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Impact of biochar on nutrient supply, crop yield and microbial respiration on sandy soils of northern Germany

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Abstract
The application of biochar to agricultural soils to increase nutrient availability, crop production and carbon sequestration has gained increasing interest but data from field experiments on temperate, marginal soils are still underrepresented. In the current study, biochar, produced from organic residues (digestates) from a biogas plant, was applied with and without digestates at low (3.4 t ha^{-1}) and intermediate (17.1 t ha^{-1}) rates to two acidic and sandy soils in northern Germany that are used for corn (*Zea mays* L.) production. Soil nutrient availability, crop yields, microbial biomass and carbon dioxide (CO_2) emissions from heterotrophic respiration were measured over two consecutive years. The effects of biochar application depended on the intrinsic properties of the two tested soils and the biochar application rates. Although the soils at the fallow site, with initially low nutrient concentrations, showed a significant increase in pH, soil nutrients and crop yield after low biochar application rates, a similar response was found at the cornfield site only after application of substantially larger amounts of biochar. The effect of a single dose of biochar at the beginning of the experiment diminished over time but was still detectable after 2 years. Whereas plant available nutrient concentrations increased after biochar application, the availability of potentially phytotoxic trace elements (Zn, Pb, Cd, Cr) decreased significantly, and although slight increases in microbial biomass carbon and heterotrophic CO_2 fluxes were observed after biochar application, they were mostly not significant. The results indicate that the application of relatively small amounts of biochar could have positive effects on plant available nutrients and crop yields of marginal arable soils and may decrease the need for mineral fertilizers while simultaneously increasing the sequestration of soil organic carbon.

Highlights

- A low rate of biochar increased plant available nutrients and crop yield on marginal soils.
- Biochar application reduced the availability of potentially harmful trace elements.
- Heterotrophic respiration showed no clear response to biochar application.
- Biochar application may reduce fertilizer need and increase carbon sequestration on marginal soils.

KEYWORDS

black carbon, carbon sequestration, corn, digestate, heterotrophic respiration, marginal soils, microbial biomass

1 | INTRODUCTION

Global agricultural soils are not only the foundation for human nutrition but also an important net source of greenhouse gases. Land use and land cover change has caused a loss of about 116 Pg of global soil organic carbon (SOC) in the last 12,000 years, with the highest loss-rate in the past 200 years (Sanderman, Hengl, & Fiske, 2017). Changing forests to agricultural land most strongly reduces soil carbon pools (Don, Schumacher, & Freibauer, 2011; Guo & Gifford, 2002), and cropland is the land use with the lowest carbon sequestration per unit of biomass input (Ciais et al., 2010). The loss of soil organic matter (SOM) contributes substantially to anthropogenic greenhouse gas emissions and it also causes the loss of soil nutrients. Furthermore, organic matter is the substrate for soil biota, which release nutrients for plant growth by organic matter decomposition. Hence, the content of SOM is one of the key parameters determining soil fertility and productivity (Komatsuzaki & Ohta, 2007) and a multitude of measures are applied to reduce the loss of organic matter from agricultural soils (Lal, 2004).

Organic amendments have been applied to cultivated soils to improve their biological, chemical and physical functions since prehistoric times (Blume & Leinweber, 2004; Sombroek, Nachtergaele, & Hebel, 1993). Furthermore, organic amendments may again increase SOC stocks if they have prolonged turnover times. One of these amendments is black carbon, which is the residue of incomplete combustion of organic matter (Goldberg, 1985). Black carbon is a natural component of SOM and originates mainly from vegetation fires, which leave behind about 0.1 to 3.4% of the initial biomass as black carbon (Czimczik, Preston, Schmidt, & Schulze, 2003; Fearnside, Barbosa, & Graça, 2007; Kuhlbusch & Crutzen, 1995). The transformation of organic matter into black carbon greatly reduces its degradability and creates a long-term carbon sink. Black carbon from Scandinavian and central European soils may have maximum ^{14}C ages of up to 9,500 years (Gerlach, Baumewerd-Schmidt, van den Borg, Eckmeier, & Schmidt, 2006; Ohlson, Dahlberg, Okland, Brown, & Halvorsen, 2009). Because of its low degradability, black carbon is almost ubiquitous in terrestrial soils. Besides its function as a potential long-term carbon sink in soils, black carbon may also increase soil nutrient supply and soil productivity (Farkas et al., 2020; Glaser, 2007; Major, Rondon, Molina, Riha, & Lehmann, 2010; Quilliam et al., 2012), water retention and erosion stability (Piccolo, Pietramellara, & Mbagwu, 1996) and microbial biomass and activity (Gaskin, Steiner, Harris, Das, & Bibens, 2008; Lehmann et al., 2011) and reduce the availability of toxic trace elements (Nie et al., 2018). Because of these positive effects, the amendment of soils with black carbon has been proposed as a sustainable management technique for enhancing soil productivity and long-term carbon storage (Glaser, Lehmann, & Zech, 2002; Lehmann, 2007). In this context the term "biochar" was introduced, which is a man-made form of black carbon, produced by organic matter pyrolysis under an oxygen-free atmosphere.

Numerous studies on the effects of biochar application to soils were conducted in recent years with a focus on tropical soils, but little emphasis was given to field studies on temperate, marginal soils. It became obvious that these effects cannot be generalized because they depend on numerous different parameters. The properties of biochar heavily depend on the feedstock (Bamminger et al., 2016; Nguyen, Lehmann, Hockaday, Joseph, & Masiello, 2010; Uzoma et al., 2011) and the conditions of biochar production, particularly the pyrolysis temperature (Bruun, Ambus, Egsgaard, & Hauggaard-Nielsen, 2012; Nguyen et al., 2010). Moreover, the effects of biochar application depend on the properties of the amended soils (Haefele et al., 2011; Wu et al., 2018) and the biochar application rate (Dempster, Gleeson, Solaiman, Jones, & Murphy, 2012; Gomez, Denef, Stewart, Zheng, & Cotrufo, 2013). Additionally, biochar application may not only have beneficial effects but also detrimental ones, such as a reduction in nutrient availability (Karer, Wimmer, Zehetner, Kloss, & Soja, 2013; Kloss et al., 2014), crop yield (Borchard, Siemens, Ladd, Möller, & Amelung, 2014) and soil microbial activity (Dempster et al., 2012). Hence, new concepts for biochar application to agricultural soils require a careful evaluation of their effects on soil properties and crop yield, which is ideally studied under field conditions using realistic application rates.

An increasing area of arable soils is used worldwide for renewable energy production, which is one of the key approaches to reduce anthropogenic greenhouse gas emissions. In 2016, crops for biogas production were grown on 1.4 Mha in Germany, which represents 50% of the total area used for renewable primary product production or 4% of the German land mass (Umweltbundesamt, 2018). Biogas is the end product of the anaerobic decomposition of organic matter and consists mainly of methane and CO_2. It is mostly produced from a feedstock consisting of animal waste and energy crops, with corn (*Zea mays* L.) silage as the main contributor (Daniel-Gromke et al., 2018). By far most biogas is produced in tank reactors in which the feedstock is digested in a liquid. The organic residues (digestates) in the process are generally used as organic fertilizer and spread on agricultural soils. However, the massive increase of biogas production plants results in a shortage of available capacities for the disposal of these residues. Furthermore, cropping corn as a feedstock for biogas production results in a loss of SOC (Don et al., 2012), which decreases the carbon gain in the life cycle of biogas production.

