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Long-term tillage, residue management and crop rotation impacts on N_2O and CH_4 emissions from two contrasting soils in sub-humid Zimbabwe

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ABSTRACT

The respective contribution of conservation agriculture (CA) principles (no-tillage, permanent soil cover/mulch and crop rotations) on greenhouse gas (GHG) emissions is still unclear. This study was conducted at two long-term experimental sites established in 2013 in Zimbabwe, on an abruptic Lixisol at Domboshava Training Center (DTC) and on a xanthic Ferralsol at the University of Zimbabwe Farm (UZF). The purpose of the study was to unravel the individual and combined effects of tillage, mulching and rotation on N_2O and CH_4 emissions in low nitrogen (N) input maize-based cropping systems (< 60 kg N ha^{-1}) and to compare emissions within maize rows and between maize rows. We hypothesised that integrating no tillage, mulch and cereal-legume rotation would enhance N_2O emissions. Six treatments, replicated four times were investigated: conventional tillage, conventional tillage with rotation, no-tillage, no-tillage with mulch, no-tillage with rotation, no-tillage with mulch and rotation. The main crop was maize (*Zea mays* L.) and treatments with rotation included cowpea (*Vigna unguiculate* L. Walp.). Gas samples were regularly collected using the static chamber method in the maize row and inter-row spaces during the 2019/20 and 2020/21 cropping seasons and during the 2020/21 dry season. Soil moisture and mineral N were measured in the 0–20 cm soil depth. In 2019/20, cumulative total N_2O emissions were significantly higher in mulch treatments at DTC, while at UZF N_2O emissions were higher with cowpea rotation. Cumulative total N_2O emissions ranged from 215 to 496 g N_2O-N ha^{-1} yr^{-1} and from 226 to 395 g N_2O-N ha^{-1} yr^{-1}, at DTC and UZF, respectively. In 2020/21, N_2O emissions were much lower and no differences were found between treatments on both sites (145 to 179 g N_2O-N ha^{-1} yr^{-1} and 83 to 136 g N_2O-N ha^{-1} yr^{-1} at DTC and UZF, respectively). A significant relationship was found between soil nitrate and daily N_2O emissions. At UZF, highest N_2O emissions were observed at a water-filled pore space of 60–70%. There were no significant differences in yield-scaled N_2O emissions between treatments at both sites for the two seasons. DTC was a net source of CH_4 (694 g CH_4-C ha^{-1} yr^{-1} on average), while UZF was a net sink of CH_4 (−494 g CH_4-C ha^{-1} yr^{-1} on average). No evidence was found for in situ CH_4 production at DTC, and an external source is most likely. Our study indicates that for low N input cropping systems in the sub-humid tropics, N loss through N_2O is low.

1. Introduction

The Agriculture, Forestry and Other Land Use (AFOLU) sector accounts for about 13% of anthropogenic emissions of carbon dioxide (CO_2), 44% of methane (CH_4), and 81% of nitrous oxide (N_2O), representing 23% of the total net anthropogenic greenhouse gas (GHG) emissions, equivalent to 12.0 ± 2.9 Gt CO_2 eq yr^{-1} (IPCC, 2019; Le Quéré et al., 2017). Global human-induced N_2O emissions increased by 30% in the past four decades, mainly due to mineral nitrogen (N) fertilization (Tian et al., 2020). The current climate emergency has

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resulted in a growing public awareness of the detrimental effects of modern agriculture on GHG emissions, bringing about the demand for more sustainable land management (Bellarby et al., 2014). The contribution of the AFOLU sector to climate change mitigation is potentially significant. However, this can only be realised through adoption of the best land management practices and stopping deforestation or conversion of carbon-rich ecosystems such as wetlands, peatlands or mangroves (IPCC, 2019). Management of agricultural soils is an important, albeit relatively small, part of this potential (Paustian et al., 2016).

Several agricultural practices have been shown to influence soil GHG emissions, especially N_2O (Guenet et al., 2021). For instance, addition of fertilizer N (Han et al., 2017), straw residue retention (Xia et al., 2018) or application of manure (Zhou et al., 2017) were found to increase N_2O emissions, while introduction of non-leguminous cover crops (Muhammad et al., 2019) or addition of biochar (Borchard et al., 2019; Verhoeven et al., 2017) can potentially lead to decreased N_2O emissions, though magnitudes are highly variable. No-tillage (NT) or reduced tillage also usually lead to higher N_2O emissions (Huang et al., 2018; Mei et al., 2018), but results vary depending on climate (Huang et al., 2018; van Kessel et al., 2013), soil texture (Huang et al., 2018; Pelster et al., 2021) and time since implementation of NT (Six et al., 2004; van Kessel et al., 2013; Cusser et al., 2020). On the other hand, it is not clear what the effect is of NT on CH_4 emissions in upland soils (Maucieri et al., 2021; Zhao et al., 2016). Studies on GHG emissions in sub-Saharan Africa (SSA) are scarce (Huang et al., 2018; Kimaro et al., 2016) and tend to focus on the effects of land use change (Chikowo et al., 2004; Mapanda et al., 2012a; Nyamadzawo et al., 2012a, 2017) and integrated soil fertility management (combined use of organic and mineral fertilizers) (Mapanda et al., 2011; Mapanda et al., 2012b; Nyamadzawo et al., 2014, 2017; Hickman et al., 2015).

Conservation agriculture (CA) combines minimum soil disturbance, permanent soil cover with living or dead mulch, and improved crop rotations or intercropping (Farooq and Siddique, 2015; Rodenburg et al., 2020) on top of general good agriculture practices (Thierfelder et al., 2018). In SSA, a meta-analysis showed that CA has significant but limited effects on crop yields (Corbeels et al., 2020a), especially in the short term (Thierfelder et al., 2015). CA also has a positive but limited effect on soil organic carbon (SOC) stocks in SSA (Corbeels et al., 2020b), especially when the three principles of CA are applied (Corbeels et al., 2019), although larger effects are expected when diversified cropping systems are included (Powlson et al., 2016). However, CA effects on GHG emissions in SSA are largely unknown and only a few studies have focussed on it (Kimaro et al., 2016; O'Dell et al., 2015). There is wide promotion of CA in SSA (Prestele et al., 2018; Kassam et al., 2019) but studies on its environmental impacts are largely limited to investigating effects on soil quality (Naab et al., 2017; Nyamangara et al., 2020; Powlson et al., 2016; Thierfelder and Wall, 2010). Some earlier studies have shown significant reduction of N_2O and CH_4 emissions through the practice of CA in Europe (Holland, 2004; Tellez-Rio et al., 2017; Kassam et al., 2019), South Asia (Anon, 2017) and North and South America (Siqueira-Neto et al., 2021). Contrasting effects of CA on GHG emissions are probably related to factors like rainfall regime and soil type (Bayer et al., 2015; Huang et al., 2018). Therefore, more research is needed to ascertain if the benefits of CA such as improved soil quality (Nyamangara et al., 2020; Thierfelder and Wall, 2012) outweigh possible negative environmental impacts.

Soil CH_4 fluxes are a result of two sequential and antagonistic processes, namely methanogenesis (CH_4 production) and methanotrophy (CH_4 oxidation) which respectively take place under anaerobic and aerobic conditions (Morel et al., 2019; Tate, 2015; Topp and Pattey, 1997). In dryland and tropical soils, $>$ 64% of CH_4 produced in the rhizosphere is microbially oxidised before emitted (Dutaur and Verchot, 2007; Nauer et al., 2018). On the other hand, N_2O is produced through nitrification and denitrification processes under aerobic and anaerobic conditions, respectively (Butterbach-Bahl et al., 2013; Hatfield, 2017; Ji et al., 2016). CH_4 and N_2O fluxes are highly regulated by soil moisture (Butterbach-Bahl et al., 2013; Davidson et al., 2000; Tate, 2015) where reduced tillage and mulching are key factors in soil moisture retention. These two CA principles often result in higher surface soil organic carbon (SOC) and soil moisture (Zhang et al., 2015; Kodzwa et al., 2020), thus increasing the potential of methanogenesis due to availability of organic substrate (Butterbach-Bahl et al., 2013; Butterbach-Bahl and Dannenmann, 2011) and moisture (Birch and Friend, 1956; Borken and Matzner, 2009; Huang et al., 2018). Similarly, reduced tillage has been shown to increase N_2O emissions due to denitrification because of increased soil moisture which decreases soil aeration (Charles et al., 2017; Giles et al., 2012; Huang et al., 2018; Ma et al., 2013). The third CA principle, crop rotations, has been shown to decrease GHG emissions in a legume-cereal rotation due to decreased fertilizer N input when residual soil N from biological nitrogen fixation (BNF) is considered in the succeeding crop (Barton et al., 2014; Schwenke et al., 2015). However, soils under legumes can emit N_2O, mainly through N release from root exudates during the growing season and from decomposition of crop residues after harvest and belowground biomass (Tellez-Rio et al., 2015; Wichern et al., 2008).

The potential of individual or combined CA principles to mitigate GHG emissions can be informed by component omission studies. Therefore, this study seeks to unravel the effects of the different CA principles (no-tillage, crop residue mulching, maize (*Zea mays* L.)-cowpea (*Vigna unguiculate* L. Walp.) rotation) on soil GHG emissions compared to conventional agriculture (CT) practices based on tillage and monoculture of maize in sub-humid Zimbabwe on two contrasting soils. As fertilization is applied on maize rows and not broadcasted, we also compared GHG emissions between and within maize rows.

2. Materials and methods

2.1. Study sites

The study was conducted during the 2019/20 and 2020/21 cropping seasons at two long-term experimental sites established in November 2013 by CIMMYT (Mhlanga et al., 2022). The site at the University of Zimbabwe Farm (UZF) is located about 12 km north of Harare (31° 00′ 48″ E; 17° 42′ 24″ S), whilst the site at the Domboshava Training Centre (DTC) is located about 30 km north-east of Harare (31° 07′ 33″ E; 17° 35′ 17″ S). UZF soils are dolerite derived xanthic *Ferralsols* (FAO classification) and are medium textured sandy clay loams (34% clay) in the top 20 cm with a subsoil (20–40 cm) of slightly higher clay content (38%). DTC soils are granitic derived abruptic *Lixisols* (FAO classification) and are light textured sandy loams (15% clay) in the 0–20 cm layer, overlying abruptly a heavier-textured subsoil (20–40 cm) of 30% clay content. The two study sites are under a sub-tropical climate with cool-dry winters and hot-wet summers with mean annual minimum and maximum temperatures of 12°C and 25°C, respectively (Mapanda et al., 2010). The rainy season starts in November and tails off in March with a mean annual rainfall of 826 and 814 mm at UZF and DTC, respectively (Mhlanga et al., 2022).

