Biochar in temperate soils: opportunities and challenges1

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REVIEW

Biochar in temperate soils: opportunities and challenges[1]

Vicky Lévesque, Maren Oelbermann, and Noura Ziadi

Abstract: Biochar, a carbon (C)-rich material produced by the pyrolysis of organic residues, is frequently used as a soil amendment to enhance soil fertility and improve soil properties in tropical climates. However, in temperate agriculture, the impact of biochar on soil and plant productivity remains uncertain. The objective of this review is to give an overview of the challenges and opportunities of using biochar as an amendment in temperate soils. Among the various challenges, the type of feedstock and the conditions during pyrolysis produces biochars with different chemical and physical properties, resulting in contrasting effects on soils and crops. Furthermore, biochar aging, biochar application rates, and its co-application with mineral fertilizer and (or) organic amendments add further complexity to our understanding of the soil-amendment-plant continuum. Although its benefits on crop yield are not yet well demonstrated under field studies, other agronomic benefits of biochar in temperate agriculture have been documented. In this review, we proposed a broader view of biochar as a temperate soil amendment, moving beyond our current focus on crop productivity, and instead target its capacity to improve soil properties. We explored biochar's benefits in remediating low-productive agricultural lands and its environmental benefits through long-term C sequestration and reduced nutrient leaching while curtailing our reliance on fertilizer input. We also discussed the persistence of beneficial impacts of biochar in temperate field conditions. We concluded that biochar displays great prospective to improve soil health and its productivity, enhance plant stress resilience, mitigate greenhouse gas emissions, and restore degraded soils in temperate agriculture.

Key words: soil health, soil quality, crop resilience, carbon sequestration, biochar aging.

Résumé : Sous les tropiques, on utilise souvent le biocharbon, matériau riche en carbone issu de la pyrolyse des résidus organiques, comme amendement afin de rendre le sol plus fertile et d'en améliorer les propriétés. Les effets d'un tel amendement sur le sol et la productivité des plantes demeurent toutefois incertains dans les régions à climat tempéré. Le présent article fait un tour d'horizon des difficultés et des possibilités liées à l'usage du biocharbon comme amendement du sol dans les régions tempérées. Une des difficultés est que la nature des matières premières et les conditions de la pyrolyse engendrent un produit dont les propriétés chimiques et physiques varient, ce qui suscite des effets contrastants sur le sol et la culture. D'autre part, le vieillissement du biocharbon, le taux d'application et son usage avec un engrais minéral ou organique compliquent l'analyse du continuum sol-amendement-végétation. Bien qu'on n'en ait pas encore entièrement démontré les avantages pour le rendement agricole dans le cadre d'essais sur le terrain, d'autres bienfaits agronomiques du biocharbon ont été illustrés dans les régions à climat tempéré. Les auteurs proposent une vue plus large du biocharbon comme amendement pour le sol des climats tempérés en ne se limitant pas à la productivité des cultures, comme on le fait couramment. Ils se penchent plutôt sur sa capacité à rehausser les propriétés du sol. Ainsi, ils examinent les bienfaits du biocharbon sur les terres agricoles peu productives et les avantages d'une séquestration prolongée

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du carbone ainsi que d'une plus faible lixiviation des oligoéléments pour l'environnement, auxquels s'ajoutent ceux d'une moins grande utilisation des engrais. Les auteurs parlent aussi de la persistance des bienfaits du biocharbon dans le sol des régions tempérées. Ils en concluent que le biocharbon s'avère très prometteur pour améliorer la vitalité et la productivité du sol, rendre les plantes plus résilientes aux stress, réduire les émissions de gaz à effet de serre et restaurer les sols dégradés par l'agriculture en climat tempéré. [Traduit par la Rédaction]

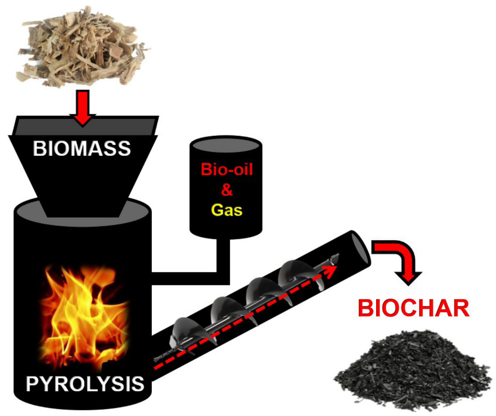
Mots-clés : vitalité du sol, qualité du sol, résilience des cultures, séquestration du carbone, vieillissement du biocharbon.

Biochar Research: A Historical Perspective

Scattered throughout the infertile soils of central Amazonia are localized patches of exceptionally fertile soil with extremely high microbiological activity. Known as Terra Preta ("do Indio"), these dark soils are particularly rich in carbon (C) compounds and minerals. They contain three times more organic matter (OM), nitrogen (N), phosphorus (P), and calcium (Ca) and 70 times more C than the surrounding soil (Glaser 2007; Barrow 2012). These Anthrosols are thought to have formed several thousand years ago from the carbonized residue (charcoal), excrement, bones, and organic waste generated by semi-intensive livestock farming activities (Glaser 2007; Barrow 2012). Over time, they evolved from an extremely poor soil to a soil with tremendous potential for modern agriculture due to their high C stability (Liang et al. 2008; Barrow 2012).

Over the last three decades, scientists have become increasingly interested in the Amazonian soils not only because of the high degree of C stability and their net negative CO_2 emissions but also because of the positive effect on soil health and consequently on crop productivity. Several studies have shown that adding highly stable C compounds such as pyrolyzed organic materials (e.g., charcoal) into highly fertile soil makes it structurally comparable to Terra Preta soils (Mao et al. 2012). Biochar is chemically same as charcoal and is only distinguished by its intended use as a soil amendment and as a mechanism for C sequestration (Lehmann and Joseph 2009). Therefore, biochar is produced in the same way as charcoal; it is an organic C-rich material produced from the pyrolysis, a process of degradation of organic biomass by heat but in the absence of oxygen (Oelbermann et al. 2020). This C-rich material has the potential to change soil properties and changes the way it hosts microorganisms and plants (Warnock et al. 2007; Gul et al. 2015; Ding et al. 2016; Harter et al. 2016). Several studies worldwide have shown that biochar not only has unique properties that promote crop health and productivity and soil fertility but is also an indispensable tool for the long-term sequestration of C in soil (Glaser 2007; Chan and Xu 2009; Lehmann et al. 2009; Barrow 2012; Schimmelpfennig and Glaser 2012). However, current methods of biochar production, including pyrolysis conditions and type of biomass feedstock used, generate biochars with very different properties and, therefore, varying effects on soils and crops (Anderson et al. 2011; Gul et al. 2015; Ding et al. 2016; Liu et al. 2016). This review presents the most up-to-date information on biochar application to temperate agricultural soils to improve soil health and crop productivity. Future opportunities and challenges on the use of biochar in Canadian agricultural soils are also discussed.

Fig. 1. Biochar production process from feedstock type to biochar generation. [Colour online.]



Biochar Production and Physicochemical Properties

Different biochars can be produced depending on the origin of the feedstock (e.g., wood, agricultural waste, animal manure, and food waste) and the pyrolysis techniques (e.g., fast versus slow) (Brewer et al. 2011; Oelbermann et al. 2020). As the biomass breaks down, pyrolysis generates three phases of products (Fig. 1), and the quantity of each product (gas, bio-oil, and biochar) varies with the pyrolysis method used (Tomczyk et al. 2020). There are several different biochar production technologies (Oelbermann et al. 2020) where slow and fast pyrolysis is the most common technique used to produce biochar for agricultural applications (Brewer et al. 2012; Bruun et al. 2012). In fast pyrolysis, the dry biomass is heated very rapidly (to more than 1000 $°C·s^{-1}$) in the absence of oxygen, and the residence time in the system is very short (about 2 s). In slow pyrolysis, the more traditional approach, dry biomass is heated very slowly (1–20 $°C·min^{-1}$) in the absence of oxygen, with a residence time from hours to days (Dutta et al. 2012). The goal in fast pyrolysis is to maximize bio-oil

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production, whereas in slow pyrolysis is to maximize biochar production. For instance, Oelbermann et al. (2020) reported that under fast pyrolysis conditions [i.e., intermediate temperatures (450–550 °C), faster heating rates (100–500 °C·s^{-1}), and short vapor residence times (<1–2 s)], biochar yields are typically of the order of 15%–20% and bio-oil yields up to 70%. With slow or intermediate pyrolysis [i.e., moderate temperatures (300–450 °C), slower heating rates (~1 °C·s^{-1}) and longer vapor residence times (>5–10 s)], biochar yields are the order of 25%–40% and bio-oil yields vary between 40% and 50% with the balance being gas.

The physicochemical properties and quality of biochar are then affected by the feedstock source, fast or slow pyrolysis, and resulting biochar particle size. Indeed, Table 1 shows several studies since 2014 to 2021 from temperate climates where the physicochemical properties of biochars are affected by feedstock source and pyrolysis conditions. Despite the efforts made in the last decade on the characterization of biochars and how they were produced, only a few information is still disclosed about that in the literature (Table 1). This lack of information confuses our decisions on the use of biochar in temperate agriculture. Therefore, there is a critical need to standardize the characterization of biochars to better understand the physicochemical properties of them according to the feedstock and pyrolysis selected, and develop recommendations for temperate agricultural soils. Based on the current information we have, the crucial physicochemical properties to consider, when using biochar as a temperate soil amendment, include its porosity, specific surface area, water-holding capacity, pH, electrical conductivity (EC), cation-exchange capacity (CEC), and the type and concentration of mineral and toxic compounds (Kloss et al. 2014; Haider et al. 2017; Brassard et al. 2019; Oelbermann et al. 2020).

Physical properties: porosity, surface area, and water-holding capacity

The porosity of a biochar is influenced by the temperature used during the pyrolysis. Dutta et al. (2012) reported that biochars produced between 350 and 400 °C had a higher total porosity than biochars produced at 300 °C. Similarly, Bagreev et al. (2001) found that a pyrolysis temperature between 400 and 600 °C increased biochar porosity considerably. The increased porosity is thought to be due to an increase in the release of water molecules following hydroxylation at high temperatures (Bagreev et al. 2001). In addition, the residence time of the biomass in the pyrolyzer may also impact biochar porosity (Novak et al. 2009*a*). For example, slow pyrolysis can cause the release of volatile OM, hemicellulose, and lignin, together with shrinkage, melting, and cracking, which improve the porosity of the biochar (Batista et al. 2018).

The specific surface area of an adsorbent defined as the surface area per unit mass (m^2·g^{-1}) is created by the biochar's micropores and mesopores. The larger the specific surface area, the larger the contact area and amount of material adsorbed. This parameter is obtained by applying the Brunauer–Emmett–Teller theory (Schimmelpfennig and Glaser 2012). The specific surface area of a biochar varies greatly with the pyrolysis temperature and conditions. For instance, Bruun et al. (2012) reported that a wheat straw biochar produced by slow pyrolysis (6 °C·min^{-1}, final temperature of 525 °C maintained for 2 h) had a lower specific surface area (0.6 m^2·g^{-1}) than a wheat straw biochar produced by fast pyrolysis (250–1000 °C·s^{-1}, final temperature of 525 °C maintained for a few seconds, 1.6 m^2·g^{-1}). Novak et al. (2009*a*) and Tomczyk et al. (2020) also found that raising the temperature caused an increase in the specific surface area. The specific surface area was also influenced by the nature of the biomass from which the biochar was produced (Tomczyk et al. 2020). For a given temperature (between 600 and 650 °C), biochar made from cow and pig manure, buckwheat husk, mulberry wood, and peanut shells had a specific surface area below 30 m^2·g^{-1}, whereas biochar made from bamboo had a specific surface area of 470 m^2·g^{-1} (Tomczyk et al. 2020). The type of feedstock can release more volatile matter during the production of biochar and then create more pores. In addition, the content of lignin and cellulose in biomass can also influence the specific surface area (Tomczyk et al. 2020). According to Schimmelpfennig and Glaser (2012), biochar with a specific surface area above 100 m^2·g^{-1} would be suitable for soil amendment and C sequestration.

