Turnover of Soil Carbon following Addition of Switchgrass-Derived Biochar to Four Soils

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Amending soils with biochar can sequester C and improve soil properties, such as nutrient holding capacity and water retention. While biochars generally have a long residence time in soil, the turnover of biochar C can be influenced by both biochar characteristics and soil properties. Biochar can also potentially alter the rate of decomposition of native soil organic matter (SOM). The turnover of switchgrass (*Panicum virgatum* L.)-derived biochar C was evaluated in the laboratory using soil from four marginally productive sites in central Pennsylvania. Carbon dioxide emissions from unamended soil, biochar-amended soil, and pure biochar were monitored during 189-d incubations, and data were fit to a two-pool exponential model to estimate the amount and mean residence time (MRT) of C in labile and stable pools. Carbon-13 signatures of emitted CO_2 were also determined to estimate the proportion of emitted CO_2 derived from the biochar. Mixing biochar with each of the soils reduced the apparent MRT of C in both labile and stable pools, but the magnitude of change depended on the soil. Overall, the biochar was largely stable in each soil, with only 1.1 to 2.1% of the added biochar C emitted during incubation. There was no measurable effect of biochar amendment on the turnover of native SOM in any of the soils. Therefore, we conclude that amendment of our soils with switchgrass-derived biochar can effectively increase net C sequestration.

Abbreviations: MRT, mean residence time; SOC, soil organic carbon; SOM, soil organic matter.

Biochar, the solid residuum following biomass pyrolysis, may return multiple benefits when used as a soil amendment (Glaser et al., 2002; Laird, 2008). In addition to benefits such as metal toxicity mitigation, reduced nutrient leaching, improved nutrient availability for plants, and enhanced crop production (Hass et al., 2012; Lehmann et al., 2003), biochar amendment may serve to sequester C in soils. However, the ratio of readily decomposed to recalcitrant organic compounds comprising the biochar will influence its residence time in the soil. The net effectiveness of biochar to sequester C depends on the longevity of the biochar while residing in the soil and the extent to which it influences the dynamics of native SOM.

The reported MRTs of biochar have varied among studies, with a number of factors, such as environmental conditions and biochar chemical and physical properties, appearing to influence biochar longevity. Biochar properties are determined by the parent material (soft woods, hard woods, grasses, manures, etc.) and pyrolysis conditions, including temperature and pyrolysis duration (Manya, 2012; Meyer et al., 2011). Bird et al. (1999) indicated that biochar could reside in a well-aerated tropical soil for decades to centuries. Singh et al. (2012) estimated that the MRT of biochar varied from 100 to 1,000,000 yr and that this variation probably depended on the biochar properties. They also reported that biochar produced from poultry

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manure turned over faster than biochar produced from wood. However, the influence of soil properties on the turnover rate of biochar is still not well understood.

In addition, relatively little is known about the effects of biochar on native SOM decomposition and thus turnover. The effects of biochar on native SOM decomposition have been reported to be positive, neutral, or even negative (Zimmerman et al., 2011; Spokas and Reicosky, 2009). Presumably this is due to variations in biochar chemical and physical properties, the variable effects of biochar on soil chemical and physical properties, and the variable nature of the native SOM.

Switchgrass, a perennial C_4 grass native to North America, is a potential biochar parent material. Because of its potential value as a biofuel crop (Parrish and Fike, 2005) and high productivity on many soils that are marginally suited for annual row-crop production (Keshwani and Cheng, 2009), switchgrass cultivation is becoming increasingly popular in the eastern United States. The marginally productive lands where switchgrass planting is being targeted are typically poorly suited for conventional cultivation because they are either excessively drained or too wet (NRCS, 1997). The application of switchgrass-derived or other biochars to those marginal sites could improve the soils and potentially lead to greater switchgrass production.

Several approaches can be used to estimate the decomposition dynamics of organic materials in soil, with selection of the most appropriate method dependent on the type of information desired. The evaluation of CO_2 emission curves during extended incubations under optimal conditions uses C mineralization by soil microbial enzymes to identify the relative proportions of labile and stable C and to estimate the MRT of C in those pools (Paul et al., 2001). When native SOM and newly added organic material possess different ratios of ^{13}C to ^{12}C, the ^{13}C signature of the emitted CO_2 can be used to quantify the sources using a simple mixing model.

