Environmental Research 212 (2022) 113544

Contents lists available at ScienceDirect

Environmental Research

journal homepage: www.elsevier.com/locate/envres

Effects of water regimes on soil N_2O, CH_4 and CO_2 emissions following addition of dicyandiamide and N fertilizer

Muhammad Shaaban [a], Muhammad Salman Khalid [b], Ronggui Hu [b], Minghua Zhou [a,*]

[a] Key Laboratory of Mountain Surface Processes and Ecological Regulation, Institute of Mountain Hazards and Environment, Chinese Academy of Sciences, 610041, Chengdu, China

[b] College of Resources and Environment, Huazhong Agricultural University, Wuhan, China

ARTICLE INFO

Keywords:
Wetting-dry cycle
Greenhouse gases
Dissolved organic C
Mineral nitrogen
Nitrification inhibitor

ABSTRACT

Water regimes strongly impact soil C and N cycling and the associated greenhouse gases (GHGs, i.e., CO_2, CH_4 and N_2O). Therefore, a study was conducted to examine the impacts of flooding-drying of soil along with application of nitrogen (N) fertilizer and nitrification inhibitor dicyandiamide (DCD) on GHGs emissions. This study comprised four experimental treatments, including (i) control (CK), (ii) dicyandiamide, 20 mg kg^{-1} (DCD), (iii) nitrogen fertilizer, 300 mg kg^{-1} (N) and (iv) DCD + N. All experimental treatments were kept under flooded condition at the onset of the experiment, and then converted to 60% water filled pore space (WFPS). At flooding stage, N_2O emissions were lower as compared to 60% WFPS. The highest cumulative N_2O emission was 0.98 mg N_2O–N kg^{-1} in N treated soil due to high substrates of mineral N contents, but lowest (0.009 mg N_2O–N kg^{-1}) in the DCD treatment. The highest cumulative CH_4 emissions (80.54 mg CH_4–C kg^{-1}) were observed in the N treatment, while uptake of CH_4 was observed in the DCD treatment. As flooded condition converted to 60% WFPS, CO_2 emissions gradually increased in all experimental treatments, but the maximum cumulative CO_2 emission was 477.44 mg kg^{-1} in the DCD + N treatment. The maximum dissolved organic carbon (DOC) contents were observed in N and DCD + N treatments with the values of 57.12 and 58.92 mg kg^{-1}, respectively. Microbial biomass carbon (MBC) contents were higher at flooding while lower at transition phase, and increased at the initiation of 60% WFPS stage. However, MBC contents declined at the later stage of 60% WFPS. The maximum MBC contents were 202.12 and 192.41 mg kg^{-1} in N and DCD + N treatments, respectively. Results demonstrated that water regimes exerted a dramatic impact on C and N dynamics, subsequently GHGs, which were highly controlled by DCD at both flooding and 60% WFPS conditions.

1. Introduction

Climate change and global warming are hot topics at global level because of increased anthropogenic greenhouse gases (GHGs) concentrations in the atmosphere (Celik, 2020). Agricultural soils are recognized as an important source of GHGs (i.e. N_2O, CH_4 and CO_2) (Liu et al., 2019). Several environmental factors affect GHGs emissions, and availability of water can substantially impact emission of these gases through triggering carbon (C) and nitrogen (N) cycling (Shaaban et al., 2014; Shurpali et al., 2019; Wang et al., 2022; Zhou et al., 2014; 2019a, b). Flooding-drying cycle of soil may expose unavailable (physically protected) soil organic matter (SOM) to microbes through breakdown of soil aggregates (Zhang et al., 2020). This previously unavailable SOM may promptly be decomposed and mineralized influencing C and N dynamics (Zhang et al., 2020). Therefore, alternate water regimes can indirectly govern the soil microbial activity and ultimately determine C and N turnover (Law and Lai, 2021; Ran et al., 2020).

Soil C and N turnover plays an imperative role in controlling GHGs emissions (Lin et al., 2021; Wu et al., 2017). Fertilizer application, particularly organic manures and synthetic fertilizers, increases the labile C and N contents in soil systems and thus enlarges GHGs emissions to the atmosphere (Ozlu and Kumar, 2018). Soil C and N dynamics vary with type and rate of fertilization, thereby, the change of C and N contents in relation to GHGs emissions is imperative to study followed by fertilizer application. Chemical fertilizers exert an intensive effect on soil N_2O emissions. Chemical fertilizers provide surplus mineral N that can ultimately lead to N_2O production. Different techniques have been proposed by researchers to mitigate GHGs emissions from soils.