The organic oxygen (O) content of biochar is useful to determine its stability and reactivity, and the O/C ratio of biochar is a potential indicator of its hydrophilicity and polarity (Schimmelpfennig and Glaser 2012). In general, as the pyrolysis temperature rises, the resulting biochar has a low O content due to its volatilization, resulting in a low O/C ratio. An increase in pyrolysis temperature can reduce the surface polarity of biochar, and thus, reduce its water-holding capacity (Wang et al. 2006). However, Kinney et al. (2012) observed low hydrophobicity in three different biochars pyrolyzed at temperatures between 400 and 600 °C. The nature of organic O could explain its low correlation between O/C ratio, pyrolysis temperature, and its hydrophobicity (Kinney et al. 2012). Bakshi et al. (2020) reported that increasing the pyrolysis temperature favored the elimination of H and O over C from the organic phase driving the elimination of H and O towards completion and promoted the formation of inorganic phases, particularly carbonates, which contain both O and C. In this case, the organic O converted to inorganic O (newly formed carbonates) would result in underestimation of the organic O content of biochars (Bakshi et al. 2020).

Chemical properties: pH, EC, and CEC

The type of feedstock used influences the pH of the biochar, which can range from 4 to 12

Table 1. Physicochemical properties of various biochar produced under temperate regions between years 2014 and 2021.

Biochar feedstock type	Pyrolysis temperature	Pyrolysis conditions	Particle size	Bulk density ($g \cdot cm^{-3}$)	Specific surface area ($m^2 \cdot g^{-1}$)	pH	EC ($mS \cdot cm^{-1}$)	CEC ($cmol_c \cdot kg^{-1}$)	Total C (%)	Total N (%)	P ($g \cdot kg^{-1}$)	K ($g \cdot kg^{-1}$)	Reference
Softwood chips of Yellow pine	500 °C	Slow	NA	0.19	NA	9.0	—	NA	NA	0.22	NA	NA	Ashiq et al. 2020
Softwood chips of Pine	500 °C	Slow	<0.15–2.0 mm	NA	22	8.3	0.3	NA	NA	NA	NA	NA	Backer et al. 2016
Poultry manure	200 °C	Slow and by batch	NA	NA	NA	7.2	8.6	58.0	39.7	3.53	33.9	10.4	Bavariani et al. 2019
	300 °C		NA	NA	NA	7.3	9.0	69.0	42.4	3.80	41.3	12.6	
	400 °C		NA	NA	NA	10.0	15.3	75.0	47.9	4.70	55.9	17.2	
	500 °C		NA	NA	NA	10.5	18.9	86.5	55.1	4.50	63.8	19.7	
Hardwood chips of Hornbeam, Beech and Oak	400 °C	Slow and by batch	<4 mm	0.30	121.6	8.4	NA	0.9	79.0	0.98	NA	NA	Bidar et al. 2019
Wood chips mixture of Oak, Elm, Hickory	600 °C	Gasification	0.1–2000 mm	NA	NA	8.8	NA	NA	63.0	0.60	0.6	8.6	Bonin et al. 2018
Wood pellets made from a mixture of Black spruce and Jack pine	516 °C	Fast	NA	NA	94.2	6.8	NA	NA	71.6	0.14	NA	NA	Brassard et al. 2018
	644 °C		NA	NA	138.1	7.6	NA	NA	80.0	0.17	NA	NA	
Switchgrass	459 °C		NA	NA	108.7	6.4	NA	NA	67.1	0.64	NA	NA	
	591 °C		NA	NA	133.2	8.8	NA	NA	79.9	0.80	NA	NA	
Solid fraction of pig manure	526 °C		NA	NA	70.9	8.6	NA	NA	51.5	4.40	NA	NA	
	630 °C		NA	NA	65.1	9.3	NA	NA	49.2	4.10	NA	NA	
Wood chips mixture of Beech and Pine	700–800 °C	Slow and by batch	<5 mm	NA	NA	8.7	NA	72.7	84.3	0.40	NA	NA	Cooper et al. 2020
Wood chips mixture of Maple, Oak, and Birch	450 °C	Fast and continuous	<2 mm	NA	NA	7.4	NA	NA	61.7	0.24	0.2	1.7	Dil et al. 2014
Corn straw	500 °C	Fast by batch	<0.25 mm	NA	NA	9.2	NA	NA	60.0	1.45	8.2	14.5	Fan et al. 2020
Wood chips mixture of 80% Douglas fir, 15% White fir, and 5% Western red cedar	450–550 °C	Slow and by batch	<5 mm	NA	NA	NA	NA	NA	69.5	0.10	NA	NA	Gao et al. 2017
Walnut shell	900 °C	Slow and continuous	54.4% >2 mm; 29.4% 0.25–2 mm; and 15.3% <0.25 mm	NA	227	9.7	NA	33.4	55.3	0.47	6.4	93.2	Griffin et al. 2017
Softwood chips of Pine and Spruce	300 °C	Flash pyrolysis	>2 mm	NA	17	8.2	NA	39.5	52.3	NA	NA	NA	Gruss et al. 2019
Wood chips mixture of Norway spruce and European Beech	550–600 °C	Slow and continuous	0.5% >6.3 mm; 31.3% 2–6.3 mm; 42.8% 0.63–2 mm; and 25.4% <0.63 mm	NA	NA	9.0	NA	19.1	74.4	0.56	1.6	6.1	Haider et al. 2017
Softwood chips of Willow	700–750 °C	Slow and continuous	80% >5 mm and 20% <0.05 mm	0.70	175	9.7	NA	20.0	81.3	0.70	NA	NA	Hangs et al. 2016

Table 1. (*continued*).

Biochar feedstock type	Pyrolysis temperature	Pyrolysis conditions	Particle size	Bulk density ($g \cdot cm^{-3}$)	Specific surface area ($m^2 \cdot g^{-1}$)	pH	EC ($mS \cdot cm^{-1}$)	CEC ($cmol_c \cdot kg^{-1}$)	Total C (%)	Total N (%)	P ($g \cdot kg^{-1}$)	K ($g \cdot kg^{-1}$)	Reference
Green waste	700 °C	Slow	NA	NA	303	9.8	0.3	10.3	51.9	0.59	NA	NA	Harter et al. 2016
Wheat straw	525 °C	Slow and by batch	<2 mm	NA	12.3	9.7	5.2	14.9	NA	NA	NA	NA	Kloss et al. 2014
Vineyard pruning	400 °C		<2 mm	NA	1.7	8.3	1.5	12.4	NA	NA	NA	NA	
	525 °C		<2 mm	NA	4.9	8.8	1.1	7.9	NA	NA	NA	NA	
Hardwood bark of Maple	400 °C	Slow and continuous	0.7% >2 mm; 67.6% 0.25–2 mm; and 31.7% <0.25 mm	0.42	NA	10.1	0.6	53.5	59.2	1.02	1.0	7.9	Lévesque et al. 2018
	550 °C		0.3% >2 mm; 58.1% 0.25–2 mm; and 41.6% <0.25 mm	0.42	NA	11.3	1.4	62.6	54.6	0.93	1.4	11.1	
	700 °C		0.4% >2 mm; 58.9% 0.25–2 mm; and 40.7% <0.25 mm	0.39	NA	11.1	1.1	60.5	54.0	0.63	1.1	9.0	
Softwood chips of Pine	700 °C	Slow and continuous	0.6% >2 mm; 67.5% 0.25–2 mm; and 31.9% <0.25 mm	0.17	NA	7.4	0.1	92.2	76.1	1.24	0.4	2.5	
Softwood chips of Willow	400 °C		2.8% >1 mm; 19.9% 0.25–1 mm; and 77.3% <0.25 mm	0.26	NA	8.2	0.4	58.4	74.5	0.78	3.8	11.9	
Tall fescue	500 °C	Slow and by batch	NA	NA	7.3	9.6	NA	NA	50.5	4.54	NA	NA	Li et al. 2021
Wheat straw	550 °C	Slow and by batch	NA	0.45	NA	9.0	NA	21.7	NA	0.59	14.4	NA	Liu et al. 2020
Softwood chips of Pine and Spruce	550 °C	Slow	37.1% >4.75 mm; 48.3% 2–4.75 mm; and 14.6% <1.00 mm	0.14	265	7.2	0.2	NA	80.0	0.47	0.3	3.0	Mechler et al. 2018
Wood chips of Norway spruce	650 °C	Slow	5–10 mm	NA	NA	8.9	NA	NA	60.6	0.29	NA	NA	Palviainen et al. 2020
Pine saw dust	350 °C	Slow and continuous	<0.5 mm	NA	3.4	5.8	0.6	56.1	52.3	0.15	NA	NA	Pariyar et al. 2020
	450 °C		<0.5 mm	NA	179.8	6.3	1.1	52.4	58.2	0.16	NA	NA	
	450 °C		<0.5 mm	NA	431.9	6.7	1.5	47.4	59.2	0.51	NA	NA	
	650 °C		<0.5 mm	NA	443.8	6.8	2.3	39.2	62.9	0.18	NA	NA	
Food waste from kitchen	450 °C		<0.5 mm	NA	0.2	9.5	12.3	22.1	62.1	2.81	NA	NA	
	550 °C		<0.5 mm	NA	2.1	9.9	20.1	17.4	63.1	2.77	NA	NA	
Poultry litter	350 °C		<0.5 mm	NA	1.8	6.3	9.3	67.2	36.8	6.31	NA	NA	
	450 °C		<0.5 mm	NA	5.7	9.5	9.0	53.5	37.2	5.14	NA	NA	
	550 °C		<0.5 mm	NA	18.1	10.0	12.6	51.5	40.8	4.28	NA	NA	
	650 °C		<0.5 mm	NA	25.3	10.1	12.7	49.7	41.4	3.52	NA	NA	
Paper mill sludge	350 °C		<0.5 mm	NA	3.2	6.3	0.6	54.3	25.3	0.22	NA	NA	
	450 °C		< 0.5 mm	NA	14.9	6.6	1.0	51.6	27.0	0.25	NA	NA	
	450 °C		<0.5 mm	NA	61.5	8.7	1.3	48.4	28.9	0.23	NA	NA	
	650 °C		<0.5 mm	NA	87.2	10.3	2.2	49.4	29.0	0.20	NA	NA	

Table 1. (*concluded*).