Soil amendment with switchgrass-derived biochar was reported to accelerate microbial growth (Ippolito et al., 2012), but the researchers did not know whether the primary C source for the microbes was biochar or native soil C. Decomposition of a range of biochars has been shown to be influenced by varying soil properties (Zimmerman et al., 2011). However, we are not aware of studies published thus far addressing variation in the turnover of switchgrass-derived biochar among soils or the effects of switchgrass-derived biochar on the dynamics of the native SOM. Our goal, therefore, was to examine the interactions of switchgrass-derived biochar and four marginal soils from Pennsylvania on the turnover of biochar C and native soil C pools.

Table 1. Pre-incubation properties of the four soils and the switchgrass-derived biochar.

Soil or biochar	Organic C	$\delta^{13}C$	pH	Particle size distribution		
				Sand	Silt	Clay
	%	‰		%		
Edom	3.77 (0.3)†	−28.15 (0.07)	6.00 (0.08)	6 (0.49)	62 (0.54)	32 (0.27)
Wharton	4.13 (0.6)	−25.43 (0.31)	5.43 (0.03)	15 (0.33)	58 (0.26)	27 (0.08)
Weikert	4.02 (0.2)	−24.08 (1.07)	5.96 (0.03)	44 (0.77)	40 (0.57)	16 (0.28)
Morrison	1.69 (0.1)	−22.74 (0.56)	5.86 (0.02)	61 (0.21)	27 (0.20)	12 (0.08)
Biochar	68.7 (2.1)	−15.71 (0.13)	9.52 (0.16)			

† Means of four measurements, with standard errors of the means in parentheses.

MATERIALS AND METHODS

Soil Sampling and Site Description

Soil was obtained from the upper 0.1 m (A or Ap horizon) of four sites that are representative of marginal land within the Appalachian Plateau and Ridge and Valley physiographic provinces in central Pennsylvania (soil and biochar details in Table 1). The Edom (a Typic Hapludalf) and Wharton (an Aquic Hapludult) soils are both silty clay loams in the surface layer and were obtained from sites that often experience extended periods of wetness in the spring and fall. The Weikert (a Lithic Dystrudept) soil is a loam with a high gravel content (~30%) and was obtained from an excessively drained site. The Morrison (an Ultic Hapludalf) soil is a sandy loam and also obtained from an excessively drained site. The soils were obtained in September 2011. Seven to 10 positions at each site were sampled, and the samples from a site were combined and thoroughly mixed (40–50 kg total soil). All soils were air dried in a greenhouse and then ground to pass a 2-mm sieve.

Biochar Production

Biochar was produced from Cave-In-Rock switchgrass by the torrefaction facility at North Carolina State University using slow pyrolysis. Four to five dry megagrams of the grass biomass was used to produce 1 Mg of biochar. The biomass was preheated to about 100°C, then transferred to the torrefaction chamber for 1 to 1.5 min. In the chamber, biomass is indirectly heated to 375 to 475°C in a low-O_2 environment with constant stirring by an auger. After removal from the torrefaction chamber, the biomass is allowed to cool to 35°C in the exit auger (about 3 min). Following pyrolysis, the biochar was hydrated to 60% water content (by weight) to prevent oxidation and self-heating. After being allowed to stand for 24 h to reach hydration and oxidation equilibrium with the atmosphere, the biochar was packaged and shipped to Pennsylvania.

Soil Incubations and Measurements

The incubation experiment was a completely randomized design with four replications. Soils from each of the four soil series were incubated, both unamended and amended with biochar (1% of dry soil by weight), in 1-L glass jars that were sealed with lids fit with rubber septa to allow gas sampling by syringe. The biochar concentration was equivalent to about 37 Mg moist biochar ha^{-1} incorporated to a 15-cm depth (assuming a soil bulk density of 1.2 g cm^{-3} and biochar water content of 49% w/w) and is comparable to application rates that we have used in field studies. Either 150 g of dry soil or a mixture of 148.5 g of dry soil and 1.5 g of dry biochar was weighed into a jar, the soil and biochar were thoroughly mixed, and water was added to bring the soil mixture in each jar to 60% water

holding capacity (Cheng et al., 2008). Four replicate jars were also prepared with 12 g of pure biochar and adjusted to 60% water holding capacity. Additionally, four replicate jars were maintained without soil or biochar to provide background CO_2 concentrations. All jars were incubated at 25°C in one growth chamber for 189 d.