* Corresponding author.
E-mail address: mhuazhou@imde.ac.cn (M. Zhou).

https://doi.org/10.1016/j.envres.2022.113544
Received 21 March 2022; Received in revised form 5 May 2022; Accepted 19 May 2022
Available online 25 May 2022

Nitrification inhibitors application in agricultural soils is one of the good strategies for mitigating N_2O emissions and enhancing N use efficiency (Lam et al., 2018).

Dicyandiamide (DCD) is widely applied nitrification inhibitor in agricultural soils which has the potential to retard the nitrification process by reducing the activities of ammonium-oxidizing bacteria (Jiang et al., 2019; Simon et al., 2018). DCD blocks active sites of ammonia monooxygenase, which is a key enzyme for nitrification, and thus hinders the nitrification by inhibiting the conversion NH_4^+ to NO_3^- (Abbasi and Adams, 2000; Di et al., 2009). The control on nitrification through DCD application can substantially reduce N_2O emissions. In the last few decades, DCD has been used to mitigate N losses as well as to increase N fertilizers use efficiencies in numerous cropping systems (O'Callaghan et al., 2010; Robinson et al., 2014; Simon et al., 2018). Depending on local climatic conditions, the type of crop being grown and the type of soil to which it is applied, N_2O emissions can normally be mitigated by 17–90% (Cahalan et al., 2015; Kelliher et al., 2008; Wang et al., 2015). DCD does not display substantial long-term negative impacts on microbial growth and soil respiration (Ruser and Schulz, 2015). Resulting products from degradation of DCD (i.e. H_2O, NH_3 and CO_2) are harmless to microbial communities. O'Callaghan et al. (2010) and Di and Cameron (2011) revealed that DCD has no substantial effects on soil microbial community. Reports regarding the effects of DCD on soil CO_2 emissions are scarce in the literature. However, a 3-year field experiment demonstrated that DCD application decreased CO_2 emissions by 7% (Weiske et al., 2001b). It is plausible that DCD may affect a specific group of soil microbes and thus disparity exists for GHGs emissions.

Based on the above stated facts, we conjectured that water regimes would alter the turnover of C and N and hence influence emissions of CO_2, CH_4 and N_2O. Further, it was hypothesized that DCD application could retard N transformation and thus N_2O emissions. Therefore, a laboratory study was conducted with the aim to evaluate the effects of different water regimes (wetting-drying cycle), N fertilizer and dicyandiamide on (i) C and N turnover and (ii) coupling their relationships with the GHGs emissions.

2. Materials and methods

2.1. Characteristics of soil, fertilizer and nitrification inhibitor

Soil used in the present study was collected from an arable field underwent the rice-rapeseed rotation. Soil samples were collected at 0–20 cm. Plant debris, stones and earthworms were separated from soil samples (5 samples from selected field, adopting a cross " × " pattern of sampling), and combined to make a composite sample. Dried soil samples were crumbled (2 mm) to use in the experiment. Soil is classified as Ultisols according to Soil-Survey-Staff (2010). The sampled field is located in Xianning, Hubei, China (30°02′16.5′N, 114°22′51.6′E). Some basic soil properties are given in Table 1. Ammonium sulphate [$(NH_4)_2SO_4$)] and dicyandiamide ($C_2H_4N_4$; DCD) were obtained from Sinopharm Chemical Reagent Co. Ltd. China.

2.2. Experimental design

Initially, air-dried soil was incubated for a week at 50% water filled pores space (using distilled water) and 25 °C to actuate microbial activities prior imposing treatments. Activated soil (after 7 days) was treated with the following treatments: control (CK), dicyandiamide (DCD), N fertilizer (N) and DCD + fertilizer (DCD + N). Dicyandiamide was applied at the rate of 20 mg kg^{-1} soil. Fertilizer [$(NH_4)_2SO_4$)] was added in N treatments at the rate of 300 mg kg^{-1}. After treating soil with the described treatments, flooding condition (1:1, v: w) was developed in all experimental treatments. Three replicates of each treatment were prepared and incubated in 1000 mL glass jars with 200 g soil (oven dry equivalent basis) in the dark for 24 days at 25 °C. First 7 days of experiment, all treatments were kept under flooding condition. After that, water in all treatments was allowed to evaporate until moisture reached to 60% WFPS and it took 8 days (Fig. 1). This period was termed as transition phase. The evaporation of soil during transition phase was achieved using the silica gel (50 g) for absorbing the water vapors. Silica gel (50 g) was put into cheesecloth and hanged in the headspace of each jar. When color of silica gel changed from blue to pink, it was replaced with dry one. After transition phase, soil was kept at moisture level equal to 60% WFPS for 9 days. Moreover, two separate but simultaneous set of treatments were prepared for gas analysis and soil properties.

