18,016
N input	–	–	90	86	373
Dissolved N input	–	–	0.27	0.10	0.31
Ca input	–	–	689	217	377
Mg input	–	–	8.0	18	48
K input	–	–	2.8	64	211
Na input	–	–	7.9	1.2	14
pH	–	–	8.9	8.3	9.8
EC	–	–	17	9.4	30

The total inputs of C, carbon; N, nitrogen, dissolved N; Ca, calcium; Mg, magnesium; K, potassium; and Na, sodium; as well as the pH and EC, electrical conductivity of the produces. The input rates are in kg ha<sup>-1</sup>.

dried and then crushed with pestle and mortar before being analyzed with varioMAX CN analyser (Elementar Company, Langensfeld, Germany) using four analytical replicates ( $N = 4$ ). Because the soil was assumed to contain negligible amount of carbonates, total soil C was assumed to equal SOC content (i.e.,  $C_{tot} = SOC$ ).

The effects of organic soil amendments on soil structure were determined by comparing them to the unamended control. Additionally, effect of spatial heterogeneity was considered by comparing the amended soil layer to the underlying unamended soil layer as well as comparing the three replicate blocks against each other using a two-way analysis of variance (ANOVA) with IBM SPSS Statistics 25.

## Vegetation growth

In 2018, plant height was measured in each plot weekly from five individual oat plants using a ruler. Above- and below-ground biomass was estimated using a systematic inventory on 1 August 2018. A total of three above-ground samples and five below-ground root samples were collected per plot. The above-ground biomass was cut ( $\phi = 0.20$ ,  $m = 0.0314$  m<sup>2</sup>) at a height of 0.02 m, whereas underground root samples were taken with a soil auger ( $\phi = 0.05$  m) down to 0.20 m. Three of the five root samples were taken on top of the plant rows and two in the middle of the rows. The above-ground biomass was frozen ( $-18^{\circ}\text{C}$ ) for 2–3 months before separating it into oat seeds, oat above-ground biomass (stems and leaves), and other biomass (weeds). Samples were then dried for 48 h at  $60^{\circ}\text{C}$  and weighed to determine their dry mass. The root samples were stored in a fridge ( $+6^{\circ}\text{C}$ ) for 3–7 days before soaking them in water for approximately 30 min and then manually separating the roots from bulk soil. The root biomass was then dried for 48 h at  $60^{\circ}\text{C}$

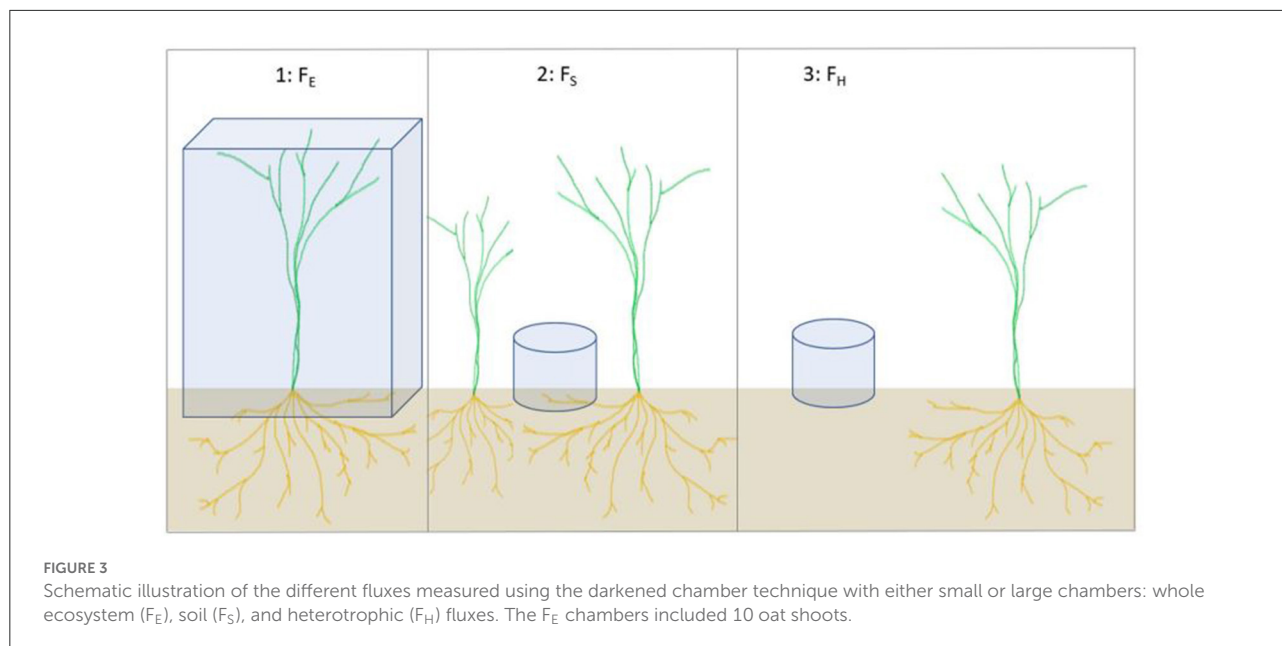
and weighed before determining loss on ignition (2 h at  $550^{\circ}\text{C}$ ). During the sampling, the seeds were still developing, whereas other biomass compartments had reached maturity. Thus, we ignored the seed biomass at this stage and used biomass yield at the time of the harvest in the further analyses. The accumulation of plant biomass measured from treatments C0, C80, SprB, and WilB was described by Kalu et al. (2022). In this article, the data is reanalyzed from the perspective of carbon balance and greenhouse gas exchange. Moreover, the data was expanded to include application of LimeS as one of the treatments.

We compared the different biomass factors as well as the final height of the oat plants using a two-way ANOVA with the treatment and the replicate block as factors in R (version 3.6.1, R Core Team, 2019). We used the Shapiro–Wilk test prior to the analysis for testing normality ( $p > 0.05$ ).

## Greenhouse gas flux measurements

### Component fluxes

We used three variations of the darkened chamber technique to measure  $\text{CO}_2$ ,  $\text{N}_2\text{O}$ , and  $\text{CH}_4$  exchange: (1) the whole ecosystem flux ( $F_E$ ) was measured during the growing season, between sowing and harvesting, using large aluminum collars installed on vegetated surface; (2) the soil flux ( $F_S$ ) was measured using smaller collars installed on surfaces without above-ground shoots but with roots presumably growing under the collars; (3) the heterotrophic flux ( $F_H$ ) was measured using large collars before sowing and after tilling, but with smaller collars during the growing season, both installed on bare soil (Figure 3, Supplementary Figures 1A,B). The non-vegetated soil was kept bare throughout the growing season by removing all growing plants manually around the collars, leaving a space of bare soil ( $p = 0.50$  m) around the collars that was used to measure  $F_H$ . In



addition, all vegetation was removed from within the collars for both  $F_S$  and  $F_H$ .