Biochar feedstock type	Pyrolysis temperature	Pyrolysis conditions	Particle size	Bulk density ($g \cdot cm^{-3}$)	Specific surface area ($m^2 \cdot g^{-1}$)	pH	EC ($mS \cdot cm^{-1}$)	CEC ($cmol_c \cdot kg^{-1}$)	Total C (%)	Total N (%)	P ($g \cdot kg^{-1}$)	K ($g \cdot kg^{-1}$)	Reference
Cattle manure and silage digestate	250 °C	Slow and by batch	NA	NA	1.4	7.9	1.2	NA	52.2	1.90	4.5	10.3	Pituello et al. 2015
	350 °C		NA	NA	2.2	8.6	0.7	NA	60.7	2.60	7.3	17.1	
	450 °C		NA	NA	2.9	10.3	1.1	NA	63.2	2.20	7.7	20.4	
	550 °C		NA	NA	58.6	10.3	1.7	NA	65.9	2.20	11.3	23.2	
Municipal organic waste digestate	250 °C		NA	NA	0.7	7.2	1.2	NA	33.7	4.10	17.8	5.6	
	350 °C		NA	NA	5.6	7.6	0.3	NA	34.8	4.00	20.5	6.5	
	450 °C		NA	NA	27.3	7.4	0.4	NA	29.4	3.00	24.7	8.0	
	550 °C		NA	NA	77.7	7.1	0.8	NA	26.2	2.70	26.1	8.7	
Dry poultry litter	250 °C		NA	NA	0.5	6.9	0.4	NA	43.7	4.10	8.7	22.3	
	350 °C		NA	NA	0.9	8.0	0.4	NA	51.2	5.60	14.4	37.7	
	450 °C		NA	NA	2.4	9.9	0.4	NA	51.2	4.50	16.4	43.2	
	550 °C		NA	NA	3.6	10.2	0.4	NA	51.1	3.70	19.4	48.0	
Vineyard pruning residues	250 °C		NA	NA	0.5	6.0	1.0	NA	48.7	0.90	1.0	4.6	
	350 °C		NA	NA	1.3	6.8	0.6	NA	65.9	1.30	2.0	9.7	
	450 °C		NA	NA	1.1	9.0	0.5	NA	69.3	1.30	2.4	11.7	
	550 °C		NA	NA	19.2	9.7	0.5	NA	75.1	1.30	3.0	14.2	
Sewage sludge digestate	250 °C		NA	NA	0.8	6.9	0.7	NA	28.3	3.80	17.3	4.2	
	350 °C		NA	NA	2.0	7.3	0.1	NA	27.5	3.60	19.6	4.4	
	450 °C		NA	NA	7.2	7.2	0.1	NA	22.5	2.80	21.9	4.9	
	550 °C		NA	NA	12.7	7.1	0.2	NA	20.1	2.30	23.6	4.2	
Wood chips mixture of Ash, Beech, and Oak	450 °C	Slow and by batch	<5 mm	0.46	39.0	10.4	4.8	12.9	84.3	0.58	1.1	4.2	Reed et al. 2017
Miscanthus straw	550–600 °C	Slow and continuous	NA	NA	864.2	10.1	2.4	NA	60.8	0.40	<0.1	<0.1	Rex et al. 2015
Mixed conifers	750 °C	Gasification	1.8% > 4 mm; 43.9% 2–4 mm; 49.8% 1–2 mm; and 4.6% < 1 mm	0.79	554.0	8.5	1.2	NA	NA	0.94	0.1	NA	Sales et al. 2020
Wood pellets made from Oak	200 °C	Slow and by batch	NA	NA	NA	4.6	NA	54.2	48.8	0.20	0.3	1.3	Zhang et al. 2015
	400 °C		NA	NA	NA	6.9	NA	61.1	42.7	0.30	0.6	3.8	
	600 °C		NA	NA	NA	9.5	NA	97.0	45.5	0.40	0.6	4.4	
Solid anaerobic digestate (a blend of dairy manure, eggplants, peppers, grape vines, tulip bulbs, animal bedding, and corn residues)	250 °C	Slow and by batch	13.7% <10 mm; 25.8% 2–4 mm; 33.2% 0.5–2 mm; and 21.2% <0.5 mm	NA	NA	7.1	1.5	NA	52.2	4.75	NA	NA	Y. Zhou et al. 2017
	350 °C		8.7% <10 mm; 23.1% 2–4 mm; 34.5% 0.5–2 mm; and 28.7% <0.5 mm	NA	NA	9.0	1.8	NA	49.6	4.33	NA	NA	
	450 °C		11.7% < 10 mm; 28.7% 2–4 mm; 32.2% 0.5–2 mm; and 22.8% <0.5 mm	NA	NA	10.0	1.6	NA	59.2	4.81	NA	NA	
	550 °C		19.8% <10 mm; 30.7% 2–4 mm; 28.1% 0.5–2 mm; and 19.3% <0.5 mm	NA	NA	10.0	3.0	NA	59.3	3.17	NA	NA	

Note: NA, not available.

(Cheng et al. 2006; Lehmann 2007, Rogovska et al. 2012). For example, biochar produced from wood, a highly ligneous material, typically has a higher pH than a biochar produced from crop residues (Singh et al. 2015). In their review paper, Ding et al. (2016), however, reported no effect of lignin content on biochar pH. Rather, biochar pH is usually proportional to the pyrolysis temperature for each type of feedstock (Ding et al. 2016; Pariyar et al. 2020). For instance, Novak et al. (2009*a*) characterized the physicochemical properties of biochars made from peanut hulls, poultry litter, pecan shells, or switch grass and pyrolyzed at temperatures ranging from 250 to 700 °C. They found that biochar produced at a high temperature (700 °C) had a higher pH than the one produced at a low temperature (250 °C) (Novak et al. 2009*a*). Rajkovich et al. (2012) evaluated 32 different biochar types and found an increase by two pH units between biochars produced at 300 °C and those produced at 600 °C. The increase in the pyrolysis temperature also increased the non-pyrolyzed inorganic elements in the feedstock (Novak et al. 2009*a*) and promoted ash production which increased base cation (Na^+, K^+, Mg^{2+}, and Ca^{2+}) content; two factors directly correlated with pH (Singh et al. 2015). Furthermore, the method of pyrolysis (fast or slow) can influence the biochar pH. For instance, Bruun et al. (2012) observed that a biochar derived from wheat straw that was pyrolyzed slowly (rate of 6 $°C{\cdot}min^{-1}$; maximum temperature of 525 °C maintained for 2 h) had a higher pH (10.1) than a biochar pyrolyzed quickly (pH 6.8; continuous pyrolysis with a residence time of a few seconds; rate of 250–1000 $°C{\cdot}s^{-1}$).

Electrical conductivity of biochar depends more on feedstock type than pyrolysis temperature. For example, the EC of biochars of animal origin (cattle and poultry manure) and those made from corn (*Zea mays* L.) residue in the study of Rajkovich et al. (2012), was not influenced when pyrolysis temperature was increased from 300 to 600 °C. However, Pariyar et al. (2020) pyrolized pine sawdust, rice husk, poultry litter, and paper sludge at 350, 450, 550, and 650 °C and found that EC values increased with increasing temperature. They made similar observations when biochar was derived from food waste and poultry manure but did not observe this effect on biochar derived from rice husks and pine saw dust. Excluding food and paper mill waste, the EC of biochars is also generally higher in biochars of animal origin (200–500 $mS{\cdot}m^{-1}$) compared with those of plant origin (3.8–203 $mS{\cdot}m^{-1}$) (Rajkovich et al. 2012). The salt content of the feedstock also influenced the EC of the biochar (Singh et al. 2015; Bavariani et al. 2019; Pariyar et al. 2020). Bavariani et al. (2019) reported that poultry biochar with high EC may not be suitable for salt-sensitive crops, and its quality was reduced when the pyrolysis temperature was more than 300 °C, especially for use in calcareous soils due to their high salt concentrations.

Cation-exchange capacity of biochar can decrease with increasing pyrolysis temperature (Mukherjee et al. 2011; Song and Guo 2012; Banik et al. 2018; Pariyar et al. 2020), due to the oxygenation of the surface functional groups (pyran, phenolic, carboxylic, lactonic, and amine groups) (Brennan et al. 2001; Cheng et al. 2006; Joseph et al. 2010). Additionally, Banik et al. (2018) found that biochar CEC varied with the nature and distribution of O-containing functional groups on the biochar's surface. Banik et al. (2018) also determined that biochar CEC varied with the nature and distribution of O-containing functional groups on the biochar's surface. For instance, negative surface charges arising from carboxylate and phenolate functional groups dominate in biochars produced at <500 °C compared with a temperature of >700 °C, and this domination may partly explain the higher biochar CEC produced at low temperature (Banik et al. 2018). Furthermore, the CEC values of biochars showed a high and positive correlation with the O/C ratios, indicating that the presence of more hydroxyl, carboxylate, and carbonyl groups contribute to higher biochar CEC (Batista et al. 2018). Other studies also reported that biochar with the high specific surface area can further increase its CEC (Liang et al. 2006) due to the loss of volatile matter (Song and Guo 2012).

Biochar mineral composition and availability

The composition and availability of minerals in biochars can vary widely according to the raw material and pyrolysis conditions (Ding et al. 2016; Pariyar et al. 2020). For instance, Chan and Xu (2009) observed a variation in P content of biochar depending on the biomass and pyrolysis conditions used, whereas N content tended to decline as the pyrolysis temperature rises. For heavy metals, the concentration of metal mainly depends on the composition of feedstock, and their concentration can increase or remain stable with increasing temperature because these compounds are weakly volatilizing (Pariyar et al. 2020). It was reported that when the pyrolysis temperature reaches 500 °C, the biomass can lose more than half of its N and S content (Bagreev et al. 2001; Lang et al. 2005; Chan and Xu 2009). The reduction in the content of some mineral elements, such as N, was caused by the release of this volatile material trapped in the biochar with increasing pyrolysis temperature (Bagreev et al. 2001; Dutta et al. 2012). Generally, biochars derived from animal sources, food waste, or corn residues have a higher mineral content than biochars made from forest products (pine, oak, and nuts) (Rajkovich et al. 2012). In addition, Chan and Xu (2009) observed that concentrations of P and N were higher in biochars produced from animal litter than in those made from plant biomass. However, Antal and Grønli (2003) noted that the C content was higher in biochar derived from hardwood than biochar derived from crop residues or poultry litter.

At high pyrolysis temperatures, aliphatic C is converted to aromatic C. For example, when the pyrolysis temperature rises from 150 to 550 °C, the OH and CH_3 groups of OM decrease, and C=C double bonds increase (Chan and Xu 2009). Furthermore, the H/C and O/C ratios of the biochar decrease as pyrolysis temperature rises (Chan and Xu 2009). Biochars produced at high temperatures, between 500 and 700 °C, are known to be well carbonized and stable (Chan and Xu 2009). These biochars also have a low H/C ratio (<0.1) and a larger specific surface area (Chan and Xu 2009). Conversely, biochars produced at low temperatures (300 and 400 °C) are partially carbonized and less stable. In such biochars, the H/C ratio and O concentration are high, and the specific surface area is low (Chan and Xu 2009). The presence of aromatic groups leads to a reduction in the C mineralization rate and consequently to a reduction in the availability of nutrients such as N, P, and S (Chan and Xu 2009; Ameloot et al. 2013; Xiao and Yang 2013). Furthermore, some biochars can be highly recalcitrant, such as those with a high proportion of C with condensed aromatic structures. The recalcitrant nature of the biochar may be useful if the main goal is C sequestration. However, to improve soil fertility as well as to increase C sequestration, the structural groups of the biochar should be oxidizable and have a low C/N ratio (Novak et al. 2009*b*). Schimmelpfennig and Glaser (2012) recommended a biochar with an O/C ratio of <0.4, H/C ratio of <0.6, and a total C content of >15% when used as a soil amendment. Pariyar et al. (2020) concluded that biochars pyrolyzed between 550 and 650 °C were most suitable for C sequestration and as an agricultural soil amendment.

