Nitrogen and Phosphorus Availability of Biologically and Thermochemically Decomposed Human Wastes and Urine in Soils With Different Texture and pH

Leilah Krounbi,[1] Harold van Es,[1,2] Nancy Karanja,[3] and Johannes Lehmann[1,2]

ABSTRACT:
Human waste contains ample plant-available nitrogen (N) and phosphorus (P), which can be used to address soil nutrient deficiencies. Biological and thermochemical sanitization methods have been proposed for safely treating the solid fraction of human waste (called human solid waste [HSW] in this study) but have rarely been compared directly. Furthermore, interactions between amendments and soil properties are also unclear, as pH, texture, and the presence of oxides all affect N and P availability. We therefore evaluated the effect of three HSW sanitization methods, thermophilic composting (60°C), torrefaction (200°C), and pyrolysis (300°C, 500°C), on N and P availability following incubation in four soils with different pH (4.6–7.4), textures (clay loam to loamy fine sand), and mineralogy. We also evaluated the effect of added urine with pyrolyzed HSW (500°C) on N and P availability in soils. A trade-off between N and P availability was observed with highest heating temperature of HSW, the magnitude of which varied with soil type. Compared with pyrolyzed HSW (500°C), composted HSW increased available soil N by 15% to 25%, whereas torrefied HSW increased available soil N by 40% to 57%. The change in P availability over time increased with pyrolyzed HSW (500°C) by 41% to 106% and 84% to 165% compared with composted and torrefied HSW, respectively. Multivariate linear regression highlighted the negative correlation between P availability and both soil oxides and amendment dissolved organic carbon. Combined fertilization with pyrolyzed HSW (500°C) and urine ameliorated low N, while preserving high soil P availability, which was 44% to 147% greater than soils fertilized with urine alone. Pyrolyzed HSW (500°C) lowered pH by 0.06 to 0.38 units compared with urine-amended soils with no pyrolyzed HSW. In conclusion, low N and P availability was observed with composted HSW (29–46 kg N ha^{-1}, 6–15 kg P ha^{-1}); ample N and moderate P were available with torrefied HSW (71–107 kg N ha^{-1}, 12–30 kg P ha^{-1}), whereas excess mineral N and ample P were provided in soils amended with pyrolyzed HSW (500°C) together with urine (404–486 kg N ha^{-1}, 33–81 kg P ha^{-1}) albeit at the expense of mild soil acidification.

Key Words: Compost, mineralization, oxides, pyrolysis, torrefaction

(Soil Sci 2018;183: 51–65)

Proper treatment can transform human wastes from noxious contaminants into soil amendments (Esrey, 2001; Schönning and Stenstroem, 2004; Drechsel, et al., 2015; Krause et al., 2016). Decentralized waste-management systems, such as those used in Haiti or Sub-Saharan Africa, are amenable for safe reuse of human waste (Katukiza et al., 2012; Berendes et al., 2015; Mawioo et al., 2016). In such systems, waste is collected separately from industrial and domestic graywater, obviating heavy metal contamination (Vinnerås and Jönsson, 2002). Most nonessential metals in domestic waste streams are not from food, but from dust, cutlery, and even chemical phosphate fertilizers (Koch and Rotard, 2000; Hammer and Clemens, 2007; Winker et al., 2009). Decentralized systems that also use urine-diversion techniques reduce the water content of collected waste, allowing for thermophilic aerobic digestion or thermochemical sanitation methods (O'Keefe et al., 2015; Orner and Mihelcic, 2018; Tarpeh et al., 2018).

[1]*Soil and Crop Sciences, College of Agriculture and Life Sciences, Cornell University, Ithaca, New York, USA.*

[2]*Atkinson Center for a Sustainable Future, Cornell University, Ithaca, New York, USA.*

[3]*College of Agriculture and Veterinary Sciences, Land Resource Management & Agricultural Technology, University of Nairobi, Nairobi, Kenya.*

Address for correspondence: Dr. Johannes Lehmann, 909 Bradfield Hall, Cornell University, Ithaca, NY 14853. E-mail: CL273@cornell.edu

Supplying plant-essential nitrogen (N) and phosphorus (P), as well as conferring soil-conditioning properties, sanitized human waste is an inexpensive and universally accessible amendment (Amoah et al., 2017). As a soil conditioner, the solid fraction of human waste (called *human solid waste* [HSW] in this publication) can enhance nutrient retention in soil through its dissolved organic carbon (DOC) content, buffering capacity, and reactive surface area (Cofie et al., 2005; Heinonen-Tanski and van Wijk-Sijbesma, 2005). Interest in P recovery from the liquid fraction of human waste (called *urine* in this publication) is growing as phosphate mines become depleted (Cordell et al., 2011; Mihelcic et al., 2011). Furthermore, N forms in urine are already plant-available and can be recovered through sorption, crystallization, or distillation processes (Maurer et al., 2006; Udert and Wächter, 2012). In their study on cabbages, Amoah et al. (2017) observed a 51% increase in marketable cabbage yield when soils were amended with fecal sludge + urine compared with poultry droppings + NPK (15-15-15) fertilizer. Lower expenditure for fertilizing cabbages with human waste earned farmers 65% more than with poultry droppings and mineral NPK.

In decentralized waste-management systems relying on urine diversion, HSW can be sanitized biologically through thermophilic composting (Niwagaba et al., 2009; Berendes et al., 2015) or thermochemically through torrefaction or pyrolysis (Shinogi et al., 2003, Shinogi and Kanri, 2003; Katukiza et al., 2012; Serio et al., 2015; Ilango and Lefebvre, 2016). Sanitization via thermophilic composting relies on aerobic biological degradation of pathogenic organisms at 55°C to 65°C. During torrefaction at 200°C to 300°C, HSW is sanitized and dehydrated, whereas pyrolysis relies on higher heat, 300°C to 700°C, and reducing conditions, fundamentally changing the chemical structure of HSW. Studies directly comparing biological and thermochemical conversion with respect to their use in producing a soil amendment have not been done.

Carbon (C), N, and P recovery, as well as plant availability, depends on the waste treatment method (Nelson et al., 2011; Hossain

et al., 2011). Thermophilic composting of urine-diverted HSW within a household latrine vault allows recovery of 92% of organic C, 100% P, and 86% N (Orner and Mihelcic, 2018). However, community-scale decentralized waste management in informal settlements requires off-site composting under ambient conditions, during which 30% to 60% of C is mineralized (Bernal et al., 2009). Concomitant with C mineralization is N mineralization generating plant-available NH_4 and nitrate (NO_3). However, microbial immobilization of available N and P is also expected as C is assimilated into microbial biomass (Tiquia et al., 1996). Lastly, thermophilic composting of HSW can extend up to 6 months (Feachem et al., 1983; Vinnerås et al., 2003; World Health Organization, 2006), during which time temperature variability within the compost pile poses a challenge to sterilization (Anand and Apul, 2014). With prolonged exposure of HSW compost to the environment, nutrient leaching and NH_3 volatilization may result in up to 40% N losses (Bernal et al., 2009).

Torrefaction is a form of mild pyrolysis that ensures sterilization from pathogenic organisms, while causing dehydration and depolymerization of hemicellulose oligomers (Ilango and Lefebvre, 2016). Torrefied biomass and the original feedstock are chemically alike, the former being increasingly "brittle" and hydrophobic, containing a higher proportion of dissolved organic C (Ilango and Lefebvre, 2016; Lohri et al., 2017). Often relied upon to "densify" biomass, torrefaction results in slightly more reduced C compounds, whereas the species and concentrations of N and P remain unchanged relative to the original feedstock. It is not clear whether torrefaction affects their mineralizability.

Pyrolysis greatly alters the nature of organic compounds in the original feedstock through depolymerization and recondensation reactions (Joseph et al., 2010; Enders et al., 2012). Increasing the highest heating temperature (HHT) concentrates ash minerals such as P relative to C in pyrolyzed biomass (biochar), although not necessarily in plant-available forms (Uchimiya and Hiradate, 2014; Zwetsloot et al., 2015). Exposure to temperatures above 300°C converts phytate and polyphosphates to orthophosphate, the latter having greater plant availability (Wang et al., 2012a; Uchimiya et al., 2015; Subedi et al., 2016). However, while increasing HHT may enhance P availability, it decreases N availability. Pyrolysis of N-rich feedstocks such as HSW chemically immobilize N within heterocyclic structures (Almendros et al., 2003; Knicker 2007; Wang et al., 2012b; Zheng et al., 2013). We therefore anticipate variable N and P availability between soils amended with biologically and thermochemically altered HSW. Yet to our knowledge, no examination has been made of the relative advantages of thermophilic composting, torrefaction, and pyrolysis as HSW sanitization methods with regard to N and P availability and the P-N trade-off with higher conversion temperatures.

One option for mitigating the P-N trade-off in pyrolyzed HSW is to coapply biochar with mineral N ($NH_4 + NO_3$) in the form of urine. Even with temporary immobilization of mineral N through electrostatic adsorption (Kizito et al., 2015; Cai et al., 2016; Wang et al., 2016; Nguyen et al., 2017), long-term availability of biochar-adsorbed N is expected. Taghizadeh-Toosi et al. (2012a, b) demonstrated that $^{15}NH_3$ from cow urine was successfully adsorbed onto woody biochar, only to be detected in ryegrass biomass in a subsequent pot trial to which $^{15}NH_3$-loaded biochar was added.

A remaining question pertaining to coapplications of urine and biochar is whether NH_4 in urine will alter P availability due to acidification from ammonia oxidation. This is especially pertinent because, as compared with compost, biochar is expected to uniquely accelerate nitrification (Prommer et al., 2014), generating soil acidity. Numerous mechanisms for biochar-enhanced nitrification have been observed such as liming (Anthonisen et al., 1976; Suzuki et al., 1974; Prosser and Nicol, 2012), the sorption of nitrification-inhibiting phenolic compounds onto biochar surfaces (DeLuca et al., 2002; Berglund et al., 2004), and an increase in the size of microbial communities including ammonia oxidizers within biochar

pores and surfaces (Lehmann et al., 2011; Nelissen et al., 2012; Prommer et al., 2014).

Furthermore, the liming effect of biochar may also maintain soil P availability even as nitrification is enhanced. A 1.6 pH increase was observed by Chan et al. (2008) after coapplication of mineral N and 450°C poultry-litter biochar (10 ton ha^{-1}), generating a significant increase in plant P uptake. It is unclear whether both nitrification and P availability could increase simultaneously when biochar is added with mineral N in the form of urine. It is also unknown whether such urine additions affect P availability from pyrolyzed HSW differently depending on soil properties.

Edaphic constraints of acidity and low cation exchange capacity (CEC), pervasive in tropical environments, and lower N and P availability (Kwabiah et al., 2003; Sanchez, 2002). Nonresponsiveness of such soils to N and P mineral fertilizers can be mitigated with organic amendments such as HSW (Chivenge et al., 2011; Krause et al., 2016). Diverse surface functional groups in compost as well as biochar help buffer the soil solution, reducing P fixation with aluminum oxides. Cation exchange capacity also increases with organic additions, prolonging the retention of cations such as NH_4 in tropical soil (Lehmann et al., 2003). Furthermore, organic acids in mineralizable organic amendments such as torrefied or composted HSW may enhance phosphate availability via anion exchange reactions (Schneider and Haderlein, 2016). Thus, complex chemical and physical interactions between soil and organic amendments such as HSW affect N and P availability over time. In their work in Tanzanian Andosols, Krause et al. (2016) observed an increase in pH from 5.3 to 5.9 together with increased P availability from 0.5 to 4.4 mg kg^{-1} following additions of pasteurized excreta and biochar. Maize grain yields were also 38% greater with excreta and biochar compared with compost made from grass and kitchen scraps. This study broadens our understanding of human waste amendments on soil N and P availability by evaluating HSW sanitized with different methods across multiple soils of varying pH, texture, and mineralogy.

Our research assessed the benefits of HSW sanitized through composting (60°C), torrefaction (200°C), and pyrolysis (300°C and 500°C) for increased N and P availability in soils. Specifically, we (1) compared plant-available N and P in soils amended with biologically and thermochemically treated HSW, (2) quantified the effects of urine application together with pyrolyzed HSW on N and P availability in soils, and (3) assessed the effect of soil properties on N and P availability after additions of HSW amendments.

MATERIALS AND METHODS

Human Solid Waste Amendments
Human solid waste was sampled from 10 "Fresh Life" squat latrines within the informal settlement of Mukuru in Nairobi, Kenya. Squat plates in latrines are fitted with two openings to allow for separation of the solid and liquid fractions (called here *HSW* and *urine*). "Fresh Life" latrines were manufactured by Sanergy (http://www.saner.gy/), a waste-management company operating out of Mukuru, Nairobi.

