Carbon dioxide and gaseous nitrogen emissions from biochar-amended soils under wastewater irrigated urban vegetable production of Burkina Faso and Ghana

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Abstract

To quantify carbon (C) and nitrogen (N) losses in soils of West African urban and peri-urban agriculture (UPA) we measured fluxes of CO_2-C, N_2O-N, and NH_3-N from irrigated fields in Ouagadougou, Burkina Faso, and Tamale, Ghana, under different fertilization and (waste-)water regimes. Compared with the unamended control, application of fertilizers increased average cumulative CO_2-C emissions during eight cropping cycles in Ouagadougou by 103% and during seven cropping cycles in Tamale by 42%. Calculated total emissions measured across all cropping cycles reached 14 t C ha^{-1} in Ouagadougou, accounting for 73% of the C applied as organic fertilizer over a period of two years at this site, and 9 t C ha^{-1} in Tamale. Compared with unamended control plots, fertilizer application increased N_2O-N emissions in Ouagadougou during different cropping cycles, ranging from 37 to 360%, while average NH_3-N losses increased by 670%. Fertilizer application had no significant effects on N_2O-N losses in Tamale. While wastewater irrigation did not significantly enhance CO_2-C emissions in Ouagadougou, average CO_2-C emissions in Tamale were 71% (1.6 t C ha^{-1}) higher on wastewater plots compared with those of the control (0.9 t C ha^{-1}). However, no significant effects of wastewater on N_2O-N and NH_3-N emissions were observed at either location. Although biochar did not affect N_2O-N and NH_3-N losses, the addition of biochar could contribute to reducing CO_2-C emissions from urban garden soils. When related to crop production, CO_2-C emissions were higher on control than on fertilized plots, but this was not the case for absolute CO_2-C emissions.

Key words: ammonia volatilization / biochar / carbon dioxide emissions / inorganic N fertilization / urban agriculture / wastewater irrigation

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1 Introduction

Most agricultural soils in sub-Saharan Africa (SSA) are limited in their ability to supply crops with adequate amounts of nutrients thus severely limiting food production. This is due to their inherently low fertility as a result of long weathering, leading to low cation exchange capacity and rapid turnover rates of soil organic carbon (SOC; *Bationo* and *Buerkert*, 2001). In these often sandy tropical SOC, which largely determines a soil's fertility, and nitrogen (N) are quickly lost through gaseous emissions and leaching (*Valentini* et al., 2014; *Kim* et al., 2016). However, the lack of available data from local management systems leads to considerable uncertainty on the magnitude of these losses (*Ciais* et al., 2011; *Hickman* et al., 2014; *Rosenstock* et al., 2016) and consequently limits the knowledge necessary to improve nutrient use efficiency for crop production.

West Africa's fast growing population and rapid urbanization lead to quickly rising market demands for produce from urban and peri-urban agriculture (UPA; *Predotova* et al., 2010b). However, the sustainability of UPA systems is debatable due to high rates of fertilization and pesticide use causing site-specific negative externalities (*Drechsel* and *Dongus*, 2010). Characterized by high rates of soil amendments, year round irrigation and high temperatures, these systems are prone to rapid mineralization of organic C and N (*Diogo* et al., 2010; *Predotova* et al., 2010b; *Lompo* et al., 2012) *via* carbon dioxide (CO_2), ammonia (NH_3), and nitrous oxide (N_2O) emissions. The magnitude of these losses depends on the availability of C and N, soil management practices, and environmental conditions (*Pelster* et al., 2012; *Kim* et al., 2016).

The rise in urban population leads to growing production of wastewater which, due to its high nutrient concentration, has

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become an important resource for irrigation throughout semi-arid Africa (*Xue* et al., 2012; *González-Méndez* et al., 2015). Fertilizer use efficiency and environmental safety are therefore important issues that need attention in UPA systems. The reduction of agronomic losses and replenishments of nutrients without adverse environmental effects are prerequisites for improving resource use efficiency and thus food security and sustainability (*Peoples* et al., 1995). In this context, the share of emissions contributed by fertilizer and wastewater in UPA has not been well addressed. Furthermore, when studying the effects of agricultural management practices on crop yield and gaseous emissions it may be useful to study emissions and yield together. Expressing emissions per unit of yield may be more meaningful than looking at values per area only (*Venterea* et al., 2011).

Biochar, a carbonized solid residue from pyrolysis (*Woolf*, 2008) has been promoted for C sequestration and enhanced N retention (*Steiner* et al., 2008) and aged biochar in soils can increase nutrient use efficiency. This may happen through the oxidation of its surface which creates exchange sites for nutrients and thus increasing cation exchange capacity (CEC) of soils (*Glaser* et al., 2002). Other mechanisms which are reported to be responsible for the functioning of biochar in nutrient retention include the formation of an organic coating (*Hagemann* et al., 2017) and its ability to reduce leaching through water retention. Due to its biological stability as well as to its effects on soil greenhouse gas exchange processes, the use of biochar as a soil amendment has been advocated to mitigate greenhouse gas (GHG) emissions, offering the potential to sequester C (*Lehmann* et al., 2006; *Woolf* et al., 2010). Therefore, applying N fertilizers in combination with biochar may be an approach to reduce N losses, enhance N use efficiency, and reduce the environmental impact of fertilization (*Cayuela* et al., 2014).

Whether biochar can accelerate the mineralization of SOC has recently been investigated by several researchers (*Kuzyakov* et al., 2009; *Wang* et al., 2016). While some studies reported increased mineralization and subsequent losses (*Maestrini* et al., 2015), others have found reduced decomposition of SOC (*Lu* et al., 2014), consequently reducing C and N emissions.

Biochar has been reported as particularly promising on poor and degraded soils (*Glaser* et al., 2001, 2002; *Steiner* et al., 2007; *Jones* et al., 2012). The small land areas in UPA systems are typically intensively cultivated with multiple crops per year. This implies high nutrient application. Biochar may be an ideal candidate for use in these systems, since the scale of application could be economically feasible.

The objectives of this study were to investigate the effects of fertilizer (manure and/or inorganic) application, wastewater irrigation and biochar amendment on gaseous C and N losses in urban gardens of Ouagadougou (Burkina Faso) and Tamale (Ghana) under typical semi-arid sub-Saharan conditions. We hypothesized that (1) farmers' fertilization practices lead to considerable gaseous C and N losses in urban gardens of Ouagadougou and Tamale, (2) the use of wastewater for irrigation at farmers' usual quantities leads to increased C and N losses compared with clean water irrigation, (3) the addition of biochar to urban gardens of Ouagadougou and Tamale will reduce gaseous losses of C and N from these gardens, and (4) high crop productivity of these high input gardens compensates for the CO_2-C emitted per unit crop yield.

2 Material and methods

2.1 Study area and soil characteristics

The study was conducted in Ouagadougou (12°24′16.3′′N, 1°28′41.0′′W, 300 m asl), the capital city of Burkina Faso, and Tamale (9°28′28.75′′N, 0°50′53.48′′W, 195 m asl) in northern Ghana, West Africa. Ouagadougou is part of the Sudano-Sahelian ecological zone, whereas Tamale is in the Guinea-Savanna, which are both characterized by a semi-arid climate with an unimodal rainfall regime < 6 months duration. The experimental field in Ouagadougou was set up in Wayalguin, while in Tamale it was set up in Zagyuri. Both are areas where vegetables for the urban market are produced at least since 1978 (*Cissé* et al., 2002; *Kiba* et al., 2012) and they are outside of flooding zones and major groundwater fluctuations.

In Ouagadougou the rainy season lasts from mid-May to mid-October (4 months > 40 mm precipitation, Bsh climate) with an average annual rainfall of 790 mm and in Tamale from mid-March to mid-October (7 months > 40 mm precipitation, Aw climate) with an average annual rainfall of 1110 mm (*Climate-data*, 2016). Weather data were recorded at the study sites using a Watchdog® weather station (Spectrum Technologies Inc., Plainfield, IL, USA). During the study period from May 2014 to April 2016, respective daily temperatures for Ouagadougou and Tamale were 10°C and 12°C (minimum temperature) in the cold dry season and 43°C and 44°C (maximum temperature) in the hot dry season. Average temperatures for the two years were 27.9°C and 28.3°C, while total rainfall for the first and second year were 730 mm and 841 mm, and 800 mm and 620 mm for Ouagadougou and Tamale, respectively. The soil in Ouagadougou was characterized as a Haplic Lixisol (Cutanic) derived from floodplain sediments, while in Tamale the soil was a Petroplinthic Cambisol (*FAO*, 2014). Topsoils (0–0.2 m) in both cities were sandy with low organic carbon contents. The pH ($CaCl_2$) was 5.9 and 5.1 for Ouagadougou and Tamale, respectively (Tab. 1).