2.3. Soil analysis

A simultaneous set of soil treatments was prepared to analyze soil properties: soil pH, nitrate (NO_3^--N), ammonium (NH_4^+-N), dissolved organic carbon (DOC) and microbial biomass carbon (MBC). For the determination of pH, soil was shaken with the distilled water (1:2.5, soil: water) for an hour. Soil mixture was awaited for 30 min prior to measure pH using a pH-meter (10-PB, Sartorius Agri., Germany) (Shaaban et al., 2013). Soil NH_4^+-N and NO_3^--N were tested by extracting soil with potassium chloride (1 M KCl). Soil mixture was placed on a mechanical shaker for obtaining homogenous mixture and passed through Whatman no. 40 filter paper. The extract was analyzed for NO_3^--N and NH_4^+-N contents using a flow injection analyzer (AutoAnalyzer-3, USA) (Wu et al., 2017). Dissolved organic carbon (DOC) content was analyzed by extracting soil with distilled water (1:5, soil: water). Mixture was placed on rotating shaker for 60 min and subsequently centrifuged at 8000 rpm for 5 min. The supernatant was passed through a 0.45 μm filter membrane and analyzed using a VarioMax Elemental Analyzer. Microbial

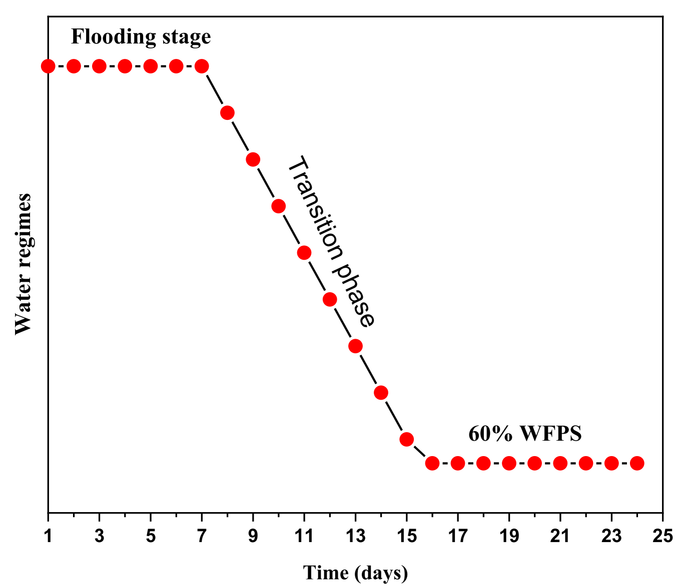

Fig. 1. Illustration of water regimes adopted in the present study.

Table 1
Major properties of soil used in the study.

pH	Total carbon (g kg^{-1})	Total nitrogen (g kg^{-1})	NH_4^+-N (mg kg^{-1})	NO_3^--N (mg kg^{-1})	Bulk density (g cm^{-1})	Clay (%)	Sand (%)	Silt (%)	Texture
5.25	1.40	0.20	75.95	32.28	1.4	30.23	56.93	11.43	Silty clay loam

biomass carbon (MBC) content was determined by chloroform fumigation extraction method (Vance et al., 1987). Water filled pore-space (WFPS) was calculated according to Lin et al. (2013).