### Whole ecosystem flux, $F_E$

$F_E$  was measured six times during the growing season, which lasted from 23 May to 14 August 2018. For the measurements, two large collars were installed in each plot, which had a surface area of  $0.60\text{ m} \times 0.60\text{ m}$  and rose to  $\sim 0.10\text{ m}$  above the soil surface. During sampling, these collars were mounted with white aluminum chambers  $0.80\text{ m}$  in height with an inbuilt fan and a HOBO temperature logger.

Sampling was conducted by withdrawing  $20\text{ mL}$  of air with a syringe every 1, 10, 20, 30, and 40 min after closing the chamber and inserting the gas sample in  $10\text{ mL}$  glass vials (Labco Exetainer, Lampeter, Ceredigion, UK). The gas samples were analyzed using a gas chromatograph (7890A, Agilent Technologies, Santa Clara, CA, USA) equipped with a flame ionization detector and a methanizer for  $\text{CO}_2$  and  $\text{CH}_4$ , and an electron capture detector for  $\text{N}_2\text{O}$  (Pihlatie et al., 2013). Kalu et al. (2022) described the  $F_E$  data measured from C0, C80, SprB, and WilB. This article reanalyzes the data, but with the addition of LimeS.

### Heterotrophic and soil fluxes, $F_H$ and $F_S$

The GHG fluxes were first measured on 8–9 May 2018. As this was done from bare soil before sowing, the collected data represented  $F_H$ . These measurements were conducted by placing an opaque chamber ( $0.20\text{ m}$  in height) on top of the

large collars, as described previously. The GHG concentrations inside the closed chamber were then measured with a Gasetm Fourier transform infrared trace gas analyser DX4015 (Gasetm Technologies, Helsinki, Finland) by sucking air out and returning it back to the chamber with the flow rate of  $2\text{ L min}^{-1}$ . The chamber also included a fan rotating at low speed to stir the air within the chamber. The measurement period was 5 min and therefore the increase in temperature was minor during the closure. The measurement was repeated three times at three different locations in each plot.

After the field was sown, six small collars ( $\phi = 0.20\text{ m}$ ) were permanently installed in each plot by inserting them to a depth of  $\sim 30\text{--}70\text{ mm}$ , which left  $\sim 80\text{--}120\text{ mm}$  above ground. Half of the collars were treated as  $F_S$  and half as  $F_H$ . The collars were removed before the harvest and afterwards placed in the middle of the harvested plots, which still included the oat stubble (Supplementary Figure 1C). Thereafter, flux measurements were carried out one to four times per month by closing the collars one by one with opaque Plexiglas and measuring the GHG concentrations inside the small air space with the Gasetm analyzer in a similar manner as with the large collars. Each measurement event lasted 3 min to avoid heating the air within the closed chamber. The small volume of chambers compared with the flowrate allowed efficient mixing of the chamber head space during these measurements due to lack of vegetation or other obstacles inside.

After plowing the soil in October 2018, we measured the GHG fluxes once during October and once during November from all treatments, except LimeS, using the large ( $0.60\text{ m} \times 0.60\text{ m}$ ) collars together with the opaque chamber. The shallower chambers were omitted because the soil surface was too rough

for their installation. The closure time was again 5 min. In further analyses, the GHG fluxes after the harvest were handled as  $F_H$  due to a lack of notable living plant biomass.

## Flux analyses

Prior to further data analysis, the development of  $CO_2$  concentration during each measurement was evaluated visually. In the  $F_S$  and  $F_H$  measurements, the increase in the  $CO_2$  concentration was observed to be unstable in 35 out of 1062 closures, indicating a leakage or other misbehavior of the measurement system. Those measurement events, including simultaneous  $CH_4$  and  $N_2O$  fluxes, were then excluded from further analyses. The remaining data were used to calculate fluxes by linear fitting over GHG concentrations measured 30–150 s after the closure.

We analyzed all gases separately, but the data were also aggregated by pooling all GHG fluxes together after transforming both  $CH_4$  and  $N_2O$  to  $CO_2$ eq according to their global warming potential using coefficients 25 and 298 for  $CH_4$  and  $N_2O$ , respectively (IPCC, 2007).

Prior to other statistical analyses, all GHG variables were tested for normality ( $p > 0.05$ ) using the Shapiro–Wilk test. Furthermore, both the  $N_2O$  and  $CO_2$  fluxes were subjected to logarithmic transformation in order to meet the normality requirements. For the  $CH_4$  fluxes, we removed outliers that were defined as data points that are located 1.5 times the interquartile range above the upper quartile and below the lower quartile. In practice, we used the boxplot function to remove 7 fluxes out of 139 as outliers before the analysis. The means of  $F_H$  in each plot were then compared among treatments using a two-way ANOVA with the treatment and the replicate block as factors in R (version 3.6.1, R Core Team, 2019), whereas with  $F_E$ , we used a two-way ANOVA with the treatment and date as factors.

## Carbon balance

We calculated the carbon gain ( $C_{gain}$ ) to the ecosystem from the atmosphere for each plot  $i$  as the difference between NPP ( $g\ C\ m^{-2}$ ) and heterotrophic respiration (RH,  $g\ C\ m^{-2}$ ), as follows

$$C_{gain} = NPP_i - RH_i \quad (1)$$

We used the biomass production as NPP calculated by summing the biomass of roots, yield, and other above-ground biomass including weeds and by assuming the mean carbon content of dry biomass to be 50%.