2.2 Experimental design, setup, and treatments

From a multi-factorial split-plot experiment (*Akoto-Danso* et al., 2018; *Manka'abusi* et al., 2019), six treatments arranged in blocks and randomly allocated in four replicates were selected for measurements of CO_2-C and N (N_2O-N and NH_3-N) emissions. The treatments comprised (1) a control (C), (2) farmers' practice (FP_1), and (3) FP_1+biochar (FP_1+BC) irrigated with either clean water (cw) or wastewater (ww) at farmers' usual irrigation quantity. Measurements on FP_1+BC+cw plots only started during the fourth and fifth cropping cycles in Ouagadougou and Tamale, respectively. Plot size was 8 m^2 (2 × 4 m) and paths of 0.7 m between plots were kept untilled throughout the study period. Adjacent, passing the city of Ouagadougou from west to east, is a

Table 1: Initial soil and biochar properties for the multi–factorial cropping experiment conducted from May 2014 to April 2016 in Ouagadougou (Burkina Faso) and Tamale (Ghana).[a]

	Soil (0–20cm)		Biochar	
	Ouagadougou	**Tamale**	**Ouagadougou (Corn cob)**	**Tamale (Rice husk)**
Sand (%)	59.6	45.7	–	–
Clay (%)	5.3	5.9	–	–
CEC (mmol$_c$ kg^{-1})	56.7	36.1	11.2	
pH ($CaCl_2$)	5.9	5.1	n.d.	n.d.
pH (1:5 v/w deionized H_2O)	n.d.	n.d.	10.3	9.1
Carbon (C) (%)	0.56	0.4	68.4	42.4
Nitrogen (N) (%)	0.06	0.04	0.9	0.6
Bray-Phosphorus (P) (mg kg^{-1})	135	7.7	n.d.	n.d.
Total phosphorus (P) (mg kg^{-1})	534.6	110.9	1406.2	861.3
NH_4Cl extr. potassium (K) (mg kg^{-1})	29.6	38.9	3296.1	977.1
BET (m^2 g^{-1})	–	–	43.5	62.9
Volatile matter (%)	–	–	19.6	23.2
Ash content (%)	–	–	18.5	45.2
H/C (molar ratio)	–	–	0.04	0.05
O/C (molar ratio)	–	–	0.16	0.27

[a]Sourced from *Häring* et al. (2017) and Atiah et al. (unpublished). BET = Brunauer–Emmett–Teller.

wastewater canal containing a mixture of urban runoff, which serves as a source of irrigation water. In Tamale, wastewater came from domestic sewage effluents of the Kamina Military Barracks. In Ouagadougou, clean water was sourced from tap water, supplied by the National Institute of Water and Sanitation (Office National de l'Eau et de l'Assainissement, ONEA) Burkina Faso or local wells while clean water in Tamale was supplied by Ghana Water Company Limited.

2.3 Biochar production and incorporation

Biochar was made from corn cobs in Ouagadougou and rice husks in Tamale, which are widely available agricultural waste materials in the regions (*Duku* et al., 2011). It was produced *via* slow pyrolysis using a local kiln at a temperature of about 500°C. The average residence time inside the pyrolsis reactor was 47 h for corn cob biochar and 43 h for rice husk biochar. Subsequently, the corn cob biochar was manually crushed to particle sizes < 2 mm prior to use, while rice husk biochar with particle size < 2 mm was applied without further alteration. Analysis revealed that corn cob biochar had a pH [1:5 (v/w) deionized H_2O] of 10.3 and a C content of 68%, while the pH of rice husk biochar was 9.1 with a C content of 42% (Tab. 1; Atiah et al., unpublished). Biochar was soil-applied at a rate of 20 t ha^{-1} dry weight basis and incorporated to a depth of 0.2 m using hand hoes.

2.4 Field management practices and agronomic inputs

Fertilization, irrigation, and other agronomic practices during the cropping cycles (cropping calendars) were in accordance with farmers' practice (*Akoto-Danso* et al., 2018; *Manka'abusi* et al., 2019). At the start of every cropping cycle, the soils were turned with hand hoes to a depth of 0.2 m. Plots were irrigated manually up to two times a day depending on crop needs. During the rainy season, irrigation was only supplementary. Over the two years a total of 11 cropping cycles were completed in Ouagadougou, while Tamale had 13. Total inputs from irrigation water wastewater differed in the two cities (Tabs. 2 and 3).

In Ouagadougou, combined use of cow manure at a rate of 9 to 20 t dry matter (DM) ha^{-1} and urea [$CO(NH_2)_2$; 46-0-0] ranging from 70 to 375 kg ha^{-1} per crop was practiced. A first fertilization with organic fertilizer took place between the first day and two weeks after planting, while the second (mineral fertilization) took place about two weeks following the first application. Total N input from urea was 1,050 kg N ha^{-1} for two years, while manure supplied was 1,850 kg N ha^{-1} and 23 t C ha^{-1} during the two year study period. In Tamale, each crop was fertilized once and NPK (15-15-15) was used except for jute mallow (*Corchorus olitorius* L.; crops 7 and 8) that was fertilized with urea. The application rate ranged from 200 to 563 kg NPK ha^{-1} per crop, with a total N input of 571 kg N ha^{-1} from NPK and 235 N ha^{-1} from urea for two years (Tab. 3).

2.5 Measurement of gaseous C and N losses

Over the two year experimental period, gas emissions were measured during eight and seven cropping cycles in Ouagadougou and Tamale, respectively, cultivated with either maize (*Zea mays* L.), lettuce (*Lactuca sativa* L.), cabbage (*Brassica oleracea* L.), amaranth (*Amaranthus cruentus* L.), jute mallow or carrot (*Daucus carota* L.).

A closed chamber system consisting of a cuvette connected to a photo-acoustic infrared multi-gas analyzer (INNOVA 1312-5, LumaSense Technologies A/S, Ballerup, Denmark) was used to determine CO_2, N_2O, and NH_3. As described by *Predotova* et al. (2010a), the cuvette was connected to the inlet and outlet of the gas monitor by a 1.5 m long standard Teflon tube® of 3.3 mm inner diameter. The cuvette consisted of a 0.3 m wide and 0.11 m high PVC ring combined with a

Table 2: Cultivated crops, carbon and nutrient inputs, fertilizer quantities and irrigation quantities in a multi-factorial cropping experiment conducted from May 2014 to April 2016 in Ouagadougou (Burkina Faso).[a]

Cropping season	**Season 1 (2014 rainy season)**		**Season 2 (2014/15 dry season)**			**Season 3 (2015 rainy season)**			**Season 4 (2015/16 dry season)**		
Crop number	1*	2*	3*	4*	5*	6	7*	8*	9	10	11*
Crop	Lettuce	Cabbage	Amaranth	Lettuce	Amaranth	Jute mallow	Amaranth	Jute mallow	Roselle	Lettuce	Carrot
Planting date	19.5.14	10.7.14	27.10.14	17.12.14	7.2.15	31.5.15	21.7.15	5.9.15	23.10.15	1.12.15	23.1.16
Harvesting date	23.6.14	10.10.14	25.11.14	27.1.15	11.3.15	28.6.15	26.8.15	2.10.15	23.11.15	13.1.16	13.4.16
Crop duration (days)	35	92	29	41	32	28	36	27	31	43	81
Gas measurement duration (d)	10	10	11	10	11		8	8	8	10	8
Full irrigation (mm)	185.3	118.6	22.6	318.5	318.5	224.3	52.0	63.4	243.8	416.0	819.0
Rainfall (mm)[b]	120.6	513.6	1.4	0.2	0.0	132.8	311.8	70.0	0.4	0.0	20.2
Mineral fertilizer – urea–N (kg ha^{-1})	87.4	87.4	174.8	87.4	174.8	87.4	74.0	74.0	32.6	84.4	85.6
Organic fertilizer – C (t ha^{-1})	2.2	1.9	1.2	4.6	3.8	2.3	1.4	1.4	0.0	1.3	3.3
Organic fertilizer – N (kg ha^{-1})	199.8	171.2	136.4	281.2	283.4	112.5	149.3	128.6	0.0	108.8	279.2
Organic fertilizer – P (kg ha^{-1})	55.2	47.3	36.8	60.0	84.3	62.5	39.7	33.2	0.0	27.9	85.2
Organic fertilizer – K (kg ha^{-1})	85.5	73.3	32.3	168.5	119.8	192.5	78.6	54.8	0.0	67.6	109.7
Organic fertilizer – Ca (kg ha^{-1})	109.2	93.6	61.1	162.8	132.3	114.2	87.8	77.4	0.0	73.4	151.5
Organic fertilizer – Mg (kg ha^{-1})	85.8	73.5	71.9	153.6	163.7	89.7	58.2	47.5	0.0	38.9	90.8
ww–N[c] (kg ha^{-1})	4.1	3.4	36.2	3.1	11.7	3.7	1.4	1.1	9.4	16.0	31.5
ww–P[c] (kg ha^{-1})	2.7	1.7	3.2	2.0	1.5	1.8	0.3	0.5	1.5	2.6	5.2
cw–N[c] (kg ha^{-1})	3.5	2.3	51.7	1.9	1.5	0.9	0.2	0.3	0.1	0.1	0.2
cw–P[c] (kg ha^{-1})	2.6	1.6	3.1	0.9	0.9	0.5	0.0	0.1	0.6	1.0	1.9

[a]The table shows all crops cultivated during the two years experimental period. Crops from which gaseous carbon and nitrogen were measured during the study period are marked with an asterisk (*). Nutrient input estimates for cw and ww are based on the full irrigation level. Data modified from *Manka'abusi* et al. (2019).
[b]Rainfall quantities are same for full and reduced irrigated plots.
[c]cw = clean water; K = potassium; N = nitrogen; P = phosphorus; ww = wastewater.