2.4. Analysis of headspace gas

Headspace gas samples were collected on daily basis throughout the study (24 days). On each sampling day, soils in jars were placed in open air for at least 20 min to make sure of filling ambient air. After that, jars were tightly closed using lids fixed with a gas-tight rubber septum and a 3-way stopcock to collect headspace gas. One sample from headspace was collected immediately after closure jars, and second was collected after 60 min using a plastic syringe. Gas samples were analyzed for N_2O, CH_4 and CO_2 concentrations using a gas chromatograph equipped with an electron capture detector and flame ionization detector (Agilent7890-A,USA). GHGs emissions were analyzed using the method as described by Yuesi and Yinghong (2003). The fluxes of gases were computed using the ideal gas law and linear regression model at a temperature of 25 °C and an average air pressure during the specified period. Emissions of gases were calculated using following equation as described by Shaaban et al. (2016).

$$F = p \times V/W \times \Delta c/\Delta t \times 273/(T+273)$$

where F is emission rate (μg kg^{-1} h^{-1} for N_2O and CH_4, and mg kg^{-1} h^{-1} for CO_2), p is density of gas at standard conditions, V is jar volume, W is the weight of the soil, Δc is the gas production during the closure time, Δt is closure time of jars, and T is temperature of experiment (25 °C).

Cumulative gas fluxes were calculated using the following equation (Shaaban et al., 2015).

$$\text{Cumulative gas flux} = \sum_{i=1}^{n} (Ri \times 24 \times Di)$$

where Ri is the gas emission rate of the sampling dates, Di is the number of days in the sampling interval, and n is the number of sampling times.

2.5. Statistical analysis

One way analysis of variance (ANOVA) was used to analyze the data. Tukey test was employed to find out significant differences of tested variables among treatments (control, DCD, N and DCD + N). Statistical software SPSS 16.0 was used to analyze data.

3. Results

3.1. Soil N_2O fluxes

Addition of DCD, N and DCD + N displayed significant ($p \leq 0.01$) effects on N_2O emissions. First 7 days of flooding, soil N_2O emission was lower as compared to 60% WFPS (Fig. 2). At the end of transition phase, N_2O emissions markedly increased and reached maximum at the initial stage of onset of 60% WFPS in N, DCD + N and the control treatments. The rise of N_2O emission in DCD only treatment was not significant at the initial stage of 60% WFPS. The highest peaks of N_2O emission were observed (3.129, 2.032 and 1.543 μg N_2O–N kg^{-1} h^{-1}) in N, DCD + N and control, respectively. Afterwards, N_2O emissions gradually decreased in 60% WFPS but remained higher than the flooding stage throughout the study period. The highest cumulative N_2O emission was 0.98 mg N_2O–N kg^{-1} in N treated soil but lowest emission as 0.009 mg N_2O–N kg^{-1} in DCD treatment (Fig. 3).

3.2. Soil CH_4 fluxes

Soil CH_4 emissions were influenced ($p \leq 0.01$) by DCD, N fertilizer and DCD + N treatments. Flooding stage showed higher CH_4 emissions while lower emissions or sometime uptake of CH_4 at 60% WFPS.

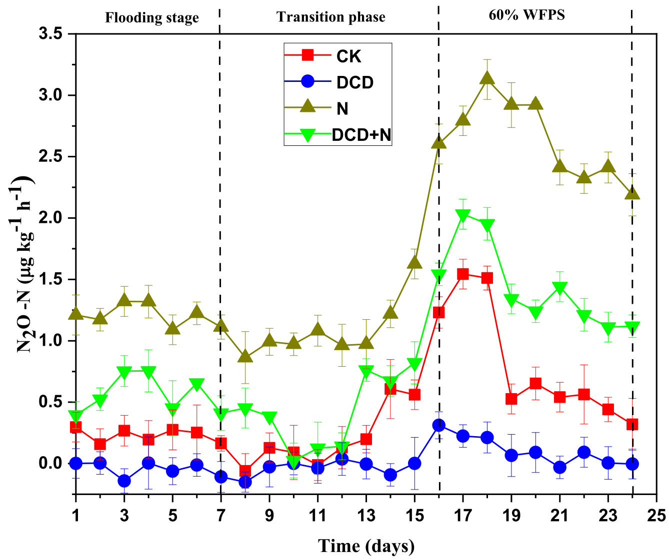

Fig. 2. Soil N_2O emissions from different treatments; CK, control; DCD, Dicyandiamide; N nitrogen fertilizer; and Dicyandiamide + N fertilizer, DCD + N. Note: kg is equivalent to dry soil basis. Vertical bars denote standard error of means (n = 3).