We estimated RH using the momentary  $CO_2$  emissions measurements from non-vegetated soil ( $F_H$ ). First, we fitted for each plot  $i$  an empirical equation describing the temperature and

moisture dependency, as follows:

$$rh_i(t) = \left( 1 + \left( \frac{1 - RWC_i(t)}{\alpha} \right)^{\nu} \right)^{-1} r_{0,i} Q_{10,i}^{0.1T(t)} \quad (2)$$

where,  $t$  is day,  $T$  is daily mean soil temperature at 100 mm depth and  $\alpha$ ,  $r_0$ , and  $Q_{10}$  are parameters obtained from the fitting. Measurements from all three collars in each plot were used to estimate the parameters for the plot. For  $\nu$ , we used a constant 11.27 according to Mäkelä et al. (2008). RWC is relative water content, calculated from the measured water content ( $M$ ,  $m^3\ m^{-3}$ ) in the topmost soil layer, as

$$RWC_i(t) = \frac{M_i(t) - WP}{FC - WP} \quad (3)$$

where, WP is the mean wilting point ( $0.15\ m^3\ m^{-3}$ ) and FC denotes the field capacity for the topmost soil layer ( $0.28\ m^3\ m^{-3}$ ) estimated from the measurements (Figure 4C). The daily  $rh_i$  values were then estimated using the measured soil moisture and temperature. Finally, the daily  $rh_i$  values were summed up to estimate heterotrophic respiration in each plot for the whole measurement season (197 days, 8 May–20 November),  $RH_i$  ( $g\ C\ m^{-2}$ ), as follows

$$RH_i = \sum_{t=1}^{197} rh_i(t) \quad (4)$$

In addition to the  $C_{gain}$  from the atmosphere, we calculated the change in the ecosystem carbon storage ( $\Delta C_{soil}$ ,  $g\ C\ m^{-2}$ ) as the difference between all the inputs and outputs, as follows

$$\Delta C_{soil} = NPP - RH - Y \quad (5)$$

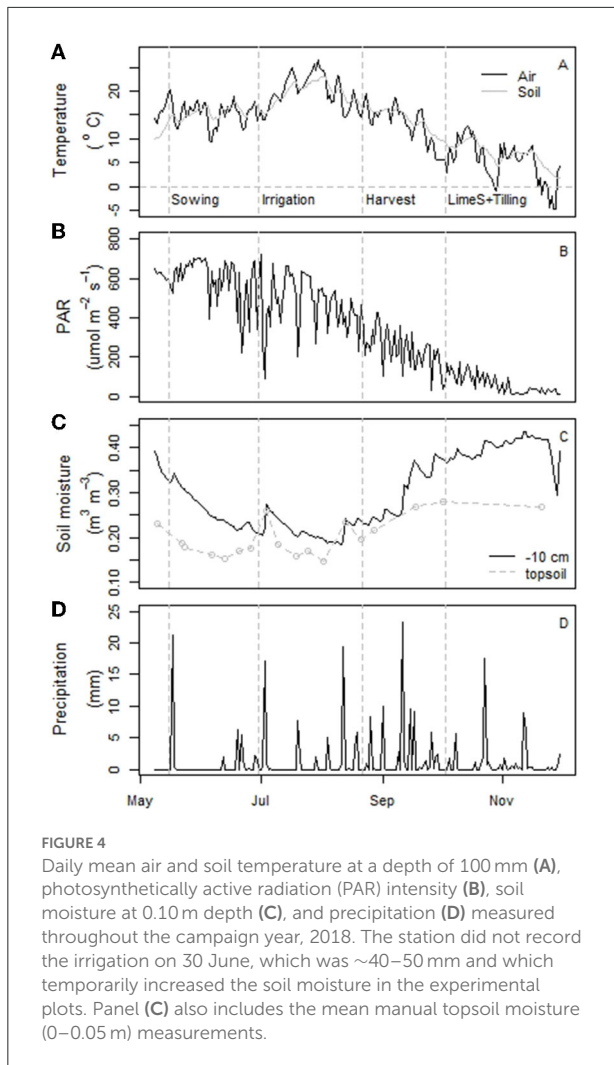
where,  $Y$  is the yield ( $g\ C\ m^{-2}$ ).

We compared  $C_{gain}$ ,  $\Delta C_{soil}$ , RH, and NPP using a two-way ANOVA with the treatment and the replicate block as factors in R. We used the Shapiro–Wilk test prior to the analysis for testing normality ( $p > 0.05$ ). In addition, we pooled all plots with amendments (LimeS, SprB, WilB) and compared those with the unamended ones (C0, C80N) using an independent 2-group  $t$ -test in R.

## Results

### General overview of the weather

In 2018, the snow melted at the study site in early April, which was still a relatively cool month before a more rapid temperature increase in early May. Thereafter, the site experienced very low precipitation in May, June, and July together with sunny weather and relatively high



mean air temperatures (Figures 4A,B,D). The precipitation in June–August was 104 mm in Qvidja, i.e., even less than in Yltöinen (140 mm) that year and approximately half of the mean precipitation measured there during 1980–2010 (210 mm). This resulted in the soil drying out especially in late June and the first half of August (Figure 4C). The moisture content measured manually from the topsoil of each plot did not differ between treatments (not shown). Moreover, the seasonal changes observed in the manual measurements equaled the automatic measurements even though the overall moisture close to the soil surface (manual data) was somewhat less than in the deeper soil layer (automatic data) (Figure 4C).

During the dry and warm latter half of July, the control treatment C0 showed slightly higher soil temperatures than the other treatments (Supplementary Figure 2A). The soil temperature in replicate block 1 was 1–2 degrees less than that measured at the other blocks during June–July, but the differences between blocks evened out after harvest (Supplementary Figure 2B).

## Soil structure and soil organic carbon

Overall, only WilB exhibited significant changes to total porosity ( $p = 0.034$ ) and bulk density ( $\rho_{\text{bd}}$ ) ( $p < 0.05$ ), having significantly denser soil structure compared to the unamended (C80N) control (Table 2). However, SprB did exhibit ~16% larger PAW compared to C80N ( $p = 0.023$ ) whereas LimeS and WilB tended to increase PAW by 9 and 10% on average. The increases in PAW had a strong positive correlation with the increase in SOC ( $R^2 = 0.72$ ) whereby SprB, WilB, LimeS, and C80N had SOC contents of 3.54% ( $\pm 0.03$ ), 2.97% ( $\pm 0.02$ ), 2.62% ( $\pm 0.01$ ), and 2.49% ( $\pm 0.02$ ), respectively. Moreover, the SprB, WilB, and LimeS amended soil layer exhibited 10, 24, and 11% larger PAW compared to the deeper soil layer, whereas, with C80N, the difference between the two soil layers was negligible. This was in addition to the general trend, whereby the deeper soil layer had less large pores ( $>30 \mu\text{m}$ ), and denser soil structure ( $p < 0.05$ ) than the managed top soil layer. In addition, replicate block 1 exhibited significantly larger PAW ( $p < 0.05$ ) when both the amended surface soil layer (0–0.05 m) as well as the deeper unamended soil layer (0.20–0.25 m) were included in the analysis. There was also a gradual increase in  $\rho_{\text{bd}}$  from one replicate block to the next (Table 2), whereby replicate block 3 had significantly denser structure ( $p < 0.05$ ).