2.2. Experimental treatments

Two identical experiments were set up at the study sites and experimental treatments were maintained every season since November 2013. The experiment can be classified as a component omission trial with eight treatments replicated four times. For this study, we only considered six treatments as described in Table 1 and a total of 24 experimental plots at each site were used. The main crop was maize (*Zea mays* L.), and cowpea (*Vigna unguiculata* L. Walp.) was rotated with maize in treatments that included rotation. The experimental plots were 6 m wide and 12 m long. All treatments with rotation were split into 6 m x 6 m subplots where maize and cowpea were grown interchangeably every season.

Maize was sown at an inter-row spacing of 90 cm and an in-row

Table 1
Treatment description and management of the long-term experiments at the University of Zimbabwe Farm (UZF) and the Domboshava Training Centre (DTC) in Zimbabwe.

Treatment name and abbreviation	Treatment description and management
Conventional tillage (CT)	Land preparation was done through digging with a hand hoe and maize was sown as a sole crop in riplines that were created afterwards using an animal-drawn Magoye ripper (a traditional plough with the mouldboard replaced by a ripper tine) at DTC and in planting basins (approximately 15 cm in diameter and 15 cm deep) created using a hand hoe at UZF. All crop residues were removed after harvesting.
Conventional tillage with rotation (CTR)	Land preparation was done as in the CT treatment and maize was rotated with cowpea. All crop residues were removed after harvesting.
No-tillage (NT)	Maize was sown in riplines created using an animal-drawn Magoye ripper (no further soil disturbance was done) at DTC and in planting holes (approximately 10 cm deep) created using a hand hoe at UZ. All crop residues were removed after harvesting.
No-tillage with mulch (NTM)	Maize was sown as in the NT treatment and maize residues were retained on the soil surface at planting at a rate of 2.5 t DM ha^{-1} every year.
No-tillage with rotation (NTR)	Maize was sown as in the NT treatment and rotated with cowpea. All crop residues were removed after harvesting.
No-tillage with mulch and rotation (NTMR)	Maize was sown as in the NT treatment, rotated with cowpea and maize residues were applied as in NTM treatment.

spacing of 25 cm whilst cowpea at an inter-row spacing of 45 cm and an in-row spacing of 25 cm to achieve plant populations of 44,444 and 88,888 plants ha^{-1}, respectively. Three seeds per plant station were planted and thinned to one after emergence. Nitrogen, phosphorus (P) and potassium (K) were spot applied within 5 cm of the seed at planting in the form of compound fertilizer for both maize and cowpea at 11.6 kg N ha^{-1}, 10.6 kg P ha^{-1} and 9.6 kg K ha^{-1}, respectively. N top dressing was applied within 5 cm of the maize stems in two equal splits at 4 and 8 weeks after emergence (WAE) when soil moisture was adequate, but no N top dressing was applied to cowpea. However, due to mid-season dry spells in the 2019/20 cropping season at both sites and in the 2020/21 cropping season at UZF, top dressing was delayed (Table 2). Ammonium nitrate was applied at 23.1 kg N ha^{-1} for each of the top dressings and placed within 5 cm radius from the maize stems. Immediately after sowing, weeds were controlled by spraying glyphosate [N-(phosphonomethyl) glycine], as a pre-emergent non-selective herbicide, at the rate of 1.025 L active ingredient ha^{-1}. This was followed by manual hoe weeding whenever weeds were 10 cm tall or 10 cm in diameter for stoloniferous weeds.

2.3. Gas sampling, analyses, and flux determinations

We used static chambers, a common approach to measure GHG emissions from agricultural soils (Abalos et al., 2013; Rochette and Eriksen-Hamel, 2008) to determine N_2O and CH_4 emissions from the 24 experimental plots at both sites. The static chamber method is based on trapping gases emitted from the soil by a chamber and collecting gas samples from the chamber headspace at regular intervals for analysis by gas chromatography. GHG fluxes are then calculated by measuring the change in gas concentration over time (Collier et al., 2014).

Base rings made of PVC (height of 0.1 m and inside radius of 0.1 m) were permanently driven 0.07 m into the soil at planting to avoid possible gas leakage or contamination by lateral diffusion (Abalos et al., 2013; Clough et al., 2020). At both sites and in each of the four replicates of the six treatments, two base rings were installed at different locations, one within maize rows (“row”) and one between two maize rows (“inter-row”). The row and inter-row chambers were centrally positioned between maize plants and maize rows such that they left a distance of about 2.5 cm and 35 cm from the maize plants and maize rows, respectively. In treatments with rotation (CTR, NTR, NTMR), the chambers were installed in the sub-plot cropped with maize, no chamber was installed in the cowpea crop. GHG emissions in these treatments therefore reflect the preceding effect of cowpea. A total of 96 static chambers was therefore used for this study, 48 per site. Cylindrical PVC lids (height of 0.2 m) with an air tight and self-sealing rubber septum on the top centre to enable gas sampling using a syringe were designed in a manner that 0.02 m of the lid can be inserted into the base rings before gas sampling. The contact area between the base rings and the lids was always smeared with a thin layer of petroleum jelly to avoid possible leakage of trapped gas. Both the base ring and the lids were painted white to avoid excessive heating. Each chamber covered an area of 0.0314 m^2 and had a headspace volume of 0.006 m^3.

At each site, the row and inter-row chambers per replicate were closed simultaneously by two persons. The gases were collected into pre-evacuated 12 mL Exetainer glass vials (Labco Ltd., Lampeter SA48, United Kingdom) using a graduated 20 mL syringe, immediately after securing the chamber and after 48 min of gas trapping. Gas diffusion theory predicts that an increase in gas concentration during chamber deployment has a negative impact on emission rate at soil surface (Rochette et al., 2008; Venterea et al., 2020). We therefore carried out a linearity test at the two sites by collecting gas at times 0, 15, 30, 48 and 60 min after securing the gas trapping chamber. Results showed that N_2O and CH_4 emissions increased linearly with time, even at 60 min, suggesting that two gas samplings at 0 and 48 min were appropriate for this study as emissions were linear and no saturation was observed (data not shown). On each sampling day, gas sampling was done between 10

Table 2
Planting, fertilizer application and harvesting dates in the long-term experiments at the University of Zimbabwe Farm (UZF) and the Domboshava Training Centre (DTC) in Zimbabwe.

Cropping season	Activity	UZF	DTC
2019/20	Planting; basal fertiliser and glyphosate application	19/11/2019	21/11/2019
	First ammonium nitrate top dressing	06/01/2020	09/01/2020
	Second ammonium nitrate top dressing	20/02/2020	31/01/2020
	Harvesting	14/04/2020	15/04/2020
2020/21	Planting; basal fertiliser and glyphosate application	25/11/2020	25/11/2020
	First ammonium nitrate top dressing	07/01/2021	13/12/2020
	Second ammonium nitrate top dressing	02/02/2021	18/01/2021
	Harvesting	15/04/2021	25/04/2021

am and 12 pm.

Gas sampling was carried out from 26 November 2019–24 March 2020 in the 2019/20 cropping season and from 28 November 2020–23 April 2021 in the 2020/21 cropping season. In 2020/21, gas sampling was also extended into the dry season, from May to September 2021. Sampling for gas was done at least every two weeks during the cropping season, with additional sampling following key events (i.e. after fertilizer applications and major rainfall events). This resulted in nine gas sampling events in the 2019/20 cropping season at both sites, and respectively, 13 and 12 sampling events at DTC and UZF in the 2020/21 cropping season, with 3 and 4 more sampling events after maize harvest during the dry season at DTC and UZF, respectively.

Vials with gas were kept in the dark for at most 3 weeks before shipping to ETH Zürich in Switzerland for analysis. Storage and shipment were assumed to result in negligible changes in gas concentration of the samples based on previous experiences with sealed Exetainer glass vials (Mapanda et al., 2011; Kennedy et al., 2013; Barthel et al., 2022). N_2O and CH_4 were quantified by gas chromatography using the electron capture and flame ionisation detectors, respectively. Gas fluxes were calculated as the difference in concentrations between the 0 and 48-minutes sampling times using the following equation:

$$F = \frac{(GC_f - GC_o) \times V}{T \times A} \quad (1)$$

where F is the gas flux (μg N_2O or μg CH_4 m^{-2} hr^{-1}), GC_f and GC_o are the gas concentrations (ppm) at end and start of chamber closure, V is the chamber volume (mL), T is the duration of the chamber closure (hours) and A is the surface area covered by the static chamber (m^2).

2.4. Cumulative greenhouse gas emissions

Cumulative N_2O and CH_4 emissions were determined using linear interpolation between sampling points by multiplying the mean flux of two successive sampling dates by the length of the period between sampling and adding that amount to the previous cumulative total (Dorich et al., 2020).

As fertilizers were only applied on maize rows, it is important to have chambers both in rows and in inter-rows. With the fertilizers being applied within 5 cm radius from the maize stems, we assumed that the chambers (0.2 m in diameter) in the maize rows were representative of the rows only. On the other hand, the inter-row static chambers were assumed to be representative of the remaining 0.7 m where there were no maize plants growing and no fertilizer applied. Cumulative emissions per chamber position (row or inter-row) were then weighted according to these proportions, i.e. 22% and 78% contribution from row and inter-row emissions, respectively (Fig. S1), to get the total cumulative emissions per treatment and per hectare.

2.5. Maize yields and yield-scaled N_2O emissions

Net maize plots measuring 5 m x 3.6 m were harvested every season at physiological maturity (Table 2) which was approximately 4 months from emergence. Maize stover and cobs were weighed separately, recorded and subsamples taken for moisture determination by oven drying at 60°C for 72 h. After shelling maize grain from the cobs, a grain moisture meter was used to measure grain moisture content and adjust the fresh weight to 12.5% standard maize grain moisture. Yield-scaled N_2O emissions were determined as N_2O-N emissions per unit of grain yield (g N_2O-N per kg of grain) to quantitatively assess the potential trade-offs between N_2O emissions and crop yield (Abalos et al., 2016).