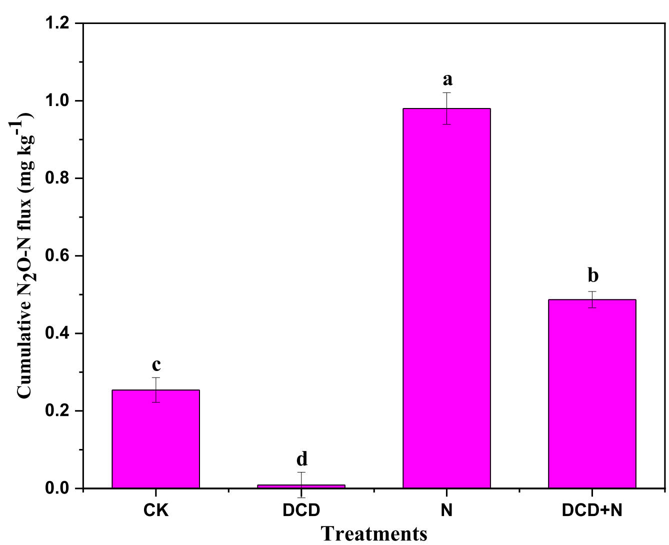

Fig. 3. Cumulative soil N_2O emissions from different treatments; CK, control; DCD, Dicyandiamide; N nitrogen fertilizer; and Dicyandiamide + N fertilizer, DCD + N. Vertical bars denote standard error of means (n = 3).

Addition of fertilizer enlarged CH_4 emissions, and the highest peak (0.36 μg CH_4–C kg^{-1} h^{-1}) was observed in N treatment at 5th day of the experiment (Fig. 4). In case of the control and DCD treatments, CH_4 emissions were lower than N treatment. As moisture decreased (during transition phase), CH_4 emissions declined and CH_4 uptake occurred with maximum of −0.065 μg CH_4–C kg^{-1} h^{-1} in the DCD treatment. The highest cumulative CH_4 emissions of 80.54 and 41.9 mg CH_4–C kg^{-1} were observed in N and DCD + N treatments, respectively, while uptake of CH_4 was observed in DCD only treatment (Fig. 5).

3.3. Soil CO_2 fluxes

Soil CO_2 emissions were affected ($p \leq 0.01$) by N fertilizer, DCD and DCD + N treatments. At the initial stage of flooding, CO_2 emissions were high in all treatments but declined steadily after 5th day. As soil moisture reached 60% WFPS, CO_2 emissions gradually increased in all

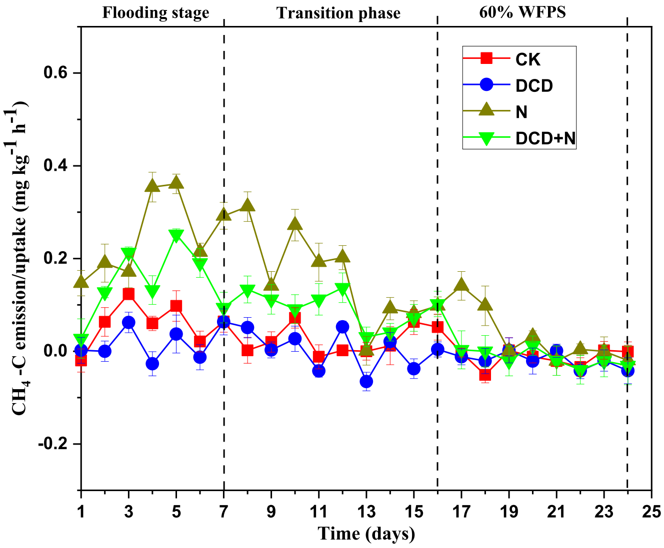

Fig. 4. Soil CH_4 emission/uptake from different treatments; CK, control; DCD, Dicyandiamide; N nitrogen fertilizer; and Dicyandiamide + N fertilizer, DCD + N. Note: kg is equivalent to dry soil basis. Vertical bars denote standard error of means (n = 3).