The jars remained sealed during the incubation, and periodic cumulative CO_2 production was determined by sampling the jar headspace CO_2 concentrations 1, 4, 8, 15, 25, 39, 53, 69, 88, 111, 139, and 189 d after the start of the experiment. The interval between sampling was steadily increased with time throughout the incubation period because CO_2 production rates steadily declined following high initial emissions. Headspace gas samples (1 mL) were removed from the jars by syringe, and the CO_2 concentration was determined using an infrared gas analyzer (LI-COR Model LI-6262). After headspace sampling, the jars were opened for 1 h to restore O_2 levels. If needed, water was added to restore the substrate to its initial moisture content before resealing the jars. Additional 25-mL gas samples were removed from the jars 1, 8, 25, 53, 111, and 189 d after the initiation of the incubations, transferred to pre-evacuated 12-mL glass vials, and immediately shipped overnight to the Cornell University Stable Isotope Laboratory for ^{13}C-CO_2 determination using a Thermo Delta V isotope ratio mass spectrometer interfaced to a Gas Bench II (Thermo Fisher Scientific).

The initial concentration of total C and the ^{13}C signature of both soils and biochar were also determined by the Cornell University Stable Isotope Laboratory by combustion in a Thermo Delta V isotope ratio mass spectrometer interfaced with a Carlo Erba NC2500 elemental analyzer (four replicate measurements per soil). The ^{13}C measurement precision of both mass spectrometers is ±0.02‰. The soil pH was measured using a 1:1 soil/deionized water ratio (by mass), while the biochar pH was measured using a 1:20 ratio (Nguyen and Lehmann, 2009) because biochar has a greater water sorption capacity than soil. The soil particle size distribution was determined using the pipette method (Gee and Bauder, 1986). Particle size cutoffs for sand, silt, and clay were 2.0 to 0.05, 0.05 to 0.002, and <0.002 mm, respectively. Each measurement was made on four replicates of each soil and the biochar. Dry sieving indicated that 3% of the biochar was >2.0 mm in diameter, 93% of the biochar was in the 0.05- to 2.0-mm range, and 4% was <0.05 mm.

Carbon Partitioning Models

The proportion of emitted C derived from either biochar (f_{BC}) or soil organic C (f_{SOC}) was estimated using the model presented by Liang et al. (2008) and Zimmerman et al. (2011):

$$\delta^{13}C_{(\text{mixture } CO_2)} = f_{BC}\delta^{13}C_{(BC\text{-}CO_2)} + (1 - f_{BC})\delta^{13}C_{(SOC\text{-}CO_2)} \quad [1]$$

and

$$f_{BC} + f_{SOC} = 1 \quad [2]$$

where $\delta^{13}C_{(\text{mixture } CO2)}$ is the delta ^{13}C of CO_2 emitted from the soil–biochar mixture; $\delta^{13}C_{(BC\text{-}CO2)}$ is the delta ^{13}C of CO_2 derived from pure biochar; and $\delta^{13}C_{(SOC\text{-}CO2)}$ is the delta ^{13}C of CO_2 emitted from the unamended soil. The calculated portions of emitted C derived from biochar vs. incubation time for each ^{13}C sampling date were fit to a two-component exponential decay equation using Sigmaplot 10 (Systat Software). The resulting curve was used to estimate the biochar contribution to CO_2 emissions on the sampling dates when $\delta^{13}C$ was not quantified. Isotope data from each source was corrected for the background ^{13}C signature using data from jars without soil or biochar ($n = 4$) and Eq. [1] and [2].