0.3 m wide and 0.07 m high tightly fitting PVC ring that was pushed 0.05 m into the soil. During measurements, the system was closed for an accumulation time of 3 min to avoid disturbances of measurements due to rapid increase in temperature and moisture within the closed chamber under given climatic conditions (*Buerkert* et al., 2010; *Predotova* et al., 2010b). The cuvette was lifted and ventilated for 2 min between measurements to minimize errors from carryover contamination. Air temperature and humidity inside the cuvette were monitored using an air-tightly installed thermo-hygrometer (PCE-313 A, Paper-Consult Engineering Group, Meschede Germany). Soil temperature was recorded at a depth of 0.1 m using a digital thermometer (Carl ROTH GmbH + Co. KG, Karlsruhe, Germany), while soil moisture was measured with a FieldScout® TDR 100 (Spectrum Technologies Inc., Plainfield, IL, USA) up to a depth of 0.05 m.

To capture the diurnal change of gas emissions, gas measurements were conducted once in the morning between 1 am and 8 am, and once in the afternoon from 11 am to 4 pm. Baseline emissions were determined the day before fertilizer application, and on a daily basis for 4–5 d after fertilization (*Lompo* et al., 2012). Measurements were also conducted the day before harvest when emissions were assumed to have dropped to their baseline rates. Three subsamples were used per plot in order to account for variation within each plot. The

Table 3: Cultivated crops, irrigation quantities (mm) and nutrient inputs in (kg ha^{-1}) of a multi–factorial vegetable growing experiment conducted from May 2014 to April 2016 in Tamale (northern Ghana).[a]

	Season 1 (2014 rainy season)			**Season 2 (2014/15 dry season)**			**Season 3 (2015 rainy season)**				**Season 4 (2015/16 dry season)**		
Crop number	1*	2*	3	4*	5*	6*	7	8	9*	10	11	12	13*
Crop	Maize	Lettuce	Cabbage	Amaranth	Lettuce	Amaranth	Jute mallow	Jute mallow	Amaranth	Jute mallow	Roselle	Lettuce	Carrot
Planting date	9.5.14	19.6.14	26.7.14	21.10.14	15.12.14	4.2.15	24.4.15	4.6.15	25.7.15	8.9.15	20.10.15	18.12.15	18.1.16
Harvesting date	9.6.14	17.7.14	6.10.14	20.11.14	1.2.15	6.3.16	25.5.15	4.7.15	28.8.15	13.10.15	25.11.15	12.1.16	18.4.16
Crop duration (days)	31	28	72	30	48	30	31	30	34	35	36	35	91
Gas measurement duration (d)	5	4		5	5	5			4			5	6
Full irrigation	198.0	339.6	204.9	242.0	431.8	176.0	200.8	160.9	38.5	8.3	264.0	242.0	540.4
Rainfall (mm)[b]	42.2	29.7	542.3	10.4	0.0	37.3	18.8	72.7	146.4	170.6	13.8	0.0	125.9
Fertilizer – N (kg ha^{-1})	84.4	85.5	58.8	31.9	54.1	31.9	115.1	119.5	30.6	45.4	45.2	46.1	57.2
Fertilizer – P (kg ha^{-1})	36.1	36.5	25.1	13.6	23.1	13.6	0.0	0.0	11.6	17.2	17.1	17.5	21.7
Fertilizer – K (kg ha^{-1})	52.3	53.0	36.5	19.8	33.6	19.8	0.0	0.0	15.0	22.3	22.2	22.6	28.1
ww–N[c] (kg ha^{-1})	30.8	52.9	19.3	32.7	172.0	55.0	86.9	91.2	15.0	2.3	55.0	95.6	258.8
ww–P[c] (kg ha^{-1})	4.6	7.9	3.5	12.5	53.8	28.5	14.7	13.8	1.4	0.1	33.4	44.8	87.3
ww–K[c] (kg ha^{-1})	7.2	12.7	5.9	7.3	20.1	9.1	8.0	7.8	4.0	1.2	37.3	36.0	80.4
cw–N[c] (kg ha^{-1})	0.5	0.9	0.5	1.7	3.0	1.2	1.1	0.5	0.1	0.0	1.3	1.2	2.6
cw–P[c] (kg ha^{-1})	0.0	0.0	0.1	0.1	0.2	0.1	0.0	0.0	0.0	0.0	0.5	0.1	0.3
cw–K[c] (kg ha^{-1})	2.3	3.9	1.5	2.7	4.8	2.0	2.1	1.4	0.7	0.2	5.8	5.3	11.8

[a]The table shows all crops cultivated during the two years experimental period. Crops from which gaseous carbon and nitrogen were measured during the study period are marked with an asterisk (*). Nutrient input estimates for cw and ww are based on the full irrigation level. Data modified from *Akoto-Danso* et al. (2018).
[b]Rainfall quantities are same for full and reduced irrigated plots.
[c]cw = clean water; K = potassium; N = nitrogen; P = phosphorus; ww = wastewater.

rings were positioned 0.5 m from the edge of the plot, with one on each 2 m side, while the other ring was placed on the 4 m side of the plot. Plants within rings were cut to base before measurement to eliminate aboveground plant respiration.

2.6 Data processing and statistical analyses

Calculations of gas flux rates for N_2O and NH_3 were conducted by R (*R Core Team*, 2017) using the package gasfluxes (*Fuss*, 2017) with a linear model. Emission rates for CO_2 were calculated by subtracting the gas concentration measured at the beginning of the accumulation period from the concentration at the end of the accumulation period and dividing the result by the time period elapsed. Emission rates were calculated by using an accumulation interval of 1 min. Daily emission rates were then calculated by averaging of the morning and afternoon emission values. Cumulative emissions for each cropping cycle were determined by successive linear interpolation of emissions on the sampling days, assuming that emissions were linear between two measurements. Subsequently, the area under the curve was calculated (*Das* and *Adhya*, 2014). At low rates, emissions of N_2O-N and NH_3-N could not be detected under our tropical conditions, and non-linear data were excluded from the calculations (*Silva* et al., 2015). This reduced our data set, N_2O-N emissions were eventually calculated from six cropping cycles for each city, while losses of NH_3-N were calculated from two cropping cycles in Ouagadougou only. To determine the effects of different treatment combinations on CO_2-C emissions and plant-captured (fixed) C, we divided the cumulated CO_2-C over different cropping cycles by the net rates of total fixed carbon for these cropping cycles (*Sehy* et al., 2003).

Statistical analyses were carried out with SAS (SAS Institute Inc., Carey, NC, USA) using the mixed model procedure (PROC MIXED), which accounted for effects of fertilization, water quality, and biochar. The analysis of variance (ANOVA) with type III test of fixed effects took into consideration the factorial design of our experiments. Block was included in the model as a fixed effect, while the randomized units were added as random effects and tested for their main effects and interactions where applicable. Prior to ANOVA, data residuals were tested for the assumptions of normality of residuals and homoscedasticity using the Shapiro–Wilk test and graphical checks. Significant differences between means were determined by the LSMEANS procedure and adjusted using Tukey's posthoc honest significant difference (HSD) at $p < 0.05$.

3 Results

3.1 Emission rates of CO_2-C, N_2O-N, and NH_3-N

Our data show that the time of the day during which measurements were done had a significant effect on emission rates in both cities ($p < 0.001$). Average CO_2-C emission rates across all cropping cycles and treatments were 203.0 ± 9.3 and 173.4 ± 7.3 mg m^{-2} h^{-1} in the afternoon, while morning hours showed rates of 135.3 ± 5.1 and 98.1 ± 3.5 mg m^{-2} h^{-1} for Ouagadougou and Tamale, respectively.

In Ouagadougou, afternoon emissions of CO_2-C were significantly higher than early morning rates for all fertilized plots ($p < 0.001$), but not for unamended controls. In Tamale, significant differences in emissions between the two daily periods were observed only when wastewater was used for irrigation ($p < 0.001$), while emission rates from plots under clean water treatment did not differ during the day (data not shown). Across treatments, N_2O-N and NH_3-N emission rates were significantly higher in the afternoon than in the morning ($p < 0.001$), except for unamended controls (Fig. 1; data for NH_3-N not shown). In Ouagadougou, N_2O-N morning fluxes were significantly lower than afternoon emissions for all treatments, except for C+cw plots. For N_2O-N in Tamale, emissions were significantly lower in C+ww, FP_1+ww, and FP_1+BC+ww in the morning than in the afternoon.

Emissions rates also differed between cropping cycles and seasons (Tab. 4). In Ouagadougou, averaged morning and afternoon CO_2-C emissions were significantly higher across treatments under amaranth (crop 3) than in all other cropping cycles. Mean emissions reached 450 mg m^{-2} h^{-1} in FP_1+ww plots. CO_2-C fluxes were significantly lower during the cold and dry period (lettuce and carrot cropping cycles) in fertilized treatments. In Tamale, CO_2-C emissions were significantly higher during the first three measured cropping cycles (which included maize, lettuce and amaranth) for all treatments ($p < 0.05$), except for C+cw. This effect was more pronounced in biochar amended plots with emissions reaching 271 mg m^{-2} h^{-1} on FP_1+BC+ww (lettuce; crop 2). Mean emissions were lowest for lettuce (crop 5). Overall, emissions peaks occurred soon after fertilization and were pronounced in Ouagadougou (data not shown).