Production of toxic compounds

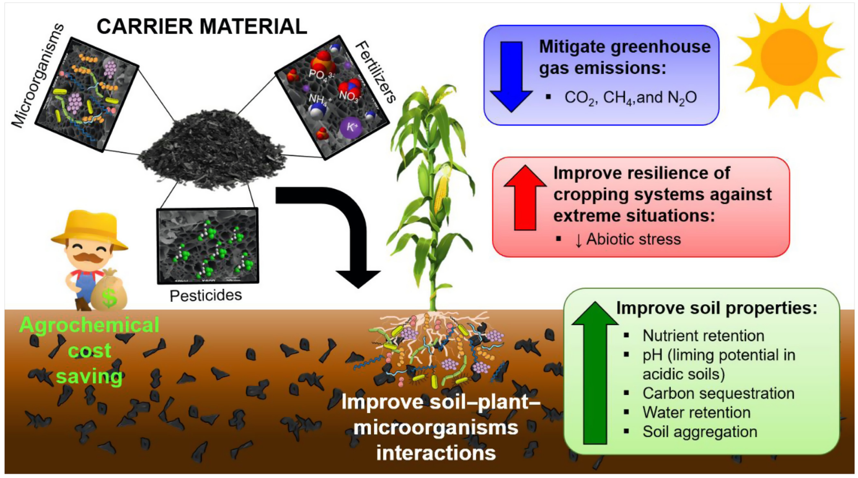
The type of biomass and the pyrolysis conditions can influence the concentration of polycyclic aromatic hydrocarbons (PAHs) and dioxins in the biochar (Brown et al. 2006; Schimmelpfennig and Glaser 2012). During pyrolysis, the organic compounds in the biomass are partially fragmented into smaller, unstable compounds. These fragments are composed of highly reactive free radicals that combine into a new, more stable compound through recombination reactions and form PAHs (Hale et al. 2012). Dioxins are mainly formed when the pyrolysis temperature is between 200 and 400 °C (Stanmore 2004), whereas PAHs are mostly generated above 700 °C (Hale et al. 2012; Manya 2012). Considering this variability, Schimmelpfennig and Glaser (2012) recommended that PAHs content in biochar must be lower than the soil it is added to when using as an agricultural amendment.

Biochar as a Soil Amendment

The effect of biochar on soil physicochemical properties and its positive outcome on soil fertility and crop productivity have been documented extensively for tropical agriculture (Ye et al. 2019). However, its impact in temperate agriculture is less impressive, and its long-term effect on soil and crops remains uncertain (Sänger et al. 2017; Ye et al. 2019). Jeffery et al. (2017*a*), in a global-scale meta-analysis, reported that biochar raised soil pH via a liming effect, which in response increased soil fertility and crop yield. This response has been observed consistently in acidic and degraded soils with low fertility in tropical environments (Jeffery et al. 2017*a*). In contrast, arable soils in temperate regions have a moderate pH and therefore a higher level of fertility compared with tropical soils (Jeffery et al. 2017*a*). Furthermore, temperate soils typically receive a source of fertilization via mineral fertilizers and (or) manure (Jeffery et al. 2017*a*). This likely explains the lower benefit of biochar in temperate agriculture (Jeffery et al. 2017*a*). Although the effect of biochar addition to tropical and temperate soils has varying outcomes, it is suggested that the management approach between the two biomes should be different (Liu et al. 2013). Blanco-Canqui (2020) synthesized which ecosystem services are or are not positively impacted by biochar amendment. According to his review, biochar benefits on ecosystem services could be in this order: C sequestration > N_2O emissions > NO_3 leaching > available water > soil biology > soil fertility > crop yields > runoff. However, the author reported that biochar does not always improve all ecosystem services and that depends on different factors such as biochar feedstock, pyrolysis temperature, application rate, properties of biochar and soil, and climatic conditions. Despite certain authors reported some benefits of biochar to improve crop productivity, several studies suggested that in temperate agriculture, biochar management should focus on non-yield benefits (Fig. 2) such as reducing nutrient leaching, controlling greenhouse gas (GHG) emissions, C sequestration, increasing soil pH, and reducing fertilizer costs (Kloss et al. 2014; Prommer et al. 2014; Haider et al. 2017; Jeffery et al. 2017*a*; Sänger et al. 2017). Other biochar agronomic benefits should also be considered (Fig. 2), including the use of biochar as a carrier material for microbial inoculants, agrochemicals (fertilizers and pesticides), or bio-fertilizers (Li et al. 2020; Sashidhar et al. 2020). Ye et al. (2019) suggested that biochar may also enhance the resilience of cropping systems against extreme climate events such as drought events (Fig. 2), although this effect might only be only evident in specific years for which longer-term field studies are required. Therefore, the use of biochar as a temperate soil amendment should thus target to improve soil health and its productivity, enhance the resilience of cropping systems to climatic change, mitigate GHG emissions, and restore degraded soils.

Regardless of soil condition zones, some positive effects of biochar on soil characteristics were reported in a few field studies of 1–4 yr of experiment length in

Fig. 2. Non-yield benefits of biochar as an agricultural soil amendment in temperate biomes. [Colour online.]



temperate environments since 2014 (Table 2). Among these field studies (Table 2), the biochar benefits on soil properties and plant productivity seem to be more important when the amount of biochar was high (9 Mg·ha^{-1}) and when it was combined with fertilizer (Table 2). However, beneficial effects of biochar seem to be a case-by-case due to the low long-term field studies available in temperate climates.

Biochar effect on soil physical properties

Biochar can improve soil physical properties through enhancing its porosity, water-holding capacity, and aggregate stability (Ding et al. 2016). Multiple and interactive factors ranging from soil and biochar properties to soil management practices, however, influence the beneficial effects of biochar amendments on soil physical properties. For example, the rate of biochar application influences bulk density, porosity, and water retention (Blanco-Canqui 2017). In general, bulk density decreases while soil porosity and water retention increase with increasing rates of biochar; however, these effects depend on the soil texture (Blanco-Canqui 2017). Soil particle size distribution is also influenced by biochar addition. For instance, coarse-textured soils (e.g., sandy soils) benefit more, compared with fine soils (e.g., clay soils), when biochar with small particles size (<2 mm diameter) was used as a soil amendment (Dil et al. 2014; Blanco-Canqui 2017). This is because the fine particles, compared with large particle size (5–8 mm diameter), of biochar can readily fill in the large pore spaces in sandy soils increasing water retention, and it may provide organic binding agents improving soil aggregation. The co-application of biochar with mineral fertilizer and (or) organic amendments may also enhance soil aggregate stability than adding biochar alone (Watts et al. 2005; Blanco-Canqui 2017). Gul et al. (2015) reported that biochars produced between 400 and 600 °C by slow pyrolysis improved soil aggregation of coarse-textured soils (e.g., sandy to sandy loam) with low OM content. Aggregation can occur when there is an interaction between OM and microbial activity (Warnock et al. 2007) because the microorganisms function as a bonding agent for aggregate formation (Watts et al. 2005). Demisie et al. (2014) observed a positive correlation between microbial biomass and macroaggregates in degraded red clay soil amended with bamboo [*Phyllostachys edulis* (Carrière) J. Houz.] and oak [*Quercus phillyraeoides* (A. Gray)] wood biochars (600 °C) in a 372 d incubation study. Although soil microbial activity and macroaggregation were similar between the bamboo and oak biochar, macroaggregate formation was superior in the soil with bamboo biochar (Demisie et al. 2014). Comparatively, a 6 yr field study showed that only biochar addition significantly increased organic C storage in aggregates and increased all aggregate size fractions ranging from <0.053 to 2 mm (Cooper et al. 2020). This suggested that biochar can increase soil organic C (SOC) sequestration in temperate biomes with inherently high fertility (Cooper et al. 2020). However, Cheng et al. (2006) found that the effect of biochar on soil aggregation was linked to its CEC and soil pH. Like soil with a high clay content, biochar with a high CEC can be conducive to soil aggregation.

Biochar effect on soil chemical properties

The beneficial effect of biochar on soil chemical properties is influenced by the type of soil to which it is applied (Unger and Killorn 2011). Results obtained after applying 10 Mg·ha^{-1} of a biochar made from pulp and paper waste to a ferrous soil in Australia improved soil quality and crop yield (Van Zwieten et al. 2010). The application of that same biochar to a calcareous soil had, however, a negative effect, particularly in the presence of mineral fertilizer. The liming value of this biochar was thus believed to be highly beneficial in acidic soils, where plant growth was restricted by a high

Table 2. The impact of biochar on soil properties, greenhouse gas (GHG) emissions, and crop productivity in temperate agricultural soils between years 2014 and 2020.

Biochar impacts	Location (country and province)	Biochar feedstock	Pyrolysis conditions	Application rate	Soil texture	Duration of the study	Major findings	Reference
Soil properties	Austria (Traismauer)	Hardwood biochar mixture (80% beech, rest derived from other hardwoods)	Slow pyrolysis at 550 °C	24 Mg·ha^{-1} 72 Mg·ha^{-1}	Sandy to loamy silt soil classified as a calcareous Chernozem on loess	2 yr	The addition of 72 Mg·ha^{-1} of biochar increased total organic C but decreased the extractable organic C pool and soil NO_3^-; The increased application rate of biochar accelerated gross nitrification rates more than twofold, reduced by 50%–80% gross rates of organic N transformation but did not affect, gross N mineralization of organic N in soil.	Prommer et al. 2014
Soil properties	Canada (Quebec)	Hardwood (Dynamotive) Softwood bark (Pyrovac) Hardwood (Basques)	>500 °C	5 Mg·ha^{-1} 10 Mg·ha^{-1}	Soil surface (0–12 cm) was a black loam above a compact gray-white-sandy loam texture	3 yr	The three biochars did not alter soil physical properties enough to change water-holding capacity, runoff volume, or infiltration rates; Runoff contained less ortho-P and particulate P in the field amended with certain biochars; Biochar increased organic C and total P content in macroaggregates >2 mm and could contribute to soil P retention.	Sachdeva et al. 2019
Soil properties	Germany (Linden)	*Miscanthus* straw	Pyreg GmbH at 550–600 °C	9.3 Mg ha^{-1}	Loamy soil classified as a Haplic Stagnolsol	2.6 yr	Biochar increased all microbial community groups (bacteria and fungi) in equal proportions, leading to a significantly higher total microbial biomass.	Rex et al. 2015
Soil properties	United Kingdom (Abergwyngregyn)	Mixture of chipped trunks and large branches of *Fraxinus excelsior* L., *Fagus sylvatica* L., and *Quercus robur* L.	Slow pyrolysis at 450 °C for 48 h	10 Mg·ha^{-1}	Sandy clay loam soil classified as an Eutric Cambisol	1 yr	Biochar promoted soil quality by enhancing nutrient availability (P and K), moisture retention, and increased total soil carbon, soil organic matter levels, and soil pH; Biochar had no effect on soil N cycling, microbial biomass, and microbial community structure.	Reed et al. 2017
Plant productivity and soil properties	Germany (Gross-Gerau)	Mixture of wood chips of Norway spruce (*Picea abies* L., 70%), and a deciduous tree European Beech (*Fagus sylvatica* L., 30%)	Pyreg GmbH at 550–600 °C	15 Mg·ha^{-1} 30 Mg·ha^{-1}	Silty sand soil	4 yr	Biochar amendments showed a higher moisture and higher NO_3^- concentration in soil, but this positive effect did not translate into higher yields; Application of large amounts of biochar to temperate sandy soils may provide environmental benefits (e.g., C sequestration and reduction of NO_3^- leaching).	Haider et al. 2017

Table 2. (*continued*).