To reduce the loss of SOC during corn cropping and increase the application of biogas production residues to agricultural fields, a novel concept of digestate treatment was tested. Digestates were dried with the waste heat of the biogas production plant, charred to biochar and subsequently applied to the corn fields. Such a treatment is expected to increase carbon sequestration on agricultural soils, improve soil nutrient availability and increase crop yield by simultaneously reducing the rate of mineral fertilizers applied. The main research questions were whether charring of digestates from biogas plants and the subsequent amendment to arable soils is a feasible management praxis to increase (a) nutrient supply, (b) crop yield and (c) carbon sequestration in the soils used for feedstock production. Therefore, field experiments were conducted with digestate and biochar application rates close to the current agricultural praxis, and basic soil chemical and biological parameters as well as CO_2 fluxes were determined over 2 years.

2 | MATERIALS AND METHODS

2.1 | Field experiment

A field experiment was conducted between October 2012 and September 2014 in northern Germany at two different sites, one fallow field (54.36445 °N, 9.152486 °E) and a cornfield (54.35400 °N, 9.172694 °E). The cornfield was used for energy crop production for an adjacent biogas plant, which was operated with a feedstock consisting of about 55% corn silage, 15% grass silage, 5% sugar beets, 5% bruised grain and 20% cattle manure (all wet weight). The cornfield received digestates from the biogas plant twice a year, whereas the fallow field did not receive any organic fertilizer prior to the field experiment. The applied biochar was produced from digestates of the adjacent biogas plant. The digestates were first air-dried with the waste heat of the biogas plant and then pyrolysed under oxygen-free conditions at 650 °C (PYREG GmbH, Dörth, Germany).

Five different treatments (Table 1) were established at the two field sites, with three replicate field plots (3 x 3 m) in a completely randomized design (Figure S1). Biochar and digestate application rates represent the current agricultural praxis of the farmer, who applies before each cropping season a total of about 3.8 t (dry weight) of digestate ha^{-1}, corresponding to 1.5 t carbon ha^{-1}. In the case of biochar, the same amount of carbon was applied, resulting in 3.4 t (dry weight) biochar ha^{-1}. One treatment with higher amounts of biochar (D:BH, Table 1) was established to enable the comparison with previous studies using elevated biochar application rates. Digestates and biochar were applied only once at the start

TABLE 1 Summary of the different treatments including abbreviations, and digestate and biochar application rates at the two field sites

Site	Treatment (abbreviation)	Amount organic amendment
Fallow	Control (C)	No amendment
Fallow	Biochar (B)	3.4 t biochar ha^{-1}
Cornfield	Digestate (D)	3.8 t digestate ha^{-1}
Cornfield	Digestate biochar low (D:BL)	3.8 t digestate ha^{-1} and 3.4 t biochar ha^{-1}
Cornfield	Digestate biochar high (D:BH)	3.8 t digestate ha^{-1} and 17.1 t biochar ha^{-1}

of the experiment in October 2012 and incorporated into the soil to a depth of about 20 cm using a small power-tiller. All field plots were cropped during the experiment with winter wheat (*Triticum aestivum* L.) and winter rye (*Secale cereale* L.) in the winter seasons 2012/2013 and 2013/2014, respectively, and with corn (*Zea mays* L.) in the summer seasons 2013 and 2014, mirroring the crop rotation applied by the farmer on the surrounding fields. The cornfield plots received 68 kg N ha^{-1} as ammonium-nitrate prior to each of the two summer crop seasons, which is the rate applied to the surrounding fields, whereas the fallow plots received no additional fertilizer. The mean annual temperature and precipitation in the area between 1988 and 2017 were 9.1 °C and 877 mm, respectively (German Weather Service, 2018).

2.2 | Soil sampling and analysis

Surface soil samples (20 cm depth) were collected from all 15 experimental plots every month during the first year of the experiment (2013) and every second month during the second year (2014). Three samples (ca. 250 cm^3) per plot were collected at random spots from the top 20 cm and combined to a composite sample that was subsequently air-dried and sieved (< 2 mm). Soil pH was measured in a suspension of 10 g of air-dried soil in 25 mL of 0.01 M $CaCl_2$ solution with a pH meter (CG820, Schott, Germany). The pH of digestates and biochar was measured in a suspension of 25 mL of organic sample in 25 mL of a $CaCl_2$ solution as described above. Total soil water content was calculated from the weight difference after drying field fresh soil samples at 105 °C. Total soil carbon and nitrogen were quantified with an elemental analyser (VarioMAX Elementar Analysensysteme GmbH, Hanau, Germany) after the sieved samples had been milled, and dried at 105 °C. Because all pH values were below 6.5, no inorganic carbon was present and total carbon represents total organic carbon (TOC). Soil texture was determined by the pipette method (van Reeuwijk, 2002) using a Sedimat 4–12 (Umwelt-Geräte-Technik GmbH, Müncheberg, Germany).

Plant available soil phosphorus (P) and potassium (K) were extracted from 1 g air-dried soil with a calcium-lactate solution (0.2 M, pH 3.6) for 90 min (Egner & Riehm, 1955). Nitrate (NO_3^-) and ammonium (NH_4^+) were extracted with a 12.5 mM $CaCl_2$ solution (30 g fresh soil in 60 mL). The cation exchange capacity (CEC) and exchangeable cations of calcium (Ca), magnesium (Mg), sodium (Na) and potassium (K) were quantified using the unbuffered salt extraction method (5 g of air-dried soil in 50 mL 1 M NH_4Cl) according to Sumner and Miller (1996). Total trace element concentrations of zinc (Zn), copper (Cu), nickel (Ni), chromium (Cr), lead (Pb) and cadmium (Cd) in soil samples and organic amendments were analysed after extracting air-dried samples with aqua regia (4:1 HCl (30%):HNO_3 (65%), Suprapur, Merck, Germany) in a microwave (MarsXpress, CEM, Matthews, North Carolina, USA) for 15 min at 160 °C. Plant available concentrations of the trace elements Zn, Pb, Cd and Cr were measured after extracting 10 g air-dried soil with 25 mL of a 1 M NH_4NO_3 solution (DIN ISO 19730, 2009). Ammonium and P concentrations in the extracts were quantified photometrically (DR5000, Hach Lange, Berlin, Germany), NO_3^- was quantified by HPLC (1,200 Series, Agilent, Santa Clara, California, USA), Ca, Mg, Na, K cations were quantified by flame atomic absorption spectroscopy (AAS) (1100B, Perkin-Elmer, Waltham, Massachusetts, USA), and Zn, Cu, Ni, Cr, Pb and Cd cations by electrothermal AAS (4100ZL Perkin-Elmer).

2.3 | Plant and microbial parameters

The crop yield was determined at the end of the cropping season (June 2013, May 2014 for winter seasons, October 2013 and September 2014 for the summer seasons) by harvesting total aboveground plant biomass from 1 m^2 in the centre of each of the different treatment plots. Plants were cut about 10 cm above the soil surface. In the summer season of 2014, the biomass of roots and stubbles was also harvested to enable the calculation of total biomass production. Therefore, roots were carefully excavated from the same plots sampled for aboveground biomass and washed in the laboratory with demineralized water to remove soil particles. After weighing the fresh plant biomass (fresh biomass yield), a representative subsample from each plot was dried at 70 °C and dry mass as well as total carbon and nitrogen concentrations were determined. The root:shoot ratio was calculated from the dry biomass of above- and belowground (roots and stubbles) plant material.