Following each sitting, latrine users customarily cover waste with 0.25 L sawdust, producing a 1:1 dry volume mixture of HSW and sawdust. Sawdust was a mixture of exotic tree species: eucalyptus (*Eucalyptus grandis, Eucalyptus saligna*), pine (*Pinus ponderosa, Pinus sylvestris, Pinus patula*), cypress (*Cupressus lusitanica*), and grevillea (*Grevillea robusta*). The initial water content of HSW (amended with sawdust) was 72.0% ± 6.8% (wt wt^{-1}). After sun-drying for 1 week on plastic tarpaulins covered with mesh screens, the water content dropped to 23.2% ± 12.4% (wt wt^{-1}). Human solid waste was then homogenized through mixing and stored in airtight plastic drums until pyrolysis and torrefaction.

The amendments evaluated in our incubation included composted HSW (60°C), torrefied HSW (200°C), pyrolyzed HSW (300°C,

500°C), and synthetic urine. Torrefaction of HSW entailed heating HSW under anoxic conditions in an atmosphere of argon gas to 200°C in a muffle furnace (Fisher Isotemp Model 126; Thermo Fisher Scientific, Waltham, MA) fitted with a drum and rotating paddle, whereas pyrolysis was carried out at 300°C and 500°C. The ramping rate was set to 2.5°C min^{-1}, and each batch was held at the HHT for 30 min.

For our trial, we used Sanergy's commercial HSW compost produced during 2013. The compost was filtered through a magnetic sieve (Circular Grid Magnet; Eclipse Magnets, Sheffield, England) prior to bagging for commercial sale. We used this sieved compost for our preincubation analyses (Table 1) and our incubation experiment. For the thermophilic phase (60°C) of composting at Sanergy, HSW is layered with carbonaceous materials such as rice hulls or sugar cane residue within rectangular wooden boxes. During the curing stage, box contents are piled into windrows and regularly turned. Both composting stages are carried out on bare earth. Composted, torrefied, and pyrolyzed HSW were crushed and sieved to 150 to 850 μm.

Synthetic urine was used instead of human urine because of transportation constraints from Nairobi to the United States. Also, 6 months passed from the time of waste collection and pyrolysis in Nairobi to the commencement of the incubation trial at Cornell, during which time collected urine would have required stabilization with acid and cold storage. Instead, we prepared a synthetic urine solution with urea (Shmaefsky, 1990), which readily hydrolyzes to NH_4 after contact with soil (Estiu and Merz, 2006). In addition to

N, synthetic urine contained P, as well as potassium and sodium. Solutions were acidified to pH 6 prior to the start of the experiment.

Amendments with HSW were applied to soils at a weight ratio of 0.5% wt wt^{-1}, based on previous incubation experiments with pyrolyzed wood (Whitman et al., 2014). This translates into 6.9 to 12 ton ha^{-1} according to tapped soil bulk densities (ASTM D7481-09). These application rates are approximately 4 times greater than the farmyard manure (~68% water content) application recommended by the Kenyan Ministry of Agriculture, 2.5 – 7.5 wet tons ha^{-1} (National Farm Information Service of Kenya, 2018). The ratio of soil to urine was 1:0.033 kg mL^{-1}, based on daily collection rates of urine and HSW in latrines. The daily collection rate of sun-dried unpyrolyzed HSW to urine in Sanergy latrines during April to July 2014 was 1:1.93 kg L^{-1} (Fausto Marcigot, pers. comm.; Sanergy, Nairobi, Kenya). This translates to an equivalent ratio of 1:6.4 kg L^{-1} pyrolyzed HSW (500°C) to urine, based on an average mass yield of 29.24% ± 8.67% (wt wt^{-1}) after pyrolysis at 500°C. Together, soil and HSW amendment mixtures contained 15 g soil + 0.075 g amendment, whereas urine-treated mixtures contained an additional 0.49 mL urine, which is equivalent to 302 mg N kg soil^{-1}.

Soil Properties

To determine the effectiveness of HSW amendments in mitigating soil fertility constraints common in the tropics, such as soil acidity and low CEC (Jeffery et al., 2017), four soils characterized by a

TABLE 1. Physical and Chemical Characteristics of Composted, Torrefied (200°C), and Pyrolyzed (300°C and 500°C) HSW, With and Without Urine							
Analysis	Units	200°C	300°C	500°C	Compost (60°C)	500°C + urine	Urine
pH	Log (mg L^{-1})	6.9 ± 0.0	8.8 ± 0.0	10.7 ± 0.0	6.4 ± 0.1	8.0 ± 0.0	6.0 ± 0.0
EC	dS m^{-1}	2.6 ± 0.0	1.5 ± 0.1	1.7 ± 0.02	0.3 ± 0.0		
CEC	mol$_c$ kg^{-1}	0.17 ± 0.01	0.32 ± 0.02	0.22 ± 0.02	0.40 ± 0.01		
SSA	m^2 g^{-1}	28.2 ± 2.4	53.9 ± 2.9	132.6 ± 3.6	28.6 ± 1.9		
CEC/SSA	mmol$_c$ m^{-2}	6.0 ± 7.1	6.0 ± 7.9	1.6 ± 6.1	14.0 ± 5.6		
N	% (wt wt^{-1}) [g L^{-1}]	3.1 ± 0.6	4.1 ± 0.3	3.0 ± 0.6	1.5 ± 0.1	3.1 ± 0.6	[9.3 ± 0.0]
C$_{org}$	% (wt wt^{-1})	48.6 ± 1.5	59.9 ± 0.7	65.0 ± 1.5	16.4 ± 0.5	53.5 ± 1.2	
C$_{inorg}$	% (wt wt^{-1}) [g L^{-1}]	0.11 ± 0.0	0.05 ± 0.0	0.28 ± 0.0	0.21 ± 0.0	0.31 ± 0.0	[4.5 ± 0.0]
H	% (wt wt^{-1})	48.7 ± 1.5	58.2 ± 2.0	63.3 ± 4.6	16.6 ± 0.5		
O	% (wt wt^{-1})	34.3 ± 1.9	21.3 ± 2.6	14.2 ± 0.3	23.0 ± 0.0		
C$_{org}$/N	mol mol^{-1}	18.2 ± 0.1	17.2 ± 0.1	25.3 ± 0.1	12.7 ± 0.1	7.5 ± 0.1	
H/C$_{org}$	mol mol^{-1}	1.5 ± 0.1	1.0 ± 0.0	0.4 ± 0.1	1.7 ± 0.1		
CaCO$_3$ eq.	% (wt wt^{-1})	0.5 ± 0.7	2.0 ± 0.2	7.3 ± 0.2	4.0 ± 1.3		
Ash	% (wt wt^{-1})	8.1 ± 0.3	15.5 ± 0.1	20.8 ± 0.1	63.7 ± 1.1		
Volatile matter	% (wt wt^{-1})	75.5 ± 0.3	48.9 ± 0.1	18.5 ± 0.1	28.3 ± 0.0		
DOC	g kg^{-1} [mg L^{-1}]	26.4 ± 1.0	5.3 ± 0.2	0.18 ± 0.01	2.3 ± 0.1	26.2 ± 0.9	[874.4 ± 38.4]
NH$_4$-N [urea-N]	mg kg^{-1} [mg L^{-1}]	776.6 ± 4.5	25.0 ± 1.9	3.4 ± 4.8	244.3 ± 1.2	1,990.0 ± 5.0	[302.3 ± 0.0]
NO$_3$-N	mg kg^{-1} [mg L^{-1}]	2.9 ± 0.4	1.8 ± 0.6	1.1 ± 0.2	184.7 ± 0.9	0.0	[0.0]
P (DIH$_2$O)	g kg^{-1}	5.9 ± 0.3	3.3 ± 0.01	2.2 ± 0.03	0.6 ± 0.0		
P (Mehlich-III)	g kg^{-1}	7.7 ± 0.5	6.7 ± 0.3	8.1 ± 0.4	1.4 ± 0.1		
P (HClO$_4$ + HNO$_3$)	g kg^{-1} [mg L^{-1}]	11.2 ± 1.3	20.1 ± 0.4	24.7 ± 0.5	7.8 ± 1.5	54.1 ± 0.5	[4.5 ± 0.0]

Values are means (n = 4) ± S.D.
EC, electric conductivity.

TABLE 2. Location and Classification of Soils Collected From Three Different Watersheds in Western Kenya

Soil Description	Clay Loam pH 4.6	Loamy Fine Sand pH 6.1	Clay Loam pH 7.4	Sandy Loam pH 6.5
Watershed	Mid Yala	Lower Yala	Lower Nyando 1	Lower Nyando 2
GPS coordinates	702,156.85 mE, 17,930.34 mN	357,389.22 mE, 958.79 mN	729,603.12 mE, 9,963,733.53 mS	725,201 mE, 9,969,083.00 mS
Soil moisture/temp regime	Udic/thermic	Udic/isohypothermic	Ustic/thermic	Ustic/thermic
Food and Agriculture Organization classification (Kenya Soil Survey, 2013)	Haplic Acrisol	Haplic Acrisol	Luvic Phazeozems	Eutric Cambisol
USDA classification (USDA 1998)	Typic Rhodudult	Psammentic Hapludults	Typic Argiudolls	Aquertic Eutrudept

range in pH and texture were collected from three maize-growing watersheds in Western Kenya: an isohyperthermic Psammentic Hapludult from Lower Yala, a thermic Typic Rhodudult from Mid Yala, and two soils, a thermic Typic Argiudoll and a thermic Aquertic Eutrudept from Nyando (Table 2). Sampling sites were chosen from a georeferenced soils database created at the World Agroforestry Center (ICRAF) during the Western Kenya Integrated Ecosystem Management Project implementation and are representative of the region

(Boye et al., 2008). U.S. Department of Agriculture (USDA) soil taxonomic classes (USDA, 1998) were determined by first matching GPS points of sampled locations to the ISCRIC Soil and Terrain Database for Kenya using Food and Agriculture Organization soil classification (Kenya Soil Survey, 2013), and then converting these into USDA classes using the wall chart *Soils of the World* (Lof et al., 1987).

Yala soils have an acidic pH, whereas the Nyando soils have neutral pH (Table 3). One soil from each location was characterized as a

TABLE 3. Physical and Chemical Characteristics of Soils Used for the Incubation Experiment

Analysis	Units	Clay Loam pH 4.6	Loamy Fine Sand pH 6.1	Clay Loam pH 7.4	Sandy Loam pH 6.5
Sand	% (wt wt^{-1})	22.2 ± 0.9	77.6 ± 1.2	20.8 ± 0.2	72.8 ± 1.5
Silt	% (wt wt^{-1})	37.4 ± 1.0	13.7 ± 0.6	38.6 ± 1.4	17.3 ± 1.0
Clay	% (wt wt^{-1})	40.4 ± 1.3	8.7 ± 1.0	40.6 ± 1.4	9.9 ± 1.5
Texture		Clay loam	Loamy fine sand	Clay loam	Sandy loam
Bulk density	g cm^{-3}	1.1 ± 0.0	1.4 ± 0.0	0.92 ± 0.0	1.6 ± 0.01
pH	Log [mg L^{-1}]	4.6 ± 0.1	6.1 ± 0.0	7.4 ± 0.1	6.5 ± 0.2
EC	dS m^{-1}	69.8 ± 0.4	51.6 ± 0.7	86.6 ± 1.3	17.4 ± 0.3
CEC	mmol$_c$ kg^{-1}	79.7 ± 7.5	41.4 ± 8.0	179.2 ± 10.7	62.1 ± 12.0
SSA^c	m^2 g^{-1}	34.4 ± 12.6	9.1 ± 5.3	154.4 ± 63.7	81.1 ± 35.0
CEC/SSA	mmol$_c$ m^{-2}	2.3 ± 0.6	4.5 ± 1.5	11.6 ± 0.2	0.7 ± 0.3
N	% (wt wt^{-1})	0.15 ± 0.00	0.06 ± 0.32	0.08 ± 0.00	0.06 ± 0.00
C$_{org}$	% (wt wt^{-1})	1.5 ± 0.0	0.4 ± 0.0	0.8 ± 0.1	0.6 ± 0.1
C$_{inorg}$	% (wt wt^{-1})	0.06 ± 0.1	0.04 ± 0.0	0.11 ± 0.0	0.05 ± 0.0
H	% (wt wt^{-1})	0.83 ± 0.00	0.12 ± 0.00	0.77 ± 0.00	0.64 ± 0.00
O	% (wt wt^{-1})	8.1 ± 0.0	1.2 ± 0.0	7.3 ± 0.0	6.7 ± 0.0
C$_{org}$/N	mol mol^{-1}	11.6 ± 0.0	10.0 ± 0.0	10.9 ± 0.0	11.5 ± 0.0
DOC	mg kg^{-1}	40.1 ± 3.0	33.6 ± 8.1	25.1 ± 4.9	55.5 ± 20.6
NH$_4$-N	mg kg^{-1}	8.6 ± 0.4	7.2 ± 1.6	6.8 ± 0.3	9.5 ± 0.7
NO$_3$-N	mg kg^{-1}	31.1 ± 0.2	19.1 ± 0.1	11.0 ± 2.0	4.7 ± 2.9
P (Mehlich-III)	mg kg^{-1}	6.08 ± 0.60	3.33 ± 0.47	2.90 ± 0.26	0.49 ± 0.20
P (HClO$_4$ + HNO$_3$)	mg kg^{-1}	376.1 ± 99.1	276.1 ± 105.8	186.1 ± 10.6	359.1 ± 27.6
Al (NH$_4$Ox)	g kg^{-1}	2.0 ± 0.3	0.75 ± 0.1	1.2 ± 0.01	1.6 ± 0.2
Fe (NH$_4$Ox)	g kg^{-1}	3.0 ± 0.4	1.6 ± 0.1	2.5 ± 0.2	6.8 ± 0.8
Mn (NH$_4$Ox)	g kg^{-1}	0.43 ± 0.06	0.81 ± 0.04	2.40 ± 0.08	10.48 ± 1.13

Values are means (n = 3) ± S.D.
EC, electric conductivity.

finer textured and one as a coarser textured. Soils were taken from the top 0.15 m in a 10-m^2 area. Following sampling, soils were air dried for 1 week and then sieved to a size less than 2 mm. The Inceptisol taken from Lower Nyando was sieved to greater than 52-μm particle size to artificially create a coarser-textured soil.

