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Crop residue decomposition in Minnesota biochar amended plots

S. L. Weyers[1] **and K. A. Spokas**[2]

[1]USDA Agricultural Research Service, North Central Soil Conservation Research Lab, Morris, MN, USA
[2]USDA Agricultural Research Service, Soil and Water Management Unit, University of Minnesota, Saint Paul, MN, USA

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Correspondence to: S. L. Weyers (sharon.weyers@ars.usda.gov)

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Abstract

Impacts of biochar application at laboratory scales are routinely studied, but impacts of biochar application on decomposition of crop residues at field scales have not been widely addressed. The priming or hindrance of crop residue decomposition could have a cascading impact on soil processes, particularly those influencing nutrient availability. Our objectives were to evaluate biochar effects on field decomposition of crop residue, using plots that were amended with biochars made from different feedstocks and pyrolysis platforms prior to the start of this study. Litterbags containing wheat straw material were buried below the soil surface in a continuous-corn cropped field in plots that had received one of seven different biochar amendments or a non-charred wood pellet amendment 2.5 yr prior to start of this study. Litterbags were collected over the course of 14 weeks. Microbial biomass was assessed in treatment plots the previous fall. Though first-order decomposition rate constants were positively correlated to microbial biomass, neither parameter was statistically affected by biochar or wood-pellet treatments. The findings indicated only a residual of potentially positive and negative initial impacts of biochars on residue decomposition, which fit in line with established feedstock and pyrolysis influences. Though no significant impacts were observed with field-weathered biochars, effective soil management may yet have to account for repeat applications of biochar.

1 Introduction

Biochar is the solid product that comes from a variety of thermolytic conversion processes creating a carbon-rich material, which is intended for carbon sequestration purposes. Biochar, when used as a soil amendment, has been hypothesized to provide nutrients for plant growth, counteract soil acidity, or induce positive effects on soil properties such as cation exchange capacity, bulk density and water holding capacity (Atkinson et al., 2010; Sohi et al., 2010; Dai et al., 2013). Biochar can have positive effects on soil biota as well (Lehmann et al., 2011). In general, biochar is perceived as a beneficial soil amendment product with multiple advantages.

Addition of biochar might alter properties that regulate soil organic matter (SOM) decomposition, which are: decomposer organism diversity and abundance, resource availability, and the physio-chemical environment, particularly soil aeration and moisture content (Swift et al., 1979; Heal et al., 1997). Microorganisms are the primary decomposers of SOM. The majority of studies evaluating biological effects of biochars observe positive stimulation of microbial abundance, which has been correlated with the improved soil conditions (Lehmann et al., 2011). Laboratory studies indicate biochar addition can change resource availability and induce priming effects, which are short-term changes in the mineralization of SOM due to stimulated microbial processing (Luo et al., 2011; Zimmerman et al., 2011).

Variable effects on residue decomposition dynamics can be expected when evaluating dissimilar biochars applied to the same or similar soils. Nutrient composition, pH, volatile components, density, porosity and other characteristics of biochar are affected by the feedstock and the conditions of the thermolytic conversion process used (Spokas et al., 2012; Lee et al., 2013; Sigua et al., 2014). In particular, the soluble, leachable components also differ among biochars (Jaffé et al., 2013). Different biochars affect microbial community composition by promoting different components of the microbial community (Lehmann et al., 2011). For instance, some biochars might stimulate bacteria and others fungi (Steinbeiss et al., 2009). Altered microbial community composition in this sense could have cascade effects on higher levels of the soil food web, such as that observed under different tillage regimes (Hendrix et al., 1986). Further, biochar may increase nutrient availability (Noguera et al., 2010). In particular for N, biochar may reduce the N limitation that results in slower C mineralization rates (Vitousek and Howarth, 1991).