Soil microbial biomass carbon (MBC) was quantified between January and October 2013 using the fumigation-extraction method (Jenkinson & Powlson, 1976; Vance, Brookes, & Jenkinson, 1987). The amount of MBC was calculated by:

$$\mathrm{MBC} = \frac{\mathrm{DOC_f} - \mathrm{DOC_c}}{\mathrm{k_{EC}}}, \tag{1}$$

where DOC_f = dissolved organic carbon in fumigated samples, DOC_c = dissolved organic carbon in non-fumigated control samples, and k_{EC} = extraction efficiency factor for converting extractable carbon to MBC. A k_{EC} factor of 0.45 was used (Joergensen, 1996). Fumigated and non-fumigated samples were extracted with a 50 mM

K_2SO_4 solution and DOC concentrations in the extract were measured with a TOC Analyzer (TOC-L, Shimadzu, Japan).

2.4 | Field measurements

To test if biochar application has an effect on heterotrophic soil respiration, CO_2 fluxes were measured in the field plots with a LI-COR 8100 soil flux system (LI-COR, Lincoln, Nebrasca, USA) and a 20-cm Survey Chamber (LI-COR) that was placed on pre-installed plastic frames. The increase of the CO_2 concentration inside the chamber was measured for 2 min between 12:00 and 15:00 CET. Two plastic frames were installed in each field plot and two measurements were conducted for each frame, resulting in 12 CO_2 flux measurements per treatment and sampling day. No corn plants were growing inside the frames and, if necessary, weeds were removed by hand prior to each measurement, to consider only heterotrophic soil respiration. Carbon dioxide fluxes were quantified about every month between November 2012 and August 2014. Cumulative fluxes were calculated by summing the linear interpolations of gas emissions between two sampling times, assuming constant emissions over this period according to the following equation:

$$f_{CO2} = \sum_{i=1}^{n} \left(\frac{f_i + f_{i+1}}{2} \right) \cdot D_i, \qquad (2)$$

where f_{CO2} = cumulative CO_2 flux, f_i = CO_2 flux at day i, f_{i+1} = CO_2 flux at the sampling date following day i, and D_i = time period between day i and day i + 1.

2.5 | Statistical analysis

To test datasets for normal distribution, the Shapiro-Wilks tests was used. Mean values of datasets were compared using either a *t*-test or a one-way ANOVA followed by a Tukey honestly significant difference (HSD) post-hoc test if significant differences ($p < 0.05$) were detected. In the case of a non-normal distribution, the medians were tested using a Mann–Whittney rank sum test. All statistical analyses were conducted using IBM SPSS Statistics 26 (IBM Corporation, Armonk, New York, USA).

3 | RESULTS

3.1 | Soil, digestate and biochar properties

The fallow soil was classified as Gleyic Podzol (WRB, 2014), with pH values increasing from extremely acidic at the surface to slightly acidic at the bottom of the soil profile, a texture dominated by sand and relatively low TOC concentrations and CEC (Table 2, Figure S1). The cornfield soil was classified as Gleyic Plaggic Umbrisol (WRB, 2014), with two surface horizons characterized by organic matter accumulation and anthropogenic disturbances due to ploughing (Ap1) and the amendment of the soil with sods and other organic material ("Plaggen") during historic times (Ap2, Figure S1). The organic carbon content was about twice as high as in the Ah horizon of the fallow soil but also decreased steeply in the underlying horizons. The texture of the cornfield soil was dominated by sand, and it was characterized by low pH values and low CEC (Table 2).

TABLE 2 Basic soil characteristics of the two experimental field sites in Drage, northwest Germany

Site	Horizon	Depth (cm)	TOC[a] (%)	N (%)	C:N	pH	CEC[b] ($mmol_c\ kg^{-1}$)	Texture[c]
Fallow (Gleyic Podzol)[d]								
	Ah	0–20	0.68	0.07	9.2	4.5	22.3	Loamy sand
	Bl-Bh	20–55	0.07	0.01	6.6	5.0	6.33	Sand
	Bhs	55–65	0.06	0.01	7.5	5.3	7.53	Sand
	Cw	> 65	0.04	0.01	8.4	6.2	6.76	Sand
Cornfield (Gleyic Plaggic Umbrisol)[d]								
	Ap1	0–20	1.37	0.10	13.4	5.3	26.7	Sand
	Ap2	20–40	1.37	0.11	12.9	5.8	33.8	Sand
	Bh-Bl	40–70	0.49	0.03	15.3	5.7	15.3	Loamy sand
	Brl	>70	0.22	0.01	22.1	6.1	8.33	Sand

[a]Total organic carbon.
[b]Cation exchange capacity.
[c]Texture classes according to FAO (2006).
[d]Soil classification according to WRB (2014).

The biochar yield was 29% of the pyrolized digestates (dry weight based). The carbon concentration in the biochar (43.9 ± 2.8%) was substantially larger than in the digestates (39.5 ± 0.3%) but the opposite was the case for nitrogen (1.2 ± 0.1% in biochar, 2.0 ± 0.04% in digestates, Table S1). To evaluate the change in element composition between digestates and biochar, a concentration factor was calculated, which is the ratio between the element concentration in biochar and in digestates (Table S1 and S2). Based on the biochar yield of 29%, a concentration factor of 3.4 (1/0.29) is expected if no loss of a certain element occurred during pyrolysis. As expected, carbon and nitrogen showed concentration factors substantially below 3.4, but also K, Na, Mg and Cd, indicating a partial volatilization of these elements during pyrolysis (Table S1 and S2). In contrast, P, Ca and Zn showed concentration factors between 3.1 and 3.3, demonstrating that these elements were retained in the biochar. Surprisingly, the concentration factors of Cu, Ni, Cr and Pb were substantially above 3.4, reaching up to 10.6 (Ni), indicating a contamination with these elements during the pyrolysis process. However, total trace element concentrations in the biochar were substantially below the maximum permissible concentration for farm-produced fertilizers (Table S2).

Furthermore, pyrolysis caused a pH rise from 6.6 in the digestates to 8.0 in the biochar.

3.2 | Effect of biochar application on soil properties

In the fallow soil, the application of biochar to the surface soil resulted in a significant increase of pH, TOC, total nitrogen, and concentrations of plant available NO_3^-, NH_4^+, P, K and all of the exchangeable cations except Na over the whole period of the field experiment (Figure 1). The differences between the pH values and nutrient concentrations of the control and the biochar plots were greatest directly after biochar application and diminished over time, whereas TOC concentrations remained almost constant (Figure S2). Addition of the moderately alkaline biochar resulted in an increase of the soil pH from 4.4 to 4.8. Furthermore, the concentrations of plant available K, Mg and NO_3^- increased considerably and reached 6.2, 4.8 and 2.3 times the concentrations of those in the control plots, respectively (Figure S2). The increase of NO_3^- and NH_4^+ concentrations represented 33 and 6.7% of the nitrogen added as biochar, respectively. Furthermore, the median of the CEC in the