A two-component exponential model can be used to partition C between labile and stable pools and estimate the MRT of the C in both biochar-amended and unamended soils (Grandy and Robertson, 2007). The basic partitioning model is

$$C_{total} = C_1 \exp(-k_1 t) + C_s \exp(-k_s t) \quad [3]$$

where C_{total} is total organic C in the soil or soil–biochar mixture; C_1 and k_1 are the size and decay constant, respectively, of the labile C pool; C_s and k_s are the size and decay constant, respectively, of the stable C pool; and t is the time since the initiation of the incubation. To estimate the pool sizes and decay rates, the quantities of C emitted as CO_2 were fit to a differential version of the equation using SigmaPlot 10:

$$\frac{dC}{dt} = C_1 k_1 \exp(-k_1 t) + (C_{total} - C_1) k_s \exp(-k_s t) \quad [4]$$

where dC/dt is the rate of change in C at time t (i.e., CO_2 emission rate). The inverse of k_1 and k_s provided the MRT for the labile and stable C pools. The amount of ambient CO_2 was subtracted from the total CO_2 in each jar before applying data to the partitioning equations.

Statistical Analysis

Both observed and estimated parameters were analyzed with ANOVA using JMP 8 (SAS Institute). Variation was expressed as standard error in all cases. When ANOVA indicated a significant treatment effect at $P \leq 0.05$, the Tukey honestly significant difference test was used to test for significant differences among treatment means. The comparison of CO_2 emissions from soils with and with biochar addition were made used a paired t-test for each soil on each sampling date ($\alpha = 0.05$).

RESULTS

Biochar Decomposition as Affected by Soils

Mixing biochar with each of the soils significantly increased biochar CO_2 evolution compared with the incubation of biochar alone (Fig. 1), with biochar accounting for 3.2, 7.7, 7.8, and 12.1% of the total CO_2 emissions (Eq. [1]) in the Edom, Wharton, Weikert, and Morrison soils, respectively. The proportion of the total respired CO_2 that was derived from biochar decreased rapid-

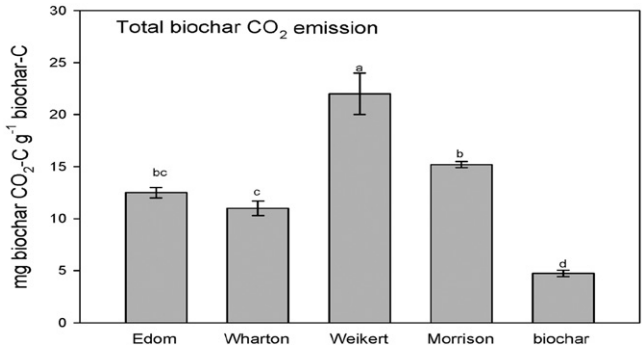

Fig. 1. Total CO_2 emitted from biochar alone and biochar mixed with four soils during a189-d incubation period. Errors bars reflected standard error of the means (n = 4). Different letters indicate significantly different ($P < 0.05$) means according to Tukey's honestly significant difference test.

ly during the first few weeks of incubation, with the proportion of CO_2 contributed by biochar dropping from between 6.5 to 30.5% to nearly zero during that time (Fig. 2). The proportion of total emitted CO_2 derived from biochar was not significantly different from zero after 90, 181, and 172 d of incubation with the Edom, Weikert, and Morrison soils, respectively. The proportion of CO_2 derived from biochar in the Wharton soil remained significantly different from zero during the entire incubation period.

The pattern of C loss from biochar depended on the soil (Fig. 3). More CO_2 evolved from biochar in the Weikert soil than the other soils at $P \leq 0.001$ (Fig. 1 and 3). The least biochar C was respired from the higher clay soils, the Wharton and Edom series. Biochar C evolution was intermediate in the Morrison soil.

Fitting CO_2–C emission data to the two-pool decay model indicated that the labile pools comprised a small portion of the total biochar C. Estimated concentrations of labile C ranged