Analyses of Soils and Amendments Before Incubation

A suite of laboratory analyses on soils and amendments was carried out prior to incubation. For all analyses requiring filtration, we used 5-μm paper filters (smooth fluted 313-folded; VWR, Radnor, PA). All chemicals were reagent grade, purchased from Sigma-Aldrich (St Louis, MO), Fisher Scientific (Fair Lawn, NJ), or VWR (Cambridge, MA). The pH (Symphony SP20 pH meter; Thermo Orion, Beverly, MA) and electrical conductivity (Model 115 A+; Orion Research, Jacksonville, FL) of HSW amendments were measured in 1:10 g mL^{-1} deionized water (DIH$_2$O) suspensions (NANOpure Diamond, Barnstead; Thermo Fisher Scientific). A ratio of 1:5 g mL^{-1} was used to measure the pH and EC of soils. Total C, N, hydrogen, and oxygen were measured by combustion on an elemental analyzer (Flash 1112; CE Elantech, Lakewood, NJ). We determined C$_{inorg}$ with a Bernard Calcimeter (Lamas et al., 2005), which measures the volume of carbon dioxide (CO$_2$) emitted from a sample treated with 4 M hydrogen chloride (HCl).

The specific surface area (SSA) was determined from gas adsorption isotherms between relative pressures (p/p^0) of 0.05 and 0.3 (Model-ASAP 2020; Micromeritics Inc., Norcross, GA), using the Brunauer Emmett Teller equation for multilayer adsorption. Dinitrogen (N$_2$) was used to measure the surface area of soils, and CO$_2$ was used for amendments with larger volumes of mesopores and micropores. To determine CEC, both soils and amendments were double extracted with 2 M KCl after saturation with 1 M ammonium acetate (NH$_4$OAc, pH 7) at a ratio of 1:50 g mL^{-1}. The NH$_4$ concentration was determined colorimetrically on an autoflow analyzer (AA3 HR AutoAnalyzer; Seal Analytical, Mequon, WI).

Amendments and soils were digested with perchloric and nitric acid (HClO$_4$ + HNO$_3$) at a ratio of 0.5:40 g mL^{-1} in an automated workstation (Vulcan, 84; Questron Technologies, Mississauga, Ontario, Canada), after which the resuspended digest was analyzed for total elements via photoemission inductively coupled plasma atomic emission spectroscopy (ICP-AES; Spectro Arcos, Ametek Materials Analysis, Kleve, Germany). Plant-available nutrients in amendments and soils were determined via ICP-AES following a Mehlich-III extraction at a ratio of 1:10 g mL^{-1} (Mehlich, 1984). Total elemental analysis from acid-digested and Mehlich-III extracted amendments is presented in the Supplementary Material (Supplementary Tables 2 and 3, Supplemental Digital Content 1, http://links.lww.com/SS/A53).Water-extractable P in amendments was also determined in DIH$_2$O extracts at a ratio of 1:10 g mL^{-1}. To determine DOC, soils and amendments were extracted with DIH$_2$O at a ratio of 1:40 g mL^{-1}. Extracts were then acidified with 1 M HCl and analyzed by combustion (Shimadzu TOC 5000, Kyoto, Japan). Plant-available, mineral N was extracted from soils and amendments with 2 M KCl at a ratio of 1:10 g mL^{-1} and analyzed colorimetrically on an autoflow analyzer.

A rapid soil texture analysis was performed on soils (Schindelbeck and Kurtz, 2016), as well as a tapped bulk density (ASTM D7481-09, 2009). The soil concentrations of amorphous aluminum (Al), iron (Fe), and manganese (Mn) oxides, hydroxides, and noncrystalline or poorly crystalline aluminosilicates were determined via ICP-AES following dissolution in 3 M ammonium oxalate (NH$_4$Ox; pH 3) at a ratio of 1:200g mL^{-1} as per Soukup et al. (2008). Soil mineralogy was qualitatively identified using XRD (Ultima IV X-ray Diffractometer; Rigaku, Tokyo, Japan) and is described in the Supplementary Online Material (Supplementary Table 1, Supplemental Digital Content 1, http://links.lww.com/SS/A53).

The liming potential of amendments, or CaCO$_3$ equivalency, was measured as per Ahern et al. (1998; Method 19A1) using a rapid titration procedure whereby samples were acidified with HCl at a ratio of 1:10 g mL^{-1} and back-titrated with sodium hydroxide. Lastly, the ash, volatile matter, and fixed C contents of all HSW amendments were tested through the ASTM D1762-84 (2013) proximate analysis method "Chemical Analysis of Wood Charcoal."

Incubation Experiment

Two separate experiments were designed to evaluate the effect of waste-based amendments on the soil contents of available P and on N mineralization. In the first experiment, composted, torrefied (200°C), and pyrolyzed HSW (300°C, 500°C) were each incubated in four soils. In the second experiment, pyrolyzed HSW (500°C) was added to four soils with and without urine. The unamended soil was included in both experiments as a control treatment.

Soil and amendment mixtures were incubated in darkness at 30°C. Samples were destructively sampled for C and N mineralization four times: day 0 (immediately after mixing soils and amendments), day 7, day 30, and day 100. Experimental units consisted of one 0.5 L mason jar (Newell Rubbermaid, Atlanta, GA), which housed a smaller jar containing the soil and amendment mixture (0.125 L straight-sided glass jars; VWR), as well as a 15-mL vial of potassium hydroxide (KOH) serving as a CO$_2$ trap (scintillation vial; Wheaton Glass, Millville, NJ). We interpolated the concentration of trapped CO$_2$ in our KOH vials using a calibration curve of the EC versus a known concentration of injected CO$_2$ (Ultrapure grade CO$_2$; Airgas, Radnor, PA) (Woo et al., 2016). To prevent CO$_2$ saturation of KOH vials, we increased the molarity for successive sampling times; 0.09 M KOH was used for day 7, 0.18 M for day 30, and 0.5 M for day 100.

Soil and amendments were mixed dry and homogenized by shaking. We then pipetted CO$_2$-free, DIH$_2$O into soil jars to 55% of the soil porosity. The porosity of soils was calculated using the gravimetric, saturated water content, and the tapped bulk density. An additional 5 mL of DIH$_2$O was added to the mason jars to prevent sample desiccation. A fifth set of jars containing a larger quantity of soil and amendment (50 g soil, 0.25 g char, and 1.63 mL urine) was installed as a replicate for day 100 to guarantee sufficient soil for Mehlich-III, pH, and EC analyses at the end of the incubation period.

Sample Analyses After Incubation

After incubation mineral N was extracted from the entire sample with 2 M KCl, at a ratio of 1:2 g mL^{-1} (Wang et al., 2012b), and analyzed colorimetrically on an autoflow analyzer. It is important to note that urea in synthetic urine treatments must undergo hydrolysis to NH$_4$ to be detected colorimetrically. While urease, the enzyme that hydrolyzes urea, is assumed to be ubiquitous in soils, we did not detect NH$_4$ in urine-treated samples, which were immediately extracted on day 0. At days 0 and 100 of incubation, the pH and EC of soil and amendment mixtures were measured in suspension with DIH$_2$O at a ratio of 1:5 g mL^{-1}, whereas soils were extracted with Mehlich-III at a ratio of 1:10 g mL^{-1}, and the supernatant measured for total elements using ICP-AES.

Statistical Analyses

Exploratory data analysis, hypothesis testing, means pairwise comparisons, linear regressions, and analysis of variance (ANOVA) were carried out using RStudio (RStudio Team 2015). Data analysis and graphics were carried out with the *ggplot2* package (Wickham, 2009). Least squares of treatment means (LS means) were calculated using the *lsmeans* package (Lenth, 2016) through the *lm* command. Order-independent P values determined with the Student *t* test were adjusted using Tukey method for comparing a family of

five estimates at the α = 0.05 threshold. The *cld* command from the package *multcompView* (Graves et al., 2015) was used to generate a compact letter display of all pairwise comparisons between amendments for each soil type for a significance level of $P < 0.05$ and was embedded within graphical figures. Type I ANOVA using an order-dependent F test was also carried out within the *lsmeans* package using the *anova* command. All mention of "significant differences" refers to probability of observing an F ratio greater than 0.05 given the null hypothesis, $\text{Pr}(>F)$, or $P < 0.05$.

We also conducted regressions of four response variables: (1) C mineralization (CO_2-C) at day 100, (2) mineral N ($NH_4 + NO_3$) at day 100, (3) available P (P) at day 100, and (4) the change in available P between days 0 and 100 (ΔP), against four continuous predictor variables: (1) soil pH, (2) the soil CEC per unit surface area (CEC/SSA), (3) NH_4Ox-extractable oxides, and (4) amendment DOC. For regressions, we evaluated only torrefied HSW (200°C), pyrolyzed HSW (300°C, 500°C), and HSW compost.

We initially tested all amendment characteristics listed in Table 1 pertaining to pH, surface reactivity, and the initial available C, N, and P content; we tested all soil characteristics listed in Table 3 related to soil pH, texture, and mineralogy. Variables that demonstrated strong co-correlation, indicated by variance inflation factors greater than 4, were eliminated, as well as variables that did not demonstrate a significant effect on any two of the four response variables, CO_2-C, $NH_4 + NO_3$, P, and ΔP. Variance inflation factors were calculated using the *vif* command in the *car* package (Fox and Weisberg, 2011).

All variables, predictor and response, were standardized prior to model evaluation using the *scale* command in RStudio to allow comparisons between predictors having different units of measurement (Marks et al., 2014). Coefficients (β) were unstandardized in the presentation of results by multiplying coefficients by the S.D. of response variables (β′*S.D.$_y$). This term, β′*S.D.$_{-y}$, represents the change in response variable y, in units shown, per change in 1 S.D. of each predictor variable.

RESULTS

Torrefied, Pyrolyzed, and Composted HSW Amendments

Plant-available P and pH

After 100 days, the concentration of plant-available P in soils was significantly greater with HSW amendments compared with unamended soils and increased with amendment HHT (Fig. 1A). Moreover, the availability of N and P following incubation of HSW amendments mimicked the plant-availability proxy determined through chemical extractions (Table 1). Pyrolyzed HSW (500°C) contained 3.2-fold more available P compared with composted HSW and contributed between 1.9- and 2.6-fold available P after 100 days' incubation in soils (Supplementary Table 4, Supplemental Digital Content 1, http://links.lww.com/SS/A53). All soils amended with pyrolyzed HSW (500°C), except the acidic clay loam, had significantly higher available P after 100 days than with other amendments. Available P in all soils with compost was not significantly different than with torrefied HSW, despite twice as much total P and fivefold more available P in the latter amendment. Furthermore, 69% of total P in torrefied HSW was plant available compared with only 18% availability in compost (Table 1).

A positive correlation was observed between ΔP of soils and the HHT of thermochemically treated HSW, except for the sandy loam soil. In this soil, available P at the start of the incubation became immobilized (ΔP < 0) with additions of torrefied and pyrolyzed HSW after 100 days, but neither with composted HSW nor with unamended soil (Fig. 1B). Similarly, additions of torrefied HSW caused P immobilization in the clay loam (pH 7.4) after 100 days (ΔP = −12.2 mg kg^{-1}), whereas P slightly increased in unamended soil to ΔP = 4.9 mg kg^{-1} (Supplementary Table 4, Supplemental Digital Content 1, http://links.lww.com/SS/A53).