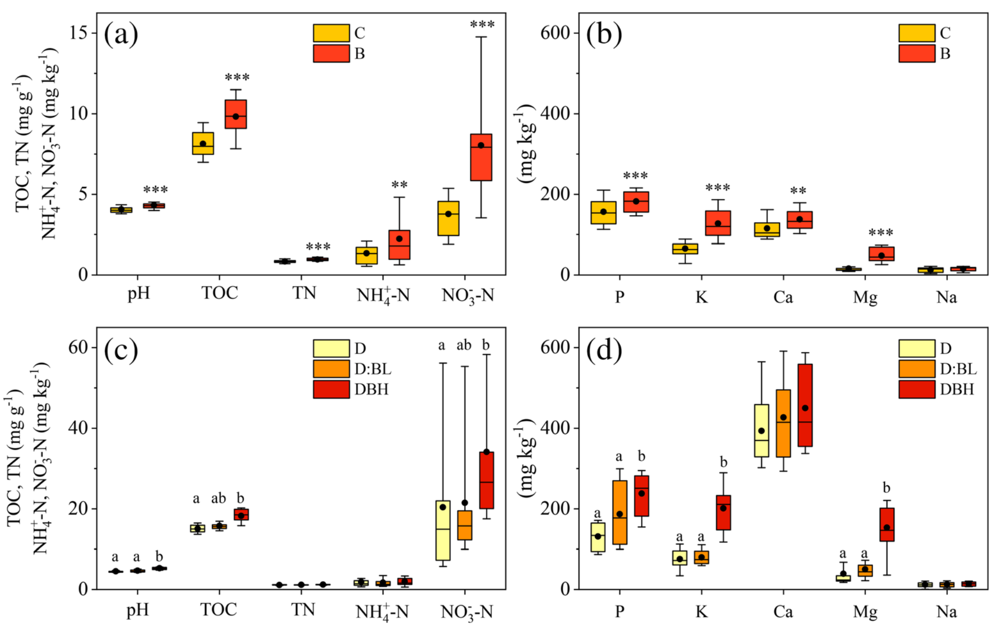

FIGURE 1 pH, total organic carbon (TOC), total nitrogen (TN), ammonium (NH_4^+-N) and nitrate (NO_3^--N) concentrations (a, c) and plant available nutrient concentrations (b, d) in the surface soils of the different treatments at the fallow (a, b) and the cornfield (c, d) sites. Presented are data from all sampling times during the field experiment lasting from winter 2012/2013 to summer 2014. Boxes represent the 25th to 75th percentiles and whiskers are the 10th to 90th percentiles. The horizontal line in each box represents the median value; the closed circle associated with each box represents the mean value (n = 24–45). The asterisks indicate a significant (t-test, ** = $p < 0.01$, *** = $p < 0.001$) difference between the control (C) and the biochar treatments (B). The different lowercase letters indicate significantly different subgroups (ANOVA, Tukey honestly significant difference [HSD] post-hoc, $p < 0.05$) [Color figure can be viewed at wileyonlinelibrary.com]

biochar treatment was substantially, although not significantly (Mann–Whitney *U*-test, $p = .28$), higher than in the control plots (Figure S3).

Biochar application also caused increasing values of almost all measured soil parameters at the cornfield soil (Figure 1), although significant increases were found only after high biochar applications (D:BH). Soil nutrient concentrations in the cornfield plots were generally larger than in the fallow plots. Similar to the fallow, the greatest effects of biochar application were found directly after biochar application, with a pH increase in the D:BH plots from 5.2 to 5.9 (Figure S2). The concentrations of exchangeable Mg and plant available K and P increased to 8.3, 5.1 and 2.0 times of those in the digestate plots, respectively. Although NH_4^+ concentrations were almost unaffected by biochar application at the cornfield, NO_3^- concentrations increased in both treatments and this increase represented 7.8 and 18% of the nitrogen added as biochar in the D:BL and D:BH treatments, respectively. The mean CEC was higher in plots amended with biochar (D:BL and D:BH) than in those without (D), with a significant increase in the case of high biochar application (D:BH, Figure S3). In contrast to nutrient concentrations, the soil water content was not affected by the application of biochar in any of the treatments (Figure S4).

The soils amended with biochar and digestates showed generally slightly larger concentrations of the trace metals Zn, Ni, Cu, Cr, Pb and Cd than soils without biochar amendment, but these differences were only significant in the case of Zn and Cr in the cornfield plots (Table S2).

Although biochar application had only a minor effect on total trace element concentrations, it substantially reduced the plant availability of trace elements (Table 3). In the fallow plots, this decrease was significant for Cd and Cr, whereas in the cornfield plots Zn, Pb and Cd concentrations decreased significantly while Cr concentration non-significantly increased.

3.3 | Crop yield

The crop yield in the biochar treatments was generally substantially larger than in plots without biochar (Figure 2). The fresh corn biomass yield on the fallow biochar plots during the two summer seasons (39.2–45.5 t ha^{-1}) increased significantly ($p < 0.05$) by 33 to 37% compared with the yields on the control plots (28.6–34.1 t ha^{-1}). The yield of winter crops was even more increased by biochar application (52–72%) but total fresh yields were very low (< 4 t ha^{-1}). At the cornfield site, the effect of biochar application was less pronounced and a significant effect on biomass yield was only detected in the second year of the experiment in the D:BH plots (Figure 2). The biomass yields on the D:BH plots was 13% (2013) and 38% (2014) greater than in the D plots. Furthermore, the yield of winter crops was significantly larger in the cornfield than in the fallow plots. The corn yield of all treatments with biochar (B, D:BL and D:BH) did not differ significantly in the summer of 2013. However, in the summer of 2014, the corn yield in the biochar treatment of the fallow site was significantly larger ($p <$ a) than that in the D:BL treatment, but significantly less than in the D:BH treatment of the cornfield site.

Biochar application significantly increased the root biomass in the B treatments of the fallow plots and the D:BH treatments of the cornfield plots. For the D:BH treatments at the cornfield plots, the root:shoot ratio was also significantly larger than that at the plots without biochar (Table S3).

3.4 | Soil microbial biomass

Soil MBC concentrations showed a clear seasonality in all treatments, with the smallest values from September to March and largest values in June and July (Figure 3). Without biochar addition, MBC concentrations ranged at the fallow plots between 50.6 ± 5.1 mg kg^{-1} in October and 217 ± 27.3 mg kg^{-1} in June. At the cornfield plots,

TABLE 3 Plant available trace element concentrations in surface soils of the two different experimental sites in Drage, north Germany. Values are means ($n = 3$, ± *SD*). Asterisk indicates significant differences (*t*-test, $p < 0.05$) between biochar and control treatments at the fallow plots; different lowercase letters indicate different subgroups (ANOVA, Tukey honestly significant difference [HSD] post-hoc, $p < 0.05$) between the cornfield treatments

	Zn	Pb	Cd	Cr
Site/treatment	mg kg^{-1}	μg kg^{-1}	μg kg^{-1}	μg kg^{-1}
Fallow/C	1.81 ± 0.16	86.3 ± 28.3	21.8 ± 2.11	5.24 ± 0.64
Fallow/B	1.62 ± 0.27	47.4 ± 13.8	15.1 ± 2.92*	3.02 ± 0.71*
Cornfield/D	2.32 ± 0.43 a	42.1 ± 7.71 a	10.3 ± 1.68 a	1.77 ± 0.85
Cornfield/D:BL	1.20 ± 0.26 b	26.2 ± 8.75 ab	5.43 ± 1.24 b	2.33 ± 1.15
Cornfield/D:BH	0.25 ± 0.15 c	24.1 ± 2.65 b	1.67 ± 0.52 c	3.29 ± 0.86

Abbreviations: B, biochar; BH, biochar high; BL, biochar low; C, control; D, digestate.