A majority of studies to evaluate biochar's impact on organic matter decomposition have been conducted in the laboratory. Most of these studies use freshly made biochar, small amounts of finely ground or sieved organic material, and short time frames in laboratory incubations. For example, Novak et al. (2010) determined that a fresh pecan shell-derived biochar primed the mineralization of 0.25 mm sieved switchgrass residues in a 67 day incubation. Similarly, Awad et al. (2012) also observed an increased rate of maize residue decomposition in a laboratory study following biochar addition, with the observed rate a function of the soil texture and biochar production temperature (Awad et al., 2013). On the other hand, Bruun and EL-Zehery (2012) found an insignificant increase in laboratory C mineralization of un-charred barley straw in the presence of fresh barley straw-derived biochar (0.15 % w/w). It is already known that biochar's surface chemistry and reactivity changes with time, largely believed due to the reactivity to oxygen (Puri et al., 1958) and water (Pierce et al., 1951) at ambient conditions. However, only limited field based studies have been conducted. Wardle et al. (2008) evaluated mass loss of humus encapsulated with fresh wood charcoal (1 : 1) in mesh bags in field plots over ten years. They observed that charcoal mixed with humus possessed a greater synergetic mass loss over the ten years than expected from charcoal and soil humus alone (Wardle et al., 2008). From the laboratory studies, fresh biochar appears to prime the decomposition of soil organic matter. In the limited field experiments, biochar had a long-term impact on humus decomposition, resulting in overall greater cumulative mass loss over time. Despite these findings, the impact of aged biochar on the decomposition of freshly added organic matter, in particular crop residue in agricultural soils, is still unknown.

The objectives of this study were to determine (1) if field-weathered biochar can affect the field decomposition of freshly added crop residue, (2) if any impact on field decomposition rates can be related to biochar feedstock or pyrolysis method, and (3) if microbial biomass was influenced by biochar applications. Based on the findings of Wardle et al. (2008), Novak et al. (2010) and others, accelerated decomposition of freshly added organic material was expected in field-weathered biochar plots. We further hypothesized that there would be differences in observed decomposition rates in field plots as a function of biochar type.

2 Materials and methods

2.1 Site description and biochar treatments

The research site is located at the University of Minnesota Research and Outreach Center in Rosemount, MN USA (44° N, 93° W). Soil at the site is a low slope ($< 2\,\%$) Waukegan silt loam (fine-silty over skeletal mixed, super active, mesic typic Hapludoll) containing approximately 22 % sand, 55 % silt, and 23 % clay with a pH of 6.4 and total organic C of 26 $\mathrm{g\,kg^{-1}}$. Seven different biochar treatments, a raw biomass (non-charred wood pellet), and a zero-amendment control treatment were applied in triplicate to 27 completely randomized plots (Table 1). The plots measured 4.88 m on a side with a 3 m buffer zone between plots. Feedstocks for these biochars were hardwoods, pine chips, macadamia nut shells, and wheat mids, which were produced by thermal pyrolysis (Table 1). All biochars and the wood pellet amendment used in the test plots were applied at a rate of 22.4 Mg (as received) $\mathrm{ha^{-1}}$, thus providing total C additions ranging 14.4 to 19.9 $\mathrm{Mg\,C\,ha^{-1}}$.

Amendments were incorporated into the soil by rotary tillage to a 15 cm depth starting in the fall of 2008. After incorporation, plots were annually planted with corn (*Zea mays*) and the residue was managed with spring rotary tillage prior to planting. Fertilization was applied uniformly and annually to all test plots, according to the control plot soil test rates. This amounted to between 100 and 125 $\mathrm{kg\,N\,ha^{-1}}$ (urea) being broadcasted prior to tillage and planting. This fertilization and corn planting occurred prior to residue bag placement.