2.6. Soil sampling and analysis

Inter-row and row soil samples were taken from the respective replicates to 0–20 cm depth using a bucket auger at each gas sampling event. Soil samples were used to determine soil moisture content, soil ammonium (NH_4^+-N) and soil nitrate (NO_3^--N) concentrations. In the field, each soil sample was sieved (< 2 mm) to discard gravel and stones (> 2 mm) and to disaggregate the soil. Five grams of the sieved soils was immediately put into 250 mL plastic bottles containing 50 mL of 2 M KCl. The mixture was then hand-shaken for approximately 10 s to stop any soil microbial activity and to extract mineral N (NH_4^+-N and NO_3^--N). The bottles and the soil samples were put in cooler boxes with ice for transportation. In the laboratory, gravimetric moisture content was determined as soil weight loss upon oven drying the soil samples for 48 h at 105°C. The soil samples in the KCl solution were further shaken mechanically for an hour and filtered through Whatman No.1 filter paper (Cytiva, Marlborough, United States). A 10 mL aliquot of the filtrate was steam distilled in MgO during which NH_4^+-N was trapped in boric acid with bromocresol-methyl red indicator solution. The distillate (50 mL) was titrated with 0.005 M H_2SO_4 in a micro-burette. Nitrate-N was determined in the same sample by adding Devarda's alloy to reduce NO_3^--N to NH_4^+-N and distilling again into fresh boric acid and then titrating with 0.005 M H_2SO_4 (Okalebo et al., 1993). Soil bulk density (BD) was measured at 0–5, 5–10, 10–15 and 15–20 cm soil depth using cylinders (5 cm height, 5 cm radius, volume = 25.7 cm^3) in May 2021. BD at 0–20 cm was estimated as a weighted average of BD measured at the four depths and was used to calculate water-filled pore space (WFPS) at 0–20 cm for each day of gas sampling using the following equation:

$$WFPS \quad (\%) = \frac{\theta_g \times BD}{\left(1 - \frac{BD}{PD}\right)} \quad (2)$$

where θ_g is the gravimetric water content, BD is the soil bulk density and PD is the soil particle density, generally given as 2.65 g cm^{-3}.

2.7. Data analysis

The full dataset is available in the CIRAD repository (Shumba et al., 2022). Statistical analyses were performed using R software, version 4.0.0 (R Core Team 2020). Prior to analysis, data were checked for normality by both visual inspection (quantile-quantile plots and density distributions) and with the Shapiro-Wilk test. Where needed, data were log transformed if there were no negative values or cube root transformed in case of some negative values (e.g for daily N_2O and CH_4 emissions).

Linear mixed effect models were fitted on daily N_2O and CH_4 emissions using the *lmer* function from the *lme4* package (Bates, 2010), using as fixed effects the site (DTC, UZF), the season (2019/20, 2020/21), the treatment (CT, CTR, NT, NTM, NTR, NTMR) and the chamber position (row vs inter-row). The chamber number was considered as random factor. The final models were chosen based on the lowest Akaike information criterion (AIC) and on the lowest Bayesian information criterion (BIC). An analysis of variance (ANOVA) was then done on the fitted models. Separation of means was done using the post hoc Tukey test at 5% significance level using the *emmeans* function from the *emmeans* package (Bolker et al., 2009).

A one-way analysis of variance was carried out to establish any significant treatment effects on hourly N_2O and CH_4 fluxes, on total soil mineral N, soil nitrate, soil ammonium, maize grain yield and grain yield-scaled N_2O emissions for each season and site followed by a mean separation using Tukey's test.

3. Results

3.1. Rainfall, soil water-filled pore space and soil mineral nitrogen

In the 2019/20 cropping season, DTC and UZF received a seasonal rainfall of 474 mm and 551 mm, respectively (Fig. 1), whilst in the 2020/21 cropping season, seasonal rainfall was 932 mm and 637 mm at DTC and UZF, respectively (Fig. 2). The temporal variations in WFPS

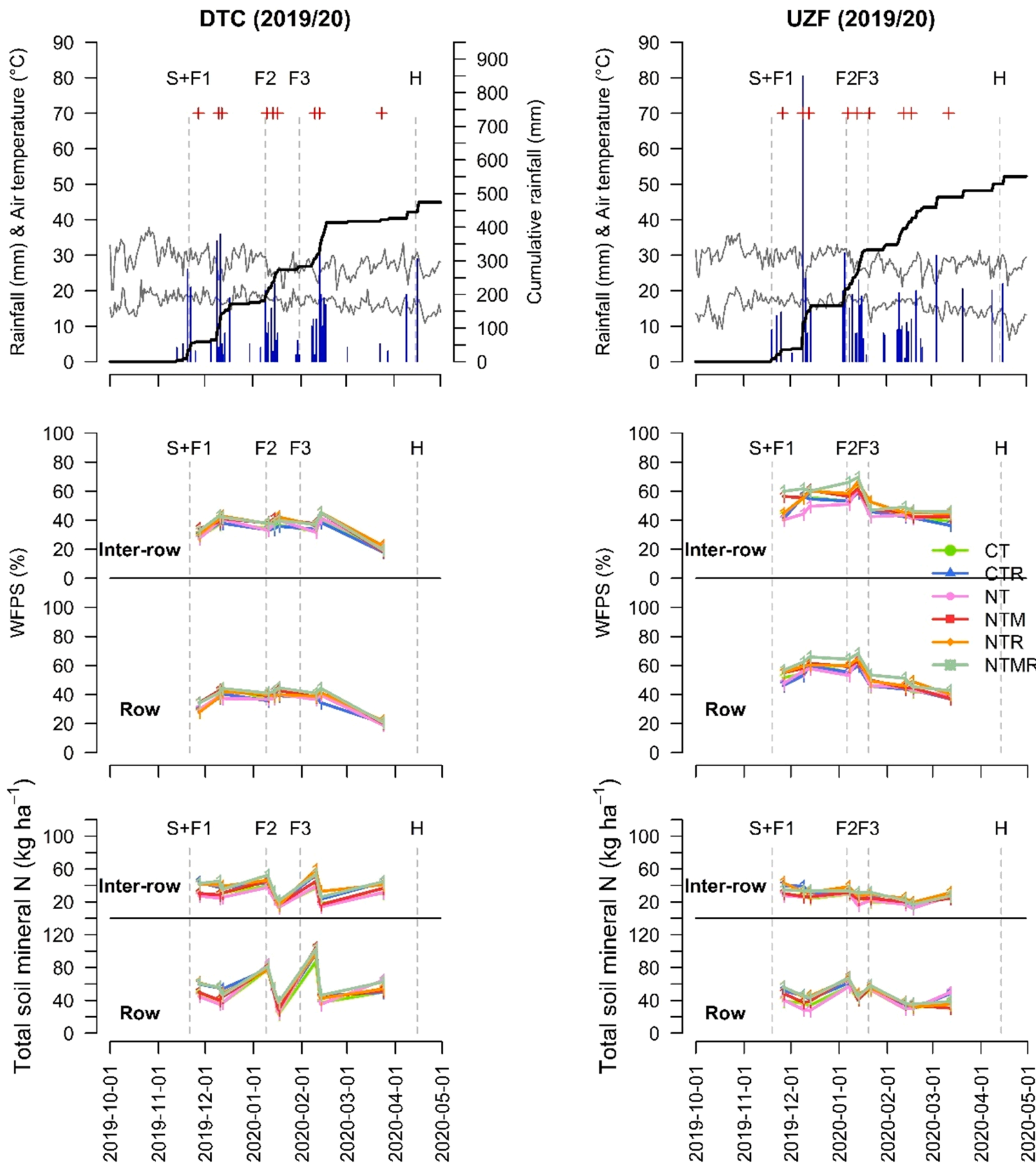

Fig. 1. Daily rainfall received during the 2019/20 season at DTC and UZF sites, water-filled pore space (WFPS) and total soil mineral N. S+F1: sowing + basal fertilizer application, F2: first NH_4NO_3 topdressing, F3: second NH_4NO_3 topdressing, H: harvesting, CT: conventional tillage, CTR: conventional tillage with rotation, NT: no tillage, NTM: no tillage with mulch, NTR: no tillage with rotation, NTMR: no tillage with mulch and rotation. The red crosses indicate gas sampling dates. The upper grey lines represent the maximum air temperature. The lower grey lines represent the minimum air temperature.

during the season were closely related to the rainfall patterns such that treatments effects were not significant. Due to less rainfall, WFPS was lower in the 2019/20 cropping season than in the 2020/21 cropping season at both sites. WFPS increased from the onset of the rainy season and tailed off at the end of it. It was generally higher at UZF than at DTC, as expected from the soil texture differences.

As expected, total soil mineral N always spiked after N fertilizer application (F1, F2 and F3), and was generally higher in the row than the inter-row, especially after N application (Figs. 1 and 2) and tailed off at least one week after F3. There were no significant ($p > 0.05$) differences in daily total soil mineral N between treatments for both seasons and sites.

Average seasonal total soil mineral N content was significantly ($p < 0.01$) higher in the row space than in the inter-row space for both sites and for the two cropping seasons (Fig. 3). However, the proportion of ammonium and nitrate substantially changed from 2019/20 to 2020/21. While the nitrate proportion ranged from 41% to 53% in the 2019/20 season, it was much lower in the 2020/21 season, ranging from 14% to 18%.

3.2. Daily soil N_2O and CH_4 emissions

Season, experimental treatment and position of the chamber had a significant ($p < 0.0001$) effect on daily N_2O emissions (Table S1). A significant ($p = 0.008$) interaction effect between treatment and site was also found (Table S1). The general pattern of N_2O fluxes was largely the same for both sites and seasons, with fluxes regularly higher in the row than inter-row space. As expected, N_2O emissions always peaked shortly after fertilizer application (F1, F2 and F3) and tailed off towards the end of the rainy season (Fig. 4). At DTC in the 2019/20 season, N_2O fluxes ranged as high as 38 (inter-row) - 48 (row) μg N_2O-N m^{-2} hr^{-1} in the no-tillage plus mulch and rotation (NTMR) treatment within 3 weeks after sowing and basal fertilizer application and declined to as low as 1.2 μg N_2O-N m^{-2} hr^{-1} in all treatments approximately 3 weeks before