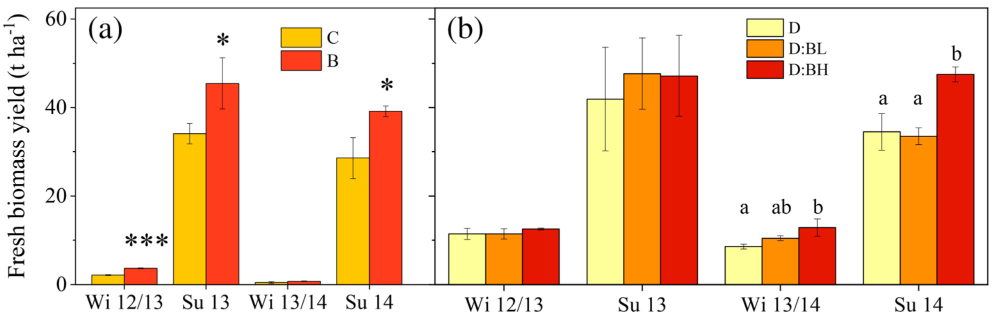

FIGURE 2 Yield of fresh aboveground plant biomass during the four cropping seasons of the field experiment at the fallow (a) and the cornfield (b) sites. During the winter seasons of 2012/2013 and 2013/2014 the fields were cropped with winter wheat and winter rye, respectively. Corn was grown during the two summer seasons in 2013 and 2014. Given are mean values ($n = 3$) with one standard deviation. The asterisks indicate a significant (t-test, * = $p < 0.05$, *** = $p < 0.001$) difference between the control (C) and the biochar (B) treatments at the fallow site. The different lowercase letters indicate significantly different subgroups (ANOVA, Tukey honestly significant difference [HSD] post-hoc, $p < 0.05$) at the cornfield site. BH, biochar high; BL, biochar low; D, digestate [Color figure can be viewed at wileyonlinelibrary.com]

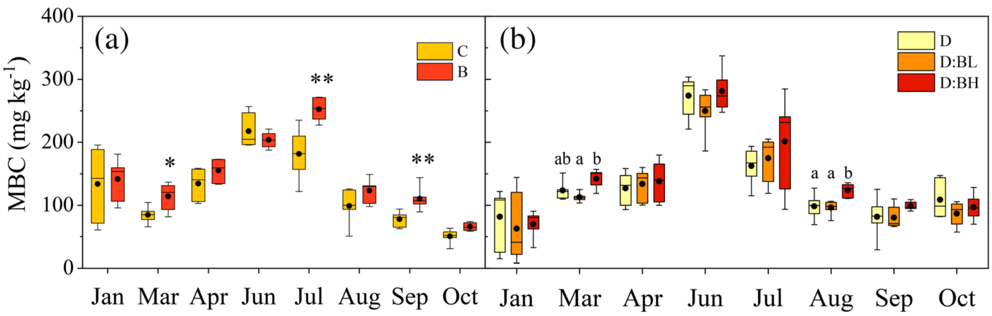

FIGURE 3 Soil microbial biomass carbon (MBC) in surface soils of the fallow (a) and the cornfield (b) plots between January and October 2013. The boxes represent the 25th to 75th percentiles and whiskers the 10th to 90th percentiles. The horizontal line in each box represents the median value; the closed circle associated with each box represents the mean value. The asterisks in (a) indicate a significant (t-test, * = $p < 0.05$, ** = $p < 0.01$) difference between the C and B treatments. The different lowercase letters in (b) indicate significantly different subgroups (ANOVA, Tukey honest significant difference [HSD] post-hoc, $p < 0.05$). BH, biochar high; BL, biochar low; D, digestate [Color figure can be viewed at wileyonlinelibrary.com]

MBC concentrations ranged between 81.5 ± 31.9 mg kg^{-1} in September and 274 ± 33.5 mg kg^{-1} (mean ± standard deviation [SD], $n = 6$) in June.

Biochar amendment resulted in a clear increase of MBC concentrations at the fallow plots, which was significant in March, July and September (Figure 3). In contrast, the application of the same amount of biochar to the cornfield (D:BL treatment) resulted on average in a slight but insignificant decrease of microbial biomass. Only the larger biochar applications in the D:BH plots resulted generally in an increase of MBC with significant differences to the plots without biochar (D treatment) in August (Figure 3b).

3.5 | Carbon dioxide fluxes

Carbon dioxide fluxes from heterotrophic respiration showed a clear seasonal trend in all of the treatments, with greatest emissions in May/June 2013 and August 2014 and very low to zero fluxes in winter (Figure 4). At the fallow site, maximum mean emissions in the control plots reached 4.7 ± 0.3 g CO_2-C m^{-2} d^{-1} and 5.7 ± 2.3 g CO_2-C m^{-2} d^{-1} in summer 2013 and summer 2014, respectively, whereas maximum CO_2 fluxes from the biochar plots were considerably larger (5.5 ± 2.0 g CO_2-C m^{-2} d^{-1} and 7.3 ± 2.4 g CO_2-C m^{-2} d^{-1}). However, the median CO_2 fluxes from the control and the biochar plots (2.15 g CO_2-C m^{-2} d^{-1} and 2.11 g CO_2-C m^{-2} d^{-1}, respectively) across the whole measuring period between November 2012 and August 2014 were not significantly different (Mann–Whitney U-test, $p > 0.05$). Also, no significant difference was found when considering the CO_2 fluxes of the years 2013 and 2014 separately, with a median CO_2 flux of 2.3 g CO_2-C m^{-2} d^{-1} for both treatments in 2013 and a median flux of 1.8 g CO_2-C m^{-2} d^{-1} and 1.7 g CO_2-C m^{-2} d^{-1} for the C and the B treatment in 2014, respectively. The seasonal cumulative CO_2 fluxes

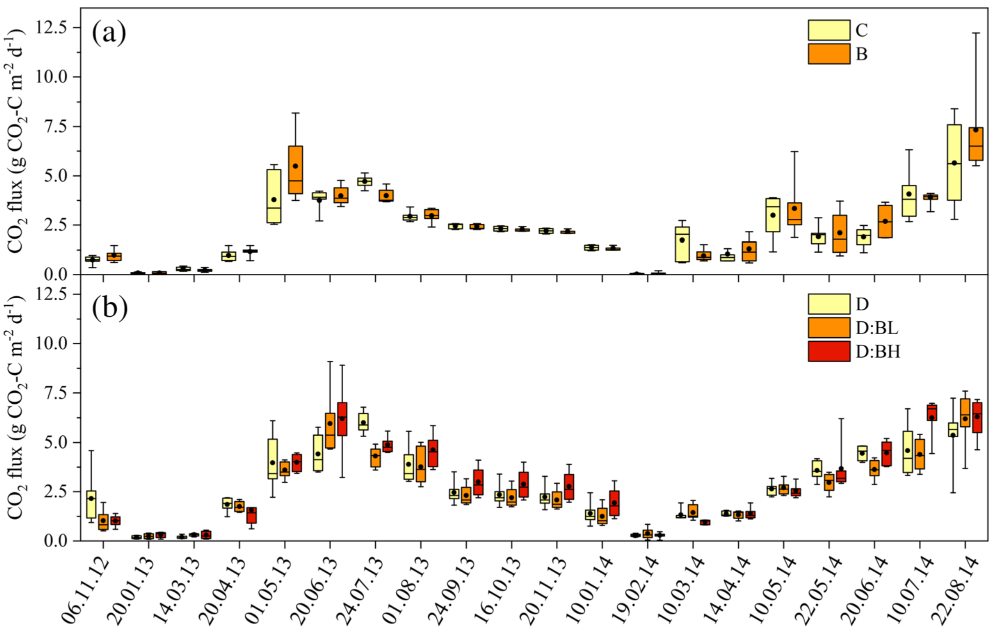

FIGURE 4 Carbon dioxide fluxes between November 2012 and August 2014 from the different treatments at the fallow (a) and the cornfield (b) sites. The boxes represent the 25th to 75th percentiles and whiskers the 10th to 90th percentiles. The horizontal line in each box represents the median value; the closed circle associated with each box represents the mean value. B, biochar; BH, biochar high; BL, biochar low; C, control; D, digestate [Color figure can be viewed at wileyonlinelibrary.com]

from the biochar and the control plots at the fallow site did not differ significantly in any season (t-test, $p > 0.05$, Table S4).