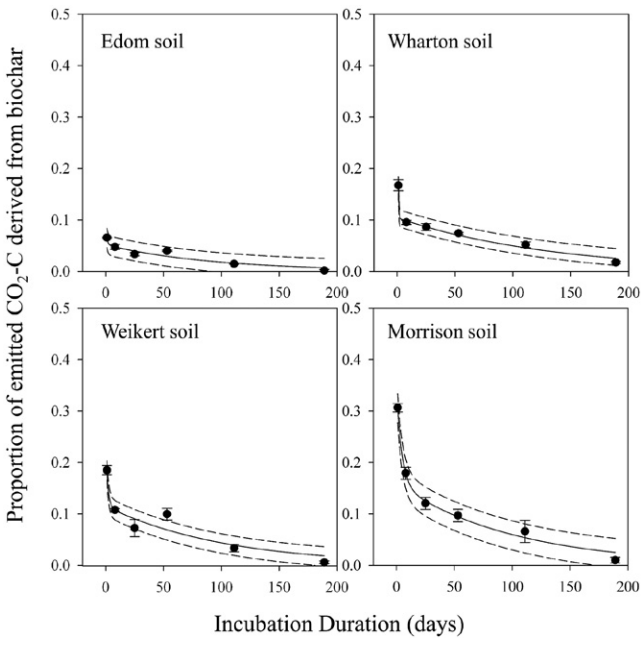

Fig. 2. Proportion of total emitted CO_2–C derived from biochar mixed with each of the four soils. The solid lines represent the plots of the best-fit, two-component exponential models. The dashed lines are the upper and lower 95% confidence limits. Error bars are the standard errors of the mean (n = 4).

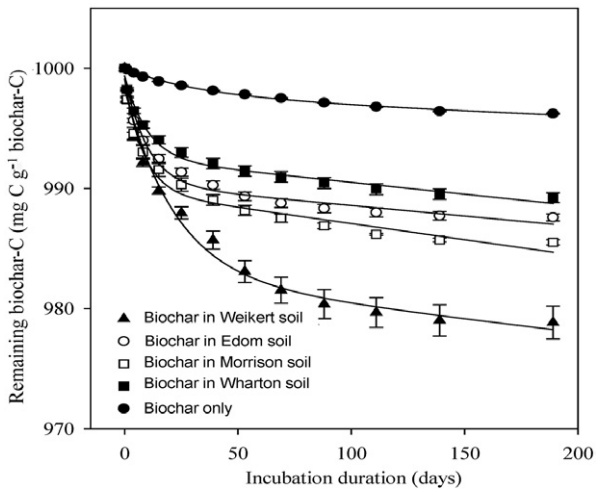

Fig. 3. Patterns of biochar C loss during incubation from pure biochar and biochar mixed with each of the four soils. Individual points are averages (n = 4) and error bars are standard deviations.

from 1.9 to 16.1 g kg^{-1}, while the majority of the biochar C (>980 g kg^{-1}) resided in the stable pools (Table 2). Soil series had a significant effect on the size of each pool of biochar C. In general, mixing biochar with any of the four soils resulted in significant increases in the concentration of the labile biochar C pool as well as a decrease in the concentration of the stable biochar C pool compared with incubation of biochar without soil. The labile biochar C concentration was significantly larger in the Weikert soil than in the other soils. The stable biochar C pool was correspondingly significantly smaller in the Weikert soil than the other soils (Table 2).

The MRT of the two pools differed significantly among soils (Table 2). The labile biochar C pool MRT for biochar in soil varied from 9 to 22 d, with the Weikert soil having a significantly longer labile biochar C MRT than the other soils. The stable biochar C pool MRT varied from approximately 113 to 163 yr, with no significant differences among soils. In general, mixing biochar with soil decreased both the labile and stable biochar C pool MRT.

Native Soil Organic Carbon Decomposition as Affected by Biochar

The patterns of native soil C decomposition in each of the soils was largely unaffected by the presence of biochar (Fig. 4). Biochar addition had no significant effect on the total C loss from native SOM for any of the four soils during the incubation period (Fig. 5). However, total CO_2 emissions from native SOM differed significantly among soils (Fig. 5). The Edom soil emitted 6.2% of total native soil organic C (SOC). In contrast, the Wharton soil emitted only 2.1% of the total native SOC. Native SOC-CO_2 loss was intermediate for the Weikert and Morrison soils.

Neither the effect of biochar amendment nor the interaction between biochar amendment and soil were significant with respect to the native labile and stable C pool sizes or MRTs (Table 3). Different soils, however, did possess significantly different sizes and MRTs for both the labile and stable native SOC pools. The labile pool was largest in the Edom soil, followed by the Weikert,

Table 2. Labile and stable C pool concentrations and mean residence times (MRTs) for biochar mixed with four soils and for pure biochar. Pure biochar was added for comparison, but it should be noted that differences in pool sizes and MRTs between pure biochar and biochar mixed in soil are probably impacted by the presence of a greater microbial population in the soils.