3.2 Effects of fertilization on cumulative losses of carbon (CO_2-C) and nitrogen (N_2O-N and NH_3-N)

Fertilizer application significantly ($p < 0.001$) increased cumulative CO_2-C emissions for all cropping cycles in Ouagadougou, but only during amaranth (crops 4 and 9) and lettuce (crop 5) cropping cycles in Tamale (data not shown). Over all cropping cycles, mean cumulative CO_2-C fluxes in Ouagadougou increased by 103% in FP_1+cw (1.8 t C ha^{-1}) compared with C+cw (0.9 t C ha^{-1}) plots and those of FP_1+ww increased by 86% compared with C+ww (Fig. 2a). A total of 14 t C ha^{-1} was lost from fertilized plots in Ouagadougou, representing 73% of the total C applied with organic fertilizers. In Tamale, average CO_2-C fluxes were 42% higher in FP_1+cw (1.3 t C ha^{-1}) compared with C+cw (0.9 t C ha^{-1}) plots (Fig. 2b).

Regardless of irrigation water quality, fertilizer application increased N_2O-N and NH_3-N losses in Ouagadougou ($p < 0.05$) during all cropping cycles from which losses were calculated (data not shown). Losses of N_2O-N in Ouagadougou significantly increased on FP_1+cw compared with C+cw plots for all crops by 37% to 360% (Fig. 2c). Total N_2O-N emissions from unfertilized plots over all cropping cycles amounted to 44 kg ha^{-1}, while fertilized plots reached 114 kg ha^{-1}. In addition, cumulative NH_3-N emissions in Ouagadougou significantly increased on FP_1+cw compared with C+cw plots during amaranth (crop 3) by 1739% and amaranth

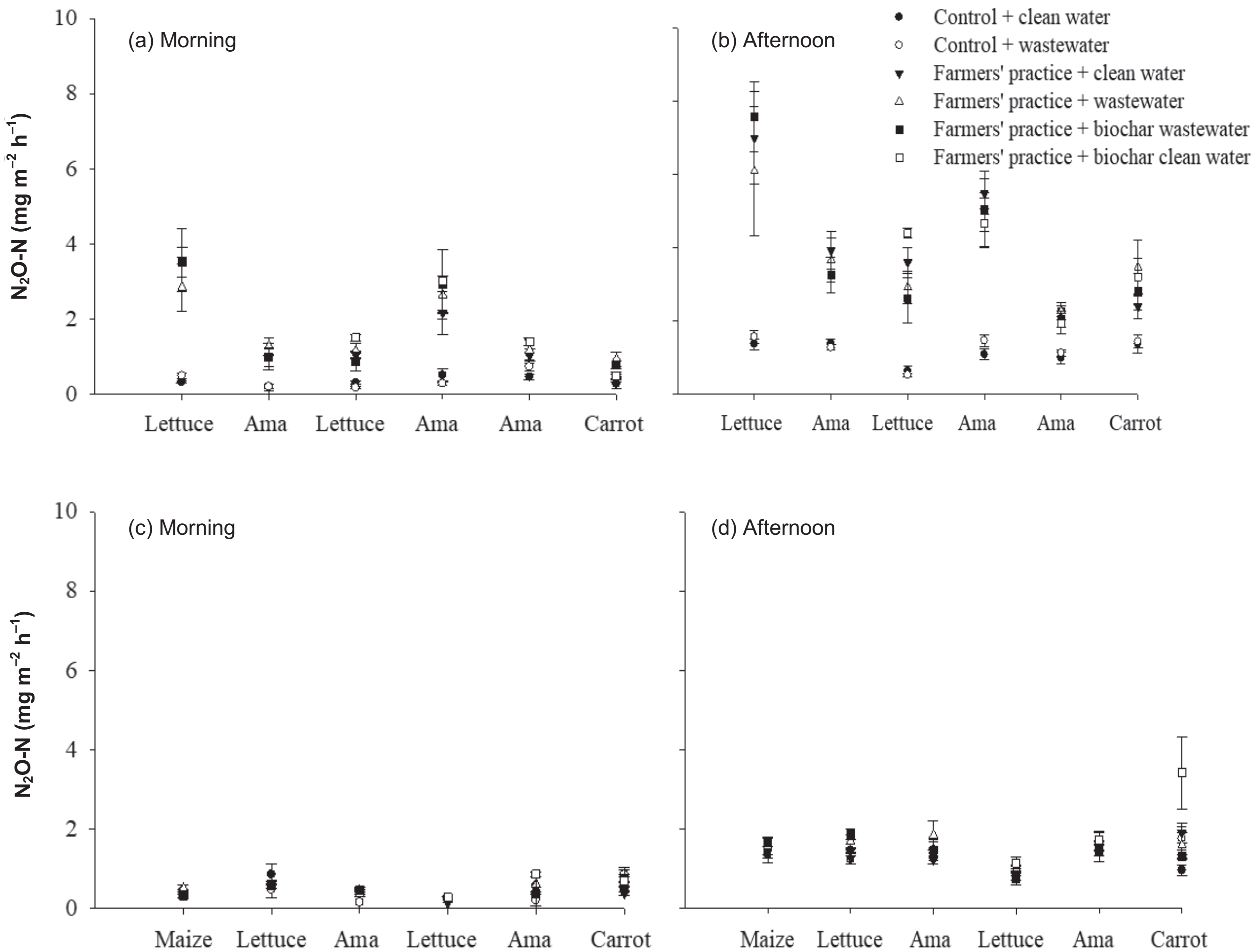

Figure 1: Effects of time of the day on emission rates of N_2O-N in (a, b) Ouagadougou, Burkina Faso and (c, d) Tamale, Ghana. Data points are means of all measurements carried out during each cycle ± one standard error, in a multi-factorial cropping experiment conducted from May 2014 to April 2016. Ama = amaranth.

(crop 5) by 225% (Fig. 2e). Average NH_3-N volatilization from C+cw for two cropping cycles from which emissions were calculated amounted to 2 kg ha^{-1}, while average fluxes from FP_1+cw amounted to 18 kg ha^{-1}, equivalent to a 670% increase in volatilization after fertilizer application. For these two cropping cycles, a total of 5 kg ha^{-1} of NH_3-N was therefore lost from unfertilized plots while up to 38 kg ha^{-1} was lost from fertilized plots.

In Tamale, after fertilizer application, no significant increases were recorded for N_2O-N, when compared with the unfertilized control plots (Fig. 2d). Total losses of N_2O-N from fertilized and unfertilized plots in different cropping cycles amounted to 55 kg ha^{-1} each. NH_3-N losses from different cropping cycles in Tamale could not be detected as a result of low volatilization rates.

3.3 Wastewater irrigation effects on cumulative losses of carbon (CO_2-C) and nitrogen (NH_3-N and N_2O-N)

Irrigating with wastewater increased CO_2-C emissions mainly in Tamale. Regardless of fertilization, four of seven cropping cycles showed significant increases in cumulative CO_2-C emissions, whereas in Ouagadougou only two of eight cropping cycles showed higher cumulative CO_2-C emissions under wastewater irrigation. Emissions on wastewater irrigated plots in Tamale were more pronounced in the dry season. Excluding treatments with fertilizer, there was no significant difference between C+ww and C+cw in Ouagadougou (Fig. 2a), while in Tamale emissions from C+ww plots were significantly higher ($p < 0.05$) during most cropping cycles with an average increase of 71% in C+ww (1.6 t C ha^{-1}) compared with C+cw plots (0.9 t C ha^{-1}; Fig. 2b). In Ouagadougou, total CO_2-C emissions from C+cw plots for all cropping cycles were 7 t ha^{-1}, while C+ww plots emitted 8 t ha^{-1}. Whereas in Tamale total CO_2-C losses of 6 t ha^{-1} were recorded from C+cw plots, emissions from C+ww plots reached 11 t ha^{-1}. Wastewater irrigation had no significant effect on N_2O-N in both cities (Fig. 2c, d) and also on NH_3-N losses in Ouagadougou when compared with clean water (Fig. 2e).

Table 4: Average morning and afternoon CO_2-C emissions during different cropping cycles and seasons in Ouagadougou (Burkina Faso) and Tamale (Ghana) during multifactorial cropping cycles from May 2014 to April 2016. Data show mean of the averaged morning and afternoon fluxes during different measurement times ± one standard error.[a]