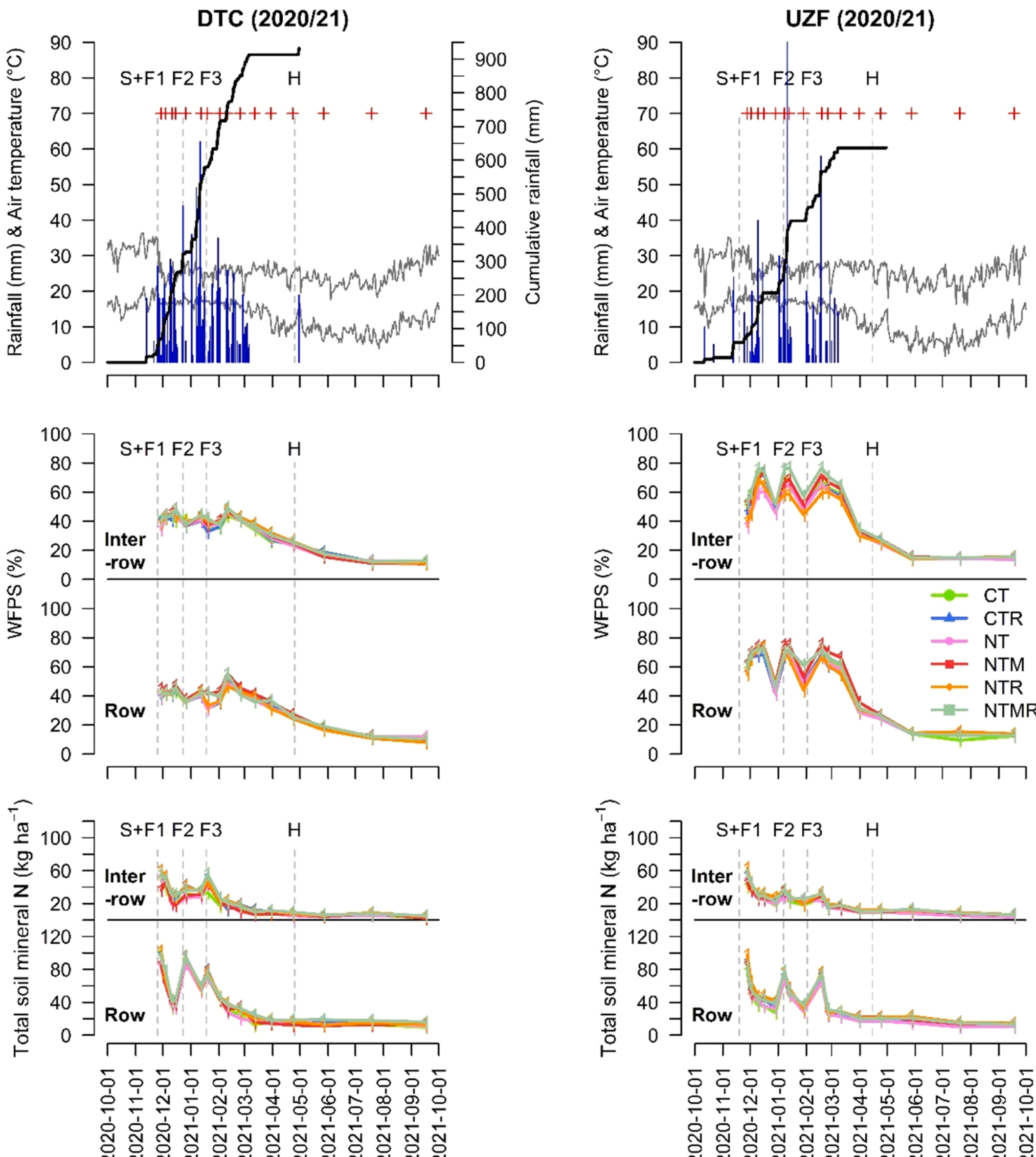

Fig. 2. Daily rainfall received during the 2020/21 season at DTC and UZF sites, water-filled pore space (WFPS) and total soil mineral N. S+F1: sowing + basal fertilizer application, F2: first NH_4NO_3 topdressing, F3: second NH_4NO_3 topdressing, H: harvesting, CT: conventional tillage, CTR: conventional tillage with rotation, NT: no tillage, NTM: no tillage with mulch, NTR: no tillage with rotation, NTMR: no tillage with mulch and rotation. The red crosses indicate gas sampling dates. The upper grey lines represent the maximum air temperature. The lower grey lines represent the minimum air temperature.

maize harvesting. N_2O fluxes were generally lower in the 2020/21 season compared to the 2019/20 season (Fig. 4).

At UZF in the 2019/20 season, N_2O fluxes ranged as high as 39 (inter-row) to 66 (row) μg N_2O-N m^{-2} hr^{-1} in the no-tillage with rotation (NTR) treatment shortly after the second N top dressing, about 7 weeks from planting. N_2O fluxes were generally lower in the 2020/21 season compared to the 2019/20 season (Fig. 4). Nitrous oxide fluxes tailed off in all treatments after about 12 weeks from planting and ranged between − 0.7–1.5 μg N_2O-N m^{-2} h^{-1} (Fig. 4). Generally, there was net emission of N_2O in both seasons.

Site and season had a significant effect on daily CH_4 fluxes (Table S1). CH_4 fluxes showed high temporal variability (Fig. 5) and had no distinct pattern for both seasons and sites. At DTC, by and large, there was CH_4 emission in the two seasons, but with a few sampling days (< 5 sampling days) when there was CH_4 consumption (Fig. 5). CH_4 fluxes respectively ranged from − 101.8–86.06 μg CH_4-C m^{-2} hr^{-1} and − 97.2–117.7 μg CH_4-C m^{-2} hr^{-1} in the row and inter-row spaces in the 2019/20 season and from − 128.2–99.7 μg CH_4-C m^{-2} hr^{-1} and − 117.8–142.9 μg CH_4-C m^{-2} hr^{-1} in the row and inter-row spaces in the 2020/21 season. In contrast, UZF soils were by and large, net CH_4 sinks in both seasons, with fluxes as low as − 94 μg CH_4-C m^{-2} h^{-1}, though there were positive fluxes early in the 2019/20 season ranging from 3.5 to 17.2 μg CH_4-C m^{-2} hr^{-1}. CH_4 fluxes ranged from − 65.5–17.2 μg CH_4-C m^{-2} hr^{-1} and − 49.7–52.0 μg CH_4-C m^{-2} hr^{-1} in the row and inter-row spaces, respectively, in the 2019/20 season and from − 94.3–0.6 μg CH_4-C m^{-2} hr^{-1} and − 84.3–0.84 μg CH_4-C m^{-2} hr^{-1} in the row and inter-row spaces, respectively, in the 2020/21 season.

3.3. Cumulative soil N_2O and CH_4 emissions

Cumulative N_2O emissions in the row and inter-row were on average not significantly ($p > 0.05$) different between treatments for both sites and seasons. However, within individual treatments, cumulative N_2O

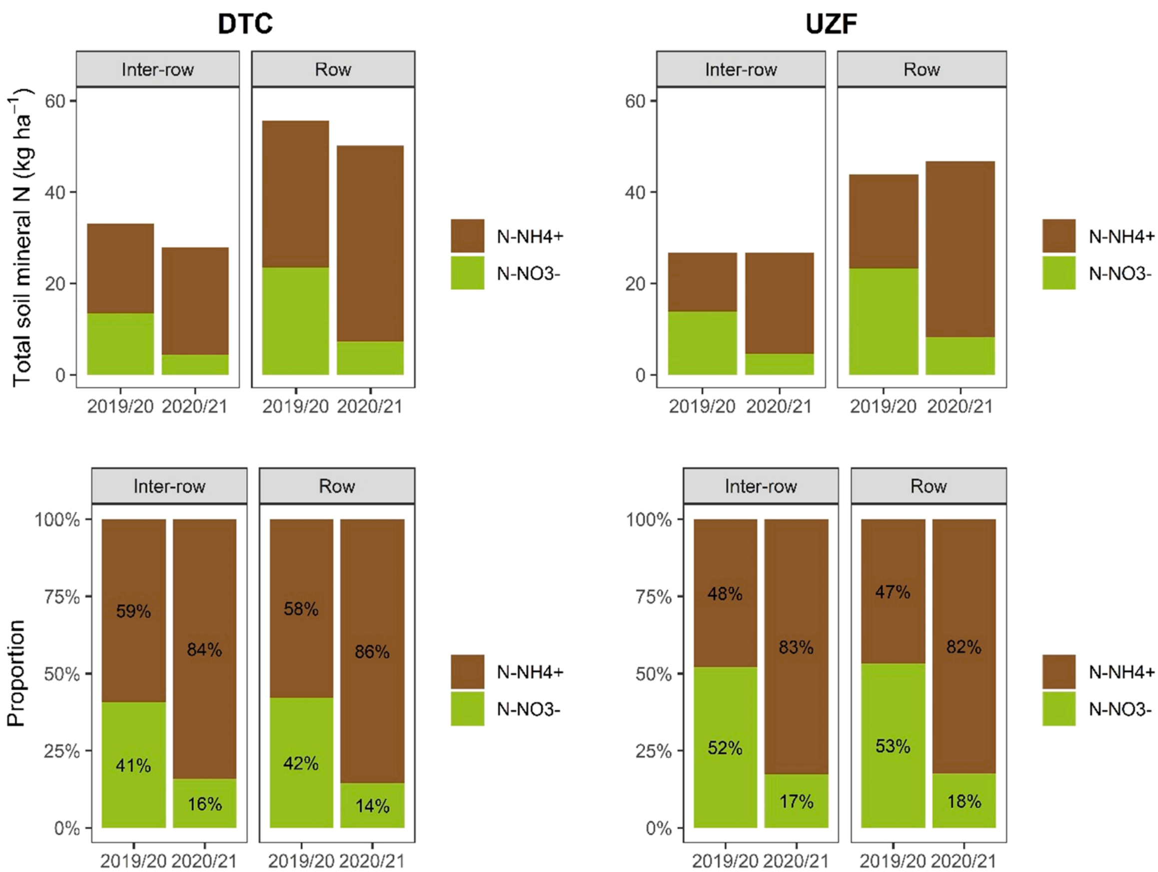

Fig. 3. Average seasonal total soil mineral N in the maize inter-row and row spaces during the two cropping seasons at DTC and UZF, and proportions of nitrate-N and ammonium-N.

emissions were significantly ($p < 0.05$) higher in the row than the inter-row, by > 100% for NT at DTC and > 33% for CT, NTM and NTMR at UZF in the 2019/20 season (Fig. 6). The same trend was observed in the 2020/21 season for NT and NTR at DTC and CT, NT and NTM at UZF where cumulative N_2O emissions were at least 20% higher in the row than the inter-row.

Cumulative total N_2O emissions, considering the proportions of row and inter-row emissions, were significantly ($p < 0.05$) higher in the 2019/20 season than in the 2020/21 season for both sites across all treatments. In the 2019/20 cropping season, mulching (NTM and NTMR) emitted significantly higher N_2O per season (Fig. 7) compared to CT and NT at DTC. In contrast, during the 2019/20 cropping season at UZF, cumulative total N_2O emissions were significantly higher in treatments with rotation (CTR, NTR and NTMR) compared to CT. On the other hand, in the 2020/21 cropping season there were no significant differences in cumulative total N_2O emissions between treatments at both sites. Cumulative total N_2O emissions ranged from 125 to 160 g N_2O-N ha^{-1} yr^{-1} at DTC and 90 to 130 g N_2O-N ha^{-1} yr^{-1} at UZF (Fig. 7). When considering the whole year (October 2020 to September 2021) instead of only the cropping season (October 2020 to April 2021), cumulative total N_2O emissions increased by approximatively 14% (Fig. 7).