2.2 Litterbag preparation and processing

Freshly harvested and baled wheat (*Triticum aestivum* L.) straw was the organic material used in this study. Straw was cut into 10 cm lengths and included stem nodes, but not grain or grain heads. Air dry litter weights were corrected to a 50 °C oven dry weight equivalent. Approximately 3.0 ± 0.3 g dry weight equivalent of wheat straw material was placed in 15 cm × 15 cm fiberglass mesh (ca 1.5 mm) bags. At the beginning of July (approximately 45 days post maize planting), 10 bags were inserted into 15 cm deep vertical slits in the ground along a center transect in each plot. Bags were randomly retrieved after 1, 3, 5, 7, 10 and 14 weeks in the field. On week 5 and 14, three replicate bags per plot (nine per treatment) were retrieved. For all other weeks only one bag per plot (three per treatment) was retrieved. Bags were brushed free of dirt and dried at 50 °C before processing. Litter material was manually cleaned of extraneous dirt, roots and other visible contaminants. Following this final cleaning, litter was dried again at 50 °C to obtain final oven dry weights. Mass loss was calculated as initial weight minus final weight of individual litter bags. To account for differences in initial weights among litterbags, data were analyzed as a percent litter mass remaining (%LMR), where %LMR = ((initial weight – final weight)/initial weight) × 100.

2.3 Microbial biomass

Soil sampling of the surface 0–10 cm in each plot was conducted in the fall prior to the litterbag decomposition study. Microbial biomass ($\mu g\,C\,g^{-1}$ soil) in all treatment plots was determined by the chloroform fumigation-incubation technique (Anderson and Domsch, 1978) with soil respiration measured by GC (Koerner et al., 2011). The microbial biomass carbon was calculated as the $\mu g\,CO_2\text{-}C\,g^{-1}$ soil of fumigated soil minus the $\mu g\,CO_2\text{-}C\,g^{-1}$ soil from un-fumigated soil divided by an efficiency factor of 0.411 (Anderson and Domsch, 1978).

2.4 Statistical analysis

The decomposition constant, k, and 95 % confidence intervals were determined across the experiment, by treatments and by replicates within treatments using the non-linear platform in JMP 10.0 software (SAS Institute, 2012). The data were fit to a simple first order decomposition equation, $\%LMR = 100e^{-kt}$, where %LMR is the percent of litter mass remaining over time for each treatment, k is the unknown simple first order decomposition constant, and t is time (Karberg et al., 2008). Percent litter mass remaining for each sampling week, calculated decomposition rate (k), and microbial biomass were analyzed by a one-way analysis of variance (ANOVA) on treatment (Wider and Lang, 1982) with PROC GLM in SAS 9.2 software (SAS Institute, 2009), using an $\alpha = 0.05$. Differences of means were tested with Bonferoni adjustment to p values of multiple comparison tests, Tukey's honestly significant difference, and with Dunnet's test for comparison to control. The correlation between microbial biomass and k was determined using the pairwise estimation procedure in JMP 10.0 software (SAS Institute, 2012).

3 Results

Despite the short duration of this study (14 weeks), the average mass loss over all the treatments was greater than 50 % (Fig. 1). The rate of litter mass loss was fit to a first-order decomposition kinetics model (Aber et al., 1990), resulting in decomposition constants, k, ranging from $7.5 \times 10^{-3}\,\mathrm{d}^{-1}$ to $9.8 \times 10^{-3}\,\mathrm{d}^{-1}$ (Table 2). Compared to the control, decomposition rates were stimulated in the wood pellet amendment (WP; +18 %) and the fast pyrolysis hardwood sawdust biochar (BC1; +18 %), 16 % faster in the slow pyrolysis pine chip biochar (BC6), and 11 % faster in the slow pyrolysis wood pellet biochar (BC2). On the other hand, a decrease in the rate of decomposition was observed in the fast pyrolysis macadamia nut biochar (BC7; −10 %). However, the differences in the k or %LMR were not significant across all treatments due to high spatial variability among replicates, which exists in natural field settings. Nonetheless, lack of overlap in the 95 % confidence intervals for k determined across replicates for whole treatments suggest the likelihood of differences between biochar treatments for BC1 and BC6 with BC7 (Table 2).

Contrary to the hypothesis that pyrolysis conditions and feedstock are deterministic variables for biochar, the decomposition dynamics did not display distinct overall patterns related to feedstock or pyrolysis methods. The fast pyrolysis hardwood biochar

(BC1) did possess the highest k rate and 6.4 % less litter mass remaining than the slow pyrolysis wood-based biochars. However, with only one fast pyrolysis biochar in the field plots the universality of this observation requires further scrutiny.

