Investigating the Influence of Biochar on the Soil N-Cycle, C Sequestration, and Stabilization with Soil Minerals

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Executive Summary:

Due to increased urbanization and population growth during the 20th and 21st centuries, the need for higher food production on smaller land areas has lead to a marked increased in the use of N fertilizers for agronomic production. The use of biochar, the byproduct of the pyrolysis of biomass in biofuel product, has received widespread attention as an agricultural amendment with the potential ability of reducing N leakage (increase N-fixation and decrease N₂O emissions) while sequestering carbon (stabilize soil organic C and decrease CH₄ emissions), improving soil fertility (increase cation retention), and increasing water retention in soil [1]. With the thrust for biochar use gathering steam, many in the agricultural field will soon be faced with evaluating if biochar application to their soil is prudent. Therefore, in order to assess the impacts of biochar soil amendments, on greenhouse gas (GHG) emissions and C and N sequestration in soil, our functional research objectives were to:

- 1) Characterize the physical and chemical properties of biochar originating from various organic residue sources and pyrolysis conditions;
- 2) Examine the ability of biochar to influence soil N-cycles and C sequestration when applied to agricultural soils and under conditions favoring denitrification via lab incubations:
- 3) Elucidate molecular-level interactions involved in the stabilization of biochar by soil minerals.

For objective 1, elemental composition of the biochar feedstock, functional group content, and surface area of the raw feedstock were determined. To achieve objectives 2, soil (Yolo silt loam) and various biochars [wood: 410° (WF410), 510° C (WF510); walnut shell: 900° C (WA900)], at an agriculturally relevant application rate of 10 tons ha⁻¹, were incubated for 28 days, in the absence and presence of acetylene (C_2H_2). Biochar only incubations and dissolved C and N sorption isotherms were also conducted to determine the effect of the biochar on native organic matter (NOM) derived from compost. Long term incubations (60d, 180d and 365d) of biochar with soil and clay minerals (i.e., gibbsite, goethite, hectorite) with various spectroscopic analyses were used to achieve objective 3.

Substantial differences in pH, ash content, cation exchange capacity (CEC) and C:N, aromaticity, surface chemistry, and surface area were apparent for biochar from different source material. Biochar soil amendments, under conditions favoring denitrification, did not significantly affect CO₂ and N₂O emissions when compared to the addition of compost and inorganic fertilizer (urea) at an agriculturally relevant application rates 120 kg applied N ha⁻¹ (100 mg N kg⁻¹ soil). The biochar with the highest pH, ash content, surface area and basicity (WA900) appears to stabilize the NOM pool most, while treatments with the biochar with the highest H:C and lowest pyrolysis temperature and surface area (WF410) had the greatest emissions of the GHGs.

Our research is the first to show the changes that occur in functional forms of C on the biochar surface when incubated in soil as a function of time at a resolution of a few nanometers. The observed decrease in aliphatic C and increase in aromatic C on the biochar surfaces as a function of time suggests that, initially, fresh biochar has readily mineralizable material at the surface that can be removed (abiotic or biotic) to reveal a core that is essentially aromatic and more resistant to degradation and thus a potential vehicle for C sequestration. We anticipate that our study will assist a wide range of stakeholders in the biochar arena (e.g., manufacturers, farmers, agricultural extension advisors, policy makers) to make more informed decisions on biochar use in agroecosystems and particularly its role in C and N cycling.

Specific Results:

Objective 1. Given the lack of standardized analysis methods for biochar characterization, as well as the broad range of products labeled as "biochar", many in the agricultural field are faced with the dilemma of what constitutes "good biochar" as well as its applicability to their soils. Biochar has been made from a variety of feed stocks, including orange peels, rice husks, nut shells, pine shavings, manure and straws [2,3]) but mostly on a small scale. Table 1 gives the manufacturing information and physical properties of 12 commercially available biochar which have been studied in our lab. The most prevalent feedstock used is wood, and the biochar produced typically has a lower moisture and ash content. Comparison of biochar made from similar feedstock (wood for example) show that surface area varies with pyrolysis temperature, with a low pyrolysis temperature corresponding to a low surface area.