There was net cumulative CH_4 emission at DTC and net cumulative CH_4 consumption at UZF in both seasons. Across all treatments, CH_4 consumption at UZF was significantly ($p < 0.01$) higher in the 2020/21 season than in the 2019/20 season (Fig. 7). In the 2019/20 cropping season at DTC, cumulative CH_4 emissions were significantly higher in the NTMR treatment (1025 g CH_4-C ha^{-1} yr^{-1}) compared to the NTR treatment (148 g CH_4-C ha^{-1} yr^{-1}). No other significant differences between treatments were observed. (Fig. 7). At UZF, cumulative CH_4 consumption throughout the whole year was at least twice higher than throughout the cropping season only.

3.4. Maize grain yield and yield-scaled N_2O emissions

Maize grain yield ranged from 1.8 to 5.0 t ha^{-1} and 4.0 to 5.6 t ha^{-1} at UZF, and 1.8 to 2.9 t ha^{-1} and 1.4 to 2.3 t ha^{-1} at DTC in the 2019/20 and 2020/21 seasons, respectively (Table 3). At DTC, there were no significant ($p > 0.05$) differences in grain yield across treatments in both the 2019/20 and 2020/21 seasons. However, at UZF, rotation treatments (CTR, NTR, NTMR) significantly increased ($p < 0.001$) grain yield (by at least 24%) compared to CT in the 2019/20 season (Table 3). Besides, the full CA treatment, NTMR, significantly ($p < 0.001$) increased maize grain yield by at least 22% and 29% compared to the conventional (CT, CTR) and no-tillage (NT, NTM, NTR) treatments, respectively. It is also worth noting that CTR significantly ($p < 0.001$) improved yield by almost 1.0 t grain ha^{-1} compared to NTM. In the 2020/21 season at UZF, yield increased in all treatments compared to 2019/20, but no difference was found between treatments ($p > 0.05$).

Grain yield-scaled N_2O emissions (g N_2O emission per unit grain yield) showed no significant ($p > 0.05$) treatment effects at both sites and in both cropping seasons (Table 3). At DTC, yield-scaled N_2O emissions ranged from 0.10 g to 0.19 g N_2O-N kg^{-1} grain in the 2019/20 cropping season and 0.07 to 0.12 g N_2O-N kg^{-1} grain in the 2020/21 cropping season, whilst at UZF, they ranged from 0.07 g to 0.18 g N_2O-N kg^{-1} grain in 2019/20 and 0.02 to 0.03 g N_2O-N kg^{-1} grain in the 2020/21 cropping season. Yield-scaled N_2O emissions were significantly ($p < 0.001$) lower in the 2020/21 cropping season than in the 2019/20 cropping season.

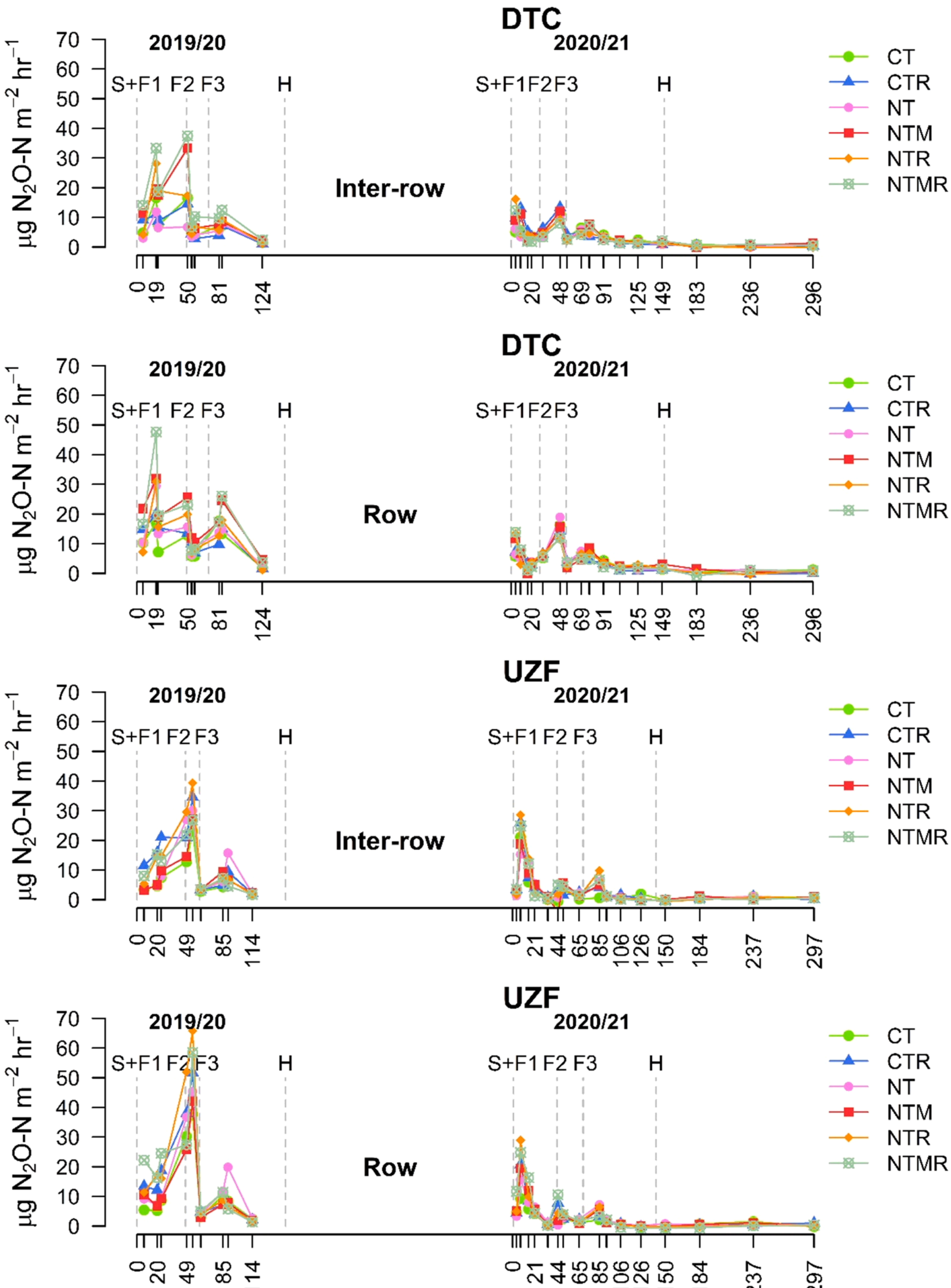

Fig. 4. Hourly N_2O fluxes in the maize inter-row and row spaces during the 2019/2020 and 2020/21 seasons at DTC and UZF. S+F1: sowing + basal fertilizer application, F2: first NH_4NO_3 topdressing, F3: second NH_4NO_3 topdressing, H: harvesting, CT: conventional tillage, CTR: conventional tillage with rotation, NT: no tillage, NTM: no tillage with mulch, NTR: no tillage with rotation, NTMR: no tillage with mulch and rotation.

3.5. N_2O fluxes and soil factors

N_2O fluxes were positively and significantly ($p < 0.0001$) related to total soil mineral nitrogen (Fig. S2) and soil NO_3^--N for both sites (Fig. 8). However, the R^2 values (< 0.30) of the regressions were low. At DTC, on a Lixisol, maximum N_2O emissions were observed at 40% WFPS (0–20 cm soil layer), while at UZF, on a Ferralsol, they occurred at 60–70% WFPS.

4. Discussion

4.1. Low N_2O emissions

The generally low N_2O fluxes (seasonal averages of 12.1 and 4.8 µg N_2O-N m^{-2} hr^{-1} at DTC and 13.7 and 4.2 µg N_2O-N m^{-2} hr^{-1} at UZF in 2019/20 and 2020/21, respectively) and corresponding low cumulative total seasonal N_2O emissions (< 500 and < 200 g N_2O-N ha^{-1} yr^{-1} at DTC and < 400 and < 150 g N_2O-N ha^{-1} yr^{-1} at UZF in 2019/20 and 2020/21, respectively) can be primarily attributed to the low N input. N was applied in the form of mineral fertilizer at planting