Soil series	Labile biochar C pool		Stable biochar C pool	
	Concentration	MRT	Concentration	MRT
	g C kg^{-1} biochar C	d	g C kg^{-1} biochar C	yr
Edom	9 b†	10 b	991 b	163 ab
Wharton	7 b	10 b	993 b	138 b
Weikert	16 a	22 a	984 c	129 b
Morrison	9 b	9 b	991 b	113 b
Pure biochar	2 c	27 a	998 a	258 a
ANOVA *P* values	<0.01	<0.01	<0.01	<0.01

† Within a column, means followed by different letters are significantly different ($P < 0.05$) according to Tukey's honestly significant difference test.

Morrison, and Wharton soils. The MRT of the labile pool was highest in the Weikert soil and lower and similar in the remaining three soils. The stable pool was largest in the Wharton soil, followed by the Morrison, Weikert, and Edom soils. The MRT of the stable pool was highest in the Wharton soil, followed by the Weikert soil, and lower still in the Edom and Morrison soils, which were not significantly different from each other.

DISCUSSION

Biochar Decomposition as Affected by Soils

Our results suggest that switchgrass biochar produced at approximately 450°C was highly stable in the soil because only a small proportion (1.1–2.1%) of the biochar C was lost during the 189-d incubations. Singh and Cowie (2010) reported that 0.3 to 6% of added biochar C was lost during 2.3-yr incubations conducted at 22°C and a water content of 70% water holding capacity. For comparison, the total proportions of the biochar in this study projected to be lost during 2.3 yr would be between 2.3 and 3.4% for the four soils. The stability of biochar in soil can be attributed to its chemical structure. Biochar is primarily composed of stable aromatic rings (Sohi et al., 2010) linked together to form multiple layers (Lehmann and Joseph, 2009). A small component of biochar is composed of more easily decomposed small, organic molecules, such as sugars and fats, that were not totally transformed at pyrolysis and bio-oil adsorbed or trapped on biochar particles (Cole et al., 2012). Because the aromatic compounds are largely resistant to decomposition, we suggest that the labile biochar C respired throughout the incubations in the current study were derived from the remaining sugars and fats and potentially from a small, readily decomposed fraction of the aromatic compounds (Hilscher et al., 2009). Greater mineralization of biochar C when mixed with soil is not surprising and probably a result of considerably greater microbial populations in the soils than in the biochar-only control.

While switchgrass biochar was highly stable in each of our soils, we observed differences in biochar decomposition among the four soils. For example, the total quantity of biochar C respired during the incubations differed among the four soils. While biochar decomposition occurs through both biotic and abiotic

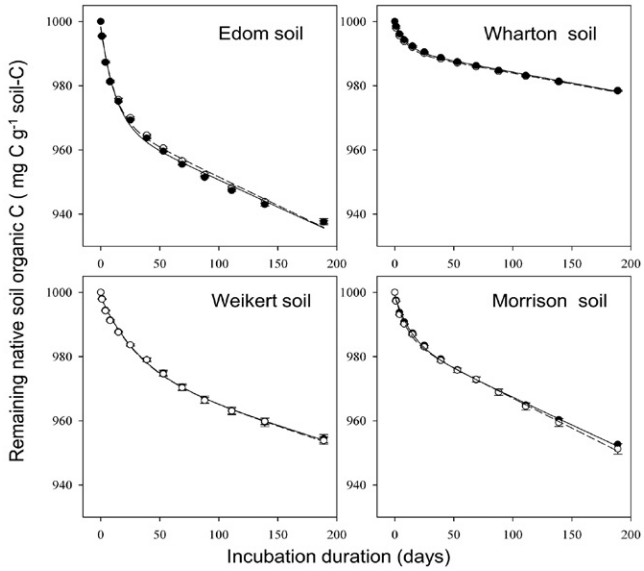

Fig. 4. Pattern of native soil organic matter decomposition from the four soils as affected by biochar additions. Filled circles and solid lines correspond to biochar-amended soils. Open circles and dashed lines correspond to unamended soils. Individual points are averages ($n = 4$) and error bars are standard deviations.