Carbon dioxide fluxes from the D and the D:BL treatments at the cornfield were similar and reached maximum values of 6.0 g CO_2-C m^{-2} d^{-1} in summer 2013 and 5.4 ± 1.6 g CO_2-C m^{-2} d^{-1} and 6.2 ± 1.4 g CO_2-C m^{-2} d^{-1} in summer 2014, respectively (Figure 4b). Also, the median values of CO_2 fluxes from the D and the D:BL plots were 2.24 g CO_2-C m^{-2} d^{-1} and 2.26 g CO_2-C m^{-2} d^{-1}, respectively, which is very similar. In contrast, the D:BH treatment showed generally higher CO_2 fluxes than the D and the D:BL treatments, with a median value of 2.65 g CO_2-C m^{-2} d^{-1}. However, the fluxes from the different treatments at the cornfield site were not significantly different (Mann–Whitney U-test, $p > 0.05$), both when considering the whole period between November 2012 and August 2014 or the annual data of 2013 and 2014 separately. Although the cumulative CO_2 fluxes from the D:BL treatments were generally higher than those from the D plots, these differences were never statistically significant (Table S4). In contrast, cumulative CO_2 fluxes from the D:BH plots between summer 2013 and summer 2014 were 47 to 102 g CO_2-C m^{-2} higher than those from the D plots.

4 | DISCUSSION

The application of biochar to agricultural soils has repeatedly been shown to increase soil nutrient concentrations, pH, crop yield and carbon sequestration. However, field trials with biochar from organic waste applied to marginal, temperate soils are still under-represented (Verheijen et al., 2017). The current field trials on sandy and acidic soils in northern Germany used biochar from biogas plant digestates at two application rates (3.4 and 17.1 t biochar ha^{-1}), with the lower rate representing the organic fertilizer load applied under the current field praxis.

In our experiments, biochar application caused a general increase of pH and of plant available nutrients. However, the magnitude of the effect depended on the intrinsic properties of the treated soils. Although a significant effect was observed at the fallow site with low biochar application rates, a similar effect was detectable at the cornfield site only at the substantially higher rate. The reason for the higher biochar application rate needed to raise plant available soil nutrient concentrations and pH in the cornfield is likely to be related to the overall higher nutrient, CEC and TOC concentrations in this soil in comparison to the fallow soil. An increase of plant available nutrients after biochar application has repeatedly been shown in field experiments (Farkas et al., 2020; Major, Rondon, et al., 2010; Quilliam et al., 2012) and was identified as the reason for increased crop productivity on agricultural soils (Major, Rondon, et al., 2010; Olmo et al., 2014). But the effects of biochar have been shown to depend on the application rate (Major, Rondon, et al., 2010; Quilliam et al., 2012; Rogovska, Laird, Rathke, & Karlen, 2014) and soil properties, with greatest effects on acidic and nutrient-poor soils (Farkas et al., 2020; Haefele et al., 2011). Several factors may contribute to increasing nutrient concentrations in the biochar treatments. First of all, biochar acts as a fertilizer. In the current experiment, all biochar-amended plots received substantially larger amounts of plant nutrients than plots without biochar (Table S1). Furthermore, the application of biochar may increase nutrient retention in the soil (e.g. by increasing soil pH (Hossain et al., 2020) and CEC (Glaser et al., 2002)), which was also observed in the current experiment. This may reduce the loss of NO_3^- through leaching (Hagemann, Kammann, Schmidt, Kappler, & Behrens, 2017; Yao, Gao, Zhang, Inyang, & Zimmerman, 2012). The stronger increase of plant available NO_3^- compared with NH_4^+ concentrations after biochar application supports previous findings, showing increasing nitrification activity after biochar application, which is likely supported by the pH increase in the acidic soils (Abujabhah, Doyle, Bound, & Bowman, 2018; Gao & DeLuca, 2020). Furthermore, microbial nitrogen fixation, which was shown to increase after biochar application (Rondon, Ramirez Orozco, Hurtado, & Lehmann, 2007), may contribute to higher plant available nitrogen concentrations in the biochar-amended plots.

However, this nutrient increase after biochar application seems transient. Quilliam et al. (2012) found that 3 years after applying 50 t biochar ha^{-1} to a loamy Cambisol there was no remaining effect on soil nutrient concentration, which rose again after repeated biochar application. A similar trend was seen in our results as nutrient concentrations decreased over the course of the experiments. Under the current field praxis, the digestates from the biogas plant are returned before the seedbed preparation (i.e., twice per year) to the same soils that are used for the production of the feedstock of the biogas plant. If this repeated input is provided as biochar, this would reduce the requirement for mineral fertilizers. Furthermore, the relatively low but repeated dose of biochar has a lower probability to negatively affect the availability of micronutrients and nitrogen, as has been observed at high (≥ 30 t ha^{-1}) biochar application rates (Borchard et al., 2014; Haider, Steffens, Moser, Mueller, & Kammann, 2017). An increase of NO_3^- and NH_4^+ concentrations was observed in the current field experiment after biochar application, providing no evidence for a reduction of plant available nitrogen through nitrogen immobilization by the applied biochar as observed in previous studies (Borchard et al., 2014; Karer et al., 2013).

The impact of biochar application on corn crop yield was similar to its effect on available nutrient concentrations. Although the corn yield was significantly higher on the fallow biochar plots in both years after applying low amounts of biochar (3.4 t ha^{-1}), a significant effect was only observed at the cornfield plots after larger biochar application (17.1 t ha^{-1}) and in the second season. Because biochar application did not result in increasing soil moisture, which has been identified as one reason for increasing yield in some field experiments (Haefele et al., 2011; Yang et al., 2015), the most likely reason for the observed yield increase is rising nutrient concentrations through biochar application, as this was also found in several field experiments on nutrient-poor tropical soils (Cornelissen et al., 2018; Haefele et al., 2011; Kätterer et al., 2019; Major, Rondon, et al., 2010). For temperate soils, the effect of biochar on crop yield is not unambiguous and a meta-analysis showed on average no effect of biochar on crop yield in temperate soil but an overall positive effect in tropical soils (Jeffery et al., 2017). However, field experiments on a temperate soil in northern Germany with a combination of biochar, mineral fertilizers and compost showed application-rate-dependent crop yield increases (Glaser, Wiedner, Seelig, Schmidt, & Gerber, 2015), which were similar to those detected in the current field experiment. However, if large biochar amounts (40 t ha^{-1}) were applied, the biochar effect was either positive, negative or neutral, depending on the additional application of mineral or organic fertilizers, or biochar treatment before application (Glaser et al., 2015). The lack of a biochar effect on crop yields in temperate soils was assigned to sufficient nutrient availability in the studied soils even without biochar (Güereña et al., 2013; Sänger et al., 2017). In this case, high biochar application rates may rather reduce nutrient supply (e.g., by nitrogen immobilization), and hence cause decreasing crop yields if no additional nitrogen fertilizer is provided (Borchard et al., 2014; Karer et al., 2013). Furthermore, the immobilization of micronutrients was held to be responsible for the decline in crop yield after biochar application (Kloss et al., 2014). However, these effects were observed at biochar application rates of 30 t ha^{-1} and higher. Such high biochar amounts may not repeatedly be applied, because they most likely violate the current restriction on maximum annual nitrogen loads on European agricultural soils, which is 170 kg N ha^{-1} yr^{-1} (CEC, 1991).