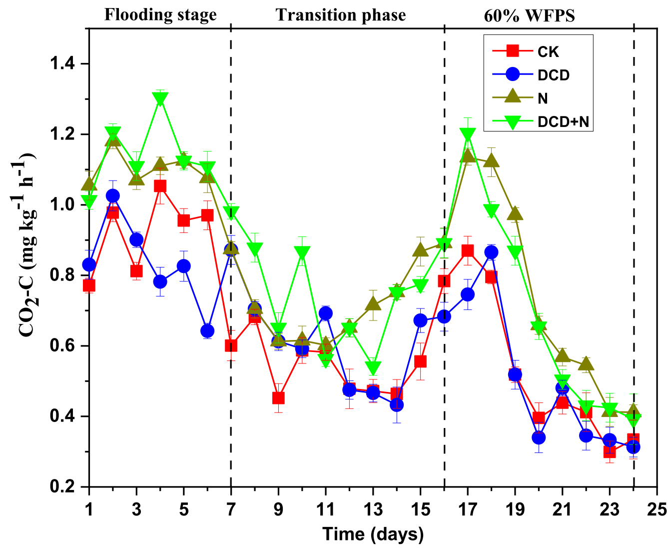

Fig. 6. Soil CO_2 emissions from treatments; CK, control; DCD, Dicyandiamide; N nitrogen fertilizer; and Dicyandiamide + N fertilizer, DCD + N. Note: kg is equivalent to dry soil basis. Vertical bars denote standard error of means (n = 3).

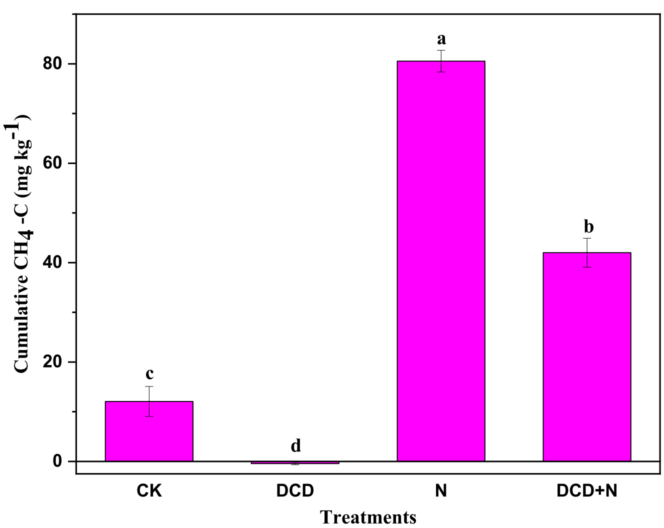

Fig. 5. Cumulative soil CH_4 emissions from different treatments; CK, control; DCD, Dicyandiamide; N nitrogen fertilizer; and Dicyandiamide + N fertilizer, DCD + N. Vertical bars denote standard error of means (n = 3).

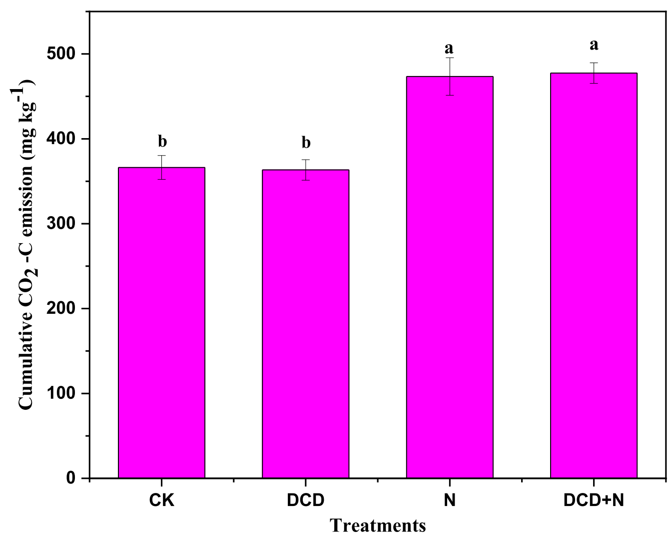

Fig. 7. Cumulative soil CO_2 emissions from different treatments; CK, control; DCD, Dicyandiamide; N nitrogen fertilizer; and Dicyandiamide + N fertilizer, DCD + N. Vertical bars denote standard error of means (n = 3).

treatments, but the peaks were observed at 17th day in N (1.135 mg kg^{-1} h^{-1}) and in DCD + N (1.204 mg kg^{-1} h^{-1}) treatments (Fig. 6). The maximum cumulative CO_2 emission was 477.44 mg kg^{-1} in DCD + N treatment (Fig. 7).