mechanisms (Cheng et al., 2006; Liang et al., 2008), the substantially greater respiration of biochar C when mixed with soil suggests that differences in soil microbial activity were largely responsible for the variation in biochar decomposition in this study. Because the four soils used in the study came from diverse sites that differed in the concentration of organic matter, fertility level, vegetation type, and management history, the size, species composition, and activity of the soil microbial communities are likely to differ among the soils. Other studies have shown significant effects of soil amendments, ryegrass (*Lolium* sp.) addition (Luo et al., 2011), and glucose addition (Hamer et al., 2004; Kuzyakov et al., 2009) on biochar decomposition, suggesting that the quantity and quality of organic substrates in the vicinity of the biochar can influence soil microbial activity and subsequent biochar metabolism. Additional studies to evaluate microbial populations will be needed to further explore this possibility.

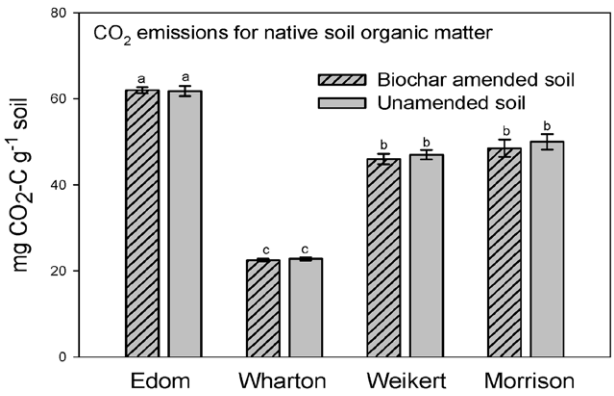

Fig. 5. Total CO_2 emitted from native soil organic C during a 189-d incubation period. Error bars are standard errors of the mean ($n = 4$). Different letters indicate significantly different ($P < 0.05$) means according to Tukey's honestly significant difference test.

Table 3. Total (biochar + native C) labile and stable C pool concentrations and mean residence times (MRTs) as affected by biochar amendment.

Soil series	Biochar amendment	Labile total C pool		Stable total C pool	
		Concentration	MRT	Concentration	MRT
		g C kg^{-1} C	d	g C kg^{-1} C	yr
Edom	no	28.8 a†	12 b	969.6 e	14.7 c
	yes	30.4 a	13 b	967.8 e	15.4 c
Wharton	no	8.8 de	10 b	990.5 a	41.6 a
	yes	8.5 e	10 b	991.1 a	39.5 a
Weikert	no	22.1 b	31 a	976.4 c	21.9 b
	yes	23.1 b	33 a	975.5 c	23.2 b
Morrison	no	13.2 cd	10 b	985.9 b	14.2 c
	yes	14.2 c	14 b	984.8 b	15.3 c
			ANOVA		
Soil series		<0.01	<0.01	<0.01	<0.01
Biochar amendment		0.25	0.09	0.29	0.66
Interaction		0.79	0.64	0.70	0.18

† Within a column, means followed by different letters are significantly different ($P < 0.05$) according to Tukey's honestly significant difference test.

The period of measurable (significantly different from zero) emissions of biochar C varied from 90, 181, and 172 d in the Edom, Weikert, and Morrison soils, respectively, to longer than 189 d in the Wharton soil. This does not mean that biochar decomposition stopped completely beyond these periods, but the proportion of emitted CO_2 derived from biochar became insignificant relative to that derived from SOM. This indicates that the labile fraction of biochar is oxidized over different periods of time in the various soils. Some of this undoubtedly is due to variation in the effective size of the labile C pools.

The amount and type of clay could also potentially influence the biochar decomposition rate because interaction with clays can protect organic materials from microbial attack (Glaser et al., 2000; Liang et al., 2008). Biochar encapsulation within soil microaggregates is believed to enhance the MRT of biochar in soils (Brodowski et al., 2006). We did not see a significant correlation between biochar decomposition and clay content (data not shown), possibly because the duration of the incubation or incubation conditions was not sufficient for interactions to develop.