Table 2 reveals that most biochars are alkaline (6.8-10.9) with the highest pH in the nonwood biochars (i.e., turkey litter, walnut shell). These biochar also have a higher ash content and alkali/alkaline elements. The higher contents of total P (0.64 and 6.61% for WA900 and TL respectively) and K (9.3 and 7.0 % for WA900 and TL respectively) of these chars could make them beneficial soil amendments; however, the associated increase in soil pH may be of concern for some soils. Significant differences in the elemental content of C (15-88%), N (0.21-1.96%), O (1.6-31.7%) and H (0.83-4.38%) are also observed with the biochar grouping by feedstock material and pyrolysis temperature, with the highest C content in the wood derived biochar and the lowest in the manure biochar. All the biochar have C:N that are >20, with the highest ratios in the wood derived biochars, which according to is a good predictor for an organic material with capabilities of net N immobilization [4]. The H:C, an approximation of aromaticity, decreases with increasing pyrolysis temperature when comparing the wood derived biochar. This ratio is important because it is a good indicator of the susceptibility of a material to rapid degradation by soil microorganisms [5]. Additions of "enhancers" such as algae (EBC) or beneficial soil microorganisms (IC) to wood biochar seem to have a positive effect on properties such as CEC and mineral content when compared to the wood only biochars. The surface basicity of the walnut shell (11.7 meg g⁻¹) was much greater than all the other biochar while the surface acidity of the biochar with additions (HWB, EBC and IC) was greater than the biochar made purely from wood.

Analysis of the FTIR spectra of the biochar, used to determine chemical functionality of the biochar (Fig. 1), showed a separation of the biochar into two groups wood derived versus non wood derived. Table 3 shows assignments of peaks of these functional groups as well as a comparison ratio of these peaks with a peak at 870 cm⁻¹ (aromatic C-H stretch). Also in Table 3 are peak height ratios from Raman spectroscopic analysis. The measurements were taken from the D (1350 cm⁻¹) and G (1580 cm⁻¹) bands, two bands typically observed in the Raman spectra of carbon which relate to sample aromaticity. Due to availability and difference in

characteristics, WF410, WF510 and WA900 were selected for use in the soil incubations to achieve Objective 2.

Objective 2. Application of biochars WF410, WF510 and WA900 at an agriculturally relevant application rate of 120 kg applied N ha⁻¹ (100 mg N kg⁻¹ soil) to a Yolo silt loam, under conditions favoring denitrification (i.e., 90% WFPS, C₂H₂), did not significantly influence the emission of both CO₂ and N₂O from the soil when compared to the addition of compost and inorganic fertilizer (urea) (Fig. 2 and 3). Despite the differences in cumulative values, the large standard errors resulted in acceptance of the ANOVA null hypothesis of equal means. We hypothesize that particulate organic matter present in the soil and biochar was the source of the large variation. According to Parkin [6], particulate organic matter supports higher specific rates of denitrification than soil material and due to its natural patchy distribution in soils, can result in a high degree of small scale variability in emissions of N₂O.

Figures 2 and 3 indicate that C_2H_2 increased the emission of both GHGs from the treatments. The greatest increase occurred in the N_2O fluxes because nitrifying inhibitors such as C_2H_2 , nitrapyrin and dicyandiamide are commonly used to prevent the conversion of N_2O to N_2 [7,8] resulting in a greater emission of the former [9] and thereby providing a measurement of total denitrification based on N_2O emissions. C_2H_2 also inhibits the conversion of N_2O (nitrification), reducing loss of soil NO_3 -N from leaching. The N_2O flux without the addition of C_2H_2 is attributed to natural incomplete denitrification [10].

Observation of the GHGs alone would suggest there was no difference between the treatments; however, analysis of the soil chemical analysis data shows otherwise. The large differences in the DOC data (Table 3), observed at the beginning of the incubations, can be explained by sorption of the soil and compost DOC by the biochar. Ongoing research in our lab has shown a similar trend of sorption of DOC to the different biochar (as seen in the treatments in the absence of C_2H_2) correlating well with biochar surface area, and also the observed increase in DOC in the SWF410 treatment from the breakdown of this biochar in solution. This phenomenon makes use of C mineralization data as an indicator of microbial activity in the various treatments complex. Despite this, the calculated C mineralization rates did decrease as expected in the presence of C_2H_2 evident in the soil only treatment. The high DOC measured in the soil only treatment (in the absence of C_2H_2) was due to accumulation of mineralized organic compounds in the soil solution that could not be assimilated by the low microbial population present.