Biochar impacts	Location (country and province)	Biochar feedstock	Pyrolysis conditions	Application rate	Soil texture	Duration of the study	Major findings	Reference
Plant productivity and soil properties	Canada (Quebec)	Pine wood (*Pinus* spp.)	500 °C for 12 min	10 Mg·ha^{-1} 20 Mg·ha^{-1}	Loamy sand soil Sandy clay loam soil	3 yr	Three years after the biochar application in loamy sand, addition of 20 t biochar·ha^{-1} increased corn yield by 14.2% compared with the control (0 t biochar·ha^{-1}); In both soil textures, biochar did not alter yield or nutrient availability in soil on soybean (*Glycine max* L.) or switchgrass (*Panicum virgatum* L.); After 3 yr, soil organic carbon (SOC) concentration was 83% and 258% higher after 10 and 20 t·ha^{-1} biochar application, respectively, than the control in sandy clay loam soil under switchgrass production; A 67% higher SOC concentration was found after 20 t·ha^{-1} biochar application in sandy clay loam under corn (*Zea mays* L.) production.	Backer et al. 2016
Plant productivity	Canada (Labrador)	Hardwood biochar	Slow pyrolysis at 550 °C	80 Mg C·ha^{-1}	Sandy soil classified as a Humo-Ferric Podzol	3 yr	Biochar combined with chemical fertilizers produced significantly higher yields of beet (*Beta vulgaris* L.) than only fertilizers; Biochar facilitated the plant uptake of both naturally occurring and added micronutrients in an acid boreal soil.	Abedin and Unc 2020
Plant productivity	Canada (Labrador)	Hardwood biochar	Slow pyrolysis at 550 °C	20 Mg C·ha^{-1}	Sandy soil classified as a Humo-Ferric Podzol	2 yr	Biochar applied to acidic, sandy soil alone cannot support crop establishment, growth, and biomass production, but biochar along with fertilizer or fishmeal boosted crop establishment, growth, and yields; Plant tissue nutrient concentrations varied depending on biochar amendment, crop, cultivars, and types of plant tissue; Biochar application in the top 0–15 cm soil layer increased soil pH and the availability of Ca, K, and Mn.	Abedin 2018

Table 2. (*concluded*).

Biochar impacts	Location (country and province)	Biochar feedstock	Pyrolysis conditions	Application rate	Soil texture	Duration of the study	Major findings	Reference
Plant productivity	United-States (Oklahomas)	Southern yellow pine (*Pinus* spp.)	NA	5 $Mg{\cdot}ha^{-1}$ 10 $Mg{\cdot}ha^{-1}$ 15 $Mg{\cdot}ha^{-1}$	Silty clay loam (Efaw site) Sandy loam (Lake Carl Blackwell site)	2 yr	Nitrogen (N) uptake and N use efficiency (NUE) under nitrogen-biochar combinations increased by 25%, 28%, and 46%, respectively, compared with nitrogen fertilizer alone, but this depends on the trial site; The combination of 10 or 15 t $biochar{\cdot}ha^{-1}$ with inorganic nitrogen fertilizer had a significant and positive response on maize (*Zea mays* L.) grain yield, N uptake, and NUE in sandy loam but not in silty clay loam.	Omara et al. 2020
Plant productivity and GHG emissions	Canada (Ontario)	Biochar mixture (50/50 mix of pine (*Pinus* spp.) and spruce (*Picea* spp.)	Slow pyrolysis at 500 °C for 15 min	3 $Mg{\cdot}ha^{-1}$	Loamy soil classified as a Grey-brown Luvisol	2 yr	Biochar combined with poultry manure and (or), fertilizer had no effect on total biomass of soybean and maize crops; Biochar influenced C and N transformations in the soil–plant–atmosphere system and caused seasonal changes in GHG emissions.	Mechler et al. 2018
GHG emissions	Canada (Labrador)	Pine wood (*Pinus* spp.)	Slow pyrolysis at 500°C for 30 min	20 $Mg{\cdot}ha^{-1}$	Sandy soil classified as an Orthic Humo-Ferric Podzol	2 yr	Biochar showed great potential to offset the negative effects of dairy manure applications on GHG emissions from a silage corn cropping system.	Ashiq et al. 2020

Note: NA, not available.

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aluminum (Al^{3+}) concentration. In contrast, the liming value of biochar seems, however, to be less beneficial for calcareous soils (Van Zwieten et al. 2010). Some alkaline biochars can contain high concentrations of salts and cause an increase in salinity in calcareous soils, potentially inhibiting plant growth and microbial activity (Van Zwieten et al. 2010; Hussain et al. 2016).

Although biochar amendments can improve soil fertility and N, P, K, and total C content (Biederman and Harpole 2013; Mukherjee et al. 2014), this C-rich material is generally not considered as a fertilizer source. The co-application with an organic or inorganic source of fertilizer is often recommended to increase the availability of nutrients in agricultural soil (Abedin 2018; Sánchez-Monedero et al. 2019; Abedin and Unc 2020; Manirakiza et al. 2020; Omara et al. 2020; Ziadi et al. 2020). For example, Abedin (2018) added biochar derived from hardwood at a rate of 20 Mg C·ha^{-1} with half or the full recommended dose of fertilizers and fishmeal. He found that treatments with biochar increased the availability of Ca, K, and Mn, and the soil's pH by 0.5 units, but he reported that biochar alone cannot support crop establishment, growth, and crop productivity due to the low nutrient content of the biochar used in this study (Abedin 2018). In a field study, Prommer et al. (2014) investigated the effects of hardwood biochar (80% beech, rest derived from other hardwoods) made at 500 °C and applied at three different rates (0, 24, and 72 Mg·ha^{-1}) in a calcareous Chernozem soil in Austria, on soil C and N dynamics. They found a significant increase of SOC in biochar-amended soil and accelerated rates of gross nitrification, suggesting that biochar with N fertilizer may increase soil organic N and enhance soil C sequestration in temperate soil. According to a meta-analysis on the effects of biochar on soil available inorganic N, Nguyen et al. (2017) reported that plant-derived biochars with higher carbohydrate content should be combined with organic fertilizer to offset the diminution of available sources of N (e.g., NH_4^+ and NO_3^-). They also recommended when applying a high quantity of biochar to soil, it should be combined with N fertilizer to minimize N immobilization. In addition, Nguyen et al. (2017) suggested that biochar should be applied 1 mo prior to the sowing date to avoid N deficiency.

Phosphorus, a highly mobile nutrient, can prone to runoff during repeated precipitation events. However, biochar can mitigate this effect. A field study using a portable rainfall simulator on a poorly drained podzolic soil amended with three different biochars at two application rates (5 and 10 Mg·ha^{-1}) showed runoff contained less ortho-P and particulate P in treatments amended with biochar, confirming that biochar could indirectly contribute to soil P retention by improving the formation of macroaggregation (Sachdeva et al. 2019). Furthermore, these authors found that biochar-amended soil presented a greater microaggregate stability and a higher total P content in macroaggregates >2 mm compared with soil without biochar.

Several studies also proposed to use biochar as slow-release fertilizer as it has the capacity to retain nutrients on its surface (El Sharkawi et al. 2018; Dong et al. 2020; Liao et al. 2020). This strategy charges biochar with mineral elements, such as N and P. Once charged biochar applied to the soil, it can prevent nutrient loss by leaching and improve crop nutrient-use efficiency (El Sharkawi et al. 2018; Dong et al. 2020). This strategy could also limit N losses through N_2O emissions by increasing the abundance of *nosZ* reducers in soil amended with biochar (Liao et al. 2020).

Biochar effect on soil biological properties

Biochar can cause major changes to the composition of microbial communities, including mycorrhiza, enzyme activity, and other soil organisms at higher trophic levels (Warnock et al. 2007; Anderson et al. 2011; Ning et al. 2019; Lévesque et al. 2020*a*, 2020*b*; Liu et al. 2020; Manirakiza et al. 2021). The biochar mechanisms interact with microorganisms in the soil are, however, complex, and this will depend on the pyrolysis conditions, biochar composition, and soil properties (Gorovtsov et al. 2020). Overall, the proportion of labile C in the biochar, its porosity, pH, volatile organic compounds, and adsorption capacity are often reported as key factors affecting positively or negatively soil biological processes (H. Zhang et al. 2014; Gruss et al. 2019; Fan et al. 2020; Gorovtsov et al. 2020; Lévesque et al. 2020*a*). According to Anderson et al. (2011), the addition of biochar can also influence the growth of microorganisms involved in soil C, N, and P cycling. Related to that, it is reported biochar could limit the use of C by microorganisms but promote their growth due to the co-location of various nutrients in and around biochar particles (H. Zhou et al. 2017). Biochar can also reduce temporal variability in microbial growth, and thereby reduce temporal fluctuations in C and N dynamics during the growing season. Indeed, a field study showed after four consecutive years of applying biochar (crushed corncob pyrolized at 360 °C) at a rate of 9 Mg·ha^{-1}·yr^{-1} to a sandy loam, significantly increased microbial biomass of C (MBC) compared with the soil without biochar (Q.-Z. Zhang et al. 2014). Additionally, the MBC in soil amended with biochar had fewer seasonal fluctuations compared with soil without biochar, suggesting that biochar provides a more suitable habitat for soil microorganisms throughout the season (Q.-Z. Zhang et al. 2014). Rex et al. (2015) reported that adding biochar derived from *Miscanthus* straw (pyrolized between 550 and 600 °C) to a loam soil at 9.3 Mg·ha^{-1} increased MBC in a grassland evaluated over 2.6 yr. These authors noted that the presence of biochar stimulated the soil microbial activity which helped improve soil aggregation and increased SOC content over the long term.

The pH and bioavailability of certain nutrients such as C, N, P, and Na^+ in biochar-amended soil can, however, adversely influence microbial populations, and some biochars can release toxic compounds to the soil fauna (Warnock et al. 2007; Ding et al. 2016; Gruss et al. 2019; Gorovtsov et al. 2020). For instance, the addition of biochars in soil with high pH can accelerate the mineralization of organic substances and CO_2 emissions, and those substances may have negative effects on microbial soil communities (Gorovtsov et al. 2020). In addition, PAHs formed during biochar production could increase the antibiotic resistance of the soil bacteria due to the horizontal transfer of PAH degradation genes (Gorovtsov et al. 2020). The International Biochar Initiative (IBI 2015) established standards for using biochar as a soil amendment and proposes a maximum concentration of hydrocarbons (PAHs, dioxin, and ethylene) and heavy metals to avoid negative effects on the soil's biology.