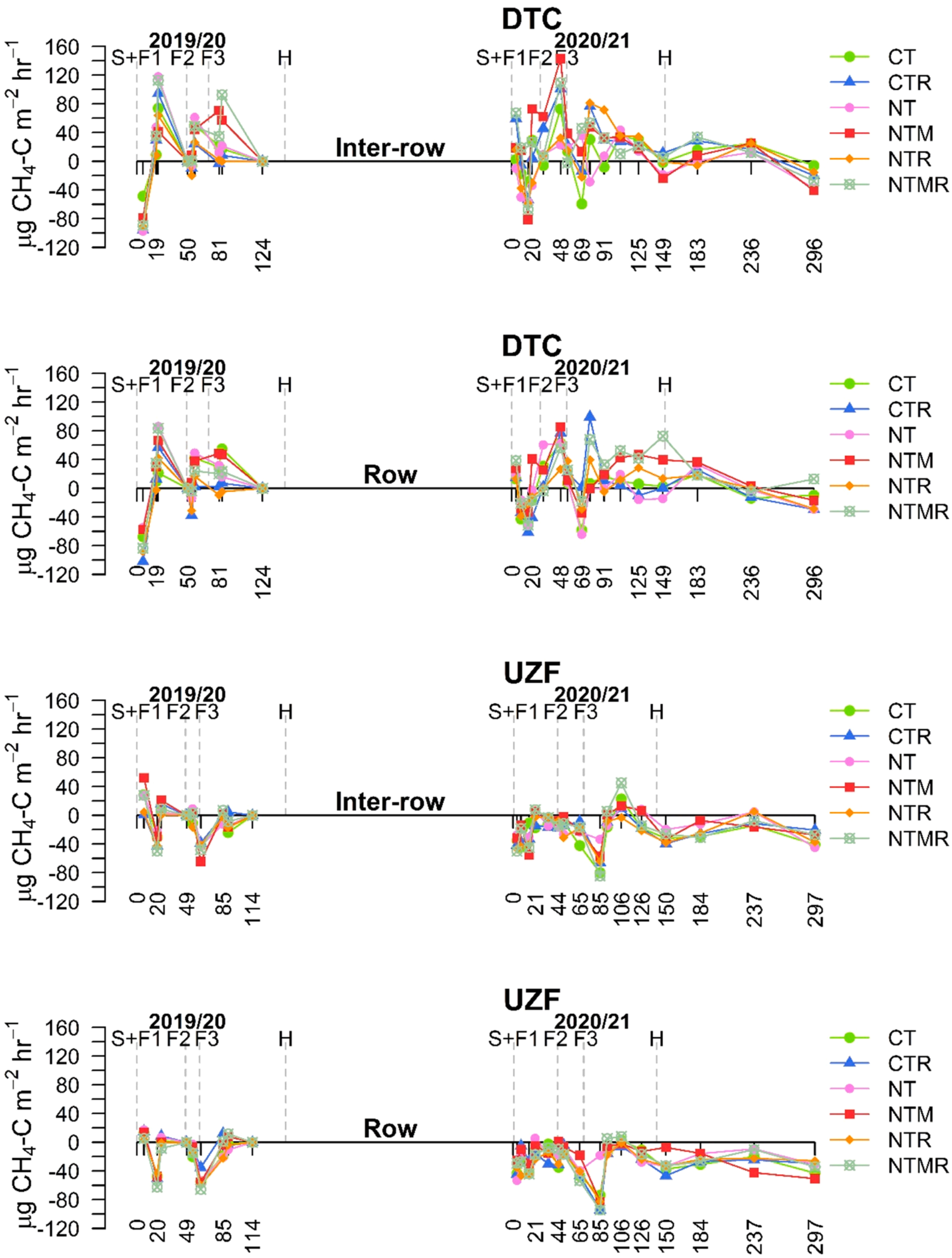

Fig. 5. Hourly CH_4 fluxes in the maize inter-row and row spaces during the 2019/2020 and 2020/21 seasons at DTC and UZF. S+F1: sowing + basal fertilizer application, F2: first NH_4NO_3 topdressing, F3: second NH_4NO_3 topdressing, H: harvesting, CT: conventional tillage, CTR: conventional tillage with rotation, NT: no tillage, NTM: no tillage with mulch, NTR: no tillage with rotation, NTMR: no tillage with mulch and rotation.

(S+F1) and as top dressing (F2 and F3), which summed up to only 57.8 kg N ha^{-1}, and was spot applied in the rows which increased N use efficiency by the maize crop. The spot application of the N fertilizer in the rows also explains why we observed generally higher N_2O emissions in the row than the inter-row space. However, to put this in context, the N fertilizer rate in our study is almost three times the rates that are usually applied by smallholder farmers in the region. On the other hand, it is lower by about a half of the recommended rates by the Ministry of Agriculture in Zimbabwe (Twomlow et al., 2010; Mapanda et al., 2011). Generally, the low N fertilizer rate in this study restrained soil N_2O emissions since the soils at both sites were weak sources of N_2O regardless of the tillage or residue management implemented (Shcherbak et al., 2014). Fertilizer use in SSA is generally low, especially compared to > 100 kg N ha^{-1} in the developed world (Motesharezadeh et al., 2017; Thierfelder et al., 2018; Wuepper et al., 2020; Menegat and Ledo, 2021). Although the Abuja Declaration of 2006 seeks to increase fertilizer use in SSA to at least 50 kg fertilizer ha^{-1} (African Union, 2006; CAAP, 2015; Winnie et al., 2022), it is expected that N_2O emissions from agricultural soils in SSA remain relatively low.

We also attribute the low N_2O emissions to good N fertilizer management in terms of timing of doses to improve N uptake efficiency by the maize crop (Cassman et al., 2002; Kennedy et al., 2013). In our study, N fertilizer was split applied in three doses (S+F1, F2 and F3) to optimize N uptake by the maize crop. In fact, post-emergence N

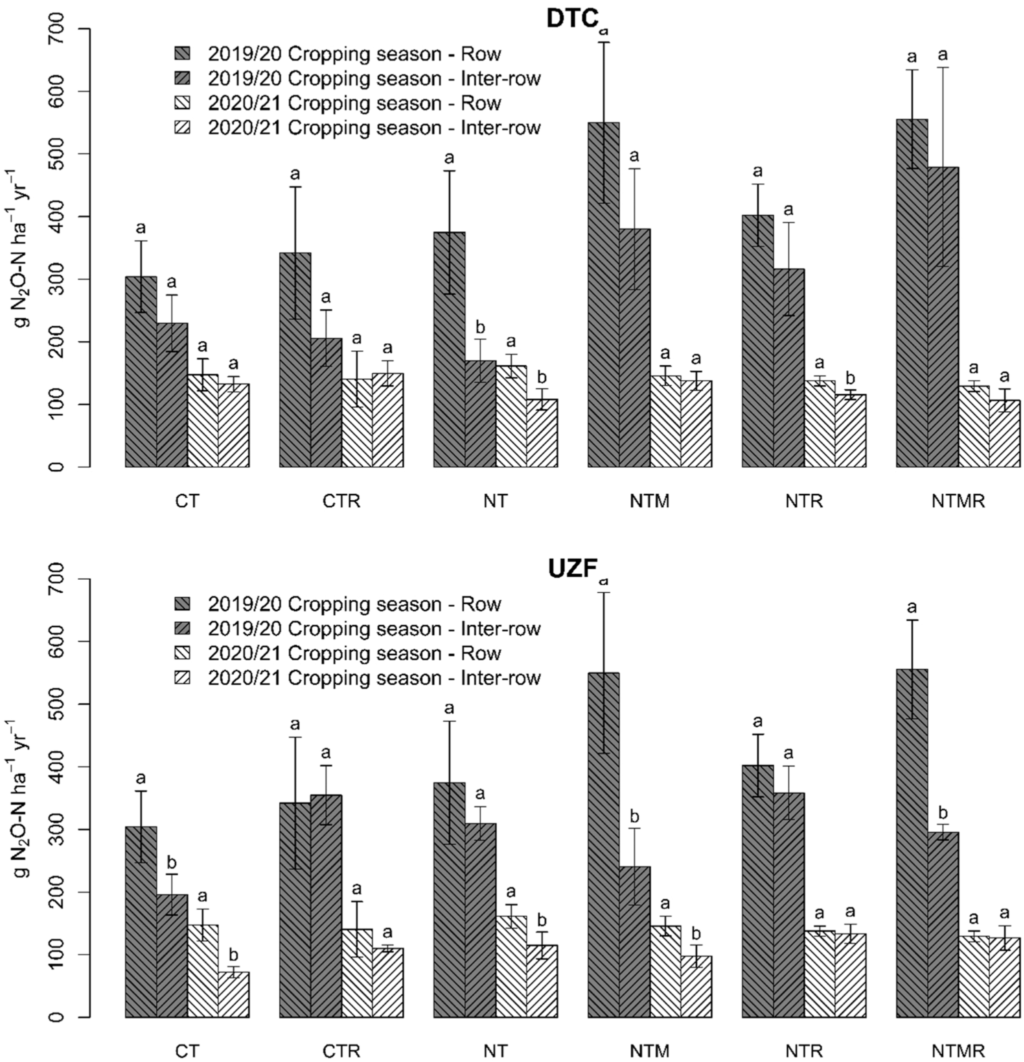

Fig. 6. Cumulative N_2O emissions in maize rows and inter-rows for the 2019/20 and 2020/21 seasons at DTC and UZF. Error bars represent standard errors (N = 4). CT: conventional tillage, CTR: conventional tillage with rotation, NT: no tillage, NTM: no tillage with mulch, NTR: no tillage with rotation, NTMR: no tillage with mulch and rotation. Different letters represent significant differences in N_2O emissions in rows compared to inter-rows within a site and a given treatment.

applications were delayed due to mid-season droughts, especially in the 2019/20 cropping season such that the N applications coincided with adequate soil moisture for crop growth (Figs. 1 and 2).

Our results indicate that there was little surplus soil N for denitrification and/or nitrification even if other soil conditions were conducive. The higher cumulative N_2O emissions in the 2019/20 season compared to 2020/21 at the two sites can be linked to the higher proportion of soil NO_3^- in the 2019/20 cropping season (Fig. 3). It is well known that soil NO_3^- has greater influence on N_2O fluxes than NH_4^+ because it is the precursor of denitrification (Mapanda et al., 2010; Baggs, 2011). The significantly higher cumulative N_2O emissions observed in the rotation treatments (with cowpea) at UZF in the 2019/20 season (Fig. 7) could, therefore, be attributed to additional soil N from biological nitrogen fixation (BNF) and from N mineralization of belowground N-rich biomass, that is susceptible to nitrification and/or denitrification. On the other hand, the introduction of legumes in the crop rotation has been shown to reduce N_2O emissions if the N contribution from the preceding legume crop is factored into the N fertilization dose of the subsequent (cereal) crop (Barton et al., 2014; Bayer et al., 2016; Hatfield, 2017). In our study, the extra N source from the preceding cowpea crop was not considered, i.e. the N fertilizer rate for the succeeding maize crop was not reduced in comparison to the treatments without cowpea rotation, hence the higher potential for N_2O emission.

Mulching significantly increased N_2O emissions at DTC in 2019/20, possibly due to enhanced moisture conservation and subsequent higher WFPS (Linn and Doran, 1984), and to a lesser extent as a result of N released from the decomposition of the maize residues (Lashermes et al., 2022). Cereal crop residues have a relatively high C/N ratio of > 30 which results in low N mineralization rates (Lupwayi et al., 2007; Abalos et al., 2022; Lashermes et al., 2022). Hence, N_2O emissions associated with cereal residues are in most cases insignificant (Lashermes et al., 2022).

At DTC, cumulative rainfall almost doubled from the first (474 mm) to the second (932 mm) season, but there were no significant maize yield changes (Table 3). We therefore postulate that most of accumulated soil NO_3^- was leached due to heavy rainfall at DTC. On the other hand, at UZF, cumulative rainfall was slightly higher in the 2020/21 season and maize yield largely increased (Table 3). We assume here that most of the soil NO_3^- was absorbed by the maize plants. At both sites, the most favourable soil moisture conditions probably led to higher soil organic matter mineralization, explaining the increase in soil NH_4^+ in 2020/21 compared to 2019/20.