In our study, the emissions of GHGs for the non-soil incubations correlated well with the H:C of the biochars (Table 3). Among the wood feedstock biochars, the WF510 biochar had a lower H:C (0.30 and 0.76 for WF510 and WF410 respectively) and was more resistant to mineralization. This is also consistent for chars made at high temperature (>700° C) due to formation of highly condensed aromatic structures and loss of more open oxidizable functional groups [10-12]. For the WF410 biochar, addition of compost and compost extract resulted in an increase in CO₂ emission when compared to treatments of the compost and compost extract alone. This would suggest this biochar enhances mineralization of the NOM pools however, the high CO₂ emission from the biochar indicates that the biochar itself was not resistant to decomposition in contradiction of other studies where it has been proposed biochar should have a limited effect on the C and N dynamics in the soil because it is recalcitrant [13]. To confirm this a short-term study was initiated to compare wet biochar versus dry biochar behavior and the results support the theory that the presence of water leads to elevated CO₂ emissions from biochar. Spokas et al, [14] observed similar emissions from a water blank treatment and

suggested biotic/ abiotic reactions of water and O_2 , as well as mineralization of pyrolysis byproducts sorbed onto the char surface as sources of the CO_2 .

For the WF510 biochar, the composite treatments all were lower than the treatments of the compost and compost extract alone (Fig. 4). This implies this biochar aides in stabilizing the compost and compost extract NOM. Explanation of this result in terms of effect of biochar on the NOM pool is complicated by possible removal of CO₂ onto the surface of the biochar. This behavior was very evident in the WA900 treatments where complete removal of headspace CO₂ occurred. Our study was not able to determine the mechanisms for the CO₂ removal from the headspace, knowledge of which would determine the length of time the CO₂ can be held by the biochar and should be investigated.

The effects of the biochar were not as significant on the N_2O emissions (Fig. 5a and b) except for the WF410 biochar which appeared to greatly stimulate N mineralization in the compost (Fig. 5b). Interestingly, similarly large fluxes of N_2O were not observed in the compost extract treatment. This might be explained by the presence of a smaller microbial population in this treatment compared to the compost. The large difference in fluxes (presence vs. absence of C_2H_2) for this treatment and WF510C show the destabilizing effects of these biochar on compost and the potentially large N_2O fluxes attainable under conditions of total denitrification instead of natural incomplete denitrification.

Sorption isotherms were conducted using the same three biochar as for the soil incubations (WA900, WF410 and WF510) and activated carbon (AC) and kaolinite (KAO), a clay mineral, as controls. Activated carbon has a very high surface area (1040 m² g⁻¹ for this sample) and has many applications such as gas and water purification and metal extraction that make use of its binding ability to inorganic and organic compounds. The data showed the AC and WA900 bound significantly more N and C than WF410, WF510 and KAO (Fig. 6a and b). The WF410 biochar was unstable in solution and this resulted in a release of N and C into the batch solution during the 48 h of pre-equilibration. A plot of % bound C versus % bound N (Fig. 7a) showed all the materials bound significantly more C than N with AC and WA900 binding the most. Specific UV Absorbance (SUVA) analysis of the compost extract before and after incubation showed all preferentially remove compounds solution the materials aromatic from [SUVA]_{final}/[SUVA]_{initial} < 1 (Fig. 7b). WA900 biochar consistently removed a greater portion of the aromatic fraction of DOC than the other materials with all these other materials showing a general trend of decreased preferential binding of aromatics with increased binding of C. The differences in the walnut shell and the two wood feedstock biochar sorption data show the feedstock had a significant effect on the biochar's ability to bind C and N. The pyrolysis temperature also had a significant effect on the behavior and properties of the biochar with the higher temperature biochar (WF510) being more stable and having significantly greater C absorption than the WF410. Due to the hydrophobic nature of biochars they tend to preferentially bind aromatic fractions of DOC. However, as concentrations of DOC increase the sorption of non-aromatic regions of DOC increases.