The application rates and size fractions of biochar could have a detrimental effect on multitrophic levels of soil fauna (Liu et al. 2020). For instance, biochar pore size distribution could play a role in preselecting the survival of some soil biota. Furthermore, macropores (diameters from 10 to 30 μm) could be a suitable size to accommodate bacteria (size range from 0.3 to 3 μm), flagellates (size range from 8 to 15 μm), some species of fungi (size range from 2 to 80 μm), and protozoa (size range from 7 to 30 μm) by providing a refuge to escape their predators (Warnock et al. 2007; Gul et al. 2015; Liu et al. 2020). But for nematodes (size range from 60 to 6000 μm) and amoebae (size range from 250 to 750 μm), macropores seem to be inaccessible due to their larger size according to the study of Liu et al. (2020). These authors hypothesized that adding biochar to soil could disrupt the soil food web due to a shortage of available food resources and reduce the abundance of some populations of soil fauna. Another study observed that biochars with a small particle size (<1 mm) had a stronger capacity to modify soil microbial community structure by promoting soil microbial groups, such as fungi and Gram-negative bacteria, due to their higher involvement in intense processes (e.g., nutrient release and mineralization) compared with biochar with a coarse particle size (2.5–5 mm) (Zhao et al. 2020). In addition, when biochar was applied with mineral fertilizer, the particle size effect of biochar on soil microbial communities was diminished or disappeared because the effect of biochar was masked by the soil acidification associated with the use of N-based fertilizers (Zhao et al. 2020).

The impact of physicochemical properties of biochar also showed varying effects on soil micro- and macrofauna community composition (McCormack et al. 2013; Domene et al. 2015; Liu et al. 2020). Abujabhah et al. (2016) found a higher population of nematodes in a sandy-loam soil amended with biochar derived from tree green waste (550 °C) compared with a soil without biochar. A shift in the community composition of eukaryotes may be associated with an improvement in macroporosity and bioturbation in soils with biochar. McCormack et al. (2013), however, reported in their review paper that large biochar particles negatively affected earthworms and enchytraeids because these organisms were not able to ingest large particles. They also reported that these particles were less likely to be consumed, transported, or aggregated by bioturbators (McCormack et al. 2013). In addition, Liu et al. (2020) observed that biochar with a high pH (pH = 8.95) decreased amoebae and nematode abundance in a cultivated acidic soil. They suggested that more studies are required to determine how biochar affects soil fauna and emphasized the need for a comprehensive evaluation of biochar from multidimensional perspectives to evaluate if whether biochar is a promising soil amendment.

Biochar effect on plant–soil–microorganism interactions

Soil microbes play an important role in plant growth and development, particularly those living in the rhizosphere and mycorrhizosphere, and biochar could help to improve their role on plant productivity. Indeed, Sashidhar et al. (2020) found that biochar had beneficial effects on the interactions between plant, soil, and microorganisms. These plant–soil–microorganism interactions limited mineral fertilizer and pesticide input, consequently resulting in a sustainable approach to temperate agriculture. For example, the combination of arbuscular mycorrhizal fungi (AMF), *Pseudomonas* spp., and biochar increased root length, root surface area, and root volume of celery (*Apium graveolens* L.) plants compared with AMF and (or) *Pseudomonas* sp. alone (Ning et al. 2019). This beneficial effect increased P uptake and crop productivity (Ning et al. 2019). In a greenhouse study, Lévesque et al. (2020*b*) found that bacterial richness, due to better accessibility of C substrates, increased when biochar (maple bark and pine chips; 700 °C) was added in peat-based growing media. They also found a greater abundance of *Agrobacterium*, *Cellvibrio*, and *Streptomyces* when it was amended with biochar, and the abundance of those bacteria showed a positive correlation with plant productivity and chemical properties of the growing medium. Graber et al. (2010) also reported an increase in *Pseudomonas* spp., *Trichoderma* spp., and *Bacillus* spp. in the rhizosphere of a biochar-amended pepper (*Capsicum annuum* L.) crop as well as improved plant growth compared with soil without biochar. However, Kolton et al. (2011) observed a divergent effect on bacterial populations, where the abundance of *Bacteroidetes* increased from 12% to 30% with biochar, whereas the abundance of *Proteobacteria* decreased from 71% to 47% with biochar. Harel et al. (2012) noted an increase in the total bacterial population and the population of *Bacillus* spp. following the addition of biochar. However, no apparent effect on the development of *Pseudomonas* spp. bacteria and *Actinomycetes* spp. fungi was observed (Harel et al. 2012).

Biochar amendments can also play a major role in inducing a systemic response in the plant against pathogenic microorganisms (Elad et al. 2010; Kolton et al. 2017). For example, a transcriptomic analysis (RNA-seq) of tomato (*Solanum lycopersicum* L.) against fusarium crown and root rot disease revealed that the addition of biochar made from greenhouse pepper plant wastes and pyrolyzed at 350 °C upregulated the pathways and genes associated with plant defense and growth in a commercial growing medium of peat and tuff (Jaiswal et al. 2020). This upregulation could then partially explain the significant improvement in plant performance and disease suppression. According to a greenhouse study, Gravel et al. (2013), however, reported that biochar made from a softwood wood mixture [softwood bark of balsam fir (*Abies balsamea* (L.) P. Mill.), white spruce (*Picea glauca* (Moench) Voss), and black spruce (*Picea mariana* (P. Mill.) B.S.P.)], pyrolyzed at 475 °C and added in an organic substrate, the growth of the pathogen *Pythium ultimum* increased in geranium (*Pelargonium hortorum* Bailey), basil (*Ocimum basilicum* L.), lettuce (*Lactuca sativa* L.), and sweet pepper. Yet, no visible adverse effect on the plant was observed during the study. Another study, where the authors evaluated the effect of eight biochar amendments on the severity of soybean root rot caused by *Fusarium virguliforme* in a silty clay loam, showed no evidence of systemic and indirect effects of biochar on severity of soybean root rot (Rogovska et al. 2017). Sales et al. (2020) also found no effect on the suppression of phytophthora root rot of highbush blueberry (*Vaccinium* hybrid 'Legacy') in a sandy soil amended with a commercial biochar manufactured from mixed conifers and inoculated with *Phytophthora cinnamomic*. The root infection by *P. cinnamomi* also increased with the increased biochar application rate, and this effect could partly be explained by the organic compounds found in biochar. The authors reported that concentrations of various organic compounds in biochar can be phytotoxic and suppress pathogens at lower rates but could damage roots and increase susceptibility to disease at higher rates.

Only few studies investigated the relationship between the type of biochar, plant, and the diversity of soil microbial populations (Gul et al. 2015; Ding et al. 2016; Harter et al. 2016). Gorovtsov et al. (2020) reported that the difference in soil properties and the complexity of biochar impact on the soil can lead to contradictory data and the difficulty of comparing the results of the different studies, due to many gaps on the biochar mechanisms interacting with the soil microbial community. In addition, biochar can alter the signaling between soil microorganisms and crops, causing a disruption in the balance of the soil food web and favoring some microbial species to the detriment of other organisms (Ding et al. 2016).

Biochar effect on crop productivity

Biochar can have a positive response in crop yield in low-fertility tropical soils (Griffin et al. 2017), especially in highly weathered soils that have a net anion-exchange capacity (Sanchez 2019); in fact, biochar addition to these soils has been shown to increase soil pH (Van Zwieten et al. 2010). Indeed, a higher soil pH can decrease Al^{3+} toxicity and nutrient leaching and cause an increase in CEC and soil microbial activity leading to a positive response in crop yield (Borchard et al. 2014; Griffin et al. 2017). Although biochar is also promoted in temperate regions for soil C sequestration and soil health (Griffin et al. 2017), it is uncertain whether similar responses in crop productivity, like those observed in tropical soils, can be obtained under temperate conditions (Borchard et al. 2014). This is because temperate soils are normally not limited by a low pH or low CEC (Rajkovich et al. 2012). Accordingly, temperate soils with high activity clays, high soil OM stocks, and lower oxide contents will respond differently to biochar addition than tropical soils (Kloss et al. 2014).

Despite these obstacles, it is necessary to maximize the crop yield of fertile temperate soils through agricultural intensification to address potential food shortages associated with a rising global population and climate change (Borchard et al. 2014). Agegnehu et al. (2017) noted that productive temperate agricultural regions will also have areas of degraded soil that will benefit from incorporating biochar with mineral fertilizers and (or) organic amendments. This is because biochar has the capacity to minimize leaching of soil N and P, and instead of making these nutrients available for crop uptake (Chan and Xu 2009). However, Oelbermann et al. (2020) reported that biochar generated from the same feedstock can vary in its physical and chemical characteristics due to differences in the origin of the feedstock and (or) conditions during pyrolysis. These discrepancies hamper optimal biochar application rates (Chan and Xu 2009). Indeed, Glaser et al. (2007) concluded that optimal application rates to maximize crop productivity will likely require an individual assessment for each crop species and soil type. A larger quantity of biochar may be needed for temperate soils to achieve a positive response in crop productivity due to their inherently greater fertility compared with tropical soils (Borchard et al. 2014). Thus, most studies in temperate regions have used high biochar application rates that ranged from 5 to 50 $Mg{\cdot}ha^{-1}$ (Nielsen et al. 2018) and as high as 100 $Mg{\cdot}ha^{-1}$ (Boersma et al. 2017). Although high biochar application rates resulted in a positive response on crop yield (Jeffery et al. 2011; Liu et al. 2013), the high cost of biochar production and transport from its point of origin to its destination is not economically feasible for most agricultural producers (Dil et al. 2014;

Latawiec et al. 2019). In addition, no economic incentive for implementing biochar use in temperate agricultural soils exists, at least for the initial few years of application (Haider et al. 2017). According to a cost-benefits analysis of biochar amendment, two studies reported that land application scenarios could be economically viable if investments in efficient biochar production techniques are used, and biochar is subsidized by low-emission incentive schemes (Homagain et al. 2016; Latawiec et al. 2019). In addition, Homagain et al. (2016) suggested that biochar amendment in Canadian soils could be economically viable with 12–13 yr of break-even time if C sequestration is credited for at least \$60CAD per tonne of CO_2 equivalent gases. Latawiec et al. (2019) also highlighted the importance of considering biochar in the context of ecosystem services to better estimate biochar potential costs and benefits, and its place in the wider context of the soil contribution to human well-being.

A Canadian study evaluated the economic feasibility of converting biomass from black spruce obtained from clear-cut forests for the construction of hydroelectric dam projects, into biochar and using it as a soil amendment to grow potatoes (*Solanum tuberosum* L.) and beets (*Beta vulgaris* L.) to improve food availability in isolated northern regions (Keske et al. 2019). According to this study, if the biochar is applied every 5, 10, or 20 yr, it would require \$1693CAD $ha^{-1}{\cdot}yr^{-1}$, \$969CAD $ha^{-1}{\cdot}yr^{-1}$, and \$621CAD $ha^{-1}{\cdot}yr^{-1}$, respectively, to cover variable costs. This provides an interesting opportunity to use biochar as an amendment; it could create a local market, boost food production as well as improve health and food security of the local population. The use of biochar in temperate soils could have a beneficial impact on health and food security in located regions.

Due to the current high cost of biochar, this generated a new research direction where lower quantities of biochar were blended with organic amendments and (or) mineral fertilizer to generate a low-cost and high-efficiency fertilizer. For example, Dil et al. (2014) evaluated the impact of preconditioning biochar derived from a mix of maple (*Acer* spp.), oak (*Oak* spp.), and birch (*Betula* spp.) feedstock with urea ammonium nitrate (UAN) applied at 1 $Mg{\cdot}ha^{-1}$. They found that UAN preconditioned biochar significantly increased corn (*Zea mays* L.) biomass, corn tissue N concentration, and N uptake in a coarse- and medium-textured soil (Dil et al. 2014). Moreover, advances in pyrolysis technology have led to the development of tailor-made biochars that can address specific plant and soil constraints (G. Perez, personal communication, Washington State University).