Phosphorus availability after 100 days and the change in P availability (ΔP) were both significantly affected by amendment type and soil type (Supplementary Table 5, Supplemental Digital Content 1, http://links.lww.com/SS/A53). We identified negative
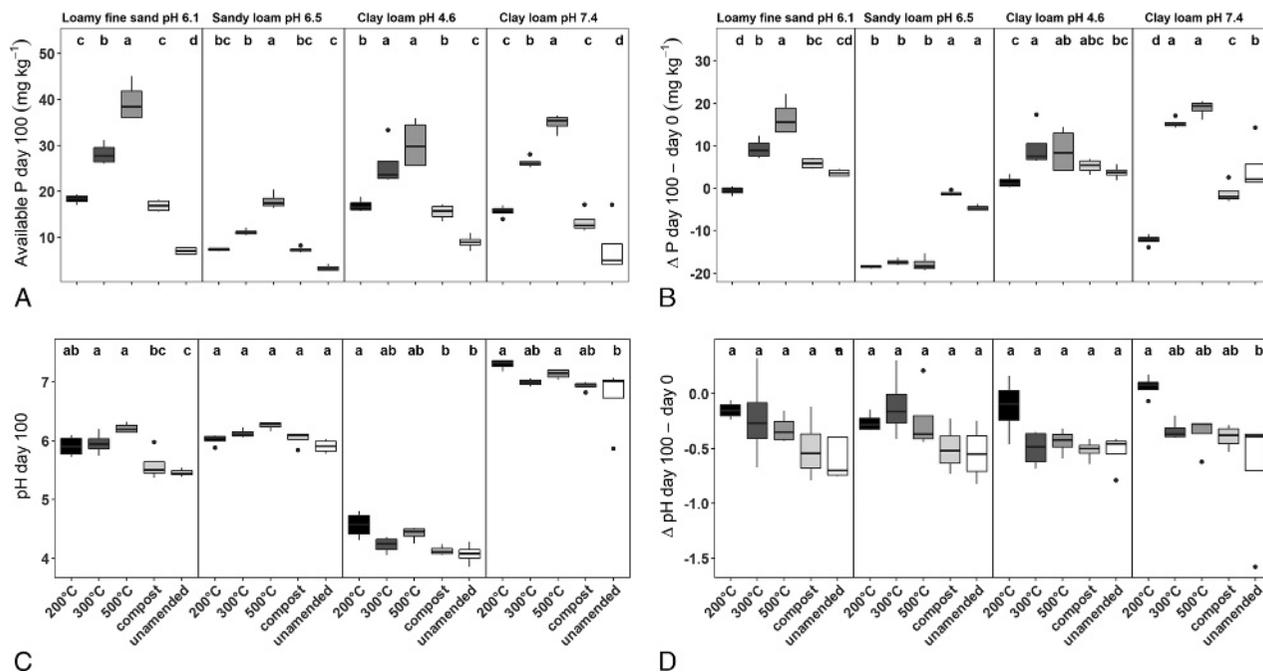
FIGURE 1. Box plots showing (A) plant-available P at day 100, (B) the change in plant-available P (△P) between day 100 and day 0, (C) pH at day 100, (D) the change in pH (△pH) between day 0 and day 100 after additions of composted HSW, HSW pyrolyzed at 200°C, 300°C, and 500°C to four different soils. Letters indicate significant differences between amendments within soils ($P < 0.05$; n = 4 ± S.D.). Results for LS means and ANOVA are shown in Supplementary Tables 4 and 5 (Supplemental Digital Content 1, http://links.lww.com/SS/A53).

TABLE 4. Variance Inflation Factors (VIF), Unstandardized Regression Coefficients (y = β$_i$*S.D.) ± S.D., and *P* Values From Regressions of Four Response Variables: C Mineralization (CO$_2$-C), Mineral N (NH$_4$ + NO$_3$), and Available P (P) at Day 100, and the Change in Available P Between Day 100 and Day 0 (ΔP) to Four Soil and Amendment Properties: Soil pH, Soil CEC/SSA, Ammonium-Oxalate Extractable Oxides (Al + Fe + Mn) in Soil, and DOC in Amendments

Order	Predictor	VIF	CO$_2$-C (g kg C^{-1})	*P*	NH$_4$ + NO$_3$-N (g kg N^{-1})	*P*	P Day 100 (mg kg^{-1})	*P*	ΔP Day 100 to Day 0 (mg kg^{-1})	*P*
1	Soil pH	1.14	16.04 ± 6.2	<0.01	−12.20 ± 1.5	0.00	0.54 ± 1.1	0.40	−0.29 ± 1.2	0.03
2	Soil CEC/SSA (mmol$_c$ m^{-2})	2.14	12.35 ± 8.5	<0.01	−12.56 ± 2.1	0.10	0.55 ± 1.5	<0.01	−1.04 ± 1.6	<0.01
3	Soil Al + Fe + Mn (mg kg^{-1})	1.97	0.13 ± 8.2	0.97	−14.43 ± 2.0	0.00	−5.35 ± 1.4	<0.01	−9.5 ± 1.5	<0.01
4	Amendment DOC (mg kg^{-1})	1.00	41.21 ± 4.0	<0.01	−5.65 ± 2.9	0.00	−3.69 ± 1.1	<0.01	−5.2 ± 1.8	<0.01

Coefficients are means of four replicate observations (n = 4).

correlations between amendment DOC (Table 1) and NH$_4$Ox-extractable oxides (Al + Fe + Mn) in soils (Table 3) with available P. Regressions revealed that an increase in 1 S.D. of amendment DOC and initial soil Al + Fe + Mn predicted a decrease in available P by 3.69 and 5.35 mg kg^{-1}, respectively (Table 4).

The pH change in soils induced by incubation with HSW amendments partially mimicked the pH of amendments. In the loamy fine sand, pH was significantly higher with the most alkaline amendment,

500°C HSW (pH 10.7), and lowest with composted HSW (pH 6.4), the most acidic amendment (Fig. 1C, Table 1). However, the pH of clay loams was highest with torrefied HSW rather than pyrolyzed HSW. In the acidic clay loam (pH 4.6), additions of torrefied HSW resulted in significantly higher pH compared with additions of compost, despite the similar pH between amendments (Table 1).

After the incubation, the pH decreased, judging by the negative ΔpH values (Fig. 1D). Furthermore, no significant differences are
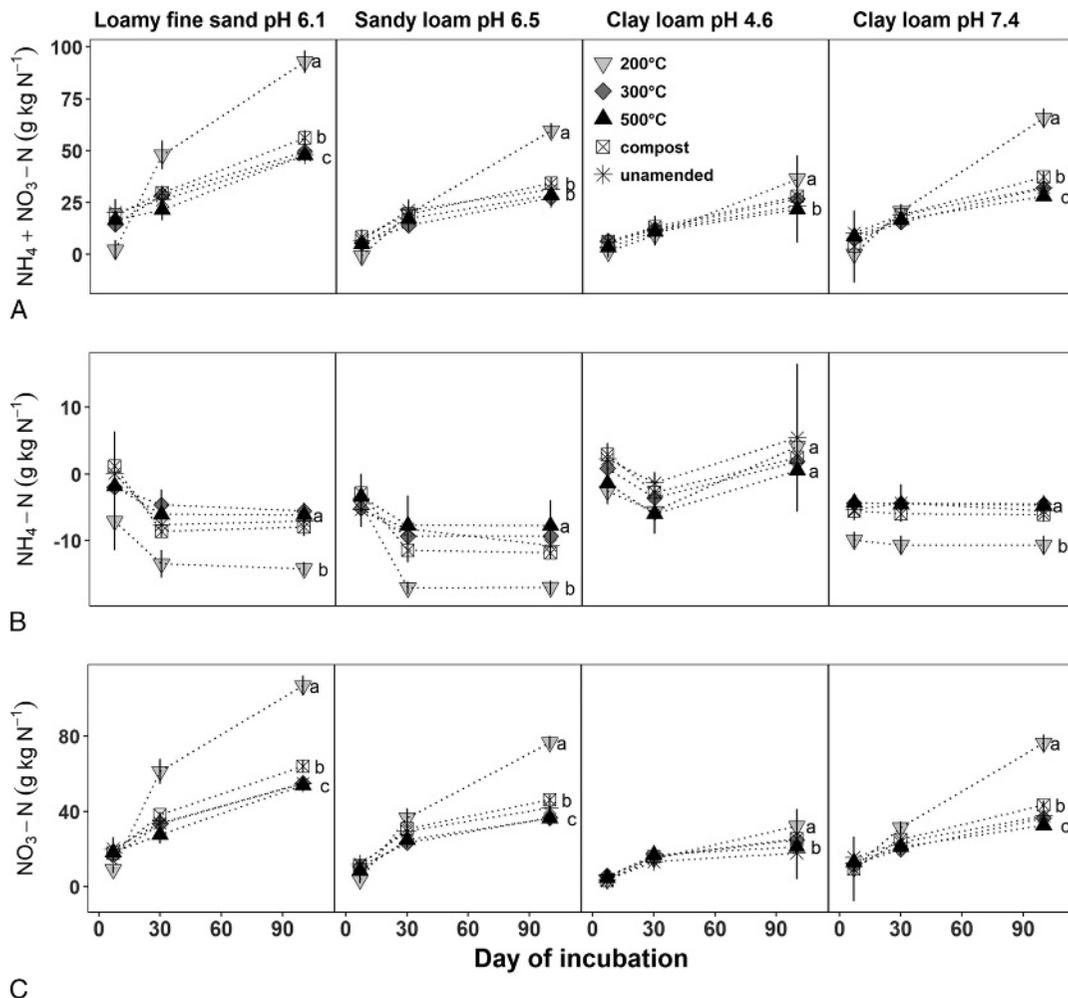
FIGURE 2. Proportion of KCl-extractable N at three time points, measured as (A) ammonium (NH$_4$-N), (B) NO$_3$ (NO$_3$-N), (C) total mineral N (NH$_4$ + NO$_3$-N) relative to the total N of four different soils amended with composted HSW, HSW pyrolyzed at 200°C, 300°C, and 500°C. Letters indicate significant differences between amendments within soils at day 100 (*P* < 0.05; n = 4 ± S.D.). Results for LS means and ANOVA are shown in Supplementary Tables 6 and 7 (Supplemental Digital Content 1, http://links.lww.com/SS/A53).

evident between soils incubated with HSW amendments or unamended soils except in the clay loam (pH 7.4), in which the ΔpH was significantly higher with torrefied HSW compared with the unamended soil.

Nitrogen mineralization

Mineral N after 100 days was greatest with torrefied HSW additions, decreasing significantly with increasing HHT (Fig. 2A). In all soils except the clay loam (pH 4.6), torrefied HSW amendments increased N mineralization above pyrolyzed HSW (500°C) by two-fold. After 100 days, 9.3% of the initial total N in the loamy fine sand amended with torrefied HSW was available, whereas only half as much, 4.8%, was recovered with HSW (500°C) additions (Supplementary Table 6, Supplemental Digital Content 1, http://links.lww.com/SS/A53). A significant increase in N mineralization was apparent in soils with torrefied HSW between days 30 and 100, relative to pyrolyzed HSW and compost. Nevertheless, while significantly lower than torrefied HSW, the N fertilization potential of HSW compost was significantly greater than pyrolyzed HSW.

In three of the four studied soils, the direction of change from day 7 to day 3 was opposite for ammonification and nitrification, possibly due to conversion of NH_4 to NO_3 (Fig. 2B,C). The only exception was the clay loam (pH 7.4), in which increasing NO_3 was observed alongside unchanged NH_4 at days 7 and 30. In all soils, NO_3 comprised the dominant species of mineral N at all time points. Ammonium production was similar at days 30 and 100 in all soils except the acidic clay loam (Fig. 2B), whereas NO_3 increased in all soils (Fig. 2C). Nitrification was impeded in the

clay loam (pH 4.6), judging by NH_4 accumulation between days 30 and 100, and the significantly lower NO_3 compared with the other three soils.

Both amendment and soil properties significantly affected N mineralization at all time points (Supplementary Table 7, Supplemental Digital Content 1, http://links.lww.com/SS/A53). Three of four predictor variables (initial amendment DOC, soil pH, soil Al + Fe + Mn) were significantly negatively correlated ($P < 0.05$) to soil mineral N concentrations after 100 days. Initial soil Al + Fe + Mn had the largest effect on soil mineral N, causing a decrease in 12.6 g kg^{-1} mineral N with 1 S.D. increase in soil Al + Fe + Mn (Table 4).