Objective 3. A key factor controlling the long term stability of biochar in soil is it interactions with soil minerals, which can serve to stabilize and protect biochar C. Scanning Transmission X-ray Microscopy (STXM) studies show that all the biochar were chemically altered with a trend towards increasing aromaticity during incubations with soil (decrease in aliphatic peak at about 287.3 eV in comparison to the aromatic peak at 285.3 eV) for all three biochars (Fig. 8). The data also show degradation of the biochar occurs quickly, with differences in the C edges evident after only 60 days of incubation in soil. C edges obtained from biochar incubated in the absence

of soil for 314 days were not significantly different from those from biochar incubated in soil for 365 days suggesting the stability of the biochar may not be entirely due to interactions with the soil. The occurrence and peak location of carboxyl C at the biochar surface (~288.8 eV), in the WA900 spectra (Fig. 8c), represents aliphatic rather than aromatic carboxyl C absorption [15].

Synchrotron-FTIR studies of the incubated biochar confirm the increase in aromaticity of the biochar through an increased prominence of the peak at 1596 cm⁻¹ corresponding to an aromatic C=C stretch node (Fig. 9). Incubation also leads to an increased functionalization on the biochar surface (Ester C-O: 1750 cm⁻¹; and C-O stretching and O-H deformation of COOH groups: 1260 cm⁻¹ (Fig. 9). This increased functionalization may facilitate bonding interactions with clay minerals and increases the stability of biochar in soil.

Potential Impacts:

Given the lack of standardized analysis methods for biochar characterization, it is very difficult for farmers to distinguish the difference between biochars which have potential for beneficial, or negative, impacts to their agroecosystem. This is particularly significant for the organic farming communities who have been encouraged by the prospects of this eco-friendly soil amendment. There is a dearth of information on the characteristics of commercially available biochar and their effects on soil, so our research will be beneficial here. However, additional studies are required to adequately address agroecosystem impacts arising from biochar soil amendments. For example knowledge of a particular biochar characteristics such as 1) cation and anion exchange is important for N cycling as it may determine the potential for processes like cation (NH₄⁺) or anion (NO₃⁻) retention and thus be important in preventing nutrient leaching; 2) surface area is important for predicting CEC and possible sorption of GHGs; 3) biochar pH is important for predicting impact on the soil pH (Fig. 10) and mobility of cations in the soil and; 4) H:C as this ratio approximates aromaticity of the biochar and is an indication of its ability to be mineralized (high H:C, very susceptible; low H:C, less susceptible).

Manufacturers of biochar will also benefit from this biochar characterization data as they can tailor biochars more suited for a particular soil given the farmers' requirements. Also decisions such as selecting feedstock prone to producing biochar with a high ash content (which is challenging to transport and incorporate into soil) and yet have a high mineral content (nutrient source) and high pH (beneficial if liming desired) will be more informed. Another group of stakeholders that stand to benefit from our study are companies looking to invest in biochar. Though highly touted, biochar is still a field in its infancy and many investors have been hesitant to invest in its production due to sparse scientifically backed information. Better knowledge of the impacts of biochar on agroecosystems will help in evaluating the logistics of biochar soil amendments, including life cycle assessments, feasibility studies and associated carbon credits.

The incubation data from this current study highlight the different effects of biochar on native organic matter dependent on the feedstock as well as the pyrolysis temperature. This will be valuable information for N management for farmers and agricultural extension officers as they try to determine biochar type and appropriate application rates of biochar into farming land with differing quantities of available native organic matter. Also of note is the reduced CO₂ emissions from the compost and compost water extracts in the presence of WA900 biochar. We hypothesize that microbial inhibition by the biochar is another potential source for the reduced levels of CO₂.

Understanding the role of biochar in C sequestration, whether as a source of stabilized black C or as a stabilizer of native soil organic carbon, is of great importance to a wide range of stakeholders concerned with climate change and emissions of GHGs. In order to fully evaluate

the fate of biochar- and native-C due to biochar amendments, an increased understanding of molecular- and atomic-level interactions with soil minerals is required. Most soil analyses are done on the bulk soil sample and though informative, they do not give a true representation of processes occurring at these small scales. Techniques such as STXM and SR-FTIR enable investigation at this scale and our data will be important in adding knowledge to scientific models predicting possible effects of biochar addition to soil, for better C management practices and climate change modeling. While this data set is certainly not exhaustive, it does stress the importance for considering biochar source material and pyrolysis conditions before applying it as a soil amendment. To further investigate the interactions of biochar with soil minerals our lab is continuing a series of synchrotron based studies to determine the specific chemical mechanisms responsible for the stabilization of C and N in soil.