Microbial biomass averaged 283 $\mu g\,C\,g^{-1}$ soil, with a high of 835 ± 53 $\mu g\,C\,g^{-1}$ soil for the wood-pellet amendment (WP), a mean of 142 ± 19 $\mu g\,C\,g^{-1}$ soil for the control, and a low of 117 ± 25 $\mu C\,g^{-1}$ for macadamia nut biochar (BC7) (Table 2). Microbial biomass was not significantly different among the treatments ($p > 0.05$). In spite of the lack of statistical significance between treatments, microbial biomass was positively correlated to k, the observed litter decomposition constant ($r^2 = 0.67$; $p < 0.05$).

4 Discussion

The decomposition rate of wheat straw observed in our control plots was similar to the rate observed by prior studies (Christensen, 1985). Wang et al. (2012) also observed similar decomposition rates in their 2 yr study, with degradation rates spanning from 3.8 to 8.1 yr^{-1}. Though particulate mass can be lost from litterbags overtime and other difficulties in the analysis of litter bag results are encountered (Wider and Lang, 1982), similarity of decomposition rates to prior studies and the condition of the wheat straw remaining over the course of the experiment indicated that the majority of the material was retained inside the litterbag and decomposed in situ.

The litterbag method was purposely chosen for its ability to integrate mesofaunal contributions, a component which has not been examined in biochar amended systems, with the microbial dynamics primarily responsible for decomposition of organic material (Coleman et al., 1999). Thus, the litterbag evaluation allowed a functional determination of biochar influence on dynamics of the decomposer community as a whole. Macrofaunal activity was evident at the field plots, in particular as visible surface earthworm activity and castings. However, a macrofaunal sampling conducted at the start of the litter decomposition study established that earthworm abundance was not significantly different at the time of litter bag placement (Weyers and Spokas, 2011).

This litterbag analysis did not investigate any further impact of biochar application on mesofauna activity.

The lack of significant differences in decomposition rates among the biochar and control treatments indicated that 2.5 yr after application biochar did not result in any statistically significant chronic priming effect for the decomposition of freshly added coarse wheat residues, since the observed differences could be attributed to natural spatial variability. Our results are in direct contrast to Wardle et al. (2008), who stated that charcoal maintained an influence on decomposition of soil humus for 10 yr. The exact reasons for these differences could be related to the fact that the Wardle et al. study was conducted in a forest soil, where the liming effect of biochar could play a more critical role than in our Midwest agricultural soil. Furthermore, upon closer inspection of their data, the mass loss rates of humus vs. humus-charcoal mixtures after the first year appear similar, suggesting that the influence was not continuous but only a carry-over effect from the initial impacts. This is supported by their own data in which their substrate induced respiration biomass assessments indicated microbial impacts likely carried through the second year, but were not significant by the fourth.

Wardle et al. (2008) cited the absorption of organic compounds on the charcoal as the leading cause of the increased microbial activity and enhanced decomposition they observed. This hypothesis can be traced back to the early 1950's, with Turner (1955) suggesting this as a potential explanation for the increased growth of clover following biochar additions. According to Bruun et al. (2011) an incomplete conversion of feedstock into biochar, as would result from a natural fire or a fast pyrolysis platform, can leave behind decomposable labile material that can sorb to the biochar. The impact of these sorbed volatiles on ash has been reviewed recently by Nelson et al. (2012). Accessibility to this labile component might stimulate soil microbial activity, which may have led to the greater turnover of soil C and N observed with fast pyrolysis biochars in comparison to slow pyrolysis biochars made from the same feedstock (Bruun et al., 2012).

In the current study, a remnant effect of sorbed labile materials could be why wheat straw decomposition was somewhat higher in the fast pyrolysis wood-based biochar treatment (BC1) than all slow pyrolysis wood-based biochar treatments. Along the same lines, Zimmerman et al. (2010, 2011) determined a greater effect on soil processes from labile components released from freshly added low temperature pyrolysis biochars made from grass and pinewood feedstocks as compared to slow pyrolysis hardwood biochars. Luo et al. (2011) also determined that this priming effect declined with increasing pyrolysis temperatures. The somewhat higher decomposition of the wheat straw in the wheat mids biochar (BC5) and pine chip biochar (BC6) treatments compared to the slow pyrolysis hardwood biochars falls in line with these evaluations.