Relationships between N and C mineralization

We evaluated the relationship between soil N mineralization ($NH_4 + NO_3$-N; N_{min}) to C mineralization (CO_2-C; C_{min}) at day 100 with HSW amendments (Fig. 3A). Additions of torrefied HSW resulted in both greater C_{min} and greater N_{min} after 100 days than other amendments (Fig. 3A). Nevertheless, in all soils except the acidic clay loam (pH 4.6), the calculated ratio for N_{min}/C_{min} was significantly lower for torrefied HSW, 0.38 to 0.53, compared with 500°C pyrolyzed HSW, 0.61 to 0.92 (Fig. 3B; Supplementary Table 8, Supplemental Digital Content 1, http://links.lww.com/SS/A53). The N_{min}/C_{min} ratio in the loamy fine sand and sandy loam was also significantly lower for torrefied HSW compared with composted HSW or no amendment, indicating a higher "carbon cost" for N mineralization with torrefied HSW compared with other amendments. Carbon mineralization for all soils and amendments is
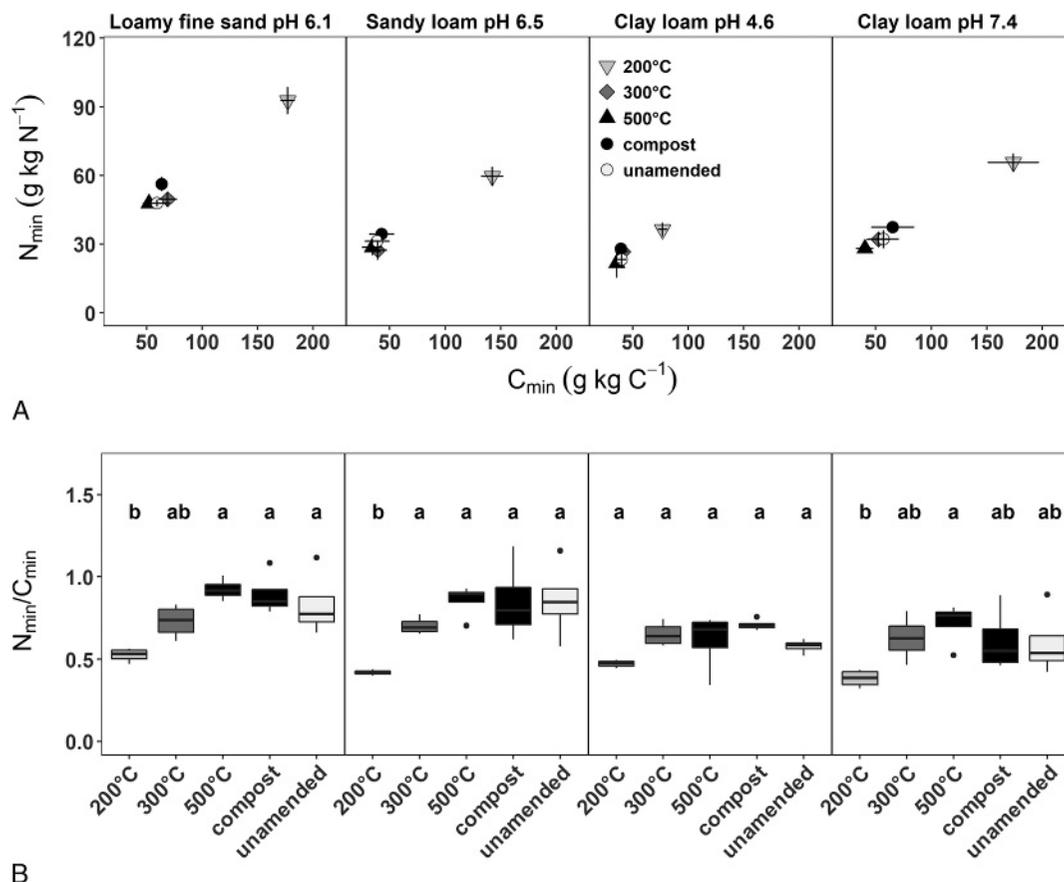
FIGURE 3. (A) Total mineral N ($NH_4 + NO_3$-N; N_{min}) versus mineralized C (CO_2-C; C_{min}) after 100 days and (B) the ratio of N_{min} to C_{min} (N_{min}/C_{min} calculated from data points in Fig. 3A) of soils amended with composted HSW, HSW pyrolyzed at 200°C, 300°C, and 500°C in comparison to an unamended control. Letters denote significant differences between amendments within soils types ($P < 0.05$; n = 4 ± S.D.). Results for LS means and ANOVA are shown in Supplementary Tables 8 and 9 (Supplemental Digital Content 1, http://links.lww.com/SS/A53).
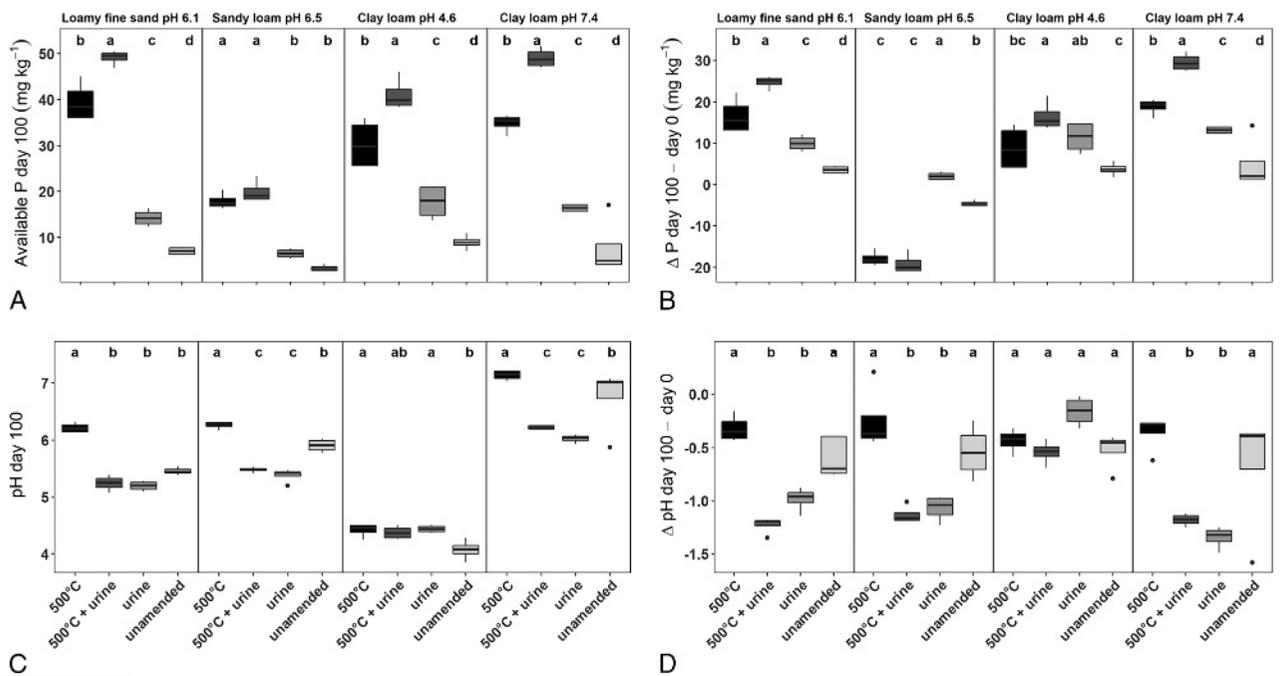
FIGURE 4. Box plots showing (A) plant-available P at day 100, (B) the change in plant-available P ($\triangle$P) between day 0 and day 100, (C) pH at day 100, (D) the change in pH ($\triangle$pH) between day 100 and day 0 of four different soils amended with urine, HSW pyrolyzed at 500°C, and their combination in comparison to an unamended control. Letters above boxes indicate significant differences between amendments within soils ($P < 0.05$; n = 4 ± S.D.). Results for LS means and ANOVA are shown in Supplementary Tables 10 and 11 (Supplemental Digital Content 1, http://links.lww.com/SS/A53).

presented in Supplementary Index (Supplementary Figure 1, Supplemental Digital Content 1, http://links.lww.com/SS/A53).

Urine Amendments

Plant-available P and pH

Available P after 100 days was significantly higher with pyrolyzed HSW (500°C) + urine compared with urine alone or unamended soil (Fig. 4A). The combination of pyrolyzed HSW (500°C) + urine also generated significantly greater P than pyrolyzed HSW (500°C) alone in all soils but the sandy loam. Across these soils, the combination of pyrolyzed HSW (500°C) + urine generated 41.1 to 49.1 mg P kg^{-1} after 100 days, whereas additions of pyrolyzed HSW (500°C) alone generated 30.3 to 39.5 mg P kg^{-1} (Supplementary Table 10, Supplemental Digital Content 1, http://links.lww.com/SS/A53). The difference in extractable P between days 0 and 100 of the incubation (ΔP) was also significantly higher with pyrolyzed HSW (500°C) + urine in the loamy fine sand and clay loam (pH 7.4) compared with other amendments. Additions of pyrolyzed HSW (500°C), with and without urine, in the sandy loam reduced P (ΔP less than −17 mg kg^{-1}; Fig. 4B; Supplementary Table 10, Supplemental Digital Content 1, http://links.lww.com/SS/A53).

In the two Nyando soils, sandy loam and clay loam (pH 7.4), urine additions with or without pyrolyzed HSW (500°C) significantly decreased pH at day 100 compared with the unamended soil (Fig. 4C). Thus, the liming equivalency of pyrolyzed HSW (500°C), 7.3% (Table 1), did not mitigate acidification induced by urine additions. Natural acidification occurred in all unamended soils between days 0 and 100 of incubation, evidenced by negative ΔpH values, except in the already acidic clay loam (Fig. 4D). This decrease was significantly lower with urine additions with or without pyrolyzed HSW (500°C), ΔpH of less than −1 than the soils without urine.

Nitrogen mineralization

Expectedly, after 100 days, mineral N was significantly greater with urine additions (with or without 500°C pyrolyzed HSW) in all soils

(Fig. 5A). In the clay loam (pH 7.4), significantly higher mineral N resulted from the combination of pyrolyzed HSW (500°C) + urine compared with urine alone (Fig. 5A). In all soils amended with urine except the acidic clay loam, NH$_4$ concentrations decreased over time (Fig. 5B), whereas NO$_3$ production increased (Fig. 5C). After 100 days, among urine-amended soils, NH$_4$ was most depleted in the clay loam (pH 7.4), 18.7 to 42.2 mg kg^{-1}, whereas NO$_3$ concentrations in this soil increased to 182.3 to 212.5 mg kg^{-1} (Supplementary Table 12, Supplemental Digital Content 1, http://links.lww.com/SS/A53). Only in the sandy loam did additions of pyrolyzed HSW (500°C) + urine reduce nitrification relative to urine alone. In the remaining three soils, NO$_3$ was greater with the combination of pyrolyzed HSW (500°C) + urine, compared with urine alone (Fig. 5C).

DISCUSSION

Biological Versus Thermochemical Sanitization of HSW and Soil N and P Availability

Composted HSW from our trial proved to be a poor source of plant-available N and P compared with all other amendments, likely due to the high ash content of HSW compost. The process of windrow composting on an earthen floor can raise the ash content of the compost product by fivefold at the expense of C contents relative to the feedstock (Matthiessen et al., 2005). However, this effect seemed exacerbated in the studied compost, which contained 67% ash minerals and 16.6% total C (Table 1), the latter value being in the lower range of literature values for fecal compost and other manure composts: 28% to 43% C for HSW compost (Germer et al., 2010), 19% to 24% C for ruminant manure, 12% to 20% C for municipal solid waste (Hargreaves et al., 2008), and 39% to 40% C for farmyard manure (Zaller and Koepke, 2004).