## Plant biomass and crop height

The height growth dynamics were generally similar among different treatments. However, the growth rate in SprB was increased in the later stages of development, whereas the non-fertilized control treatment (C0) lagged compared to the other treatments throughout the growing period (Figure 5). Consequently, SprB ended up having the tallest vegetation, whereas C0 had the shortest. Nevertheless, there were no statistical differences mainly due to the great variation between the different replicate blocks. The total mean plant biomass ( $\pm$  standard deviation) was 664 ( $\pm 364$ ), 744 ( $\pm 266$ ), 925 ( $\pm 252$ ), 964 ( $\pm 265$ ), and 1161 ( $\pm 210$ ) g dry mass  $\text{m}^{-2}$  in C0, C80N, LimeS, WilB, and SprB, respectively (Figure 6). In general, the largest biomass fractions and yield were recorded in the first replicate block and the smallest in the third block (not shown). The fraction consisting of leaves and stem was significantly larger in SprB than in the control treatments C0 or C80N. Other than this, none of the other biomass fractions nor the root/shoot ratio differed significantly between treatments due to large variation between the replicate blocks.

## Gas exchange from bare soil

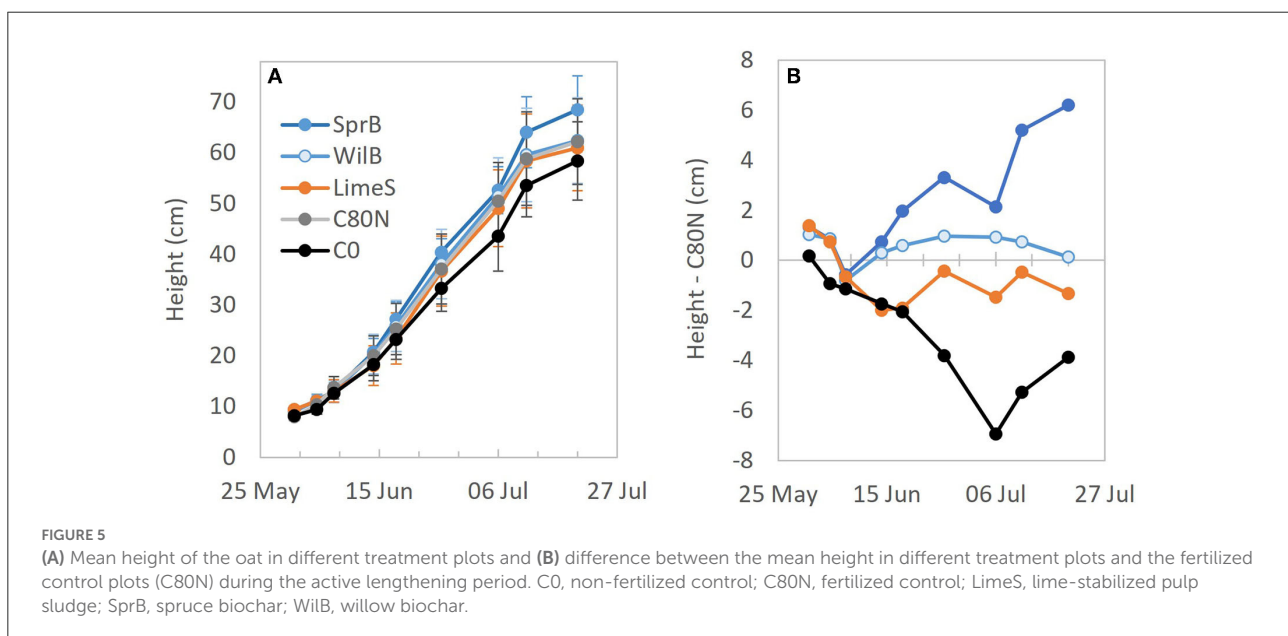
The  $F_{\text{H}}$  of  $\text{CO}_2$  peaked in mid-May, right after sowing, and again in August–September, and was smaller during the dry period in June–July (Figure 7A). Increases in soil



TABLE 2 Mean ( $\pm$  standard deviation) soil bulk density ( $\rho_{bd}$ ,  $\text{g cm}^{-3}$ ) and plant available water (PAW,  $\text{g g}^{-1}$ ) displayed as means of the three replicate blocks ( $N = 15$ ) and treatments ( $N = 9$ ) for the amended soil layer 0–0.05 m and deeper unamended soil layer 0.20–0.25 m.

		Soil layer 0–0.05 m		Soil layer 0.20–0.25 m	
		$\rho_{bd}$	PAW	$\rho_{bd}$	PAW
<b>Block</b>	1	$1.16 \pm 0.10^a$	$24.9 \pm 2.22^a$	$1.30 \pm 0.09^a$	$23.8 \pm 2.22^a$
	2	$1.18 \pm 0.06^{ab}$	$22.1 \pm 2.15^b$	$1.31 \pm 0.07^a$	$21.5 \pm 3.56^{ab}$
	3	$1.24 \pm 0.05^b$	$23.5 \pm 2.55^{ab}$	$1.36 \pm 0.08^a$	$20.1 \pm 2.09^b$
<b>Treatment</b>	C80N	$1.18 \pm 0.08^a$	$21.9 \pm 2.18^a$	$1.27 \pm 0.06^a$	$21.8 \pm 3.54^a$
	LimeS	$1.17 \pm 0.10^a$	$24.0 \pm 1.38^{ab}$	$1.33 \pm 0.11^a$	$22.1 \pm 4.00^a$
	WilB	$1.27 \pm 0.08^a$	$24.1 \pm 2.29^{ab}$	$1.40 \pm 0.05^b$	$19.4 \pm 1.21^a$
	SprB	$1.16 \pm 0.06^a$	$25.4 \pm 2.96^b$	$1.33 \pm 0.05^a$	$22.8 \pm 4.77^a$

The letters indicate significant differences between the treatments or replicate blocks. C80N, fertilized control; LimeS, lime-stabilized pulp sludge; SprB, spruce biochar; WilB, willow biochar.