Although biochar may have detrimental effects on the availability of micronutrients, in particular if applied at high rates, it may also reduce the availability of phytotoxic trace elements in the soil (Nie et al., 2018; Zhang et al., 2015). The production of biochar from digestates caused a relative accumulation of the potentially phytotoxic trace elements Zn, Cu, Ni, Cr, Pb and Cd in the biochar to concentrations that were generally well above those in soils of the two field sites but still well below permissible concentrations for farm-produced fertilizers (Table S2). In particular, organic waste from industrial livestock farming (e.g., pig manure) may contain high levels of potentially toxic trace elements such as Cu, Pb, Cd or Cr (Nicholson, Chambers, Williams, & Unwin, 1999). If these organic residues are used as feedstock for biogas production, the trace elements remain in the digestates and may cause the accumulation of phytotoxic trace metals in the soils after application to the agricultural fields (Dragicevic, Eich-Greatorex, Sogn, Horn, & Krogstad, 2018). Furthermore, the absolute amount of Cu, Ni, Cr and Pb increased during pyrolysis of digestates to biochar, indicating a contamination of biochar with these elements, probably from the kiln of the pyrolysis plant. Such contaminations need to be eliminated (e.g., by modifying the pyrolysis process) to prevent an accumulation of phytotoxic trace elements in soils. German legal regulations define upper boundaries for annual trace element loads to arable soils and a precaution value, which is the concentration of a given trace element indicating a harmful soil change (Table S2) (FMJCP 1999). Due to the relatively high Zn concentrations in the applied biochar (378 mg kg^{-1}) the maximum annual load of Zn (1,200 g ha^{-1}) would be reached with 3.2 t biochar ha^{-1}, which is consequently the maximum permitted annual load of biochar. A simple calculation shows that to reach the precaution value of Zn (60 mg kg^{-1}) in the uppermost 20 cm of soil, almost 300 t biochar ha^{-1} would need to be applied. Considering the relatively low maximum annual load of 3.2 t biochar ha^{-1}, the precaution values for trace elements will probably not be reached in several decades because trace elements also are removed from the surface soil by uptake into harvested crops or leaching. A loss of trace metals introduced by biochar application through leaching must be considered in these sandy soils characterized by a low CEC, which will be particularly problematic if biochar is produced from organic waste containing elevated concentrations of phytotoxic trace elements. In this case, the fate of trace elements should be studied in column leaching experiments or even field studies on the watershed scale. On the other side, column experiments demonstrated that biochar may reduce the export of trace elements from contaminated soils either through sorption or an interaction with metal-binding dissolved organic matter (Riedel, Hennessy, Iden, & Koschinsky, 2015).

After biochar application, a slight but significant increase of total trace element concentrations (Zn and Cr) was only observed in the cornfield plots, which might be due to the generally higher trace element concentration in the fallow plots. However, more important than the slight observed increase of total Zn and Cr concentrations in the soil after biochar application is that plant available concentrations of the elements Zn, Pb and Cd were reduced after biochar application. Although Zn is an essential trace element for plant growth (Marschner, 1995), all of these trace elements may harm plant growth at concentrations between about 0.3 μM (Pb) and 25 μM (Zn) in the soil solution (Kopittke, Blamey, Asher, & Menzies, 2010). The immobilization of phytotoxic trace elements has also been shown for other types of biochar (Park, Choppala, Bolan, Chung, & Chuasavathi, 2011; Xu et al., 2018), and a reclamation strategy to reduce plant available concentrations of toxic trace elements in contaminated soils is based on this effect (Fellet, Marchiol, Delle Vedove, & Peressotti, 2011). Therefore, the amendment of soils with biochar from digestates at the rates tested in the current field experiments not only increases soil nutrient concentrations but also reduces the risk of toxic trace element transfer from soils to the food chain, which is particularly important if problematic feedstock such as animal waste is used.

Maximum MBC concentrations in the current field experiments were similar to MBC concentrations (186–325 mg MBC kg^{-1}) reported previously for biochar-amended temperate soils, for example in a Luvisol in southern Germany (Bamminger et al., 2016; Grunwald et al., 2017) or loamy Anthrosols on the Chinese Loess Plateau (Zhang et al., 2017). Although plots amended with biochar in the current field trials showed only a slight increase in MBC, biochar application to agricultural soils caused a clear rise of MBC in previous laboratory incubations (Luo, Lin, Durenkamp, Dungait, & Brookes, 2017) and field experiments (Grunwald et al., 2017; Zhang, Cheng, Feng, et al., 2017; Zheng et al., 2016). However, latter effects were observed at biochar application rates substantially higher than those in the current experiment. Because a positive correlation between biochar application rates and MBC concentrations could be shown in incubations (Singh & Mavi, 2018), the relatively low biochar application might be the reason for the weak biochar effect on MBC in the current field experiment and could also be the reason for the absence of significant effects in previous studies that applied similarly low biochar amounts (Galvez et al., 2012; Singh et al., 2014). A positive effect of biochar application on MBC was assigned to increasing pH (Aciego Pietri & Brookes, 2008) and nutrient availability (Taylor, 1951), but also to the adsorption of microbial cells on biochar surfaces (Pietikäinen, Kiikkilä, & Fritze, 2000). However, several studies could not find an effect of biochar application on MBC even at application rates of 30 t biochar ha^{-1} and above (Bamminger et al., 2016; Castaldi et al., 2011), or even a decrease of MBC and microbial CO_2 production at relatively low application rates of 5 t biochar ha^{-1} (Dempster et al., 2012). These conflicting results indicate that the response of the soil microbial communities to biochar applications depends on the properties of the applied biochar and soil conditions. A meta-analysis showed that the largest increases in MBC were found in field experiments on vegetated, coarse-textured and acidic soils and with a biochar feedstock of waste organic matter (Liu et al., 2016).