Similarly low N mineralization with the pyrolyzed HSW was not caused by high ash contents of the HSW (that were only 16%–21%) but by the formation of polycyclic N forms (Wei et al., 2018) that
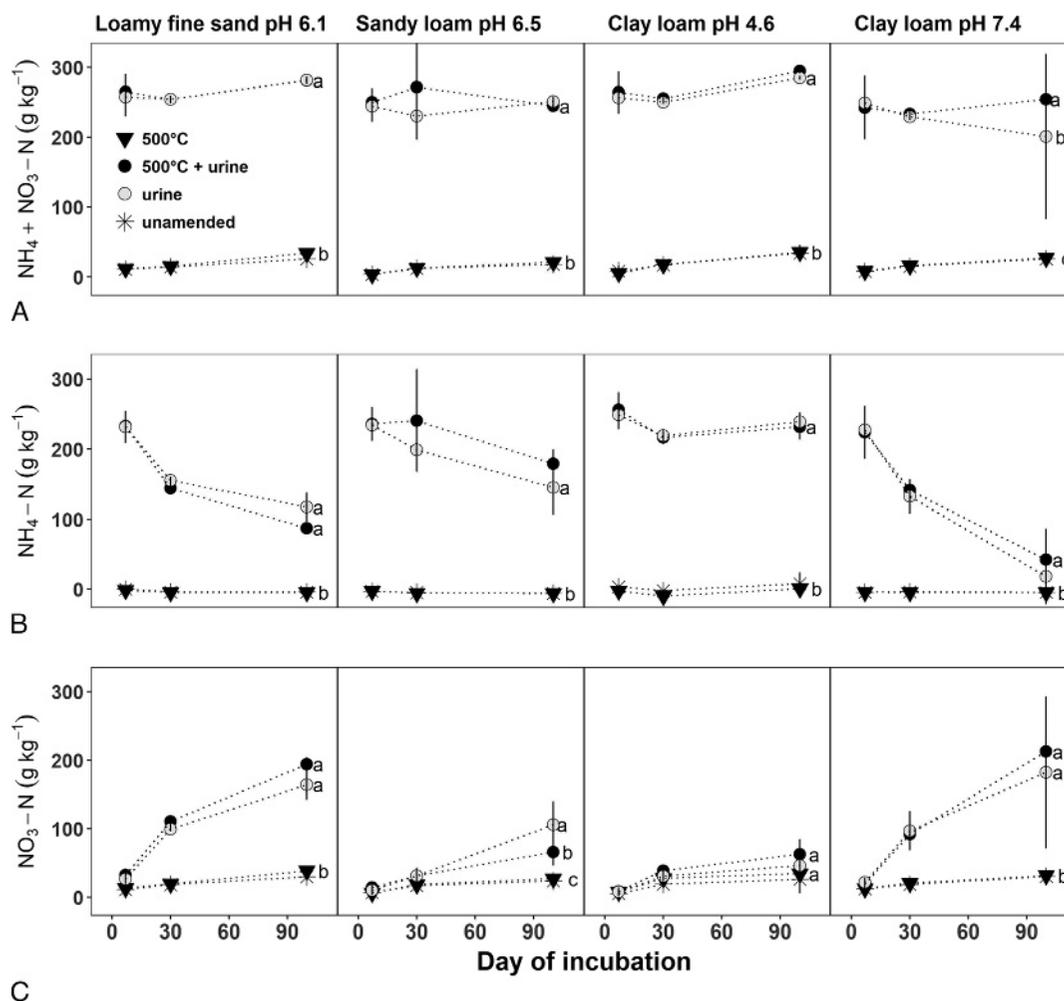
FIGURE 5. Proportion of KCl-extractable N at three time points, measured as (A) ammonium (NH_4-N), (B) NO_3 (NO_3-N), (C) total mineral N ($NH_4 + NO_3$-N) per weight of soil amended with urine, HSW pyrolyzed at 500°C, and their combination in comparison to an unamended control. Letter above boxes indicate significant differences between amendments within soils at day 100 ($P < 0.05$; n = 4 ± S.D.). Results for LS means and ANOVA are shown in Supplementary Tables 12 and 13 (Supplemental Digital Content 1, http://links.lww.com/SS/A53).

have been shown to mineralize very slowly (Wang et al., 2012b). In contrast, the high N mineralization with additions of torrefied HSW implies that torrefaction did not generate a high proportion of fused aromatic ring structures, which is confirmed by the high H/C_{org} ratio of 1.5 compared with 1.0 of HSW pyrolyzed at 300°C and 0.4 of HSW pyrolyzed at 500°C.

The lack of differences in available P between soils amended with torrefied and composted HSW cannot be explained by initial P concentrations, as torrefied HSW contained twofold more total P, and more than fivefold more available P than composted HSW (Table 1). Variable P species may partially explain low soil P availability with additions of composted HSW that was most likely enriched with the organic P species such as phytate (Uchimiya and Hiradate, 2014), which is not bioavailable. Also, inorganic orthophosphates in composted HSW may have become fixed by soil minerals such as Fe, Al, and Mn oxides, as suggested by the negative relationship between available P after incubation and poorly crystalline soil Al + Fe + Mn before incubation (Table 4). The high soil P availability after additions to soil of pyrolyzed HSW can be explained by two times higher total P concentrations in the pyrolyzed HSW than in the composted and torrefied HSW (Table 1).

Another explanation for low soil P availability with additions of composted and torrefied HSW was a lower pH of composted

HSW and torrefied HSW before incubation (Table 1). In addition, microbial P immobilization also lowered P availability specifically with torrefied HSW additions, as indicated by two observations: P availability was similar in soils amended with composted HSW and torrefied HSW, although the latter contained five times more total P, and C respiration with torrefied HSW was 2 to 3.33 times higher than with composted HSW (Supplementary Figure 2 and Supplementary Table 6, Supplemental Digital Content 1, http://links.lww.com/SS/A53).

Effect of Highest Heating Temperature During Thermochemical Processing

The trade-off between P and N availability with HHT (Figs. 6A, B) can be explained by amendment mineralizability (Table 5) as a result of their molecular properties. The decrease in P availability in the order of 500°C HSW > 300°C HSW > 200°C with an increase in N availability in the opposite direction (Fig. 6A) was first due to production of persistent organic N forms, such as heterocyclic pyridine and furan rings with greater HHT (Chen et al., 2018; Wei et al., 2018). Direct changes in P species with greater HHT may also have increased amendment P availability. At lower HHT, a greater proportion of P remains in organic form as phytate and pyrophosphate
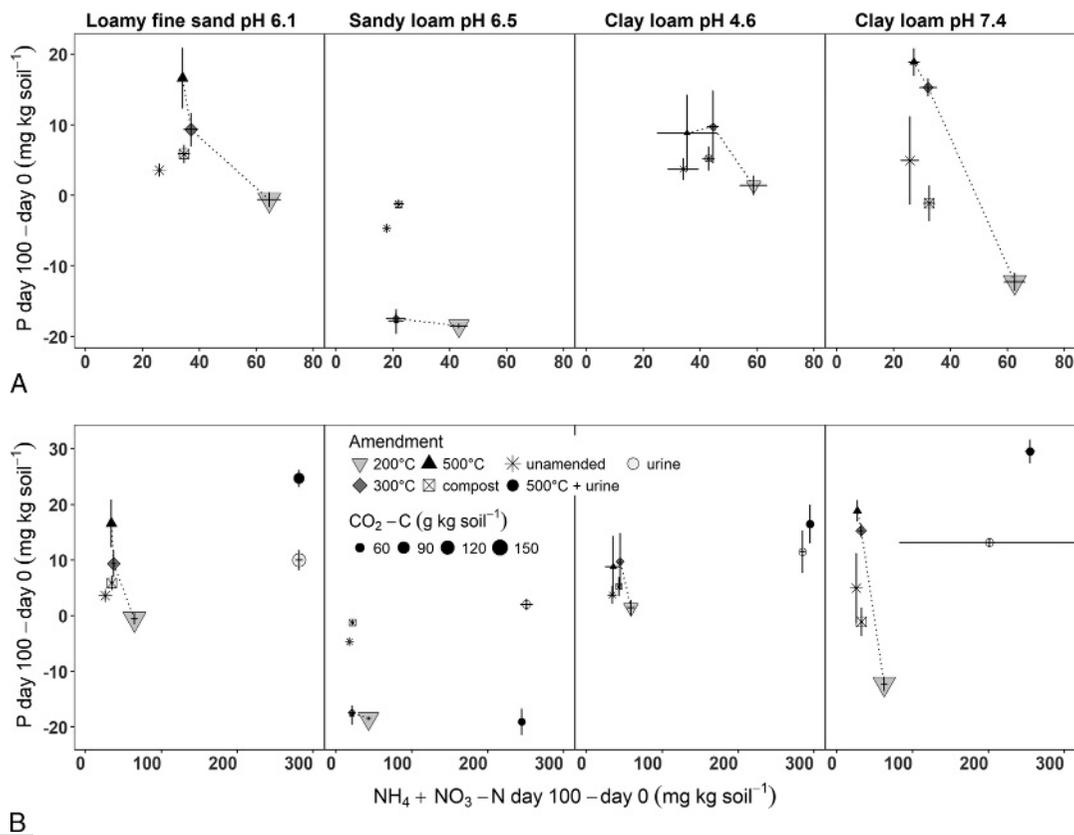
FIGURE 6. A, Available P versus available N ($NH_4 + NO_3$-N) in soils amended with torrefied HSW (200°C), pyrolyzed HSW (300°C, 500°C), composted HSW, and an unamended soil and (B) torrefied HSW (200°C), pyrolyzed HSW (300°C, 500°C), composted HSW, pyrolyzed HSW (500°C) + urine, urine alone, and an unamended soil. Marker shapes and colors correspond with amendment type. Marker size corresponds with the magnitude of C mineralization in each treatment per unit soil. Marker values are means (n = 4 ± S.D.; the dashed line connects soils to which thermochemically converted HSW was added).

(Uchimiya et al., 2015; Huang and Tang, 2015), both species having low plant availability. With increasing HHT, phytate is converted into pyrophosphate and subsequently into plant-available orthophosphates. If torrefied HSW indeed contains pyrophosphate, it may have complexed with amendment functional groups or with polyvalent soil metals, explaining the lower P availability with torrefied HSW compared with pyrolyzed 300°C HSW, despite the higher available P in torrefied HSW prior to incubation.

Another explanation is the effect of soil type on lower P, even as N availability remains high. This agrees with observations by Wu et al. (2007) of microbial P immobilization in a weathered Ultisol amended with glucose and rice straw. The two Nyando soils in which P immobilization occurs, sandy loam and clay loam (pH 7.4), contained excess Mn, 10.5 and 2.5 g kg^{-1}, respectively, compared with lower concentrations of less than 1 g kg^{-1} in the other two Yala soils. While P fixation is commonly associated with concentrations of Al or Fe, oxidized Mn can also chemically bond to P under aerobic conditions (Shahandeh et al., 2003). Moreover, coprecipitation of MnIV and phosphate has been reported in soils (Ponnamperuma, 1972), as well as in commercial mixtures of macronutrient and micronutrient fertilizers. Giordano and Mortvedt (1969) found reduced maize P uptake when amended with ammonium phosphate fertilizers containing 10% Mn.

Combined Additions of Pyrolyzed HSW and Urine

Contrary to our expectations, additions of both pyrolyzed HSW (500°C) and urine did not reduce N availability over time, even though there is evidence of reduced NO_3 extractability in biochar-amended soils, even as gross nitrification rates increase (Nelson et al., 2011; Prommer et al., 2014). Since low C respiration was observed when our soils were amended with pyrolyzed HSW (500°C) together with urine compared with HSW compost and pyrolyzed HSW (300°C), additions of pyrolyzed HSW (500°C) with urine did not lead to NO_3 immobilization as sometimes observed (DeLuca et al., 2015). On the other hand, the presence of pyrolyzed HSW as an alkaline soil amendment did not increase NO_3 concentrations in urine-amended soils over time because of the mitigation of soil acidity, which is a noted inhibitor of nitrification (Suzuki et al., 1974; Anthonisen et al., 1976).

Phosphorus availability was significantly higher with pyrolyzed HSW (500°C) + urine compared with urine without biochar. In addition, the mild acidification induced by urine additions did not affect available P in soils with pyrolyzed HSW (500°C). Thus, at the lower end of global scientifically tested biochar application rates, 10 ton ha^{-1} (Jeffery et al., 2015) and urine application in proportion to HSW collected in latrines, a combination of fertilizer that comprised pyrolyzed HSW (500°C) and urine can provide between 26 and 52 kg P ha^{-1}, across three of the four tested soil types (Table 3). This P provision alone constitutes a retail P fertilizer (diammonium phosphate) value of $17.7 to $35.4 ha^{-1} in Kisumu, Kenya, as of June 2018 (http://africafertilizer.org/local-prices/#tab-id-1, scroll to Kisumu, Kenya; Supplementary Figures 1–3, Supplemental Digital Content 1, http://links.lww.com/SS/A53).

CONCLUSIONS

Biologically sanitized HSW in this study had no advantage to thermochemically treated HSW with regard to N and P availability

TABLE 5. Stocks of Mineral N, Available P, and C Mineralization in Soil Amended With Composted HSW, HSW Pyrolyzed at 200°C, 300°C, and 500°C, and Urine With and Without HHSW Pyrolyzed at 500°C in Comparison to an Unamended Control After 100 Days (n = 4 ± S.D.)