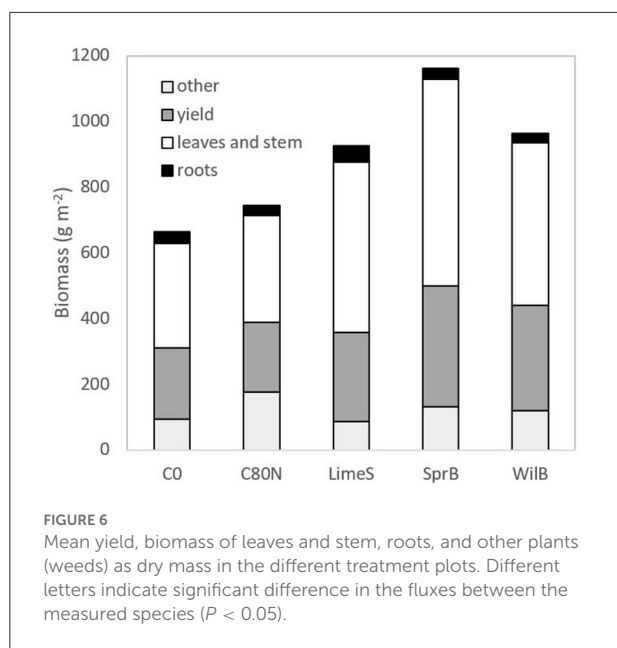
moisture, due to irrigation and precipitation along with a rather minor rain event on 20–21 June, led to a minor increase in heterotrophic  $\text{CO}_2$  fluxes in early July. The largest mean  $F_H$  was measured from plots treated with LimeS and SprB (Figure 7B), but the differences between the treatments were not statistically significant.  $F_H$  of  $\text{CO}_2$  was generally smaller than  $F_S$  from bare soil with surrounding plants, except on 31 July when the mean  $F_H$  clearly exceeded  $F_S$  in all treatments (Supplementary Figure 5).

All treatments in the bare soil acted as  $\text{CH}_4$  sources in mid-May, whereas later in the season the  $\text{CH}_4$  exchange was close to zero (Figure 7C). The mean  $\text{CH}_4$  exchange was comparable between the treatments (Figure 7D).  $\text{N}_2\text{O}$  emissions from bare soil fluctuated during the season showing highest peaks in late July, especially for the fertilized control (C80N) and LimeS treatments (Figure 7E). These

treatments showed the largest mean  $\text{N}_2\text{O}$  emissions during the whole measurement period, whereas WilB showed the smallest (Figure 7F). When all GHGs were summed up as  $\text{CO}_2\text{eq}$ , LimeS showed the highest mean emissions and WilB the lowest (Figure 7H). Again, there were no statistically significant differences between treatments. There were no clear differences between  $F_H$  and  $F_S$  considering  $\text{N}_2\text{O}$  and  $\text{CH}_4$  (Supplementary Figures 4, 5).

## Gas exchange from vegetated soil ( $F_E$ )

The  $\text{CO}_2$  fluxes were several times larger and  $\text{CH}_4$  fluxes lower from the vegetated soil compared to the non-vegetated soil (Figures 7, 8). The largest  $\text{CO}_2$  emissions from vegetated soil ( $F_E$ ) were measured in July and the smallest in early June



(Figure 8A). Overall, the only recorded statistically significant difference in  $\text{CO}_2$  fluxes from the vegetated soil was between LimeS and SprB (Figure 8B). During the measurement period, the experimental plots fluctuated between small sinks and small sources of  $\text{CH}_4$  (Figure 8C). WilB had the largest  $\text{CH}_4$  sink but due to large variation, there were no statistical differences between the treatments (Figure 8D). Some of the C80N plots exhibited occasional peaks of  $\text{N}_2\text{O}$ , especially in late July and early August (Figure 8E). Overall, C80N and LimeS exhibited significantly greater  $\text{N}_2\text{O}$  emissions than WilB, but otherwise there were no significant differences between treatments (Figure 8F). Furthermore, there were no significant differences between the treatments when all GHGs were considered together as  $\text{CO}_2\text{eq}$  (not shown).

## Carbon balance spanning over the growing season

The control treatment (C0) had the smallest, and SprB the largest, mean cumulative RH over the measurement campaign (8 May–20 November 2018, Figure 9A). Also, the  $C_{\text{gain}}$  represented by the difference between NPP and RH was most negative with C0 and the most positive with SprB (Figure 9B). In general, plots with soil amendments acted as a sink of atmospheric  $\text{CO}_2$ , whereas the plots without amendments had negative median values, indicating that they were sources of atmospheric  $\text{CO}_2$  during the measurement campaign (Figure 9C). However, the soil carbon pool decreased in all treatment plots when accounting for the C loss in the form of yield ( $\Delta C_{\text{soil}}$ , Figure 9C). Nevertheless, there were

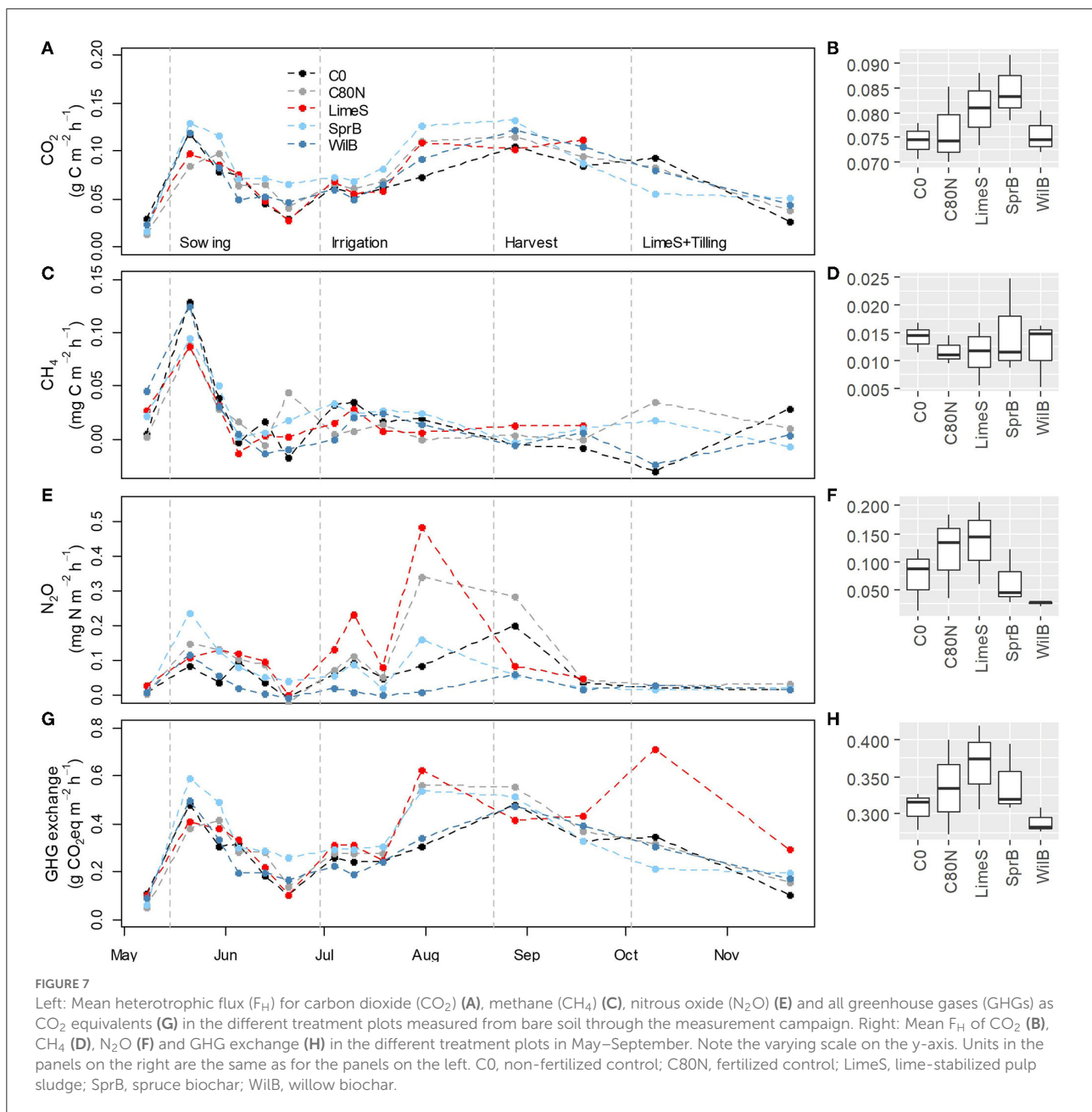
no significant differences between the treatments due to high variation between the replicates. Moreover, when all plots treated with soil amendments were pooled and compared against the unamended plots, the differences between RH,  $C_{\text{gain}}$ , and  $\Delta C_{\text{soil}}$  values were close, but still not different, with  $p$ -values of 0.055, 0.061, and 0.060, respectively.