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Tables and Figures

Table 1. Manufacturing information and physical characteristics of 12 commercially available biochars.

| Char | Source Material | Pyrolysis Temp (°C) | Moisture (wt %) | Ash (wt %) | BET Surface Area (m²/g) |
|------------------------|---------------------------|------------------------|--------------------|---------------|----------------------------|
| Biochar C (Bio C) | Wood chips | unavailable* | 2.7 | 4.2 | 301.6 |
| Biochar M (Bio M) | Wood chips | unavailable | 2.4 | 5.5 | 154.40 |
| Biochar S (Bio S) | Wood chips | unavailable | 4.7 | 2.8 | 153.10 |
| Charcoal Green (CG) | Wood chips | unavailable | 2.6 | 5.0 | 164.10 |
| Enhanced Biochar (EBC) | Wood + Algal digestate | 600-700 | 37.6 | 6.4 | 2.03 |
| Hog waste (HWB) | Soft wood chips/hog waste | 600-700 | 64.1 | 2.4 | 25.15 |
| Inoculated Char (IC) | Inoculated wood | unavailable | 12.3 | 15.5 | 95.90 |
| New Earth Pine (NEP | Wood chips | 500-650 | 3.9 | 17.0 | 4.97 |
| Turkey Litter (TL) | Turkey litter | 700-800 | 11.8 | 64.0 | 21.77 |
| Walnut Shell (WA900) | Walnut shell | 510 | 3.1 | 40.4 | 227.10 |
| Wood feedstock (WF410) | Wood | 410 | 2.0 | 2.6 | 2.82 |
| Wood feedstock (WF510) | Wood | 510 | 4.4 | 3.0 | 165.80 |

^{*}Biochar producer either did not know of exact pyrolysis temperature or considers the information as proprietary and did not provide.

Table 2. Chemical characteristics of 12 biochars commercially available biochars.

| Char | pH _w (1:2) | C (wt | N (wt %) | H (wt | O (wt %) | PO ₄ -P (wt %) | K (wt %) | S (ppm) | Fe(total) (ppm) | CEC* | Acidity | Basicity |
|-------|-----------------------|-------|----------|-------|-------------|------------------------------|----------|------------|--------------------|------|---------|----------|
| Bio C | 9.5 | 82.5 | 0.49 | 1.64 | 5.6 | 0.06 | 1.02 | 160 | 473 | 16.5 | 0.36 | 1.21 |
| Bio M | 8.8 | 85.4 | 0.55 | 2.37 | 8.9 | 0.07 | 0.48 | 140 | 606 | 3.6 | 0.60 | 0.94 |
| Bio S | 9.5 | 88.0 | 0.44 | 2.55 | 14.8 | 0.02 | 0.33 | 60 | 79 | 14.9 | 0.49 | 0.87 |
| CG | 9.2 | 87.3 | 0.59 | 2.15 | 7.4 | 0.07 | 0.85 | 140 | 203 | 9.1 | 0.41 | 0.84 |
| EBC | 6.8 | 58.1 | 0.41 | 4.16 | 31.7 | 0.08 | 0.19 | 685 | 3370 | 67.0 | 1.56 | 1.22 |
| HWB | 7.5 | 68.2 | 0.51 | 3.66 | 26.8 | 0.13 | 0.26 | 370 | 1934 | 26.2 | 1.24 | 1.02 |
| IC | 6.8 | 53.3 | 1.96 | 3.70 | 24.3 | 0.47 | 1.20 | 5920 | 1109 | 44.5 | 1.22 | 1.00 |
| NEP | 7.9 | 71.2 | 0.91 | 2.88 | 11.6 | 0.08 | 0.72 | 480 | 3517 | 3.2 | 0.79 | 1.01 |
| TL | 10.9 | 15.6 | 0.78 | 0.83 | 4.4 | 6.61 | 7.05 | 10720 | 9191 | 24.4 | 0.08 | 4.92 |
| WA900 | 9.7 | 55.3 | 0.47 | 0.89 | 1.6 | 0.64 | 9.32 | 940 | 1981 | 33.4 | - | 11.71 |
| WF410 | 7.1 | 65.7 | 0.21 | 4.38 | 23.5 | 0.02 | 0.12 | 50 | 248 | 10.0 | 0.83 | 0.40 |
| WF510 | 7.3 | 83.9 | 0.36 | 1.88 | 19.8 | 0.02 | 0.13 | 110 | 505 | 12.0 | 0.27 | 0.93 |