A global meta-analysis revealed that crop yield was influenced by the biochar type and application rate, climate, soil type, crop species, and agroecosystem management practices (Boersma et al. 2017). However, meta-analyses conducted on temperate soils found minimal or no improvement in crop yield when biochar was applied alone or in combination with mineral fertilizer and (or) organic amendments (Jeffery et al. 2011; Biederman and Harpole 2013). Nielsen et al. (2018) concluded that it is more beneficial to use biochar as a soil conditioner to augment the efficacy of mineral fertilizers or organic amendments because this will likely cause a positive response in crop yield. In addition, a long-term field study evaluated the effects of softwood biochar on crop biomass yield for eight growing seasons after the incorporation into two boreal soils (Kalu et al. 2021). Even if the results showed little effect of biochar on the crop biomass yield, possible synergies of biochar addition in certain combinations with pre-crops of N-fixing legumes in the previous year might have contributed to increasing crop yield.

Overall, the beneficial properties of biochar that contribute to the fertilizer effect and potentially enhanced crop productivity are associated with its high surface area, surface charge density and CEC, intrinsic nutrient load of N, P, K, and other cations, low bulk density, high porosity, and pH (Reed et al. 2017). For example, Van Zwieten et al. (2010) found a 30%–40% increase in wheat (*Triticum aestivum* L.) height when biochar, produced from paper mill sludge, was applied at 10 $Mg{\cdot}ha^{-1}$ to an acidic soil. The positive response of wheat was due to a reduction in Al^{3+} toxicity by the carbonates in the biochar (Van Zwieten et al. 2010). However, when the same biochar was applied to soil with a neutral pH, there was no positive response in wheat yield (Van Zwieten et al. 2010). This was also observed by Mechler et al. (2018) in a field study where biochar was blended with poultry manure or with poultry manure and N fertilizer applied at 3 $Mg{\cdot}ha^{-1}$ under corn–soybean [*Glycine max* (L.) Merr.] rotation on soil with an intrinsically high carbonate content. Rajkovich et al. (2012) evaluated corn production under optimal fertilizer conditions with and without biochar in New York, United States on a fertile Alfisol (U.S. Taxonomy). They found that feedstock type and biochar nutrient content, rather than pyrolysis temperature, contributed to a greater corn yield (Rajkovich et al. 2012). In contrast to these results, Devonald (1982) and Gaskin et al. (2010) found a reduction in crop productivity due to biochar. The negative response of crops to biochar may be due to phytotoxicity, biochar heavy metal content, soil with an intrinsically high pH, or due to N immobilization after biochar addition (Rajkovich et al. 2012). Negative results on crop yield and soil are typically only observed when biochar is derived from a low-quality feedstock and (or) was generated by inconsistencies in the pyrolysis process (Oelbermann et al. 2020). Other studies who reported a positive or no response of biochar on crop yield noted that overall, biochar caused no negative effects on crops and soil (Jones et al. 2012; Dil et al. 2014; Boersma et al. 2017; Gao et al. 2017; Griffin et al. 2017).

Biochar is also used in the field and greenhouse-produced horticultural crops, but with mixed results on crop productivity (Chan et al. 2007). For example, Chan et al. (2007) found that the addition of 100 Mg·ha^{-1} green waste biochar had no significant impact on radish (*Raphanus sativus* L.) yield. However, when biochar was blended with N fertilizer, radish yield increased due to an increase in N-use efficiency (Chan et al. 2007). However, Boersma et al. (2017) did not find an increase in cauliflower (*Brassica oleracea* var. *botrytis* L.), broccoli (*Brassica oleracea* var. *italica* L.), and pea (*Pisum sativum* L.) productivity and quality when biochar from Eucalyptus (*Eucalyptus polybractea* R.R. Baker) was added at 10 Mg·ha^{-1} to a fertile soil in northwestern Tasmania, Australia. In an organic farming system in Washington State of the United States, Gao et al. (2017) found that the sandy soils in this region responded favorably to biochar, increasing squash (*Cucurbita* spp.) yield and nutrient uptake. Y. Zhou et al. (2017) found that biochar significantly increased arugula [*Eruca vesicaria* ssp. *sativa* (P. Mill.) Thellung] shoot and root biomass in southwestern Ontario, Canada. They also observed that biochar decreased N losses over the winter months which contributed to a greater arugula yield in the following spring (Y. Zhou et al. 2017).

Biochar can suppress bacterial wilt in tomato (*Lycopersicon esculentum* P. Mill.) production systems, but this does not always correspond to an increase in tomato yield. Griffin et al. (2017) investigated the effects of walnut shell biochar applied at 10 Mg·ha^{-1} to a corn–tomato rotation in a high-fertility fine-textured soil in California's Central Valley in the United States, over 4 yr. They found that fertilizer application had a greater impact on crop yield in all year than treatments with biochar (Griffin et al. 2017). However, they observed a transient effect of biochar with an increased tomato yield only in the second year of their study and attributed this to an increase in soil pH and soil-extractable K^+, Ca^{2+}, and PO_4-P (Griffin et al. 2017).

Most studies evaluated the effect of biochar on annual crops (Borchard et al. 2014; Egamberdieva et al. 2016; Mechler et al. 2018), but the impact of biochar on perennial crops (e.g., woody and grass) is sparse and with variable outcomes (Bonin et al. 2018). For example, Edmunds (2012) observed a positive effect of biochar on switchgrass (*Panicum virgatum* L.) biomass yield, and Adams et al. (2013) reported an increase in the productivity of big bluestem (*Andropogon gerardii* Vitman). However, biochar had no effect on switchgrass height and biomass yield on a fertile soil in lowa, United States (Bonin et al. 2018). In Shandon, the addition of biochar derived from lignocellulosic biomass composed of pine, poplar, and *Caragana intermedia* with a ratio of roughly 2:2:1 improved soil properties of a saline alkali soil that led to a better *Miscanthus lutarioriparius* yield due to a reduction in excess Na^+ (He et al. 2020). In a potted experiment, biochar increased perennial ryegrass (*Lolium perenne* L.) productivity at harvest only in the first year of crop establishment, which was attributed to a N fertilization effect, and no impact on yield was observed in subsequent years (Jeffery et al. 2017*b*). When a grassland in Wales, United Kingdom, was amended with hardwood biochar applied at 10 Mg·ha^{-1}, no significant increase in grass yield (*Lolium mutiflorum* L.), height, and nutritional quality was observed compared with treatments without biochar (Reed et al. 2017).

Only a few studies examined the impact of biochar on tree growth in temperate forests (Robertson et al. 2012; Sarauer et al. 2019). In Canada, Robertson et al. (2012) concluded that tree seedlings benefited from biochar during the early phases of growth. They found that seedling biomass of pine [*Pinus contorata* (Douglas)] and sitka alder (*Alnus viridis* spp. *sinuata*) were greater in treatments with biochar. In Finland, Palviainen et al. (2020) reported that Scots pine (*Pinus sylvestris* L.) had a greater height and diameter growth in treatments with biochar, and Somerville et al. (2019) observed that biochar and (or) compost increased growth of spotted gum (*Eucalyptus maculata* Hook) in urban areas in a warm temperate climate of Australia. They attributed this positive response to a better adaptation of trees to cope with drought stress, which is frequently observed in urban environments (Somerville et al. 2019). Sarauer et al. (2019) applied biochar at a rate of 0, 2.5, or 25 Mg·ha^{-1} to a forest soil in northwestern of the United States. Although they found no impact on tree growth, they concluded that biochar was effective in soil C sequestration and had no negative impact on soil and plants (Sarauer et al. 2019).

Biochar effect on GHG emissions

To mitigate the negative impacts of agriculture on climate change in temperate agricultural soils, biochar amendments showed a positive effect on the mitigation of GHG emissions. For instance, a field study with a silage corn cropping system in a boreal climate (Podzol) showed that the application of 20 Mg·ha^{-1} of biochar made from yellow pine wood and pyrolyzed at 500 °C significantly mitigated negative effects of the dairy manure application on GHG emissions (Ashiq et al. 2020). In addition, in a 6 wk incubation experiment, Hangs et al. (2016) found that the application of 20 Mg·ha^{-1} biochar derived from shrub willow (*Salix* spp.) in two contrasting marginal loam soils in Saskatchewan, Canada, decreased N_2O and CH_4 emissions from urea fertilizer. They reported that the lower availability of $NH_4{}^+$ and $NO_3{}^-$ and the better aeration in both soils amended with biochar could be involved in the mitigation of N_2O and CH_4 emissions. However, the efficiency of biochar to mitigate GHG emissions in temperate soils varies depending on biochar feedstock source, pyrolysis conditions, and soil texture (He et al. 2017; Brassard et al. 2018; Lévesque et al. 2020*a*). Furthermore, Lévesque et al. (2020*a*) found that the ability of biochars to mitigate GHG in a clay soil depends on its effect on soil physicochemical properties (such as C dynamic, porosity, and pH), and on its impact on soil microbial communities involved in the biochemical cycling of C and N. The 2 yr field study of Mechler et al. (2018) also showed that the addition of 3 Mg·ha^{-1} biochar mixture [50/50 mix of pine and spruce (*Picea* spp.)] to a loam soil in southwestern Ontario, Canada, influenced C and N transformation in the soil–plant–atmosphere system and caused seasonal changes in GHG emissions. For example, their results showed that biochar addition in the first year of the study stimulated CO_2 emissions due to the presence of high labile C compounds that had increased soil microbial activity, whereas in the subsequent seasons, CO_2 emissions decreased in the soil amended with biochar due to the depletion of available labile C sources. The stabilization of C and N in soil amended with biochar due to biochar ageing also had a positive effect on the mitigation of N_2O in the second year in the study by Mechler et al. (2018).

A global meta-analysis compiled 91 peer-reviewed publications on biochar applications under various conditions (field and controlled environments) and the impact of these applications on GHG emissions. The results showed that, on average, biochar application increased soil CO_2 emissions by 22.1%, but decreased N_2O emissions by 30.9% and had no effect on CH_4 emissions (He et al. 2017). The authors of this meta-analysis also reported that CO_2 emissions were suppressed, whereas CH_4 emissions increased and those of N_2O were not affected in soils amended with inorganic N fertilizer. Another meta-analysis by Borchard et al. (2019) compiled 88 peer-reviewed publications to evaluate the impact of biochar on N_2O emissions in temperate, semi-arid, subtropical, and tropical biomes. This study revealed that biochar resulted in a 38% reduction of N_2O emissions, where the greatest reduction of this GHG occurred immediately after biochar application. Additionally, Borchard et al. (2019) also found that biochar had the strongest effect on reducing N_2O emissions in paddy soil and sandy soil, followed by a reduction in arable farming and horticulture but had no impact on N_2O losses from grasslands and perennial crops.