Amendment	Soil	NH$_4$ + NO$_3$-N (kg ha soil^{-1})	P (kg ha soil^{-1})	CO$_2$-C (ton ha soil^{-1})
200°C	Clay loam pH 4.6	97.0 ± 7.8	27.9 ± 2.3	2.3 ± 0.1
300°C		73.4 ± 3.1	42.5 ± 8.5	1.3 ± 0.1
500°C		58.4 ± 17.2	49.9 ± 9	1.1 ± 0.1
500°C + urine		486.2 ± 5.6	67.7 ± 5.7	1.5 ± 0.1
Urine		70.9 ± 3.3	25.6 ± 2.8	1.1 ± 0.1
Compost		56.2 ± 8.6	14.7 ± 2.6	1.0 ± 0.1
Unamended		470.4 ± 4.8	29.2 ± 6.3	1.2 ± 0.1
200°C	Clay loam pH 7.4	103.2 ± 6.0	25.8 ± 2.1	3.2 ± 0.4
300°C		52.9 ± 4.8	43.4 ± 2	1.0 ± 0.2
500°C		44.8 ± 3.1	57.6 ± 3.2	0.8 ± 0.2
500°C + urine		420.3 ± 9.6	80.8 ± 3.5	1.0 ± 0.2
Urine		53.6 ± 3.2	22.1 ± 4.2	1.0 ± 0.3
Compost		42.4 ± 5.1	12.8 ± 10.3	0.8 ± 0.2
Unamended		331.8 ± 194.7	27.1 ± 1.4	0.8 ± 0.1
200°C	Loamy fine sand pH 6.1	106.5 ± 6.7	30.2 ± 1.7	2.1 ± 0.1
300°C		61.2 ± 3.9	46.5 ± 4	0.9 ± 0.1
500°C		56.3 ± 2.2	65.1 ± 7.1	0.7 ± 0.1
500°C + urine		464.4 ± 9.2	81.0 ± 2.6	1.0 ± 0.0
Urine		57.1 ± 3.2	27.8 ± 2.1	0.6 ± 0.0
Compost		42.9 ± 1.3	11.5 ± 1.4	0.5 ± 0.1
Unamended		464.6 ± 7.3	23.5 ± 3	0.9 ± 0.1
200°C	Sandy loam pH 6.5	71.2 ± 5.0	12.1 ± 0.5	2.1 ± 0.1
300°C		34.9 ± 5.4	18.3 ± 1.2	0.6 ± 0.1
500°C		34.9 ± 4.3	29.5 ± 2.9	0.5 ± 0.1
500°C + urine		404.3 ± 5.5	32.9 ± 3.9	0.7 ± 0.2
Urine		36.4 ± 2.7	12.0 ± 1	0.5 ± 0.1
Compost		29.4 ± 0.1	5.5 ± 1.1	0.4 ± 0.1
Unamended		414.7 ± 12.5	10.6 ± 1.6	0.6 ± 0.1

in three of the four studied soils. Moreover, the mineralizability of organic C in HSW amendments affected N and P availability in all four studied soils. Also, the trade-off between N and P availability as a function of HHT for thermochemically treated HSW undermines the effectiveness of any one amendment, whether torrefied HSW or pyrolyzed HSW (500°C), as a stand-alone substitute for an NP fertilizer. Urine coapplication with pyrolyzed HSW (500°C) at equivalent rates to the daily supply of excreted HSW and urine in latrines supplied sufficient N and P to serve as an NP fertilizer. In this incubation experiment, however, Mehlich-III extractable P and KCl-extractable N were used as a proxy for plant availability. Validation of our results against actual plant nutrient uptake is the next step in this research. In addition, our experiment was carried out under soil moisture and temperature conditions considered ideal for microbial growth. N and P availability in soils amended with biologically versus thermochemically treated HSW may change under adverse conditions such as drought, anoxia, and excessive leaching.

Additional experimentation under dynamic soil conditions will further clarify the fertilization potential of different HSW amendments.

ACKNOWLEDGMENTS

REFERENCES

Ahern C. R., A. McElna, and D. E. Baker. 1998. Acid Neutralising Capacity Methods Method 19. *In:* Acid Sulfate Soils Laboratory Methods Guidelines. Ahern C. R., B. Blunden, and Y. Stone (eds.). Acid Sulfate Soil Management Advisory Committee, Wollongbar, NSW, Australia, pp. 53.

Almendros G., H. Knicker, and F. J. Gonzalez-Vila. 2003. Rearrangement of carbon and nitrogen forms in peat after progressive thermal oxidation as determined by solid-state ^{13}C-and ^{15}N-NMR spectroscopy. Org. Geochem. 34:1559–1568.

Amoah P., N. Adamtey, and O. Cofie. 2017. Effect of urine, poultry manure, and dewatered faecal sludge on agronomic characteristics of cabbage in Accra, Ghana. Resources. 6:19.

Anand C. K., and D. S. Apul. 2014. Composting toilets as a sustainable alternative to urban sanitation—A review. Waste Manag. 34:329–343.

Anthonisen A. C., R. C. Loehr, T. B. Prakasam, and E. G. Srinath. 1976. Inhibition of nitrification by ammonia and nitrous acid. J. Water Pollut. Control Fed. 48:835–852.

ASTM D1762-84. 2013. Standard Test Method for Chemical Analysis of Wood Charcoal. ASTM International, West Conshohocken, PA.

ASTM D7481-09. 2009. Standard Test Methods for Determining Loose and Tapped Bulk Densities of Powders Using a Graduated Cylinder. ASTM International, West Conshohocken, PA.

Berendes D., K. Levy, J. Knee, T. Handzel, and V. R. Hill. 2015. *Ascaris* and *Escherichia coli* inactivation in an ecological sanitation system in Port-au-Prince, Haiti. PLOS ONE. 10:e0125336.

Berglund L. M., T. H. DeLuca, and O. Zackrisson. 2004. Activated carbon amendments to soil alters nitrification rates in Scots pine forests. Soil Biol. Biochem. 36:2067–2073.

Bernal M. P., J. A. Alburquerque, and R. Moral. 2009. Composting of animal manures and chemical criteria for compost maturity assessment. A review. Bioresour. Technol. 100:5444–5453.

Boye A., L. Verchot, and R. Zomer. 2008. Baseline Report Yala and Nzoia River Basins: Western Kenya Integrated Ecosystem Management Project Findings from the Baseline Surveys. Available at: http://www.worldagroforestry.org/economics_papers/anonymous/WKIEMP/Datasets/Documents%20from%20GraMis/Baseline%20Report%20Yala%20and%20Nzoia%20River%20Basin.pdf (last accessed April 2017).

Cai Y., H. Qi, Y. Liu, and X. He. 2016. Sorption/desorption behavior and mechanism of NH_4^+ by biochar as a nitrogen fertilizer sustained-release material. J. Agric. Food Chem. 64:4958–4964.

Chan K. Y., L. van Zwieten, I. Meszaros, A. Downie, and S. Joseph. 2008. Using poultry litter biochars as soil amendments. Soil Res. 46:437–444.

Chen W., Y. Chen, H. Yang, K. Li, X. Chen, and H. Chen. 2018. Investigation on biomass nitrogen-enriched pyrolysis: Influence of temperature. Bioresour. Technol. 249:247–253.

Chivenge P., B. Vanlauwe, and J. Six. 2011. Does the combined application of organic and mineral nutrient sources influence maize productivity? A meta-analysis. Plant Soil. 342:1–30.

Cofie O., G. Kranjac-Berisavljevic, and P. Drechsel. 2005. The use of human waste for peri-urban agriculture in Northern Ghana. Renew. Agric. Food Syst. 20:73–80.

Cordell D., A. Rosemarin, J. J. Schröder, and A. L. Smit. 2011. Towards global phosphorus security: A systems framework for phosphorus recovery and reuse options. Chemosphere. 84:747–758.

DeLuca T., M. C. Nilsson, and O. Zackrisson. 2002. Nitrogen mineralization and phenol accumulation along a fire chronosequence in northern Sweden. Oecologia. 133:206–214.

DeLuca T. H., M. J. Gundale, M. D. MacKenzie, and D. L. Jones. 2015. Biochar effects on soil nutrient transformations. Biochar effects on crop yield. *In:* Biochar for Environmental Management: Science, Technology and Implementation (2nd ed) Lehmann J., and S. Joseph (eds.). Routledge, New York, pp. 421–454.

Drechsel P., B. Keraita, O. Cofie, and O. Nikiema. 2015. Productive and Safe Use of Urban Organic Wastes and Wastewater in Urban Food Production Systems. Cities and Agriculture: Developing Resilient Urban Food Systems, Earthscan, Routledge, Abingdon, pp. 162.

Enders A., K. Hanley, T. Whitman, S. Joseph, and J. Lehmann. 2012. Characterization of biochars to evaluate recalcitrance and agronomic performance. Bioresour. Technol. 114:644–653.

Esrey S. A. 2001. Towards a recycling society: Ecological sanitation–closing the loop to food security. Water Sci. Technol. 43:177–187.

Estiu G., and K. M. Merz. 2006. Catalyzed decomposition of urea. Molecular dynamics simulations of the binding of urea to urease. Biochemistry. 45:4429–4443.

Feachem R., D. D. Mara, and D. J. Bradley. 1983. Sanitation and Disease. John Wiley & Sons, Washington, DC.

Fox J., and S. Weisberg. 2011. An {R} Companion to Applied Regression (2nd ed). Sage, Thousand Oaks, CA.

Germer J., M. Y. Boh, M. Schoeffler, and P. Amoah. 2010. Temperature and deactivation of microbial faecal indicators during small scale co-composting of faecal matter. Waste Manag. 30:185–191.

Giordano P. M., and J. J. Mortvedt. 1969. Phosphorus availability to corn as affected by granulating manganese with ortho-and pyrophosphate fertilizers. Soil Sci. Soc. Am. J. 33:460–463.

Graves S., H. P. Piepho, L. Selzer, and S. Dorai-Raj. 2015. multcompVieiw: Visualizations of Paired Comparisons. R package version 0.1-7. Available at: https://CRAN.R-project.org/package=multcompView (last accessed January 2016).

Hammer M., and J. Clemens. 2007. A tool to evaluate the fertiliser value and the environmental impact of substrates from wastewater treatment. Water Sci. Technol. 56:201–209.

Hargreaves J. C., M. S. Adl, and P. R. Warman. 2008. A review of the use of composted municipal solid waste in agriculture. Agr. Ecosyst. Environ. 123:1–14.

Heinonen-Tanski H., and C. van Wijk-Sijbesma. 2005. Human excreta for plant production. Bioresour. Technol. 96:403–411.

Hossain M. K., V. Strezov, K. Y. Chan, A. Ziolkowski, and P. F. Nelson. 2011. Influence of pyrolysis temperature on production and nutrient properties of wastewater sludge biochar. J. Environ. Manage. 92:223–228.

Huang R., and Y. Tang. 2015. Speciation dynamics of phosphorus during (hydro)thermal treatments of sewage sludge. Environ. Sci. Technol. 49:14466–14474.

Ilango A., and O. Lefebvre. 2016. Characterizing properties of biochar produced from simulated human feces and its potential applications. J. Environ. Qual. 45:734–742.

Jeffery S., D. Abalos, K. A. Spokas, and F. G. Verheijen. 2015. Biochar effects on crop yield. *In:* Biochar for Environmental Management: Science, Technology and Implementation, 2. Lehmann J., and S. Joseph (eds.). Routledge, New York, pp. 301–325.

Jeffery S., D. Abalos, M. Prodana, A. C. Bastos, J. W. van Groenigen, B. A. Hungate, and F. Verheijen. 2017. Biochar boosts tropical but not temperate crop yields. Environ. Res. Lett. 12:053001.

Joseph S. D., M. Camps-Arbestain, Y. Lin, P. Munroe, C. H. Chia, J. Hook, L. van Zwieten, S. Kimber, A. Cowie, B. P. Singh, and J. Lehmann. 2010. An investigation into the reactions of biochar in soil. Soil Res. 48:501–515.

Katukiza A. Y., M. Ronteltap, C. B. Niwagaba, J. W. Foppen, F. Kansiime, and P. N. Lens. 2012. Sustainable sanitation technology options for urban slums. Biotechnol. Adv. 30:964–978.

Kenya Soil Survey. 2013. Soil and Terrain Database for Kenya (ver. 2.0) (KENSOTER). Available at: http://www.isric.org/data/soil-and-terrain-database-kenya-ver-20-kensoter (last accessed April 2017).

Kizito S., S. Wu, W. Kipkemoi Kirui, M. Lei, Q. Lu, H. Bah, and R. Dong. 2015. Evaluation of slow pyrolyzed wood and rice husks biochar for adsorption of ammonium nitrogen from piggery manure anaerobic digestate slurry. Sci. Total Environ. 505:102–112.

Knicker H. 2007. How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. Biogeochemistry. 85:91–118.

Koch M., and W. Rotard. 2000. On the contribution of background sources to the heavy metals content of municipal sewage sludge. *In:* 1st World Water Congress of the International Water Association, Conference Preprint Book 1—Industrial Wastewater Treatment. IWA, Paris, France, pp. 367–374.

Krause A., T. Nehls, E. George, and M. Kaupenjohann. 2016. Organic wastes from bioenergy and ecological sanitation as a soil fertility improver: A field experiment in a tropical Andosol. Soil. 2:147–162.