## Discussion

There is an urgent need to find management practices that decrease the negative climatic impact of traditional Nordic agriculture. Recent international studies with organic soil amendments have shown that they have generally favorable climatic impacts (Woolf et al., 2010; Griscom et al., 2017; Bai et al., 2019; Smith et al., 2020). However, studies in Finland have demonstrated only encouraging trends and many effects have remained statistically insignificant (e.g., Karhu et al., 2011; Soinne et al., 2020; Kalu et al., 2021). Our results continue this trend and show that the studied ligneous soil amendments had only moderate effects on plant biomass, soil water holding capacity and GHG exchange. However, the possible reasons for the lack of more significant differences arise from the low number of replicate blocks (three) accompanied by high variation in the studied variables due to possible spatial heterogeneity in the field. In addition, the campaign summer of 2018 was exceptionally warm and dry, which hindered both plant growth and soil microbial activity. This assumption is supported by the study by Heimsch et al. (2020) whose  $\text{CO}_2$  exchange results from a nearby grassland at Qvidja indicated that the carbon uptake was clearly hindered by the dry and warm weather in 2018. Furthermore, the mean oat yield in 2018 ( $2.8 \text{ t ha}^{-1}$ ) was  $\sim 64\%$  of the normal oat yield recorded in Varsinais-Suomi (Luke, 2020).

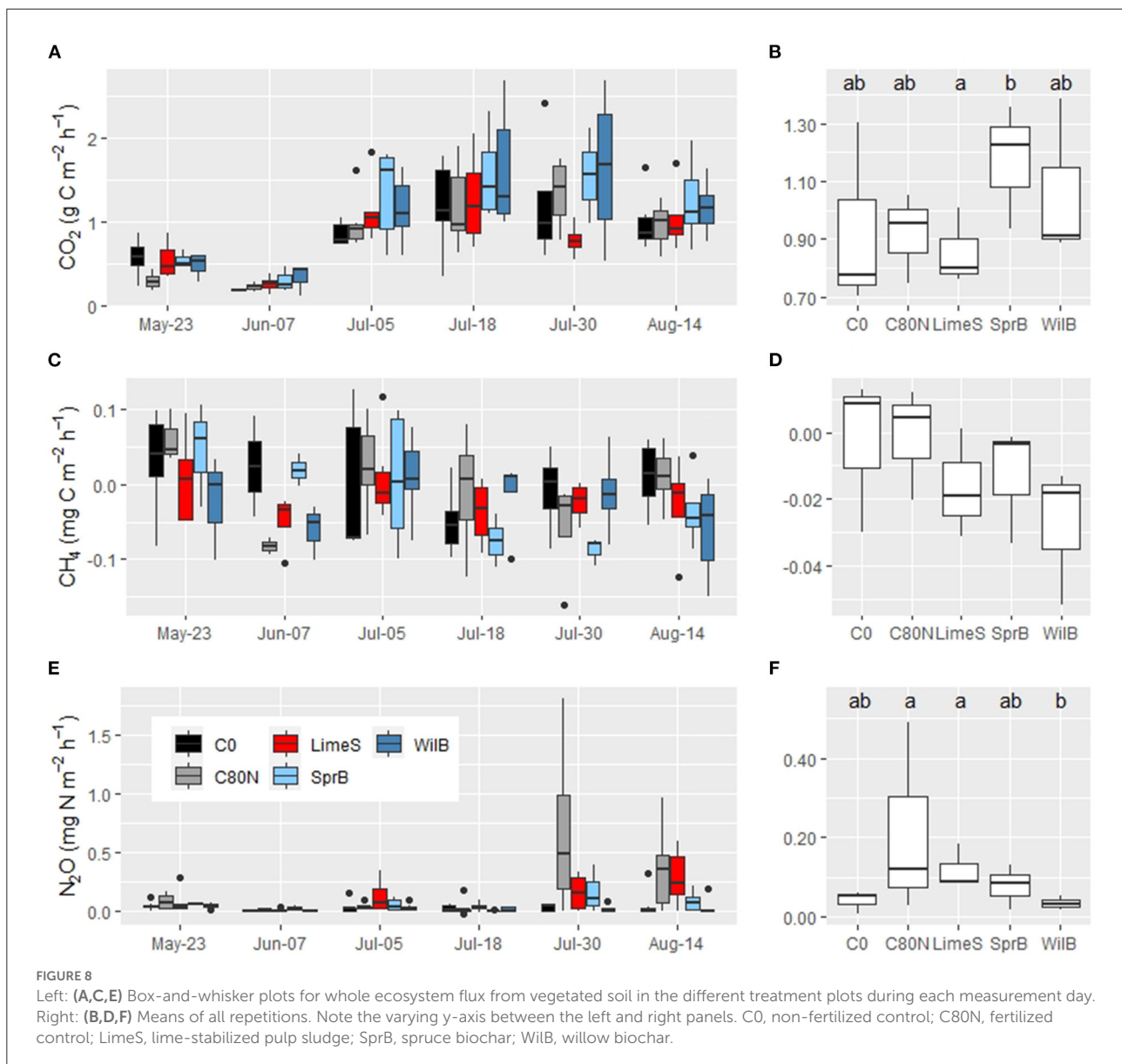
## Soil physico-chemical properties and plant growth