In our study, we observed temporal variability of N_2O fluxes (Fig. 4). Fertilizer N application was the major driver of N_2O emissions, with peaks generally occurring soon after fertilizer application (Fig. 4). The temporal increase in soil mineral N concentrations soon after fertilizer

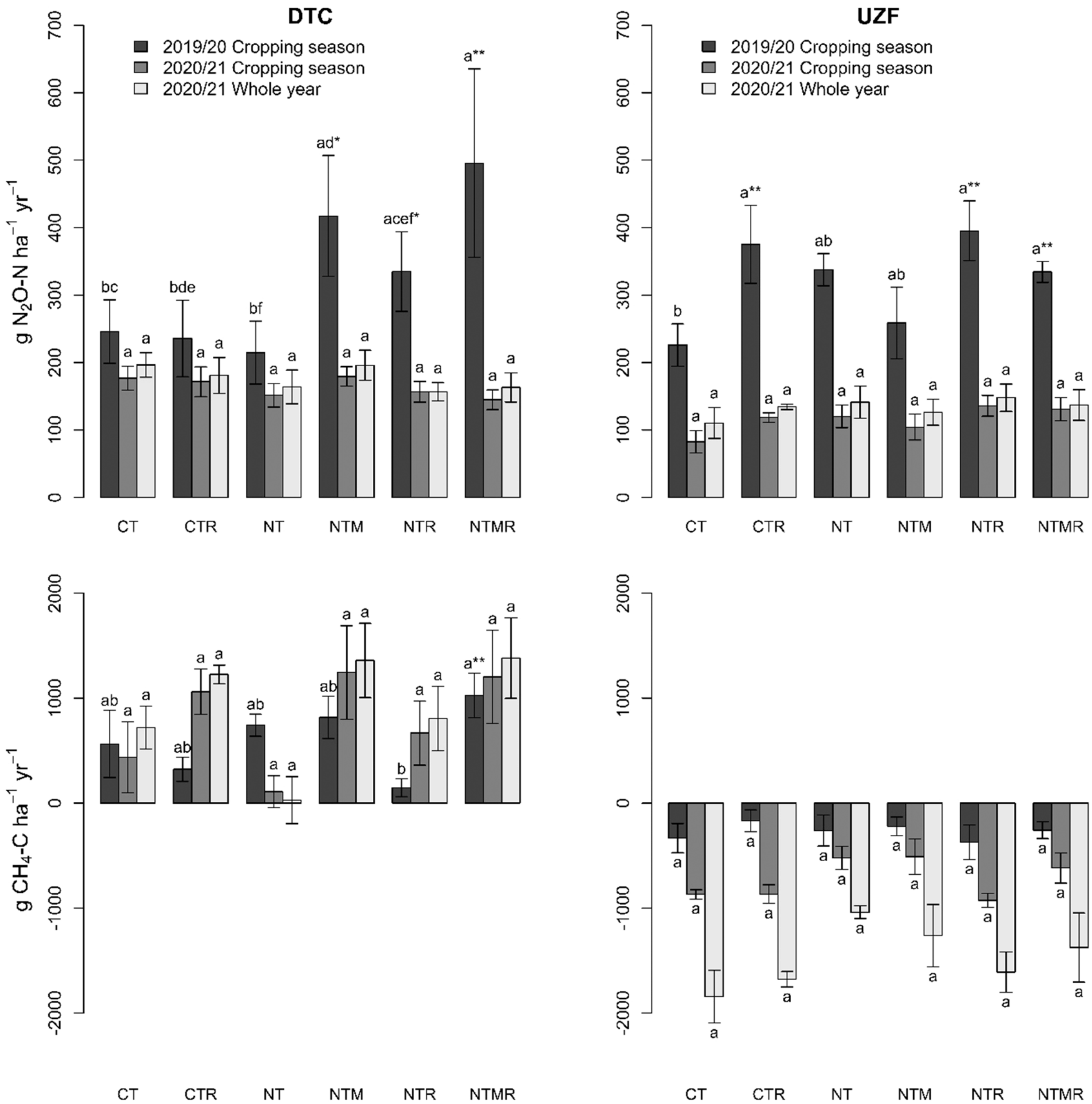

Fig. 7. Cumulative total (from row and inter-row) N_2O and CH_4 emissions for the 2019/20 and 2020/21 seasons at DTC and UZF. Error bars represent standard errors (N = 4). CT: conventional tillage, CTR: conventional tillage with rotation, NT: no tillage, NTM: no tillage with mulch, NTR: no tillage with rotation, NTMR: no tillage with mulch and rotation.

application (Figs. 1 and 2) led to the temporal peaks in N_2O emissions. The significantly positive correlation between N_2O emission and soil mineral N (Fig. 8) supports this finding. This was clearly linked to the effects of crop management activities on substrate (NH_4^+ and NO_3^-) availability and moisture for N_2O synthesis. However, in our study, the fact that rainfall events followed by fertilizer application (Figs. 1 and 2) resulted in spikes of N_2O fluxes rather than rainfall events alone clearly suggest that N availability and not soil moisture was most limiting. Rainfall that was received after the last N top dressing (F3) did not induce spikes in N_2O fluxes due to the sharp drop in soil mineral N (Figs. 1, 2 and 4). Fertilizer N induced N_2O emissions has also been reported by Burger et al. (2005), Ma et al. (2010), Kennedy et al. (2013), Cayuela et al. (2017) and Sanz-Cobena et al. (2017). Nitrification was largely responsible for most of N_2O emissions at DTC, as WFPS was always conducive (WFPS < 60%, Figs. 1 and 2) whilst both processes, nitrification (WFPS < 60%) and denitrification (WFPS > 60%) were responsible for N_2O emissions at UZF (Cayuela et al., 2017; Sanz-Cobena et al., 2017, 2016). Nitrification and denitrification are aerobic and anaerobic processes which have been found to occur at < 60% and > 60% WFPS, respectively (Linn and Doran, 1984; Sanz-Cobena et al., 2016). At DTC, WFPS was < 60% at both sites though there were periodic times of > 60% WFPS at UZF soon after rainfall (Figs. 1 and 2).

Our CA component omission study provided evidence that the different combinations of CA principles are site specific in their effects on N_2O emissions; at UZF maize-cowpea rotation (CTR, NTR, NTMR), regardless of tillage, increased N_2O emissions whilst at DTC crop residue mulching (NTM, NTMR) tended to increase N_2O emissions.

The Intergovernmental Panel on Climate Change (IPCC) guidelines for the calculation of national GHG inventories widely uses the simplest method (Tier 1) for national inventories in SSA which assumes that 1% of applied N fertilizer is lost as N_2O in direct emissions annually (Daioglou et al., 2020). In this study, without subtracting N lost as N_2O emissions from a control treatment without N fertilizer since our study did not have such a treatment, N lost as N_2O emission was, respectively, ≤ 0.43% and ≤ 0.29% in the 2019/20 and 2020/21 seasons at both sites. Earlier field studies conducted in SSA reported emission factors which were < 0.75% (Chikowo et al., 2004; Brümmer et al., 2009; Mapanda et al., 2010, 2011, 2012; Hickman et al., 2015b), except when fertilizer rates were ≥ 100 kg N ha^{-1} (Hickman et al., 2015; Kim et al., 2016). Our study confirms the finding, as suggested by Shcherbak et al.

Table 3
Maize grain yield (± standard error) and yield-scaled N_2O-N emissions (± standard error) for the 2019/20 and 2020/21 cropping seasons at DTC and UZF. CT: conventional tillage, CTR: conventional tillage with rotation, NT: no tillage, NTM: no tillage with mulch, NTR: no tillage with rotation, NTMR: no tillage with mulch and rotation.

Season	Treatment	DTC		UZF	
		Grain (kg ha^{-1})	Yield-scaled emissions (g N_2O-N kg^{-1} grain)	Grain (kg ha^{-1})	Yield-scaled emissions (g N_2O-N kg^{-1} grain)
2019/20	CT	1848.2^a ± 157.5	0.13^a (± 0.02)	1800.2^d ± 413.2	0.15^a (± 0.04)
	CTR	2291.5^a ± 373.5	0.10^a (± 0.01)	4116.6^b ± 197.4	0.09^a (± 0.02)
	NT	1815.6^a ± 47.4	0.12^a (± 0.02)	2355.3^d ± 522.9	0.18^a (± 0.05)
	NTM	2868.3^a ± 304.6	0.15^a (± 0.03)	3122.8^c ± 309.9	0.09^a (± 0.02)
	NTR	2391.1^a ± 126.5	0.14^a (± 0.02)	3880.6^b ± 174.3	0.10^a (± 0.01)
	NTMR	2572.6^a ± 233.5	0.19^a (± 0.05)	5014.1^a ± 220.8	0.07^a (± 0.01)
	Significance	ns	ns	$p < 0.001$	ns
	LSD	737.9	0.09	701.3	0.08
	CV%	21.3	41.3	13.8	46.0
2020/21	CT	2241.7^a ± 601.9	0.09^a (± 0.01)	4829.8^a ± 322.0	0.02^a (± 0.003)
	CTR	2229.8^a ± 627.5	0.09^a (± 0.02)	5616.5^a ± 333.8	0.02^a (± 0.001)
	NT	2152.1^a ± 225.4	0.07^a (± 0.02)	4004.6^a ± 352.0	0.03^a (± 0.005)
	NTM	1726.4^a ± 331.6	0.12^a (± 0.04)	4578.9^a ± 346.3	0.02^a (± 0.003)
	NTR	2279.6^a ± 595.8	0.09^a (± 0.03)	5465.7^a ± 539.7	0.03^a (± 0.005)
	NTMR	1409.5^a ± 145.4	0.10^a (± 0.01)	5544.1^a ± 281.0	0.02^a (± 0.003)
	Significance	ns	ns	ns	ns
	LSD	1427.9	0.08	1175.8	0.01
	CV%	27.2	55.0	15.6	29.4

(2014), that additional N applications in these low N inputs systems have little impact on N_2O emissions, while largely boosting crop yields if the additional N is used efficiently by the crops (Kimaro et al., 2016; Nyamadzawo et al., 2017). It should be underscored that the current inventories are clearly overestimating N_2O emissions from agriculture in SSA due to use of Tier 1 default emission factors (EF) (Rosenstock et al., 2013; Hickman et al., 2014) resulting in recommending ineffective mitigation measures (Pelster et al., 2017). Thus, more research is needed to gather more site specific (region specific) data on N_2O emissions so that a Tier II approach can be used for SSA.

For sustainable farming strategies, it is important to minimize yield-scaled N_2O emissions and not only area-scaled N_2O emissions. In this study, yield-scaled N_2O emissions were low (< 0.20 g N_2O-N kg maize grain^{-1}) and there were no significant differences between treatments for both sites and seasons. The significant maize grain yield increases due to the maize-cowpea rotation at UZF and the mulching at DTC is compensating for the higher area-scaled N_2O emissions in these cropping systems (Table 3), which is an important consideration for sustainable agriculture intensification (Zhao et al., 2017). Measured yield-scaled N_2O emissions are in the same order of magnitude as those reported in the literature. Nyamadzawo et al. (2017) for instance reported yield-scaled N_2O emissions of 0.60 and 0.27 g N_2O-N kg maize grain^{-1} after application of 60 and 120 kg N ha^{-1}, respectively, at the same DTC experimental site. In the Ethiopian Rift Valley, Raji and Dörsch (2020) reported yield-scaled emissions of < 0.06 g N_2O-N kg maize grain^{-1} after application of about 100 kg N ha^{-1} as a combination of mineral N and legume residues. Yet, low yield-scaled N_2O emissions were also observed in e.g. China with high N fertilizer rates of > 120 kg N ha^{-1} (Nyamadzawo et al., 2017) or in the USA with rates of about 250 kg N ha^{-1} (Zhao et al., 2017) which were < 0.14 g and < 0.6 g N_2O-N kg maize grain^{-1}, respectively. This is due to high maize grain yields ranging between 9 and 16 t ha^{-1}, compared to < 1.6 t ha^{-1} in Zimbabwe. It clearly highlights the need to improve nitrogen use efficiency for boosting crop yields in the low N input cropping systems that are characteristic of smallholder farms in SSA, but which could be achieved by innovations like breeding.