City	Cropping cycle	Crop type	Season	Average temperature during cropping cycle (°C)	Average CO_2-C emissions ($mg\ m^{-2}\ h^{-1}$) C+cw	C+ww	FP_1+cw	FP_1+ww	FP_1+BC+cw	FP_1+BC+ww
Ouagadougou (Burkina Faso)	1	Lettuce	rainy	31	96.3 ± 9.0de	97.9 ± 9.4de	185.1 ± 12.7fg	174.8 ± 12.6g	nd	224.4 ± 17.2ef
	3	Amaranth	dry	30	230.8 ± 9.8a	241.5 ± 10.6a	433.8 ± 24.2a	449.9 ± 25.7a	nd	425 ± 20.1a
	4	Lettuce	dry	24	39.4 ± 2.4f	51.7 ± 3.0f	200.9 ± 12.0f	187.2 ± 8.5f	190.3 ± 7.4g	225.2 ± 9.9ef
	5	Amaranth	dry	29	94.9 ± 5.3de	104.9 ± 5.8d	323.6 ± 15.9c	351.7 ± 17.8b	335.4 ± 17.0bc	340.5 ± 15.8b
	7	Amaranth	rainy	27	107.5 ± 1.8d	149.6 ± 1.8bc	258.8 ± 3.4d	277.9 ± 3.5d	231.9 ± 3.2ef	257.2 ± 3.3de
	8	Jute mallow	rainy	28	131.4 ± 12.6c	154.3 ± 10.3b	233 ± 11.3d	247.8 ± 18.0de	245.1 ± 16.9def	255.9 ± 11.8de
	11	Carrot	dry	26	48.7 ± 3.3f	67.1 ± 5.3ef	125.7 ± 9.4h	124.2 ± 9.4h	127.2 ± 9.8h	133.3 ± 10.1h
Tamale (Ghana)	1	Maize	rainy	28	137.0 ± 5.7c	183.5 ± 6.9b	182.0 ± 8.4b	173.5 ± 8.2b	nd	226.7 ± 8.9ab
	2	Lettuce	rainy	29	193.5 ± 20.6b	254.3 ± 18.0a	211.3 ± 15.6a	229.0 ± 15.9 a	nd	270.8 ± 17.2a
	4	Amaranth	dry	27	99.7 ± 4.4d	191 ± 9.3b	181.6 ± 6.9b	214.4 ± 9.3a	nd	245.9 ± 11.0ab
	5	Lettuce	dry	31	42.0 ± 1.7e	88.4 ± 4.4d	50.9 ± 3.4e	88.9 ± 5.6d	56.8 ± 4.8d	80.1 ± 5.1d
	6	Amaranth	rainy	30	54.3 ± 12.5de	149.9 ± 35.3c	101.4 ± 23.1cd	139.4 ± 31.2c	101.4 ± 22.7cd	143 ± 32.0c
	9	Amaranth	rainy	27	60.0 ± 4.3de	78.6 ± 3.5d	92.7 ± 4.6d	95.9 ± 3.1d	83.2 ± 3.0d	106.1 ± 4.0cd
	13	Carrot	dry	31	73.7 ± 7.9d	145.5 ± 12.5c	110.5 ± 13.4c	126.0 ± 11.9c	86.1 ± 7.8d	135.1 ± 12.0c

[a]nd = not determined; C+cw = control clean water; C+ww = control wastewater; FP_1+cw = farmer practice clean water; FP_1+ww = famer practice wastewater; FP_1+BC+ww = farmer practice+biochar clean water; FP_1+BC+ww = Farmer practice+biochar wastewater. Means with different letters for same treatment under different water qualities and across different cropping cycles are significantly different according to Tukey HSD, $p < 0.05$

3.4 Relative C and N losses

In Ouagadougou, relative CO_2-C losses on fertilized plots ranged between 30 and 82% of C input through organic fertilizers during dry seasons, and 65 and 152% during rainy seasons, regardless of irrigation treatment. While on fertilized plots relative N_2O-N losses amounted to 3.5 to 11% without clear seasonal differences and irrigation water effects, on control plots N_2O-N losses exceeded the input from clean and wastewater by 17 to 7300%, except on clean water irrigated plots during amaranth (crop 3) cultivation and on wastewater irrigated plots during carrot (crop 11) cultivation in Ouagadougou. In general, N balances on control plots based on N inputs from irrigation and output from N_2O-N emissions were more negative on clean water than on wastewater irrigated plots. This finding was even more pronounced in Tamale, where the N_2O-N losses in C+cw exceeded N input *via* irrigation in all measured cropping cycles by 179 to 7017%, whereas in C+ww losses accounted for 4 to 46% of input in respective cropping cycles, resulting in a positive N balance. In fertilized plots in Tamale, this difference in relative N losses between clean water and wastewater irrigation was less pronounced with ranges between 7 to 37% in FP_1+cw compared with 4 to 17% in FP_1+ww.

3.5 Biochar effects on cumulative losses of carbon (CO_2-C) and nitrogen (NH_3-N and N_2O-N)

Biochar application significantly increased CO_2-C emissions (p < 0.05) by 17 to 26% during the first four cropping cycles in Tamale, but by 24% during the first cycle in Ouagadougou. However, the tendency of this amendment to enhance CO_2-C emissions was not consistent during cropping cycles of amaranth (crops 3 and 7) in Ouagadougou and of carrot (crop 13) in Tamale (Fig. 3a, b), whereas CO_2-C emissions on biochar plots were lower than on plots without biochar.

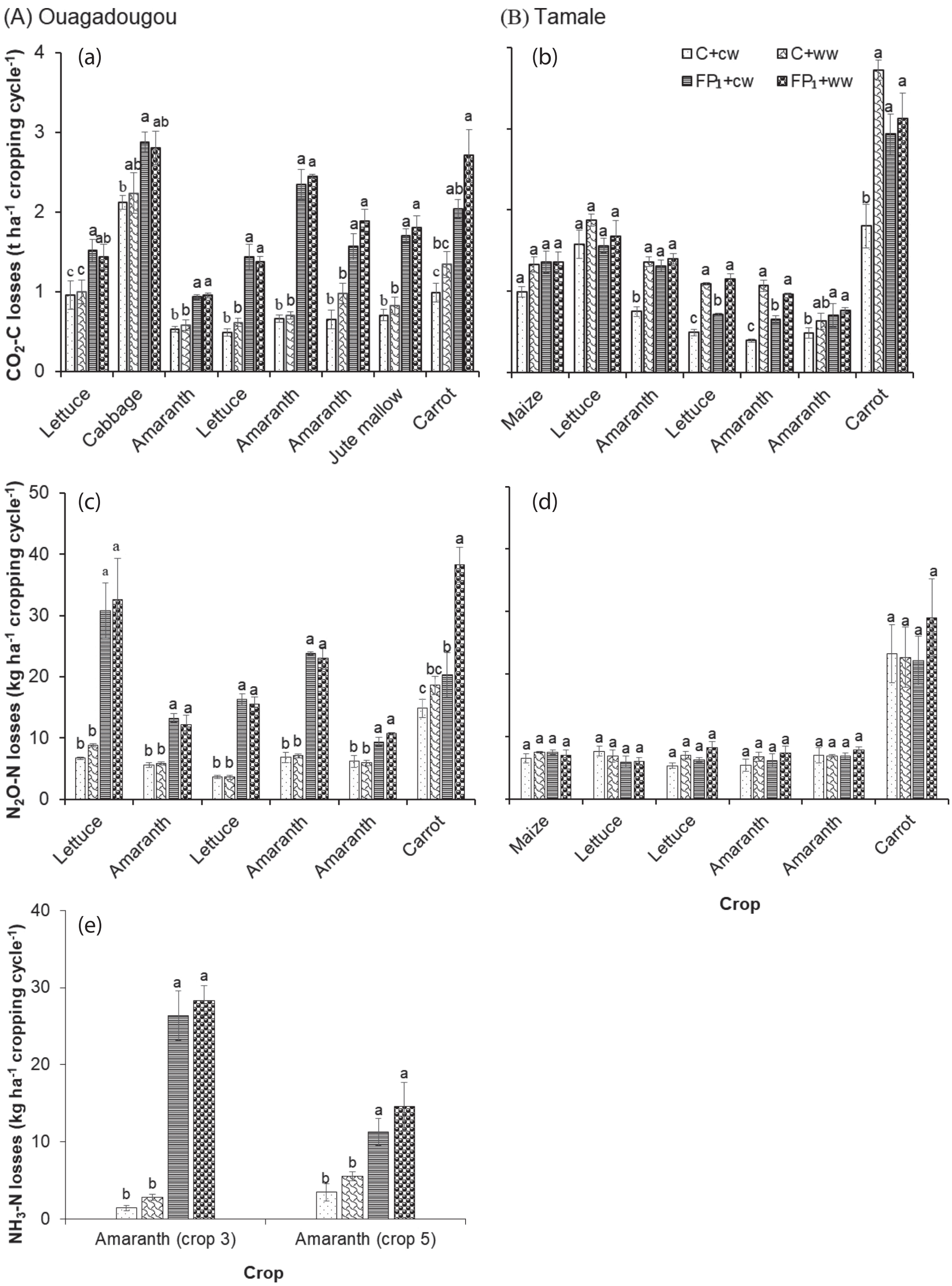

Figure 2: Effects of fertilizer and wastewater on cumulative CO_2-C (a, b), N_2O-N (c, d), and NH_3-N (e) losses for different cropping cycles in urban crop production systems of (A) Ouagadougou (Burkina Faso) and (B) Tamale (Ghana). Data show means (n = 4) ± one standard error, obtained in a multi-factorial cropping experiment conducted at both locations from May 2014 to April 2016. Means with different letters are significantly different according to a Tukey HSD, $p < 0.05$. C+cw = control clean water, C+ww = conttrol wastewater, FP1+cw = farmers' practice clean water, FP+ww = farmers' practice wastewater.