^{*}CEC = Cation exchange capacity

Table 3. Infrared analysis and Raman analysis of 12 commercially available biochars

| Char | ^x Aromatic C-H (744cm ⁻¹) | *Aliphatic ether (1029cm ⁻¹) | xAliphatic CH3 (1417cm ⁻¹) | *Aromatic C=C (1565cm ⁻¹) | ^x Aromatic carbonyl (1690cm ⁻¹) | $^{ m y}$ Raman $_{ m I_d/I_g}$ |
|-------|--|--|--|---|--|---------------------------------|
| Bio C | 1.1 | 1.01 | 1.05 | 1.22 | 0.12 | 0.71 |
| Bio M | 1.05 | 1.17 | 1.6 | 1.9 | 0.36 | 0.59 |
| Bio S | 0.71 | 1.67 | 0.27 | 1.41 | 0.36 | 0.72 |
| CG | 1 | 1.2 | 1.83 | 1.98 | 0.38 | 0.68 |
| EBC | 0.27 | 2.83 | 2.2 | 2.05 | 0.94 | 0.65 |
| HWB | 0.63 | 2.67 | 2.46 | 2.6 | 1.2 | 0.58 |
| IC | 0.53 | 3.6 | 0.38 | 0.69 | 0.39 | 0.76 |
| NEP | 1.12 | 1 | 1.71 | 1.78 | 0.41 | 0.83 |
| TL | - | 2.9 | 1 | 0.21 | 0.02 | 0.40 |
| WA900 | - | 1.1 | 2.2 | - | - | 0.34 |
| WF410 | 1.2 | 2.16 | 0.83 | 1.5 | 0.4 | 0.25 |
| WF510 | 1.2 | 2.09 | 1.7 | 2.3 | 0.28 | 0.40 |

x. Ratios of peak intensities relative to the aromatic C-H stretch at 870cm⁻¹ common to all spectra

Table 4. Variations in pH, water loss, dissolved oxygen content (DOC), nitrate (NO₃), ammonium (NH₄) concentrations and net N mineralization rates of soil treatments after 29 days of incubation. Day 0 values are from analysis of bulked samples.

| | pН | | Water Loss DOC (mg/kg) | | NO ₃ -N (mg/kg) | | NH ₄ -N (mg/kg) | | Net N- mineralization | |
|---------|-------|-----------|------------------------|-------|----------------------------|-------|----------------------------|--------------|--------------------------|------------|
| | Day 0 | Day 29 | g/day | Day 0 | Day 29 | Day 0 | Day 29 | Day 0 Day 29 | | (mgN/kg/d) |
| Soil | 7.8 | 7.8 (7.8) | 0.16 | 191 | 247 (192) | 2.9 | 38 (np) | 5.7 | 8.2 (35) | 1.3(0.9) |
| SC | 7.8 | 7.9 (7.9) | 0.15 | 216 | 197 (221) | 6.0 | 50 (np) | 5.7 | 0.6 (29) | 1.4(0.6) |
| Surea | 7.9 | 7.4 (7.9) | 0.15 | 197 | 194 (196) | 2.9 | 135 (np) | 6.7 | 1.2 (101) | 4.4(3.2) |
| SWF410C | 7.8 | 7.8 (7.9) | 0.12 | 201 | 233 (199) | 4.9 | 48 (np) | 5.6 | 5.5 (29) | 1.3(0.6) |
| SWF510C | 7.8 | 7.8 (8.2) | 0.11 | 202 | 206 (218) | 4.9 | 51 (np) | 5.6 | 3.8 (30) | 1.5(0.7) |
| SWA900C | 8.6 | 8.6 (8.6) | 0.14 | 160 | 192 (192) | 5.0 | 52 (np) | 5.7 | 0.6 (11) | 1.9(0.1) |

Values in parenthesis are for the incubations in the presence of C₂H₂

y. Ratio of peak intensities of the Carbon D (1350cm⁻¹) and G (1690cm⁻¹) bands in Raman spectra

 $np-not\;present$

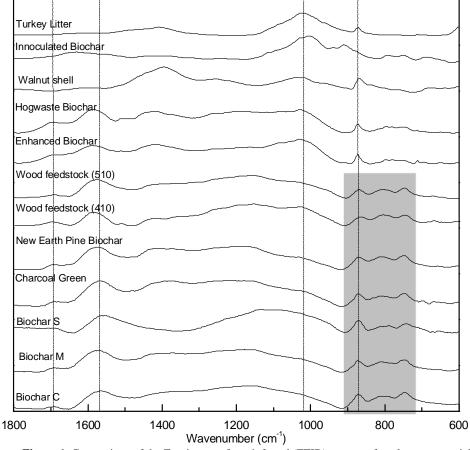


Figure 1. Comparison of the Fourier transform infrared (FTIR) spectra of twelve commercially available biochar. The shaded area shows the common peaks found in wood derived biochar.