Estimates of MRT determined under optimal laboratory conditions represent a potential decomposition rate and are expected to be shorter than actual MRT under field conditions. However, they provide a way to compare soils or C pools within an individual soil. Mixing biochar with soils shortened the MRT of both labile and stable biochar C pools. As in this study, the fastest biochar decomposition has commonly been shown to occur within the first 2 to 3 mo of incubation (Kuzyakov et al., 2009). Laboratory labile biochar C MRT estimates in the current study varied from 9 to 22 d and are consistent with MRTs of 3 to 39 d reported for various biochar types (Singh et al., 2012). Given its short MRT and the small portion of the total biochar C (<2% of the total biochar C) that it comprises, the labile C compounds appear to have limited importance in terms of C sequestration.

Laboratory determinations of the MRT of the stable pool of switchgrass biochar C mixed with soils in the current study were estimated to vary from 113 to 163 yr. These values are within the ranges reported in other studies, 62 to 248 yr (Zimmerman et al., 2011) and 90 to 1616 yr (Singh et al., 2012), depending on biochar type and incubation conditions including temperature, soil type, or plant litter addition. Under field conditions, the MRT of the currently tested biochar is expected to be much longer than the laboratory estimate because temperature and moisture in the field frequently limit microbial activity and decomposition rates. The Q_{10} is commonly used to adjust biological activity rates for changing temperature [$Q_{10} = (r_2/r_1)^{10/(t2 - t1)}$, where r_1 and r_2 are decomposition rates of biochar at times t_1 and t_2 at 25°C]. Assuming a Q_{10} value of biochar of 3.4 (Cheng et al., 2008) and a mean annual temperature of 10°C (t_1) for central Pennsylvania (Casey, 2010), the estimated MRTs of the stable biochar C pools in our four soils would range from 711 to 1020 yr under field conditions. However, it is important to note that this is only a rough estimate because factors such as soil moisture fluctuations and freezing and thawing would also influence decomposition rates in the field (Kim et al., 2012).

Native Soil Organic Matter Decomposition as Affected by Biochar

The addition of biochar to soils did not significantly impact native SOM decomposition in the current study, but overall CO_2 emissions from the native SOM varied among the soils despite similar SOM concentrations in three of the four soils. The Wharton soil emitted the least CO_2 even though it had the highest SOM concentration, indicating that factors in addition to total SOM concentration controlled decomposition rates in these soils.

Several studies have reported that biochar addition may increase (positive priming) (Wardle et al., 2008; Singh and Cowie, 2010; Luo et al., 2011; Zimmerman et al., 2011) or decrease (negative priming) (Jones et al., 2011; Zimmerman et al., 2011) the decomposition of native SOM, with the addition of grass-derived biochars often causing positive priming (Zimmerman et al., 2011). These positive priming effects are most often thought to occur when the stimulation of extracellular microbial enzyme production, in response to biochar application, also results in greater native SOM decomposition (co-metabolism) (Kuzyakov et al., 2000). However, the extent of priming varies with the type of biochar and charring temperature (Zimmerman et al., 2011). The low concentration of readily mineralized C in the switchgrass biochar may not have been sufficient to stimulate microbial activity in our soils. While the lack of priming in this study has positive implications for the stability of switchgrass biochar added to soil, further examination of biochar created from switchgrass using different pyrolysis conditions, especially temperature, is needed.

CONCLUSIONS AND IMPLICATIONS

Short-term laboratory incubation and C isotope analysis provided a valuable means to quantify the proportion of labile C contributed by biochar and to determine if interaction between

biochar and the various soils influenced the decomposition of either biochar or native SOM. As has been shown for wood-derived biochars, a large portion of the switchgrass biochar can be expected to have a long residence time in soil. The stability of the switchgrass biochar added to soil can contribute to long-term C sequestration and sustained improvement in soil properties commonly associated with increased SOC, such as nutrient retention and aggregate stability. Greater than 98% of the biochar C was partitioned into the stable pool when mixed with each of our four soils. While soil series impacted the mineralization of labile C pools, those differences represented only a very small portion of the total biochar C. We saw no evidence to indicate that amendment with switchgrass-derived biochar had any effect on the decomposition of native C pools. Therefore, although biochar production via pyrolysis results in an immediate loss of a substantial portion of the switchgrass C, the amendment of soil with switchgrass biochar can have a net positive effect on soil C sequestration.

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