The main physicochemical properties of biochar that are involved in mitigation of soil CO_2 emissions are water-holding capacity, porosity, C storage, stability, and toxic compound content (Laird et al. 2009; Jones et al. 2011; Maestrini et al. 2015; Liu et al. 2016). For example, an increase in labile C in the soil due to biochar addition can promote microbial activity and contribute to CO_2 emissions (Ameloot et al. 2013). Joseph et al. (2009) reported that there are three fractions of labile C : mineral C (water soluble); organic C (water soluble and rapidly mineralized by microorganisms); and poorly oxidizable C from the amorphous or microcrystal portion of the biochar structure. Biochar labile C is defined as the CO_2 fraction that is mineralized over a short period of time. This mineralization is believed to be triggered by abiotic or biotic effects. Biochar mineralization typically consists of two phases: rapid followed by slow mineralization. According to incubation studies, the rapid mineralization phase occurred in the first week or first few months following biochar application to the soil (Joseph et al. 2009). In addition, the rapid mineralization phase manifested especially with biochars that have been produced at low pyrolysis temperature, as their labile C content was higher than that of those produced at low pyrolysis temperature (Zhang et al. 2015; Lévesque et al. 2020*a*). Once the labile C was exhausted, it led to a reduction in CO_2 emissions in the biochar-amended soil (Case et al. 2012; Yoo and Kang 2012; Maestrini et al. 2015; Lévesque et al. 2020*a*).

Due to the high porosity of biochar and its capacity to absorb gas on its surface, biochar may increase diffusive CH_4 uptake by soil microorganisms, and thereby enhance the mitigation of CH_4 emissions to the atmosphere (Jeffery et al. 2016). Soil CH_4 emissions from biochar-amended soils, however, vary widely with the type of biochar and agricultural soils (Manya 2012; Yoo and Kang 2012; Liu et al. 2016; Lévesque et al. 2020*a*). A meta-analysis of 42 studies on the effect of biochar on CH_4 emissions, however, found that it can have an excellent potential to mitigate CH_4 emissions (Jeffery et al. 2016). For example, in anoxic environments (e.g., peat soils and flooded fields), where important sources of CH_4 are produced, biochar addition can greatly decrease the abundance, and therefore, the ratio of methanogenic to methanotrophic microorganisms, thereby lowering CH_4 emissions (Feng et al. 2012; Brassard et al. 2016). The efficiency of biochar to mitigate CH_4 emissions, however, depends on the presence of labile C sources in biochar (Jeffery et al. 2016) because this labile C fraction in biochar can serve as a methanogenic substrate that promotes CH_4 production. Furthermore, the high concentration of K in a biochar can also promote the activity of methanotrophic, but inhibit the activity of methanogenic, microorganisms (Le Mer and Roger 2001; Babu et al. 2006; Lehmann and Joseph 2009).

Several biotic mechanisms can be involved in the mitigation of N_2O emissions in biochar amended agricultural soil including (*i*) an increase in soil aeration that inhibits denitrification, (*ii*) the labile C content of the biochar promotes complete denitrification and the formation of N_2, (*iii*) an increase in pH by biochar creates an optimal environment for N_2O reductase activity, where it contributes to the formation of N_2, (*iv*) a lower availability of available N to the microorganisms involved in nitrification and (or) denitrification thereby controlling N_2O emissions, and (*v*) a release of toxic compounds inhibiting soil biological activity (Clough et al. 2013; Cayuela et al. 2014; Van Zwieten et al. 2014; Zhang et al. 2015).

Abiotic mechanisms can also control N_2O emissions in biochar-amended soils including chemical denitrification, which refers to an abiotic chemical reaction leading to the formation of nitrogen monoxide (NO),

N_2O, and N_2 (Cayuela et al. 2014). These reactions include the chemical decomposition of the following compounds: (*i*) hydroxylamine (NH_2OH), (*ii*) NO_2^-, and (*iii*) ammonium nitrate (NH_4NO_3) in the presence of light, moisture, and a reactive surface (Cayuela et al. 2014). A second plausible abiotic mechanism is N_2O adsorption to the biochar surface due to the presence of iron hydroquinone, copper, and manganese. In addition, biochar may potentially serve as an electron shuttle transporting electrons to microorganisms competing with denitrifying microorganisms, thus reducing N_2O emissions (Cayuela et al. 2013). However, biochar aging can weaken or even reverse the suppressing effects on soil N_2O emissions due to the increase of abundance of the biochar's oxidative moieties (e.g., C=O groups) (Yuan et al. 2019).

Biochar as a soil remediation method

In addition to its beneficial effect on soil fertility and quality, biochar application can be used as a valuable approach in remediating contaminated soils due to its capacity to adsorb organic and inorganic pollutants (Tang et al. 2013; Li et al. 2021). Used alone or in combination with other organic residues, biochar can reduce the mobility and the toxicity of contaminants in the soil such as heavy metals and pesticides. The efficiency of this technique, however, varied with the contaminant itself, the soil, and biochar characteristics because the involved mechanisms varied largely with those factors (Li et al. 2017; Nejad et al. 2018).

Many studies reported that biochars can reduce the phytoavailability and consequently the toxicity of metals present in contaminated soils such cadmium (Cd), chromium (Cr), lead (Pb), nickel (Ni), and zinc (Zn) throughout different mechanisms including adsorption, electrostatic interaction, complexation, and precipitation (Tang et al. 2013; Nejad et al. 2018). Furthermore, in a study conducted under controlled conditions using two contaminated soils with rye grass (*Lolium perenne*, L.), Bidar et al. (2019) concluded that wood biochar reduces the mobility and phytoavailability of Cd, Cu, Pb, and Zn. In another study, Shen et al. (2016) investigated the effects of two biochars, heavy metal-rich and -free ones, on the phytoavailability of Cd and Pb in two acidic soils under tobacco (*Nicotiana tabacum*, L.) production. They conduced that both biochars increased soil pH and consequently improved plant growth (Shen et al. 2016). However, Cd and Pb concentrations in soil and plant decreased only when heavy metal-free biochar was applied; the reverse was obtained when heavy metal-rich biochar was used. These results highlighted the importance of the quality of biochar to be used and the need for its characterization before any application.

The increase of soils contaminated by pesticides is another global environmental issue, and different biochars were used in various studies as a solution to reduce the negative effect of contamination. For instance, it was reported that the addition of biochars could be used to reduce the bioavailability and mobility of atrazine through adsorption–desorption processes which are mainly depending on soil properties (Zhao et al. 2013; Mandal et al. 2017; Cusioli et al. 2019; Ding et al. 2019). Indeed, Tao et al. (2019) highlighted the influence of soil physical and chemical characteristics, especially pH, CEC, and OM on the adsorption–desorption behavior of pesticides. These results agree with those obtained by Sun et al. (2019) who reported that the sorbed amount of atrazine was related to the OM content and pH of the three tested soils. Using five different soils ($0.58 < OM < 5.02\%$; $5.64 < CEC < 21.3\ cmol{\cdot}kg^{-1}$), Han et al. (2019) also concluded that the adsorption and desorption processes for carbendazim and thiamethoxam varied largely with soils and are positively correlated to OM and CEC. Liu et al. (2018), however, found that this mechanism was too complicated, unclear, and therefore, more studies are required to elucidate this mechanism. In addition, Gorovtsov et al. (2020) recommended further studies on biochar interactions with microorganisms in the remediation of heavy metals and organic pollutants contaminated soils, as these interactions are currently under-explored.

Biochar aging in temperate regions

Despite biochar can persist in the soil for thousands of years, and the impact of biochar on soil chemical, physical, and biological properties may be a critical factor in the plant–soil system continuum, limited information on biochar aging is available (Hardy et al., 2019; Blanco-Canqui et al. 2020; Wang et al. 2020; Kalu et al. 2021). Additionally, a limited number of long-term field studies examined the effect of biochar on crop productivity on fertile temperate agricultural soils, and most of this research was conducted under controlled laboratory or greenhouse conditions (Boersma et al. 2017). For example, a meta-analysis of 371 independent experiments in temperate and tropical regions found that the average study length was 113 d (Biederman and Harpole 2013), but only a few studies evaluated multiple crop seasons in temperate regions (Jones et al. 2012; Griffin et al. 2017; Mechler et al. 2018). The inclusion of commercial farming management practices in field studies over multiple seasons provides could thereby insight of how biochar ages in soil and how this could affect crop productivity (Griffin et al. 2017). However, the lack of long-term studies and the uncertainty surrounding biochar ageing in soil is, for now, preventing its inclusion into the agricultural policy (Jones et al. 2012). This is because once biochar is added to soil, it cannot be readily removed (Jones et al. 2012).

Wang et al. (2020) reported that when it is applied in the soil, various natural mechanisms lead to significant changes in physicochemical properties of biochar, such as freeze–thaw cycles (e.g., variations in temperature), wetting–drying cycles (e.g., rainfall events), photochemical (e.g., sunlight irradiation) degradation, and mild oxidation (e.g., atmospheric oxygen, roots exudates, or microbial activity). These modifications could have a positive or negative impact on biochar performance for field applications over time. Jones et al. (2012) also noticed that the effect of biochar on soil properties may be brief, with the greatest benefits observed only in the first year after its application. They also noted that biochar effects may persist in soil if biochar is added more frequently than one application only. Therefore, the addition of biochar should be applied more frequently than we had expected, in order to keep the desired effect of biochar in temperate soil. Indeed, Kalu et al. (2021) studied the effects of a single application of softwood biochars on two contrasting boreal agricultural soils over eight growing seasons following the application. The authors reported the initial beneficial effects of biochars on the reduction of bulk density, and improvement of plant-available water became weaker over the years. This faded effect is likely due to a downward movement of biochar in the soil profile, creating a dilution of the effect of biochar over time. Hardy et al. (2019) also found that over the long term, agroecosystem management practices had a stronger influence on the soil microbial community than biochar. However, the authors suggested that further research on the impact of biochar aging on soil biological properties is warranted. This is because biochar aging in soil is not ubiquitous among biochar types given the diversity in feedstock and pyrolysis approaches.

Because long-term in situ field trial to evaluate biochar aging is a slow process, Wang et al. (2020) proposed different artificial aging methods to help shorten the aging duration from years to months or days. The authors mentioned that these artificial pre-aging methods, such as chemical oxidation, wet–dry cycling, and mineral modification may serve to synthesize engineered biochars and improve the knowledge on the role of biochar long-term aging in agricultural ecosystems. Kalu et al. (2021) also suggested to study the properties of the field-aged biochar particles and comparing that with archived fresh biochar to facilitate in understanding the underlying mechanisms.

Conclusion

We highlighted the opportunities and challenges to using biochar in temperate soils in this review. Overall, biochar has shown multiple benefits on soil in temperate biomes, but its beneficial effect is not universal in all agroecosystems. Despite the fact that the impact of biochar on crop productivity in temperate agroecosystems is unclear, we conclude that biochar has a positive impact on soil health and productivity while showing great potential for GHG mitigation and the restoration of degraded soils. However, a lack of information associated with biochar characterization and production still exists in the literature, thereby limiting our judgment on the best type of biochar for achieving the desired ecosystems service. Additionally, a knowledge gap remains on the potential of biochar to enhance agroecosystem resilience in light of a changing environment and climate change. It is also known that the growing global population will simultaneously increase demand for food, fiber, and fuel, and that will result in greater pressure on our global natural resources and greater pressure to maintain crop productivity and food security. Therefore, long-term field studies (>3 yr) with one and more frequent biochar applications are crucial to evaluate the beneficial effects of biochar aging on agroecosystems across the atmosphere–soil–plant–microorganism continuum in temperate biomes. The usage of biochar as an agricultural amendment could then provide an opportunity to enhance agroecosystem resilience with climate change and support food security. However, current biochar production shortages result in unrealistically high costs for agricultural producers. Therefore, optimization of biochar production systems and the generation of high-quality biochar are necessary to ensure repeatability and reproducibility of biochar, necessary steps to assure agricultural producers of its benefits.

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