Total emissions of CO_2-C over all cropping cycles in Ouagadougou were similar across biochar levels. Emissions from FP_1+cw and FP_1+BC+cw (in four measured cycles) both recorded 9 t C ha^{-1}, while FP_1+ww and FP_1+BC+ww (in seven cropping cycles) amounted to 15 t C ha^{-1} and 16 t C ha^{-1}, respectively. However, in Tamale total emissions were significantly higher (+12%) in FP_1+BC+ww than in FP_1+ww; p = 0.05, but not on clean water irrigated plots. Total CO_2-C

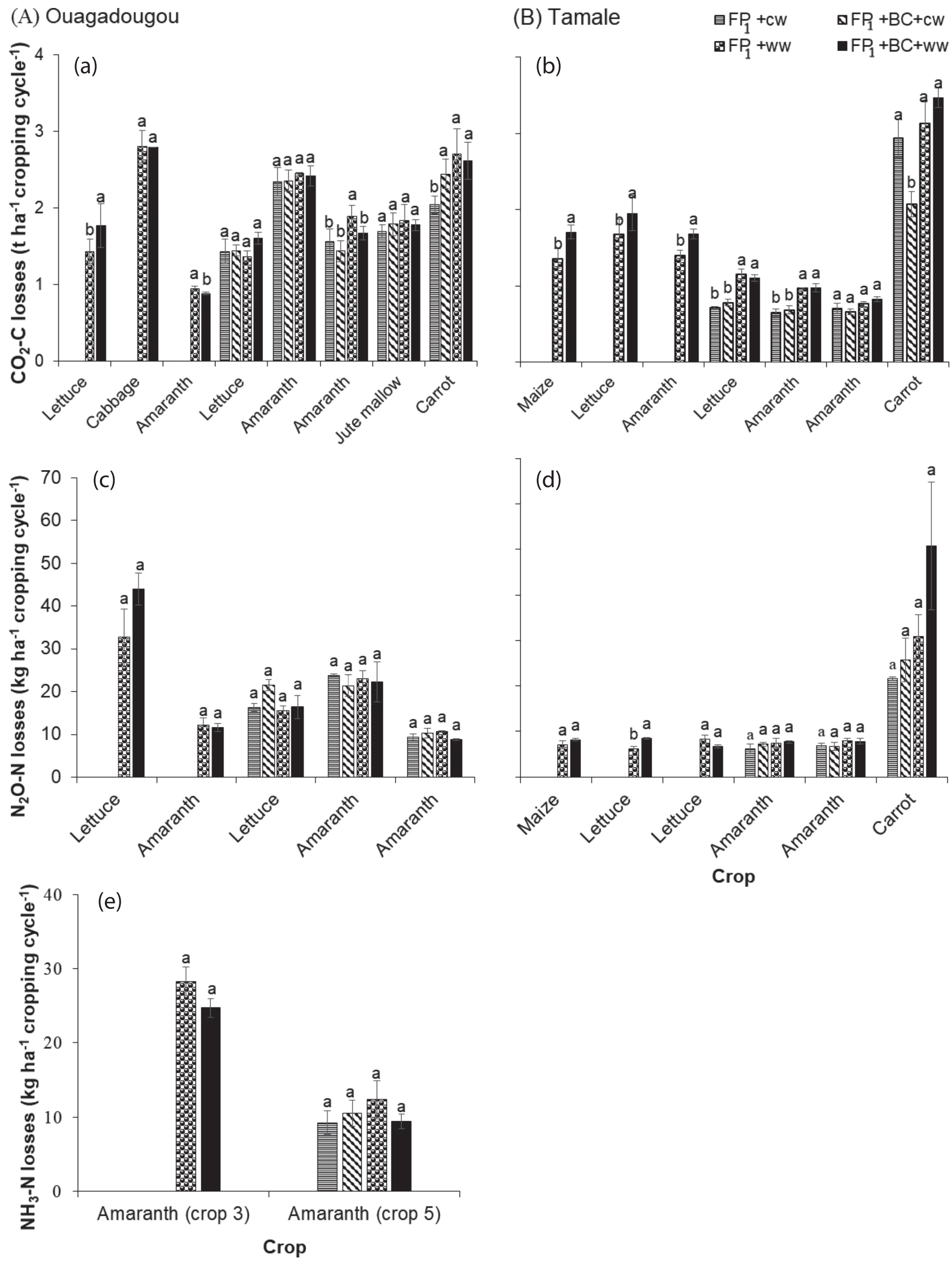

Figure 3: Effects of biochar on cumulative emissions of CO_2-C (a, b), N_2O-N (c, d), and NH_3-N (e) for different cropping cycles in urban crop production systems of (A) Ouagadougou (Burkina Faso) and (B) Tamale (Ghana). Data show means (n = 4) ± one standard error, obtained in a multi-factorial cropping experiment conducted at both locations from May 2014 to April 2016. Means with different letters are significantly different according to a Tukey HSD, $p < 0.05$. FP_1+cw = farmers' practice clean water, FP_1+ww = farmers' practice wastewater, FP_1+BC+cw = practice clean water, FP_1+BC+ww = practice wastewater.

losses from FP_1+ww plots amounted to 10 t C ha^{-1}, whereas FP_1+BC+ww reached 12 t C ha^{-1}. Biochar amendment did not affect N_2O-N emissions during different cropping cycles in Ouagadougou and inconsistent effects were observed in Tamale. Similarly, volatilization of NH_3-N did not differ significantly in Ouagadougou between biochar plots and plots without biochar (data not shown).

A plot of the relationship between soil carbon and CO_2-C emissions in Ouagadougou from three data points in time

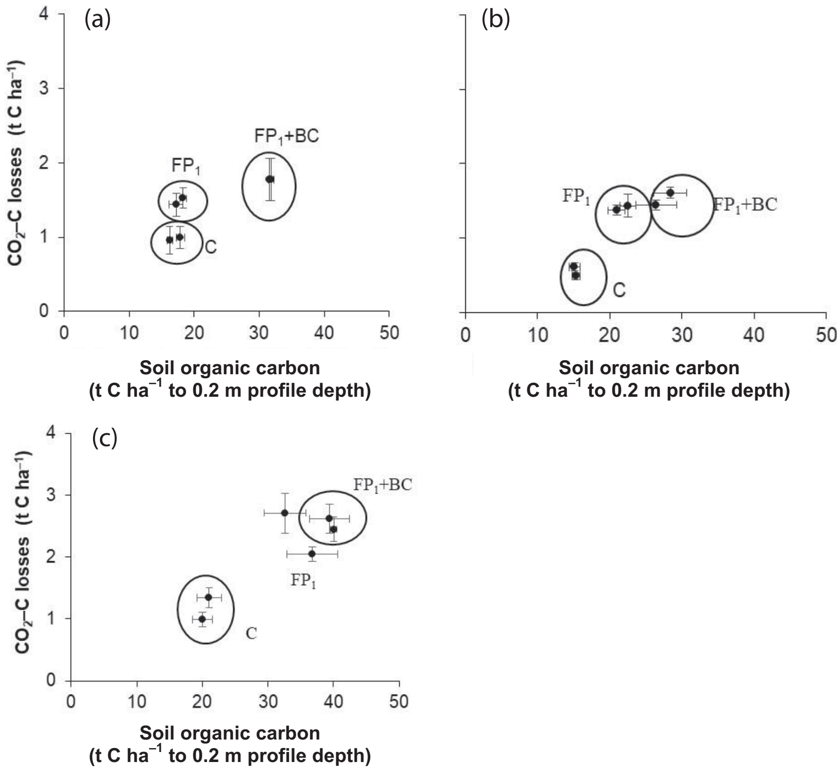

Figure 4: Relationship between soil carbon and CO_2-C losses in Ouagadougou (Burkina Faso) during (a) crop 1 (b) crop 4, and (c) crop 11. Data show mean (n = 4) ± one standard error, during a multi–factorial cropping experiment conducted from May 2014 to April 2016. Soil carbon was calculated per m^2 to a depth of 0.2 m. Circles drawn around treatments do not represent any statistical analysis but highlight the dynamics between treatments over time.

showed that the difference between farmers' practice of fertilization (FP_1) and farmers' practice *plus* biochar (FP_1+BC) decreased with time, whereas the difference between control (C) and FP_1 increased with time (Fig. 4).

3.6 Biochar, wastewater and fertilizer effects on yield-scaled CO_2-C emissions

Comparing crop production (C fixed in aboveground dry matter) to CO_2-C emissions revealed that application of fertilizer and biochar resulted in lower CO_2-C emissions per unit C produced than in the unamended control treatment. In Ouagadougou, CO_2-C emissions were significantly lower on FP_1+cw and FP_1+BC+cw plots compared with C+ww ($p < 0.05$). In Tamale, estimated ratios of CO_2-C emissions to fixed C showed that C+cw emitted significantly higher CO_2-C than all other treatments (Tab. 5).

4 Discussion

The intensive urban vegetable cultivation of Ouagadougou and Tamale with 11 and 13 cropping cycles in two years, respectively, was characterized by high application rates of agronomic inputs, continuous cropping, and year-round irrigation. However, the inputs differed in Ouagadougou from Tamale by input rates and the use of organic fertilizer (*Akoto-Danso* et al., 2018; *Manka'abusi* et al., 2019). This resulted in overall higher C and nutrient inputs from fertilizers in Ouagadougou than in Tamale. Furthermore, daily irrigation with diluted wastewater from open canals supplied only modest nutrient loads in Tamale, while irrigation with untreated sewage led to high nutrient inputs in Tamale, especially during the dry season (Tab. 2). These differences in soil management practices affected soil properties leading to a higher pH

Table 5: Average cumulative CO_2-C emitted and ratios of CO_2-C losses per captured-C during different cropping cycles in Ouagadougou (Burkina Faso) and Tamale (Ghana) during multiple cropping cycles from May 2014 to April 2016. Data show CO_2-C and fixed C for six and five cropping cycles in Ouagadougou and Tamale respectively, mean (n = 4) ± one standard error.[a]

		C+cw	C+ww	FP_1+ cw	FP_1+ ww	FP_1+BC+cw	FP_1+BC+ww
Ouagadougou (Burkina Faso)	Cumulative CO_2-C emitted (t ha^{-1})	4.0	5.0	10.0	11.2	10.4	11.4
	Cumulative fixed C (t ha^{-1})	3.1	3.5	10.0	9.5	10.6	10.3
	Ratio CO_2-C: cumulative C fixed (t CO_2-C t^{-1} fixed C)	1.3ab	1.4a	1.0b	1.2ab	1.0b	1.1ab
Tamale (Ghana)	Cumulative CO_2-C emitted (t ha^{-1})	4.2	7.8	6.4	7.4	5.2	8.1
	Cumulative fixed C (t ha^{-1})	1.0	3.3	2.7	4.0	3.1	4.4
	Ratio CO_2-C: cumulative C fixed (t CO_2-C t^{-1} fixed C)	4.3a	2.4b	2.4b	1.8b	1.7b	1.8b

in Ouagadougou than in Tamale (*Häring* et al., 2017) and consequently higher gaseous C and N emissions.