These studies all indicated that sorbed compounds and not the actual biochar structure were responsible for the impact on microbial communities. Though the present study still indicated the absence of an effect on microbial biomass and decomposition rates, the significant correlation between the two could be a residual of an impact that might have occurred when the biochar was freshly added. Regardless, the current data indicated that any potential impact from initial application is not likely to last beyond three years in the field. A lack of correlation with pyrolysis conditions and feedstocks was also concluded in a recent meta-analysis of biochar plant growth responses (Crane-Droesch et al., 2013).

The lowest rate of decomposition, correlating with the lowest microbial biomass measurement in the macadamia nut biochar treatment (BC7) was notable. A reduction of CO_2 production rates in the laboratory using fresh samples of this biochar (Spokas and Reicosky, 2009) was attributed to elevated ethylene levels (Spokas et al., 2010). Ethylene can inhibit soil microbial processes (Augustin, 1991; McCarty and Bremner, 1991; Wheatley, 2002), plant growth (Deenik et al., 2010) and soil greenhouse gas production (Spokas et al., 2009). Though weathering in the field may have reduced the impact of ethylene, such that the results were not significant, the lower decomposition rates observed here could be the residual of this earlier impact.

Changes in soil physical and chemical characteristics, such as higher moisture content, reduced soil bulk density and increased nutrient availability, have been noted with fresh biochar additions (Atkinson et al., 2010; Sohi et al., 2010; Spokas et al., 2012), though these potential changes from multiple biochars in field plots are rarely compared (Brockhoff et al., 2010; Laird et al., 2010; Meyer et al., 2012). Biochars greater than 1 cm in size are likely to influence soil bulk density, which includes some of the biochars used in this study. These effects may have contributed to the high variability in our results, thus negating our ability to detect potentially real trends.

5 Conclusions

In this study we evaluated the impact of seven different biochars and one non-biochar wood pellet amendment on the degradation rate of wheat straw in Minnesota field plots. The results indicated that 2.5 yr after application these biochars had no significant impact on the decomposition of freshly added organic residues. The variability in decomposition rates among the biochars could be correlated to impacts observed with fresh biochar (sorbed volatile components), thus providing some indication these slight differences might be of short duration as the compounds volatilize or are mineralized. Soil microbial biomass changes, reduced in the macadamia nut derived biochar plots and conversely increased in the wood pellet amendments, were the most likely drivers of the variability in the decomposition rates observed. These observations demonstrated a one-time fresh biochar application has little potential for long-term influence on the soil decomposer community. Detailed short and long-term field analyses using charred and un-charred feedstocks, fresh and weathered, are necessary to confirm this result.

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Table 1. Treatment designations by production source, feedstock type, and pyrolysis method and temperature, with volatile matter (VM), C and N content.

Treatment designation	Biochar source[a]	Feedstock	Pyrolysis method[b]	Pyrolysis Temperature (°C)	% VM	% C	% N
Control	–	–	–	–	–	–	–
WP	Somerset Wood Pellets (US)	Hardwood Pellet	Uncharred	–	23.5	76.9	0.2
BC1	Dynamotive BC (Canada)	Hardwood	fast	500	26.1	63.8	0.2
BC2	Chip Energy (US)	Hardwood Pellet	slow (updraft gasifier)	> 500	12.4	69.0	0.1
BC3	Best Energies (US)	Mixed hard and softwoods	slow	550	34.8	71.1	0.1
BC4	Cowboy Charcoal (US)	Hardwood	slow	538	32.5	88.3	0.3
BC5	ICM (US)	Wheat mids	slow	540–600	22.4	81.8	0.5
BC6	ICM (US)	Pine chip (bark + wood)	slow	600–700	45.8	64.3	3.1
BC7	Biochar Brokers (US)	Macadamia nut shell	fast	650	19.5	71.0	0.9

[a] Names are necessary to report factually on available data; however, the USDA neither guarantees nor warrants the standard of the product, and the use of the name by USDA implies no approval of the product to the exclusion of others that may also be suitable.