The microbial decomposition of biochar carbon to CO_2 has been demonstrated in numerous laboratory incubations (Herath et al., 2015; Knoblauch, Maarifat, Pfeiffer, & Haefele, 2011; Kuzyakov, Subbotina, Chen, Bogomolova, & Xu, 2009; Wu et al., 2018), but data from field experiments, in particular on temperate soils, are still scarce. No biochar effects on CO_2 fluxes were found in previous field experiments using relatively low biochar application rates (< 10 t ha^{-1}) (Karhu, Mattila, Bergström, & Regina, 2011; Mechler, Jiang, Silverthorn, & Oelbermann, 2018; Sackett et al., 2015), which is confirmed by the results from the B and D:BL treatments in the current field experiments. Also, Polifka, Wiedner, and Glaser (2018) found no significant increase of mean CO_2 fluxes from heterotrophic respiration when they applied a mixture of biogas plant digestates and biochar on a Cambisol in northern Germany at low biochar application rates (3 t ha^{-1}). In contrast, when applying high biochar application rates (40 t ha^{-1}), a strong (53%) and significant increase of CO_2 fluxes was found (Polifka et al., 2018). However, the increased CO_2 fluxes only accounted for about 0.3% of the added biochar, indicating a substantial sequestration of carbon after biochar application. Strong and significant increases of CO_2 fluxes after high (> 30 t ha^{-1}) biochar application rates were also observed in other field experiments on temperate soils (Fang et al., 2016; Zhang et al., 2012), but this response was in part transient and disappeared after 1 year (Castaldi et al., 2011). However, high biochar application rates (> 10 t ha^{-1}) do not always cause a significant increase of CO_2 fluxes (Bamminger, Poll, & Marhan, 2018; Lentz, Ippolito, & Spokas, 2014; McClean et al., 2016; Mukherjee, Lal, & Zimmerman, 2014; Zhang et al., 2017) and may even result in a significant decrease (Case, McNamara, Reay, & Whitaker, 2014), even at very high biochar application rates of 60 t ha^{-1} (Hagemann et al., 2017).

The source of increasing CO_2 fluxes after biochar application in previous field trials is unclear. Although incubation studies applied ^{13}C- or ^{14}C-labelled material to identify CO_2 fluxes from biochar carbon (Kuzyakov et al., 2009; Pan, Li, Chapman, Khan, & Yao, 2016), field experiments with labelled biochar were not conducted, and the unequivocal identification of the source of the additional CO_2 is not possible. The increasing CO_2 fluxes were assigned to the microbial decomposition of a labile organic matter fraction of the applied biochar (Castaldi et al., 2011) but also to organic matter decomposition of organic fertilizers applied together with the biochar (Polifka et al., 2018). Another possibility is the contribution of respired plant carbon (Fang et al., 2016; Major, Lehmann, Rondon, & Goodale, 2010). Although CO_2 fluxes were measured between the plant rows in the current experiment to only consider heterotrophic respiration, it cannot be excluded that a part of the emitted CO_2 also originates from root respiration, from microbial respiration in the rhizosphere or from symbiotic mycorrhiza. Corn plants were shown to release root exudates under nutrient limitation into the rhizosphere that may be decomposed by microorganisms to CO_2 (Carvalhais et al., 2011). Furthermore, an interaction between arbuscular mycorrhizae and corn plants was shown to increase biomass yields on nutrient poor soils (Willmann et al., 2013). Because the extraradical mycelium of arbuscular mycorrhiza also colonizes soil outside the rooting zone of the corn plant (Jansa, Mozafar, & Frossard, 2003) and respires plant assimilated organic matter, plant-derived carbon might contribute to the surplus CO_2 in the D:BH treatments if roots or attached mycorrhizae grew below the CO_2 flux chamber. The root biomass and the root:shoot ratios on plots with biochar were higher than on plots without biochar (Table S3) and it was observed previously that biochar applications can increase root growth (Xiao et al., 2016). Root:shoot ratios of maize plants may vary widely depending on plant age (Cai et al., 2017; Liu et al., 2017), soil properties such as pH and nutrient concentrations (Liu et al., 2017), water supply (Cai et al., 2017) and seedbed preparation (You et al., 2017), with root:shoot ratios of mature plants ranging between about 0.3 and 0.05 (Cai et al., 2017; Jia et al., 2018; You et al., 2017; Zhang et al., 2015).

If root or mycorrhizal respiration contributed to CO_2 soil fluxes, the measured CO_2 fluxes would overestimate heterotrophic soil respiration. Although the soils of the two studied sites were relatively poor in nutrients and had a low carbon content and low pH values, the mean CO_2 respiration rates during the different cropping seasons in 2013 and 2014 were in the same range as mean heterotrophic CO_2 fluxes (0.5–4.5 g CO_2-C m^{-2} d^{-1}) from agricultural fields on highly productive Mollisols or loamy Luvisols (Bamminger et al., 2018; Black, Davis, Hudiburg, Bernacchi, & DeLucia, 2017; Mechler et al., 2018). These relatively high CO_2 fluxes, even from the control treatment at the fallow site that did not receive any organic or mineral fertilizer, may further indicate a contribution of plant-derived carbon to the measured CO_2 emission rates. However, the higher cumulative CO_2 respiration fluxes from the B plots at the fallow site (13.4–52.4 g CO_2-C m^{-2}) and the D:BH plots at the cornfield site (47–102 g CO_2-C m^{-2}) were more than compensated for by the additional carbon input (150 g m^{-2} in B plots, 750 g C m^{-2} in D:BH plots) that these plots received in the form of biochar. Furthermore, biochar-amended plots received an additional carbon input through a higher biomass production than plots without biochar (Table S3), of which at least the roots and stubbles remained in the soils and contributed to their carbon pool.

The arable soils of the current field experiments were part of a management system in which biogas plant digestates, or the biochar produced from them, have to be disposed of twice per year to the soils before seedbed preparation. Such a system may only be sustained with relatively low annual doses of digestate or biochar to prevent the violation of regulations of annual trace element deposition. Hence, the current management system requires the repeated application of relatively low doses of biochar and the results from the field experiment give no evidence that these will significantly increase CO_2 emissions from the decomposition of soil organic matter.

5 | CONCLUSIONS

The results from the field experiments on temperate, acidic soils indicate that a relatively low single dose of biochar may improve soil fertility and crop yield without causing a substantial increase of soil CO_2 fluxes. However, such a positive effect of biochar was only detectable in marginal soils with initially low nutrient concentrations. In the current management system of biogas plants, feedstock production on the soils and returning of the remaining digestates to the fields requires repeated applications with organic residues from the biogas plant. Doing so in the form of biochar is not expected to be accompanied by negative effects as long as the rate of biochar application is low enough to avoid the accumulation of potential harmful trace elements in the soil. However, the response of the soils to repeated applications of biochar needs to be further investigated. The field experiments on temperate agricultural soils demonstrate that charring of biogas plant residues and the subsequent application to the soils is a valuable praxis for increasing soil fertility and soil productivity. It may contribute to increased long-term stable organic carbon pools in soils and thereby reduce the loss of organic matter from agricultural soils in the form of CO_2. Although the beneficial effects of biochar application to soils are becoming more and more clear, its application is still not widespread. The main obstacle is the production of biochar, which requires (e.g., in the case of digestates as feedstock) the drying of the feedstock and the pyrolysis of the dried feedstock under controlled conditions. Until now, this process is still too expensive, and the product, biochar, may not compete with the low costs of mineral fertilizers. This situation might change if cheaper pyrolysis systems for biochar production were available and if the reduced CO_2 emissions from agricultural fields after biochar application could be monetized in the form of carbon credits.

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AUTHOR CONTRIBUTIONS

Christian Knoblauch: Conceptualization; formal analysis; funding acquisition; investigation; methodology; project administration; resources; supervision; validation; visualization; writing-original draft; writing-review and editing. **S.H. Renuka Priyadarshani:** Conceptualization; formal analysis; investigation; methodology; writing-review and editing. **Stephan M. Haefele:** Conceptualization; funding acquisition; supervision; validation; writing-review and editing. **Nicola Schröder:** Investigation; methodology; validation; writing-review and editing. **Eva-Maria Pfeiffer:** Conceptualization; project administration; resources; supervision; writing-review and editing.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

The data presented are available from the corresponding author upon reasonable request

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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