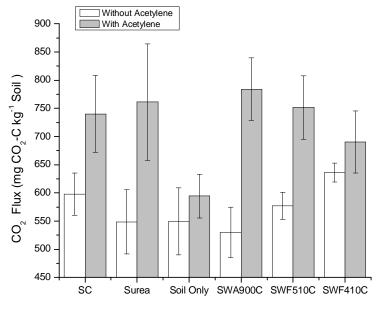


Figure 2. Cumulative CO_2 production from incubations of soil only, soil/compost, soil/urea and soil/compost / biochar of the three biochars; WA900, WF410 and WF510 in the presence and absence of C_2H_2 . Averages and standard errors of the triplicate incubations are shown. Fluxes are relative to experimental controls of ambient air and are corrected for water content.

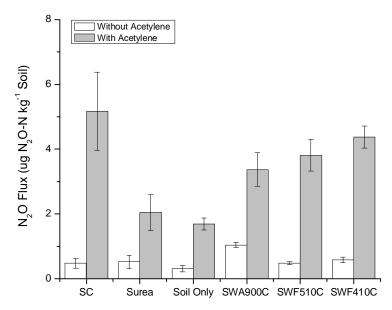


Figure 3. Cumulative N_2O production from incubation of soil only, soil/compost, soil/ urea and soil/compost/biochar of the three biochars; WA900, WF410 and WF510 in the presence and absence of C_2H_2 . Averages and standard errors of the triplicate incubations are shown. Fluxes are relative to experimental controls of ambient air and are corrected for water content.

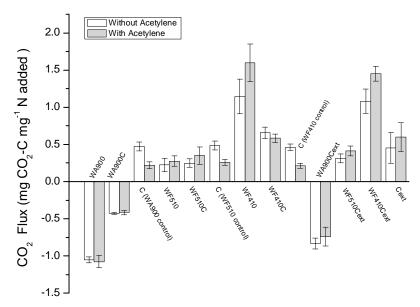


Figure 4. Cumulative CO₂ production from incubations of biochar only, biochar/compost and biochar/compost water extract of the three biochars; WA900, WF410 and WF510 under denitrifying conditions. Averages and standard errors of the triplicate incubations are shown and the fluxes are relative to experimental controls of ambient air over water. Due to differing compost additions, the compost only treatments are relative to a biochar (name in parentheses).

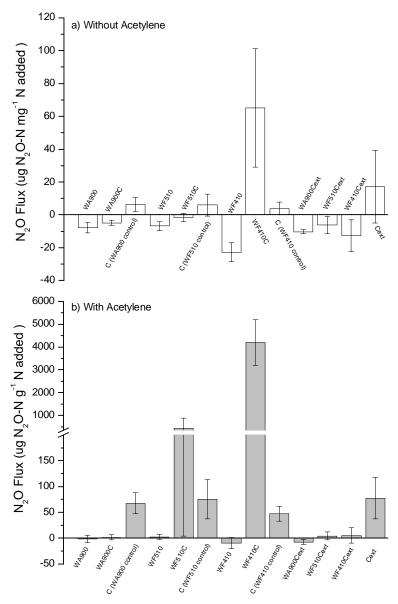


Figure 5. Cumulative N_2O production from denitrification a) in the absence of C_2H_2 and b) in the presence of C_2H_2 of biochar only, biochar/compost and biochar/compost compost water extract of the three biochars; WA900, WF410 and WF510. Averages and standard errors of the triplicate incubations are shown and the fluxes are relative to experimental controls of ambient air over water. Due to differing compost additions, the compost only treatments are relative to a biochar (name in parentheses). Due to differing compost additions, the compost only treatments are relative to a biochar (name in parentheses).