4.2. Methane consumption and emission

Our results on CH_4 emissions at the UZF site corroborate the finding that well-drained, aerobic soils generally act as sinks for CH_4, with an estimated global sink strength of 22–100 Tg CH_4 year^{-1} (Castaldi et al., 2006; Dutaur and Verchot, 2007). In general, tropical and subtropical soils are largely net sinks of CH_4 since they are in most cases well drained and well aerated (Bayer et al., 2012; Zhao et al., 2016) and thus contain methanotrophs that use CH_4 as their source of carbon and energy (Bayer et al., 2012; Chiri et al., 2020).

The large CH_4 emissions we observed at the DTC site during the two seasons are very surprising and difficult to explain. Soils emitting CH_4 are generally those that are frequently submerged or water-saturated (Le Mer and Roger, 2001). However, at the DTC site, WFPS was ≤ 60% in the 0–20 cm soil depth. On the other hand, this site has an abrupt change in soil texture from a light sandy loam in the topsoil (0–20 cm) to a medium to heavy textured (sandy clay loam and sandy clay) subsoil (20–100 cm). As a result, temporary water-logging could happen, which is corroborated by the observation of mottling and iron and manganese oxides in the subsoil (data not shown here). However, in the 2020/21 cropping season with relatively high seasonal cumulative rainfall (932 mm), moisture measurements were carried out down to 1 m depth with a neutron probe close to the GHG sampling dates and revealed that the soil was never water-saturated (Fig. S3). Whilst highest WFPS values were found at 60–75 cm soil depth (Fig. S3), and highest CH_4 emissions were found at the same time as the highest WFPS levels in this soil layer (Fig. S4), this cannot explain the constant CH_4 emissions during the season. We cannot exclude that water-logging was present below 1 m depth leading to these CH_4 emissions, but deeper soil moisture measurements would be required.

Another explanation for the CH_4 emissions may lie in the fact that termites were present within the soil profile at DTC. Termites are known to be a source of CH_4 and are responsible for 1–3% of global CH_4 emissions (Nauer et al., 2018). CH_4 emission from termites were for instance reported by Brümmer et al. (2009) in Burkina Faso and Nyamadzawo et al. (2012b) in Zimbabwe. These authors found that termite mounds are significant sources of CH_4 and that in case of land-use change from savanna woodlands to cropping, the cleared termite mounds continued to be sources of CH_4. Several other authors have reported termites as a source of CH_4 (Zimmermann P et al., 1982; Reuß et al., 2015; Chiri et al., 2020) though about 50% of the termite CH_4 emission is consumed by termite mound soils (Nauer et al., 2018; Chiri et al., 2020) before being released into the atmosphere. It should also be noted that, the reported CH_4 emissions from termite mounts are at least an order of magnitude lower than those measured at DTC (Van Asperen

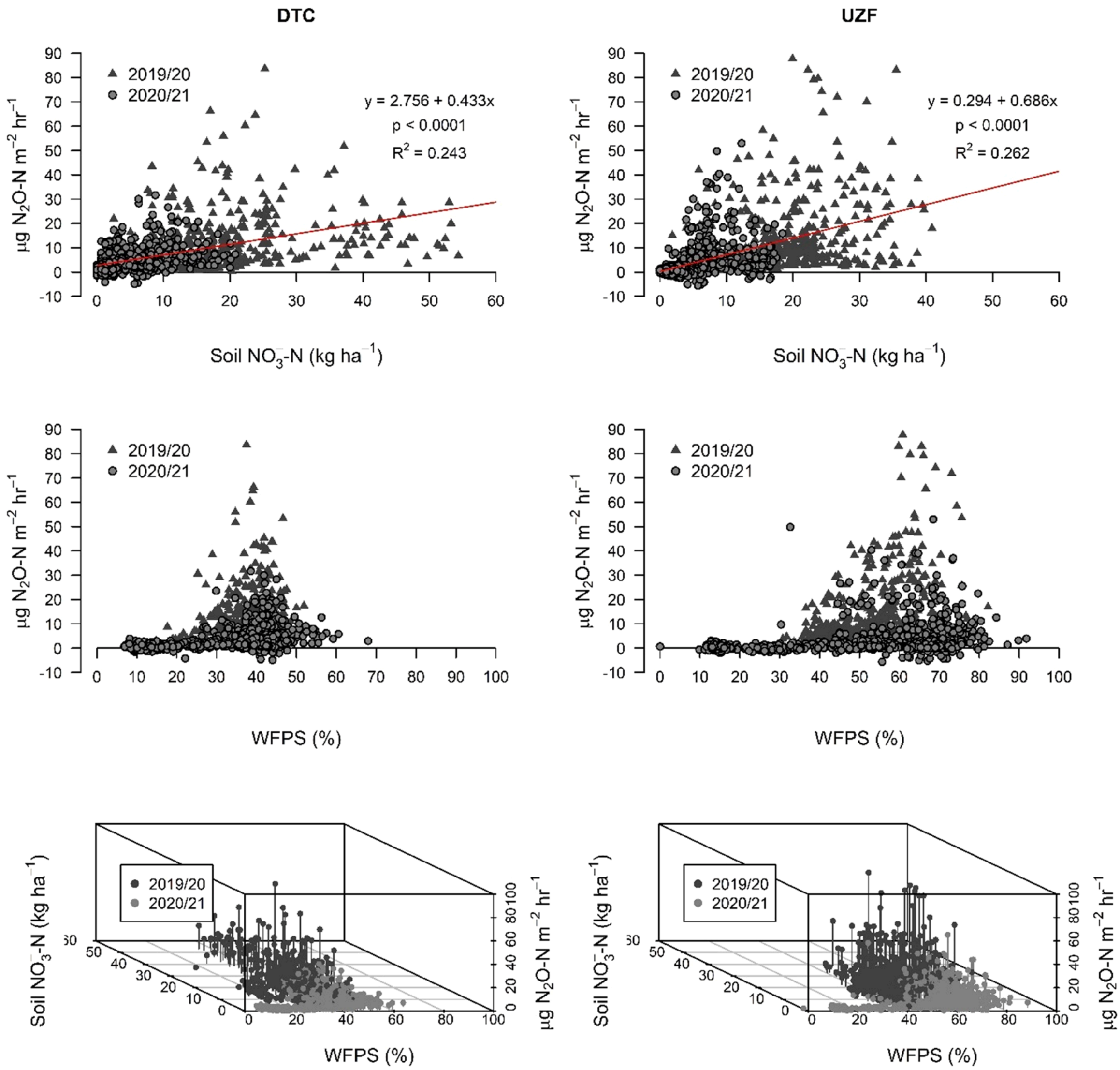

Fig. 8. Relationships between hourly soil N_2O emissions, soil nitrate (NO_3^--N) and water-filled pore space (0–20 cm) for the 2019/20 and 2020/21 seasons at the DTC and UZF experimental sites.

et al., 2021) and that termites were also present at UZF where we did not observe CH_4 emission, but CH_4 consumption. Thus, most likely, termites are not the explanation for the large CH_4 emissions observed at DTC.

Another possible explanation, even though unlikely, could be methane coming from an abiotic source on this site (Lemme and Givens, 1974). The isotopic composition of carbon and hydrogen of trapped CH_4 should, however, be determined in order to rule out this hypothesis.

Lastly, an external source such as CH_4 saturated deep water could also be an explanation. However, we could not identify a source such as effluent or faecal sludge in the neighbourhood. GHG measurements in adjacent and non-adjacent fields with the same soil type could help us resolving this mystery, with potential large implications in terms of GHG accounting.

4.3. Limitations of the study

The static chamber method used in this study has some limitations. Firstly, we used linear interpolation between sampling points to calculate cumulative N_2O and CH_4 emissions which in essence might not be linear between the sampling dates. Secondly, we might also have missed hot spots of N_2O emissions especially immediately after fertilizer application since we could not sample gases on a daily basis due to budget limitations. However, we tried to sample as frequently as possible in periods where significant peaks in gas fluxes were anticipated like following fertilizer application and rainfall events, as shown in Figs. 1 and 2. Finally, soil mineral N was monitored in the 0–20 cm soil layer only, hence subsoil mineral N was not monitored to determine possibilities of N leaching losses.

5. Conclusions

Our hypothesis that integrating the three principles of CA enhances soil non-CO_2 GHG emissions and improves maize productivity was partly confirmed. Firstly, NT in combination with mulch (NTM) and with mulch and cowpea rotation (NTMR) increased total seasonal N_2O emissions on a Lixisol (DTC) in a dry season. Secondly, rotation of maize with cowpea, regardless of tillage (CTR, NTR, NTMR), increased total seasonal N_2O emissions on a Ferralsol (UZF) in a relatively dry cropping season, whilst rotation, mulching, tillage or any combination thereof, had no significant effect on N_2O emissions in a relatively wet season. Moreover, no significant differences in yield-scaled N_2O emissions were found between treatments for both seasons and sites. This study confirms that in low N input cropping systems (approximately 60 kg N ha^{-1} yr^{-1} in our case) with fertilizer split applications to synchronize N supply with plant N uptake, N_2O emissions are low and, in fact, considerably lower than the default emission factor of IPCC. This calls for more N_2O emission assessments in SSA to allow for the use of Tier II methodology in N_2O inventories. Methane emissions were not driven by crop management in the upland soils of our study, but were linked to soil type. Finally, the full climate benefit of CA should however not be limited to non-CO_2 emissions, and must incorporate soil organic carbon sequestration, as well as biogeophysical changes such as albedo and evapotranspiration that have an impact on global warming.

Declaration of Competing Interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests, Armwell Shumba reports financial support was provided by Agropolis Foundation. Armwell Shumba reports financial support was provided by TotalEnergies Foundation. Christian Thierfelder reports financial support was provided by MAIZE CGIAR Research Program.

Data Availability

Data are fully available and the link to a dataverse was shared in the manuscript.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2022.108207.

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