Kwabiah A. B., N. C. Stoskopf, C. A. Palm, and R. P. Voroney. 2003. Soil P availability as affected by the chemical composition of plant materials: Implications for P-limiting agriculture in tropical Africa. Agr. Ecosyst. Environ. 100:53–61.

Lamas F., C. Irigaray, C. Oteo, and J. Chacon. 2005. Selection of the most appropriate method to determine the carbonate content for engineering purposes with particular regard to marls. Eng. Geol. 81:32–41.

Lehmann J., J. P. da Silva, C. Steiner, T. Nehls, W. Zech, and B. Glaser. 2003. Nutrient availability and leaching in an archaeological Anthrosol and a Ferralsol of the Central Amazon basin: Fertilizer, manure and charcoal amendments. Plant Soil. 249:343–357.

Lehmann J., M. C. Rillig, J. Thies, C. A. Masiello, W. C. Hockaday, and D. Crowley. 2011. Biochar effects on soil biota—A review. Soil Biol. Biochem. 43:1812–1836.

Lenth R. V. 2016. Least-squares means: The R Package lsmeans. J. Stat. Software. 69:1–33.

Lof P., and J. H. V. van Baren; ISRIC, FAO, and UNESCO. 1987. Soils of the World. Elsevier Science Publishers, New York.

Lohri C. R., S. Diener, I. Zabaleta, A. Mertenat, and C. Zurbrügg. 2017. Treatment technologies for urban solid biowaste to create value products: A review with focus on low-and middle-income settings. Rev. Environ. Sci. Biotechnol. 16:81–130.

Marks E. A., S. Mattana, J. M. Alcañiz, and X. Domene. 2014. Biochars provoke diverse soil mesofauna reproductive responses in laboratory bioassays. Eur. J. Soil Biol. 60:104–111.

Matthiessen M. K., F. J. Larney, L. Brent Selinger, and A. F. Olson. 2005. Influence of loss-on-ignition temperature and heating time on ash content of compost and manure. Commun. Soil Sci. Plant Anal. 36:2561–2573.

Maurer M., W. Pronk, and T. A. Larsen. 2006. Treatment processes for source-separated urine. Water Res. 40:3151–3166.

Mawioo P. M., C. M. Hooijmans, H. A. Garcia, and D. Brdjanovic. 2016. Microwave treatment of faecal sludge from intensively used toilets in the slums of Nairobi, Kenya. J. Environ. Manage. 184:575–584.

Mehlich A. 1984. Mehlich 3 soil test extractant: A modification of Mehlich 2 extractant. Commun. Soil Sci. Plant Anal. 15:1409–1416.

Mihelcic J. R., L. M. Fry, and R. Shaw. 2011. Global potential of phosphorus recovery from human urine and feces. Chemosphere. 84:832–839.

National Farm Information Service (NAFIS) of Kenya: Field management for maize. Available at: http://www.nafis.go.ke/agriculture/maize/field-management-practices/ (last accessed July 2018).

Nelissen V., T. Ruetting, D. Huygens, J. Staelens, G. Ruysschaert, and P. Boeckx. 2012. Maize biochars accelerate short-term soil nitrogen dynamics in a loamy sand soil. Soil Biol. Biochem. 55:20–27.

Nelson N. O., S. C. Agudelo, W. Yuan, and J. Gan. 2011. Nitrogen and phosphorus availability in biochar-amended soils. Soil Sci. 176:218–226.

Nguyen T. T. N., C. Y. Xu, I. Tahmasbian, R. Che, Z. Xu, X. Zhou, H. M. Wallace, and S. H. Bai. 2017. Effects of biochar on soil available inorganic nitrogen: A review and meta-analysis. Geoderma. 288:79–96.

Niwagaba C., M. Nalubega, B. Vinnerås, C. Sundberg, and H. Jönsson. 2009. Bench-scale composting of source-separated human faeces for sanitation. Waste Manag. 29:585–594.

O'Keefe M., C. Lüthi, I. K. Tumwebaze, and R. Tobias. 2015. Opportunities and limits to market-driven sanitation services: Evidence from urban informal settlements in East Africa. Environ. Urban. 27:421–440.

Orner K. D., and J. R. Mihelcic. 2018. A review of sanitation technologies to achieve multiple sustainable development goals that promote resource recovery. Environ. Sci. Water Res. Technol. 4:16–32.

Ponnamperuma F. N. 1972. The chemistry of submerged soils. Adv. Agron. 24:29–96.

Prommer J., W. Wanek, F. Hofhansl, D. Trojan, P. Offre, T. Urich, C. Schleper, S. Sassmann, B. Kitzler, G. Soja, and R. C. Hood-Nowotny. 2014. Biochar decelerates soil organic nitrogen cycling but stimulates soil nitrification in a temperate arable field trial. PLoS One. 9:e86388.

Prosser J. I., and G. W. Nicol. 2012. Archaeal and bacterial ammonia-oxidisers in soil: The quest for niche specialisation and differentiation. Trends Microbiol. 20:523–531.

RStudio Team. 2015. RStudio: Integrated Development for R. RStudio, Inc., Boston, MA, pp. Available at: http://www.rstudio.com/ (last accessed April 2017).

Sanchez P. A. 2002. Ecology. Soil fertility and hunger in Africa. Science. 295:2019–2020.

Schindelbeck R. R., and K. Kurtz. 2016. Soil Texture. *In:* Comprehensive Assessment of Soil Health—The Cornell Framework Manual (Edition 3.1 ed). Moebius-Clune B. N., D. J. Moebius-Clune, B. K. Gugino, O. J. Idowu, R. R. Schindelbeck, A. J. Ristow, H. M. van Es, J. E. Thies, H. A. Shayler, M. B. McBride, D. W. Wolfe, and G. S. Abawi (eds.). Cornell University, Geneva, NY, pp. 37–38.

Schneider F., and S. B. Haderlein. 2016. Potential effects of biochar on the availability of phosphorus—Mechanistic insights. Geoderma. 277:83–90.

Serio M. A., J. E. Cosgrove, M. A. Wójtowicz, J. Lee, K. Wignarajah, and J. Fisher. 2015. Space applications of torrefaction processing. 45th International Conference on Environmental Systems, ICES-2015-291, Bellevue, WA, July 12–16.

Shahandeh H., L. R. Hossner, and F. T. Turner. 2003. Phosphorus relationships to manganese and iron in rice soils. Soil Sci. 168:489–500.

Shinogi Y., H. Yoshida, T. Koizumi, M. Yamaoka, and T. Saito. 2003. Basic characteristics of low-temperature carbon products from waste sludge. Advances in Environmental Research. 7:661–665.

Shinogi Y., and Y. Kanri. 2003. Pyrolysis of plant, animal and human waste: Physical and chemical characterization of the pyrolytic products. Bioresour. Technol. 90:241–247.

Shmaefsky B. R. 1990. Artificial urine for laboratory testing. Am. Biol. Teach. 52:170–172.

Soukup D. A., B. J. Buck, W. Harris, A. L. Ulery, and R. L. Drees. 2008. Preparing soils for mineralogical analyses. *In:* Methods of Soil Analysis Part 5—Mineralogical Methods. Ulery A. L., and R. L. Drees (eds.). SSSA Book Ser. 5.5. SSSA, Madison, WI, pp. 13–31.

Schönning C., and T. A. Stenstroem. 2004. Guidelines on the safe use of urine and faeces in ecological sanitation systems. Report 2004-1. EcoSanRes Programme. Stockholm Environment Institute. Available at: http://www.ecosanres.org/pdf_files/ESR_Publications_2004/ESR1web.pdf (last accessed April 2017).

Subedi R., N. Taupe, I. Ikoyi, C. Bertora, L. Zavattaro, A. Schmalenberger, J. J. Leahy, and C. Grignani. 2016. Chemically and biologically-mediated fertilizing value of manure-derived biochar. Sci. Total Environ. 550: 924–933.

Suzuki I., U. Dular, and S. C. Kwok. 1974. Ammonia or ammonium ion as substrate for oxidation by *Nitrosomonas europaea* cells and extracts. J. Bacteriol. 120:556–558.

Taghizadeh-Toosi A., T. J. Clough, R. R. Sherlock, and L. M. Condron. 2012a. Biochar adsorbed ammonia is bioavailable. Plant Soil. 350:57–69.

Taghizadeh-Toosi A., T. J. Clough, R. R. Sherlock, and L. M. Condron. 2012b. A wood based low-temperature biochar captures NH_3-N generated from ruminant urine-N, retaining its bioavailability. Plant Soil. 353:73–84.

Tarpeh W. A., I. Wald, M. O. Omollo, T. Egan, and K. L. Nelson. 2018. Evaluating ion exchange for nitrogen recovery from source-separated urine in Nairobi, Kenya. Dev. Eng. 3:188–195.

Tiquia S. M., N. F. Y. Tam, and I. J. Hodgkiss. 1996. Microbial activities during composting of spent pig-manure sawdust litter at different moisture contents. Bioresour. Technol. 55:201–206.

Uchimiya M., and S. Hiradate. 2014. Pyrolysis temperature-dependent changes in dissolved phosphorus speciation of plant and manure biochars. J. Agric. Food Chem. 62:1802–1809.

Uchimiya M., S. Hiradate, and M. J. Antal. 2015. Dissolved phosphorus speciation of flash carbonization, slow pyrolysis, and fast pyrolysis biochars. ACS Sustain. Chem. Eng. 3:1642–1649.

Udert K. M., and M. Wächter. 2012. Complete nutrient recovery from source-separated urine by nitrification and distillation. Water Res. 46:453–464.

USDA. 1998. Keys to Soil Taxonomy. US Department of Agriculture, New York.

Vinnerås B., and H. Jönsson. 2002. The performance and potential of faecal separation and urine diversion to recycle plant nutrients in household wastewater. Bioresour. Technol. 84:275–282.

Vinnerås B., A. Björklund, and H. Jönsson. 2003. Thermal composting of faecal matter as treatment and possible disinfection method—Laboratory-scale and pilot-scale studies. Bioresour. Technol. 88:47–54.

Wang B., J. Lehmann, K. Hanley, R. Hestrin, and A. Enders. 2016. Ammonium retention by oxidized biochars produced at different pyrolysis temperatures and residence times. RSC Adv. 6:41907–41913.

Wang T., M. Camps-Arbestain, M. Hedley, and P. Bishop. 2012a. Predicting phosphorus bioavailability from high-ash biochars. Plant Soil. 357: 173–187.

Wang T., M. Camps Arbestain, M. Hedley, and P. Bishop. 2012b. Chemical and bioassay characterisation of nitrogen availability in biochar produced from dairy manure and biosolids. Org. Geochem. 51:45–54.

Wei F., J. P. Cao, X. Y. Zhao, J. Ren, B. Gu, and X. Y. Wei. 2018. Formation of aromatics and removal of nitrogen in catalytic fast pyrolysis of sewage sludge: A study of sewage sludge and model amino acids. Fuel. 218: 148–154.

Whitman T., Z. Zhu, and J. Lehmann. 2014. Carbon mineralizability determines interactive effects on mineralization of pyrogenic organic matter and soil organic carbon. Environ. Sci. Technol. 48:13727–13734.

Wickham H. 2009. ggplot2: Elegant Graphics for Data Analysis. Springer, New York.

Winker M., B. Vinnerås, A. Muskolus, U. Arnold, and J. Clemens. 2009. Fertiliser products from new sanitation systems: Their potential values and risks. Bioresour. Technol. 100:4090–4096.

Woo S. H., A. Enders, and J. Lehmann. 2016. Microbial mineralization of pyrogenic organic matter in different mineral systems. Org. Geochem. 98:18–26.

World Health Organization. 2006. Guidelines for the safe use of wastewater, excreta and greywater. Vol. 2. Wastewater use in agriculture. World Health Organization, Geneva, Switzerland.

Wu J., M. Huang, H. A. Xiao, Y. R. Su, C. L. Tong, D. Y. Huang, and J. K. Syers. 2007. Dynamics in microbial immobilization and transformations of phosphorus in highly weathered subtropical soil following organic amendments. Plant Soil. 290:333–342.

Zaller J. G., and U. Koepke. 2004. Effects of traditional and biodynamic farmyard manure amendment on yields, soil chemical, biochemical and biological properties in a long-term field experiment. Biol. Fertil. Soils. 40: 222–229.

Zheng H., Z. Wang, X. Deng, S. Herbert, and B. Xing. 2013. Impacts of adding biochar on nitrogen retention and bioavailability in agricultural soil. Geoderma. 206:32–39.

Zwetsloot M. J., J. Lehmann, and D. Solomon. 2015. Recycling slaughterhouse waste into fertilizer: How do pyrolysis temperature and biomass additions affect phosphorus availability and chemistry? J. Sci. Food Agric. 95:281–288.