Our first research question asked whether the soil amendments improved soil water holding capacity. Despite the natural, spatial differences in soil properties, the soil amendments were observed to benefit the soil structure. The benefits were most notable with SprB, which reduced the proportion of small pores and increased the number of large pores, resulting in an overall increase in PAW. This was attributed to increase in SOC, as there was a strong positive correlation between the increase of SOC and the increase of PAW. This indicates that the increase in SOC contributed to the increased water retention, as suggested by Foley and Cooperband (2002). Our conclusion is supported by the fact that PAW was increased only by the amendment treatments and not by the control treatment as illustrated



by the comparison of the top soil layer (0–0.05 m) and the corresponding deeper unamended soil layer (0.20–0.25 m). Therefore, it is unlikely that soil management alone was the sole contributor to improving soil structure and water retention. It is also noteworthy that despite the higher response, the initial input of spruce biochar e.g., in total biomass and in dissolved nitrogen was lower than that of willow (Table 1) highlighting that other characteristics than just the amount applied to soil, play a key role degerming the impacts of soil amendments. Here, for example the stability of the amendment might have played a role as the SOC content was the highest in the spruce biochar treatment.

Our second research question asked whether the soil amendments improved net primary production. In our study, yield and total biomass were on average 28–75 and 24–56% higher in the amended plots than in the fertilized control plots, respectively. Additionally, some differences in root biomass tie in with the soil properties. By chance, all three replicate plots amended with WilB were in the south to south-west side of the field, which suffered from compaction as indicated by the increased bulk density. Consequently, WilB had the least root biomass, while the plots amended with LimeS were located on the opposite side of the field and had the most abundant root biomass. Indeed, there was a negative but insignificant

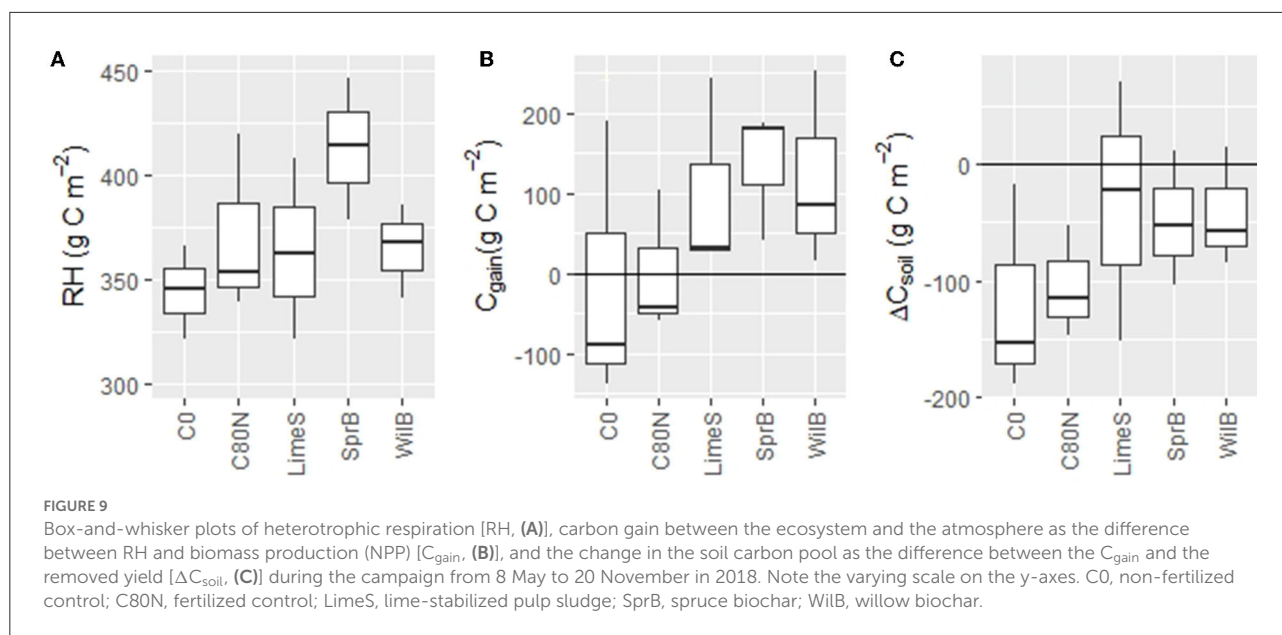


relationship between the root biomass and the bulk density (not shown). Moreover, the differences in biomass growth between the replicate blocks coincided with the soil moisture properties and soil temperatures. Greater water holding capacity at replicate block 1 may partially explain the more abundant plant biomass, whereas taller vegetation as well as greater water content likely contributed to lesser soil temperatures (Rodskjer et al., 1989; Song et al., 2013). However, overall the only significant differences were found in the fraction of leaves and stem, which was larger in spruce biochar than in the control treatments. Therefore, our results are in line with extensive meta-analyses (Biederman and Harpole, 2013; Liu et al., 2013) that in Nordic case studies biochar application has yet to significantly increase biomass or yield (Tammeorg et al.,

2014a,b; Soinne et al., 2020). Furthermore, our study reiterates the conclusion by Hakojärvi et al. (2013) that soil parameters alone are rarely enough to predict yield in case of high variation. Still, we conclude that the soil amendments provide long-term benefits to plant growth through improved soil conditions even if the benefits are not reflected directly by increases in yield (Phillips et al., 1997).