4.1 Flux rates of CO_2-C, N_2O-N, and NH_3-N

As expected, emission rates of C and N were higher in the afternoon than in the morning in Ouagadougou and Tamale, correlating with on average 10°C higher afternoon temperatures compared to morning temperatures. It has been well documented that soil temperature and moisture govern C and N mineralization rates in soils (*Sierra*, 1997; *Rey* et al., 2005), and temperatures are usually lower during the night. When moisture and other factors are not limiting, increases in temperature foster microbial metabolism as well as biochemical processes leading to higher microbial soil respiration, thus C and N decomposition (*Lloyd* and *Taylor*, 1994; *Allison* et al., 2010; *Rey* et al., 2005). The daily irrigation in the morning ensured high soil moisture contents and thus provided optimum conditions for microbial activity, triggering peak emissions observed in the afternoon. *González-Méndez* et al. (2015) further reported that repeated re-wetting of relatively dry soil favors microbial activities. While CO_2-C emission rates from control increased by on average 19% between morning and afternoon, on fertilized plots rates increased by 28% in Ouagadougou and by 74 and 81%, respectively, in Tamale. A similar trend could be observed for N_2O-N emission rates and NH_3-N volatilization rates in Ouagadougou.

The accuracy and precision of the deployed photo-acoustic infrared multi-gas analyser (INNOVA) is debatable. Although gas chromatography (GC) is acknowledged to be a more accurate and precise method, the photo-accoustic and spectroscopic equipment is gaining popularity and increasingly seen as an attractive alternative due to its portability, low maintenance and ease-of-operation (*Iqbal* et al., 2013). This equipment further has the advantage to measure multiple gases (CO_2, CH_4, N_2O, NH_3, and H_2O vapour). Comparisons on East African soils showed that INNOVA measurements of CO_2 were comparable to those of the GC (*Iqbal* et al., 2013; *Rosenstock* et al., 2013), whereas our measured CH_4 concentrations were strongly affected by water vapour and changes in temperature, and were thus excluded. Measurements of N_2O and NH_3 concentrations suffered from cross interferences with water vapour and CO_2 (*Flechard* et al., 2005; *Rosenstock* et al., 2013) under varying conditions present in the field. Therefore, although the INNOVA may be effective and efficient under well-defined conditions, results from field measurements need to be interpreted with caution. To be able to the compare treatments, care was taken to determine all comparable treatments within one day under similar environmental conditions. The measured absolute N_2O and NH_3 concentrations might have been underestimated due to cross interferences, and presented results are considered to be at the lower range of expected emissions. Due to the high temperatures and quickly increasing relative humidity within the closed chambers, a short accumulation period was chosen (see *Predotova* et al., 2010a, 2010b). Being aware that at times of low gas emissions only background noise might have been captured, all measurements that did not achieve fluxes calculated by linear regression with a p-value > 0.1 were excluded.

4.2 Fertilization effects on cumulative CO_2-C, N_2O-N and NH_3-N emissions

Irrespective of irrigation water quality, CO_2-C emissions significantly increased on fertilized plots during all cropping cycles in Ouagadougou, but were inconsistent in Tamale. Enhanced total CO_2-C losses on fertilized plots compared with unfertilized plots in Ouagadougou corroborate previous studies in fertilized grasslands of Northern China by *Xu* and *Wan* (2008). The availability of C and nutrients is paramount to microbial and plant respiratory processes, wherein organic and inorganic fertilizers play an important role (*Franzluebbers* et al., 2002). The combined use of manure and mineral N (urea) practiced in Ouagadougou, compared with only mineral (NPK) fertilization in Tamale, resulted in differences in the soils' C and N dynamics, and consequently different emission patterns in the two cities. Fertilization in Ouagadougou (mineral + organic) increased CO_2-C total emissions by 103%, while a 42% increase was observed in Tamale (mineral only). The enhancing role of organic matter (OM) input on soil C and consequently CO_2 emissions has been widely demonstrated (*Thangarajan* et al., 2013; *Liu* et al., 2014). The higher CO_2-C emissions in Ouagadougou correlated with higher input of 23 t C ha^{-1} supplied over two years *via* manure, whereas in Tamale no manure was added. This result is in accordance with studies of vegetable gardens in Kenya and Tanzania (*Rosenstock* et al., 2016) and sub-Saharan Africa (*Kim* et al., 2016), where higher emissions were attributed to C and N inputs *via* manure. However, this was in sharp contrast to lower emission rates reported from other agricultural systems like croplands and grazing lands (*Sugihara* et al., 2012; *Rosenstock* et al., 2016). Similar to this study, *Atakora* and *Kwakye* (2016) reported inconsistent results from Tamale, where major differences in CO_2 emissions from NPK fertilized plots were observed between seasons. They attributed this difference to a decrease in soil pH, an observation that was also made at our study site (*Häring* et al., 2017). Observed decreases in soil pH over time in Tamale probably affected the changes recorded in CO_2 emissions at this site, while higher soil pH in Ouagadougou than in Tamale added to the observed dynamics in CO_2 emissions across both cities. On the other hand, gaseous emissions have been suggested to not only depend on soil properties and climatic conditions but also on crop type and management (*Rosenstock* et al., 2016).

The increased CO_2-efflux during five of the seven measured cropping cycles in the fertilized plots in Tamale must therefore originate from either higher C inputs from increased crop growth (*Adviento-Borbe* et al., 2007) or from increased SOC mineralization due to the fertilizer-induced stimulation of soil microbial activity. Evidence for the latter process is found in *Häring* et al. (2017) who also observed a decline in SOC in Tamale. Nonetheless, because this reduction was also observed in control plots, they attributed the loss in SOC not only to a lack of organic fertilization and frequent tillage, but also to land use change. Particularly N addition stimulates microbial activity and subsequently soil respiration, primarily through enhancing plant growth, root activities, and consequently below ground C (*Lovell* and *Hatch*, 1997; *Xu* and *Wan*, 2008; *Peng* et al., 2011). Interestingly, in wastewater irrigated plots, the fertilizer effect was masked in all cropping cycles, which may be explained by the considerable input of N and likely labile C fractions with the untreated sewage. Introducing the application of organic fertilization in Tamale as a soil management practice may help to increase SOC at this site. However, *Barbarick* et al. (2004) reported an increase in microbial community and enzyme activity after the application of manure and *Lompo* et al. (2012) attributed large CO_2-C effluxes in an urban garden of Bobo Dioulasso (Burkina Faso) to rapid mineralization of organic matter induced by high amounts of combined C and N applied under daily irrigation and frequent tillage.

Cumulative N (N_2O-N and NH_3-N) losses from fertilized plots in Ouagadougou were 2.6-fold higher than in the control plots and represented on average 8% of the N applied, which was in a similar range as the 11% reported by *Predotova* et al. (2010b) from an urban garden in Niamey, Niger. In Tamale, cumulative N_2O-N emissions did not respond to fertilizer application, whereas relative to N application N_2O-N losses were reduced at higher N input levels. *Kim* et al. (2016) further suggested that N inputs over 150 N kg ha^{-1} y^{-1} may cause an increase in N_2O emissions on sub-Saharan African soils. Soil N emissions depend largely on soil pH, whereby NH_3 volatilization especially occurs at high pH (*van der Weerden* and *Jarvis*, 1997; *Laubach* et al., 2015). Continued application of mineral fertilizers alone has already caused soil acidification in Tamale (*Häring* et al., 2017), explaining low low NH_3 volatilization (*Oertel* et al., 2016) and most likely also affecting microbial denitrification (*Senbayram* et al., 2012).

Emissions from fertilized plots were generally higher in Ouagadougou than in Tamale. The form of N fertilizer used could also be a contributing factor to differences in emissions observed between Ouagadougou and Tamale (*Eichner*, 1990; *Bouwman*, 1996). Higher NH_3-N emissions were reported from applied urea N compared to those of non-urea based fertilizers like ammonium nitrate (*van der Weerden* and *Jarvis*, 1997; *Pan* et al., 2016). This is as a result of rapid hydrolysis of urea to NH_4^+ in substrates causing a rise in pH (*Laubach* et al., 2015), and also from NH_4^+ to NH_3 as pH increases due to higher demand for protons (*Rochette* et al., 2009). This was probably a reason for the NH_3-N volatilization observed only from cropping cycles with high urea application rates in Ouagadougou. Also, the soil CEC has been reported as an important factor determining the magnitude of NH_3 emission from N fertilizers applied to agricultural soil (*Kim* et al., 2012). While the CEC of fertilized soil increased with time in Ouagadougou, that of Tamale remained unchanged (*Häring* et al., 2017).