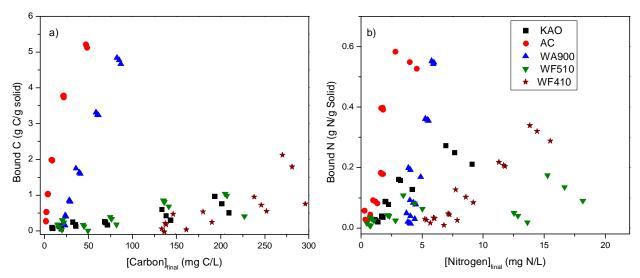


Figure 6. Sorption isotherms of a) dissolved organic carbon and b) nitrogen to kaolinite, activated carbon and three biochars, WA900, WF410 and WF510.

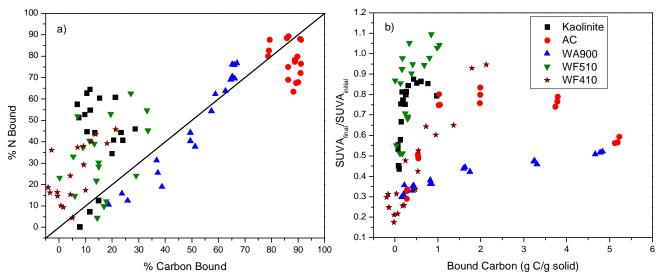
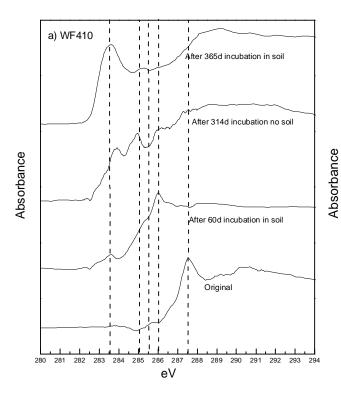


Figure 7. Plots showing a) the relationship between % N bound and % C and b) [SUVA]_{final}/[SUVA]_{initial} as a function of bound C to kaolinite, activated carbon and three biochars (WA900, WF410 and WF510).



After 365d incubation in soil

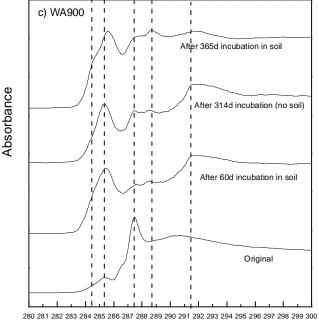
After 90d incubation in soil

Original

Original

280 281 282 283 284 285 286 287 288 289 290 291 292 293 294

Figure 8. STXM spectra of C edges of a) WF410, b) WF510, and c) WA900 biochars made from wood feedstock (WF410, WF510) and walnut (WA900) at pyrolysis temperatures of 410°, 510°, and 900° C and incubated both with and without soil at 25°C and 55-60% water filled pore space (WFPS) for differing time periods.



eV

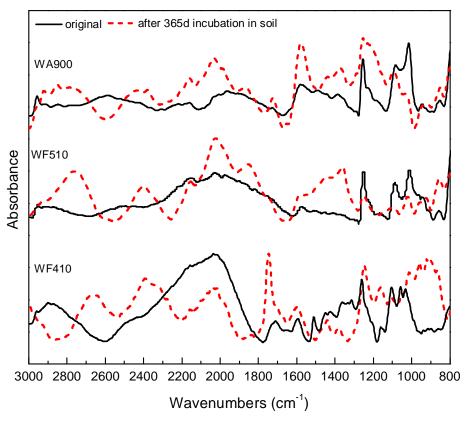


Figure 9. SR-FTIR spectra of three biochar before and after 365 days of incubation.

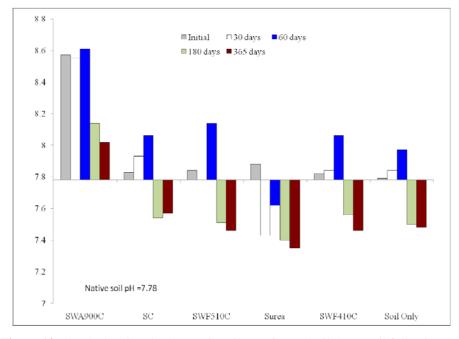


Figure 10. Graph showing the change in soil pH of a Yolo silt loam soil following incubations with biochar.