## Greenhouse gas exchange

Our third research question examined whether soil amendments affect the GHG exchange between the ecosystem and the atmosphere. Our overall conclusion is that the CO<sub>2</sub>



emissions from amended soils equaled, or at times exceeded, the unamended control, which we propose is a positive trend as it indicates that the treatments may have occasionally supported a more robust plant and soil activity during the unfavorable weather conditions. However, the CO<sub>2</sub> fluxes were inconsistent and the most pronounced heterotrophic CO<sub>2</sub> emissions were observed with LimeS and SprB, whereas the whole ecosystem CO<sub>2</sub> fluxes from LimeS were comparable with unamended treatments. In addition, the CO<sub>2</sub> fluxes from vegetated soil were many times larger than heterotrophic respiration, which is natural but may also be partly explained by the difference in the measurement method (Kohl et al., 2019). On the other hand, previous reports on GHG emissions in Finland show somewhat smaller overall CO<sub>2</sub> emissions from similar croplands located in southern Finland (Lohila et al., 2003; Regina and Alakukku, 2010; Karhu et al., 2011). We used linear fitting in the flux calculations even though it is criticized that the gas exchange must decrease during long closures as the concentration gradient between the chamber headspace and soil air decreases (Kutzbach et al., 2007). However, an exponential function would produce larger estimates than the linear regression function (Forbrich et al., 2010), whereas the latter is more stable overall, which we considered more fitting for examining treatment effects.

Significant CH<sub>4</sub> emissions require anaerobic conditions and the build-up of CO<sub>2</sub>, which often develops only in water-saturated conditions (Conrad, 2007; Van Groenigen et al., 2011). Because of the low energy yield of methanogenesis, CH<sub>4</sub> emissions increase slowly and only after more favorable options have been exhausted (Lovley and Phillips, 1987). Therefore, it is unsurprising that CH<sub>4</sub> emissions were predominantly measured in May when the soil moisture was still high after the winter frost and melting snow. In addition, the sowing

and tilling practices may have disturbed the soil, causing an outflow of accumulated gases from the soil system. A slight peak in CH<sub>4</sub> emissions was also recorded in July, after the soil moisture increased due to irrigation and a precipitation event, especially from non-vegetated soil. Throughout the rest of the growing season, CH<sub>4</sub> emissions fluctuated around zero without a clear trend and contributed little to the overall climatic effect of cultivation. However, the soil amendments seem to have a noticeable effect on the CH<sub>4</sub> exchange from vegetated soil, which were generally larger sinks of CH<sub>4</sub> even though the differences were non-significant. The reason for this remains unknown, but the results indicate soil hydrology as a major contributor, either by facilitating more abundant transpiring biomass and more even soil moisture conditions (Hiltbrunner et al., 2012), or through a liming effect. The latter explanation is supported by a recent meta-analysis on the effects of biochars on CH<sub>4</sub> emissions by Jeffery et al. (2016), which attributed pH as the predominant factor contributing to CH<sub>4</sub> exchange in non-flooded acidic soils during the growing period. However, it stated that the overall CH<sub>4</sub> exchange of non-flooded soils is outweighed by regularly flooded soils (i.e., paddy fields). In any case, the question remains, to what extent does CH<sub>4</sub> exchange during the cool and humid winter season contribute to the net CH<sub>4</sub> exchange of boreal agricultural soils?

The observed N<sub>2</sub>O peaks followed the recorded irrigation and precipitation events during the growing season, reflecting the results from a parallel incubation (unpublished) experiment in which the C80N treatment exhibited significantly larger efflux of N<sub>2</sub>O in response to rewetting than the ligneous fiber or biochar treatments. The occasional peaks may have been caused by enhanced microbial activity (Wagner-Riddle et al., 2008) or microbiological changes in respiration in response

to changing redox conditions at soil microsites (Del Grosso et al., 2000; Fidel et al., 2017). Smaller peak emissions by the amendment treatments, especially in the WilB, may be due to greater volume of microsites and more stable soil conditions, which could in theory buffer sudden changes in soil moisture or aeration (Hagemann et al., 2016; Ribas et al., 2019). On the other hand, increased biomass growth decreases soil temperature and moisture compared to less-vegetated soil possibly further undercutting the conditions for N<sub>2</sub>O production. However, the vegetation did not seem to have a clear effect on the N<sub>2</sub>O fluxes in this study, as the emissions from vegetated and non-vegetated soil were generally very modest and equal in size. The differences in N<sub>2</sub>O emissions measured from the bare soil may have been affected also by the natural variance in soil conditions, which may have been further influenced by the soil amendments, making it challenging to distinguish significant treatment effects. Our study lacked observations in midwinter and early spring whereas some studies report that in frozen soil areas, notable N<sub>2</sub>O emissions occur during wintertime (Maljanen et al., 2007) or immediately after snowmelt and thawing of soil frost (Yanai et al., 2011; Risk et al., 2013). Therefore, those periods might be important when assessing the overall impact of soil amendments in the northern region.

## Conclusions

Despite the lack of statistical differences, the numerous parallel trends with CO<sub>2</sub> and N<sub>2</sub>O fluxes, plant biomass and soil properties indicate that the studied soil amendments had a favorable climatic impact. Among them, spruce biochar was the most beneficial for climate during the campaign as the decrease in the soil carbon storage was one of the lowest and it showed one of the lowest N<sub>2</sub>O emissions. The N<sub>2</sub>O emissions were clearly diminished by willow biochar as well, but it remains unclear to what extent this was caused by spatial variability. Additionally, lime-stabilized pulp sludge improved the soil quality and its effect on soil carbon pool was comparable with the studied biochars. However, as pulp sludge is considered to contain less stable carbon than the biochars, it is deemed less favorable in the long-term.

Further research is needed to conclusively characterize the specific effects of ligneous soil amendments on GHG exchange, soil structure and plant growth. However, together with other implications this study suggests that soil amendments in boreal agricultural sites have a favorable net climatic impact, as the amendments seem to improve soil properties, such as porosity and plant available water holding capacity even after two years of single application. It is debatable whether these changes will provide immediate benefits to crop production considering the quality of Finnish

croplands. However, sustained use may help to maintain robust plant and soil activity during extreme weather events like droughts. We hope that in the future, better understanding of the soil processes will guide both the producers of soil amendments and policymakers into adopting more climate-smart management practices.

## Data availability statement

The datasets presented in this study can be found in online repositories. The names of the repository/repositories and accession number(s) can be found below: <https://doi.org/10.5281/zenodo.4401327>.

## Author contributions

LK, KP, and JH collected the empirical data. LK and KP were the main writers of the article. JH, MP, TL, JL, and AL supervised experimental design and contributed to the writing process producing the final version of the article. All authors approved the submitted version.

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## Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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## Supplementary material

The Supplementary Material for this article can be found online at: <https://www.frontiersin.org/articles/10.3389/fsufs.2022.951518/full#supplementary-material>

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