4.3 Wastewater effects on cumulative CO_2-C, N_2O-N and NH_3-N emissions

Wastewater irrigation significantly increased CO_2-C emissions in Tamale but not in Ouagadougou. The 71% increase in CO_2-C emissions from wastewater irrigated plots in Tamale were in conformity with *Häring* et al. (2017) who reported that wastewater had no effect on SOC, attributing it to C mineralization. These results were also consistent with those of *Xue* et al. (2012) who reported an increase in CO_2 emissions from soils irrigated with treated wastewater compared to those irrigated with clean water. Although in contrast to the results of *Häring* et al. (2017) for this soil, it has been well documented that wastewater irrigation can substantially increase SOC (*Singh* et al., 2012; *Bedbabis* et al., 2014; *Aydin* et al., 2015). Enhanced microbial respiration was observed by *Meli* et al. (2002), after long-term irrigation with wastewater. Therefore, we suggest that the application of sewage wastewater rich in particulate and dissolved organic carbon (DOC) enhanced microbial activity, thereby causing higher CO_2-C emissions. Contrary to Tamale, the use of canal wastewater in Ouagadougou had no significant effects on annual CO_2-C emissions, which was likely due to the low concentration of organic C in the comparatively clean canal water compared to typical sewage (*Singh* et al., 2012). Wastewater effects were thus more pronounced in Tamale due to the wastewater's higher organic carbon load. Another contributing factor could also have been the fact that the soil in Tamale is petroplinthic with much higher concentrations of iron oxides in the soil. This may have enhanced carbon sequestration making it partly inaccessible to microbial mineralization.

In both cities, fluxes of N_2O-N and NH_3-N in Ouagadougou were not significantly affected by water quality in contrast to other reports (*Neeteson* and *Carton*, 2001). This effect was expected in Ouagadougou, considering the lower N inputs from the canal wastewater used. However, in Tamale, *Häring* et al. (2017) found that despite high N inputs from wastewater and corresponding biomass return to the soil, total N stocks did not increase. We assume that one possible reason could be that the amount of N supplied by wastewater in Tamale was just enough to satisfy crop needs. Furthermore, higher water leaching observed from wastewater irrigated plots reported by *Werner* et al. (2019) was accompanied by higher N losses. The magnitude of gaseous emissions depends on complex interaction between soil processes, climate factors, and agricultural practice. Soil pH governs important components of these emissions, and as it was low in our study soil could consequently have limited emissions. Additionally, *Zhang* et al. (2014) reported that N_2O emission can be mitigated when phosphorus (P) is simultaneously added to soil which results in increased N uptake by crops. This possibly was also the case in Tamale considering the high P concentration of the wastewater there.

4.4 Biochar effects on cumulative CO_2C, N_2O-N and NH_3-N emissions

The addition of biochar to farmers' practice increased CO_2-C emissions by 21% for about half a year following application in Tamale. In Ouagadougou an increase in CO_2-C emission by 24% was only observed during the first cropping cycle (lettuce). Higher initial CO_2 effluxes after biochar addition have been demonstrated by *Kuzyakov* et al. (2009), showing that biochar is not completely biologically inert (*Ameloot* et al., 2013; *Farrell* et al., 2013). Quick decomposition of biochar partly depends on the presence of more labile organic substrate (*Das* et al., 2008; *Nguyen* and *Lehmann*, 2009; *Ameloot* et al., 2013). Corn cob biochar applied in Ouagadougou had 20% volatile organic matter, while rice husk biochar in Tamale contained 23%. With the presence of such easily degradable labile C, initial mineralization of biochar was likely to occur (*Kuzyakov* et al., 2009). *Wang* et al. (2016) also observed a higher decomposition rate of biochar in studies lasting for less than half a year compared with longer term studies. A bi-phasic mineralization rate has therefore been suggested (*Ameloot* et al., 2013; *Wang* et al., 2016). In line, *Major* et al. (2010) recorded an increase in net CO_2 emissions during the first year of their experiment, when biochar was added to a Savanna soil. Apart from the mineralization of labile C, *Major* et al. (2010) attributed this observation to mineralization of larger biomass production and greater root respiration. This also offers a partial explanation for the results of this study since higher CO_2-C emissions in Tamale coincided with higher biomass production on biochar amended plots (*Akoto-Danso* et al., 2018).

After seven cropping cycles, total CO_2-C emissions in Ouagadougou did not differ between farmers' practice (FP_1) and farmers' practice *plus* biochar (FP_1+BC) plots, while in Tamale, CO_2-C emissions were 12% higher on FP_1+BC plots irrigated with wastewater compared to FP_1+ww plots. Similar to the results in Ouagadougou, *Bamminger* et al. (2017) found that biochar did not influence CO_2 emissions in an arable field, while results in Tamale were similar to those of a meta-analysis reported by *Sagrilo* et al. (2015) with an average increase in CO_2 of 28% when biochar was added to soils.

The temporal pattern of emissions during the different cropping cycles in the two cities gave insights on how biochar affected the dynamics of C in intensive vegetable systems of SSA. In Ouagadougou, the absence of emission difference between plots with and without biochar indicated a rapid depletion of the initial labile carbon, followed by a slow mineralization of the more recalcitrant C. The relationship between soil C and CO_2-C emissions (Fig. 4) further demonstrated the stability of corn cob biochar in this study. While the difference in CO_2-C emissions from FP_1 and FP_1+BC decreased, the difference between those from C and FP_1 rather increased over time. Although SOC increased after biochar application (*Häring* et al., 2017), it did not create a corresponding increase in CO_2 emissions as manure amendments did. Instead a small increase in SOC from manure additions led to a very high increase in CO_2-C emissions. According to *Kuzyakov* et al. (2009), the contribution of biochar to CO_2 emissions is rather small compared to mineralization of soil organic matter and other plant residues.

Although more C was added with corn cobs than with rice husks biochar, the latter increased C emissions more than corn cob biochar. The decomposition of biochar was found to vary significantly with feedstock (*Wang* et al., 2016). Biochar can improve the conditions for mineralization but only if there is enough labile organic carbon (*Kuzyakov* et al., 2009; *Maestrini* et al., 2015). Thus, the labile C in the wastewater in Tamale possibly contributed to higher emissions.

Biochar can increase the soil pH and thus influence N emissions. However, in this study, the application of biochar did not have an effect on soil pH (*Häring* et al., 2017), nor on total N_2O-N and NH_3-N losses across cities. Despite studies indicating suppression of N_2O emissions (*Cayuela* et al., 2014) this was not the case in our study. In line, *Wang* et al. (2015) and *Suddick* and *Six* (2013) found that biochar application did not affect N_2O-N emissions in intensive, small scale rotation vegetable production systems. Increased porosity through decreased bulk density and consequently enhanced soil aeration has been identified as a possible mechanism for reduced emissions (*van Zwieten* et al., 2010; *Case* et al., 2012). Soil pH has been widely recognized as a parameter affecting N_2O production and consumption, whereby an increase can alter the N_2O/N_2 ratio of nitrification and denitrification in agricultural soils and enhance the subsequent reduction of N_2O to N_2 (*Cayuela* et al., 2014). But a mere change in pH does not by itself induce N_2O emissions but rather other properties of biochar intrinsically connected to pH (*Cayuela* et al., 2013). The addition of biochar to the study soil had no effect on pH and bulk density which possibly translated to the lacking response of N_2O-N emissions to biochar amendment.

The data showed that wastewater irrigation in Tamale improved agronomic efficiency (*Akoto-Danso* et al., 2018) thereby reducing N emissions per unit yield. These results were similar to those of *Mapanda* et al. (2011) who reported relatively lower N_2O emissions per unit of produce when fertilizer (NH_4NO_3) was applied to maize compared with the control treatment. An increase in CO_2-C emissions from control plots in this study was only apparent when linked to productivity and not when total CO_2-C emissions were calculated.

5 Conclusions

Estimates of C and N fluxes from intensively managed West African UPA soils are needed to improve our knowledge on C and N flows within these systems. Although fertilizer application and wastewater irrigation have been recognized for improving soil fertility, an understanding of their contribution to carbon and N use efficiency in UPA systems is of particular importance. Results of this study confirmed that high rates of fertilizer application and wastewater irrigation lead to C and N losses in urban gardens. However, the magnitude of loss depended on quantity and type of fertilizer and the nutrient composition of wastewater used. Emissions of CO_2-C, NH_3-N, and N_2O-N were considerably increased by the application of manure and high rates of urea in Ouagadougou. In Tamale, in contrast, CO_2-C losses were predominantly affected by the use of sewage wastewater with high organic loads. N_2O-N losses were unaffected by fertilization and irrigation practices due to soil acidification and a lower N application rate.

Adding biochar did not affect NH_3-N and N_2O-N losses in Ouagadougou and Tamale. Nonetheless, our results disclosed that the application of biochar can improve carbon storage in urban garden soils reducing possible losses, despite an initial increase in CO_2-C emissions due to mineralization of the labile fraction of biochar-C. The difference in CO_2-C emissions from FP_1 and FP_1+BC decreased, while the difference between those from C and FP_1 increased over time. This underlines the potential of biochar to sequester C in these intensively managed systems and thus reducing CO_2 emissions from soils of sub-Saharan West Africa. Although high rates of fertilizer application plus wastewater irrigation increased CO_2 emissions per unit of cultivated land, higher biomass production from these inputs can cause a shift in the C balance which leads to a reduction in yield-based emissions.

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