[b] Abbreviations: fast less than 2 s resident time; slow greater than 2 s.

Table 2. Decomposition rate constant, k, with standard error (s.e.) and 95 % lower and upper confidence limits (LCL, UCL), model fit (r^2), and microbial biomass carbon (MBC) with s.e.

Treatment	k ($\times 10^{-3}$ d^{-1})	s.e. ($\times 10^{-3}$ d^{-1})	95 % LCL ($\times 10^{-3}$ d^{-1})	95 % UCL ($\times 10^{-3}$ d^{-1})	r^2	MBC (µg g^{-1} soil)	s.e. (µg g^{-1} soil)
Control	8.3	0.3	7.5	9.0	0.76	142	53.4
WP	9.8	0.4	8.9	10.7	0.72	835	19.2
BC1	9.8	0.6	8.6	10.9	0.63	232	31.0
BC2	9.2	0.6	7.9	10.5	0.53	277	64.5
BC3	8.0	0.4	7.1	9.0	0.50	136	10.7
BC4	8.9	0.4	8.0	9.9	0.71	133	19.6
BC5	8.8	0.4	7.8	9.9	0.58	239	54.3
BC6	9.6	0.5	8.5	10.8	0.56	435	48.0
BC7	7.5	0.3	6.7	8.4	0.63	117	24.5
Mean						283	44.3

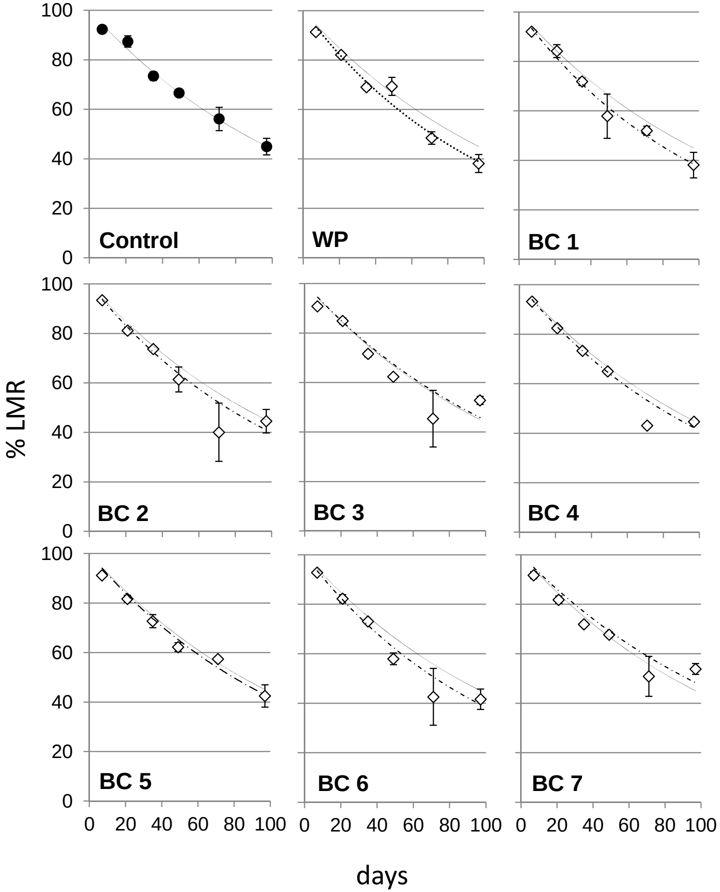

Fig. 1. Average percent litter mass remaining (%LMR), over days of incubation, by treatments given in Table 1. Modeled exponential decay curves are shown for each treatment (broken lines) compared to control (solid line). Bars indicate one standard error of the mean (*n* = 3 or *n* = 9; see text).

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