3.4. Soil environmental variables

Addition of N fertilizer and DCD + N significantly ($p \leq 0.01$) influenced NH_4^+-N and NO_3^--N contents. NH_4^+-N contents were higher under flooded conditions than 60% WFPS (Fig. 8). At 60% WFPS stage (16th day), the contents of NH_4^+-N rapidly declined in N treatment and reached at 126.34 mg kg^{-1}, nonetheless, NH_4^+-N contents remained high (163.72 mg kg^{-1}) in DCD + N treatment. Similarly, NH_4^+-N contents were higher in DCD alone treatment when compared with the control.

In contrast to NH_4^+-N, the contents of NO_3^--N were lower under flooded conditions than that of 60% WFPS. NO_3^--N contents increased with the commencement of transition phase and reached the maximum at the end of the study at 60% WFPS stage (Fig. 8). Higher contents of NO_3^--N were observed in the N treatment than that of DCD + N treatment. The maximum NO_3^--N contents were 96.41 mg kg^{-1} in N and 70.34 mg kg^{-1} in DCD + N treatment. In case of control and DCD treatments, the NO_3^--N contents remained low than all other treatments.

Dissolved organic C (DOC) contents were high at the onset of the experiment during flooding stage, but it continuously declined till mid of the transition phase, and then increased with the instigation of 60% WFPS in all treatments (Fig. 9). The maximum DOC contents were observed in N and DCD + N treatments with values of 57.12 and 58.92 mg kg^{-1}, respectively. In case of control (CK) and DCD treatment, DOC remained low throughout the study. Similar trend of MBC content was found as for DOC in all treatments (Fig. 9). MBC content was higher at flooding while lower at transition phase, and increased at the initiation of 60% WFPS stage. However, MBC content declined at the later stage of 60% WFPS. The maximum MBC contents were 202.12 and 192.41 mg

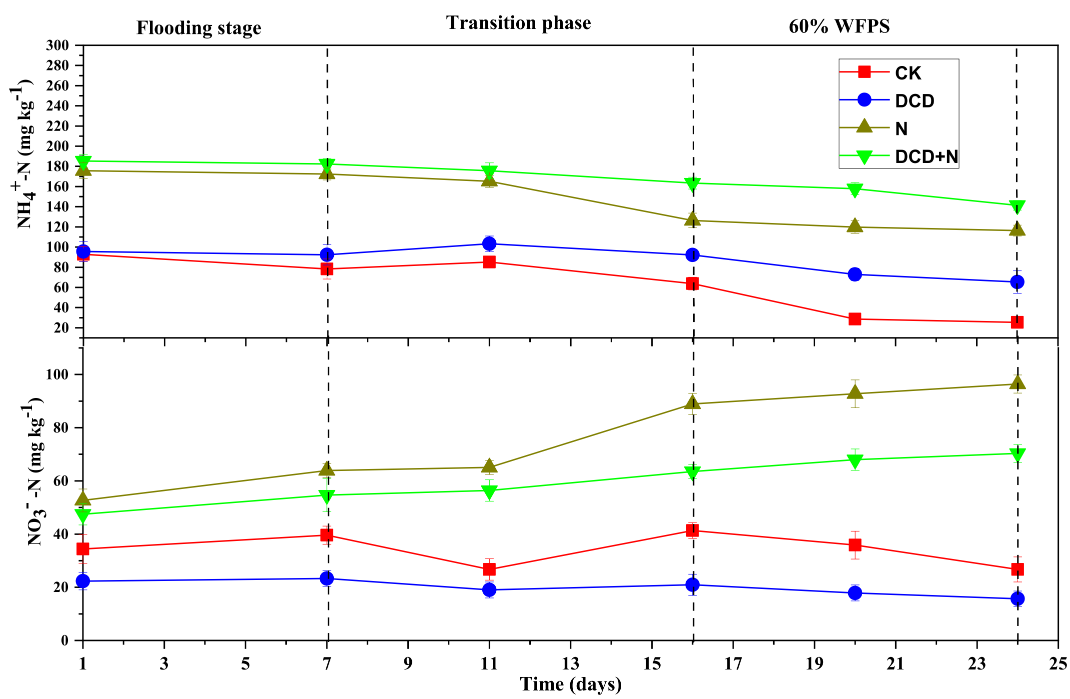

Fig. 8. NH_4^+-N and NO_3^--N contents in soil of various treatments. CK, control; DCD, Dicyandiamide; N nitrogen fertilizer; and Dicyandiamide + N fertilizer, DCD + N. Vertical bars denote standard error of means (n = 3).

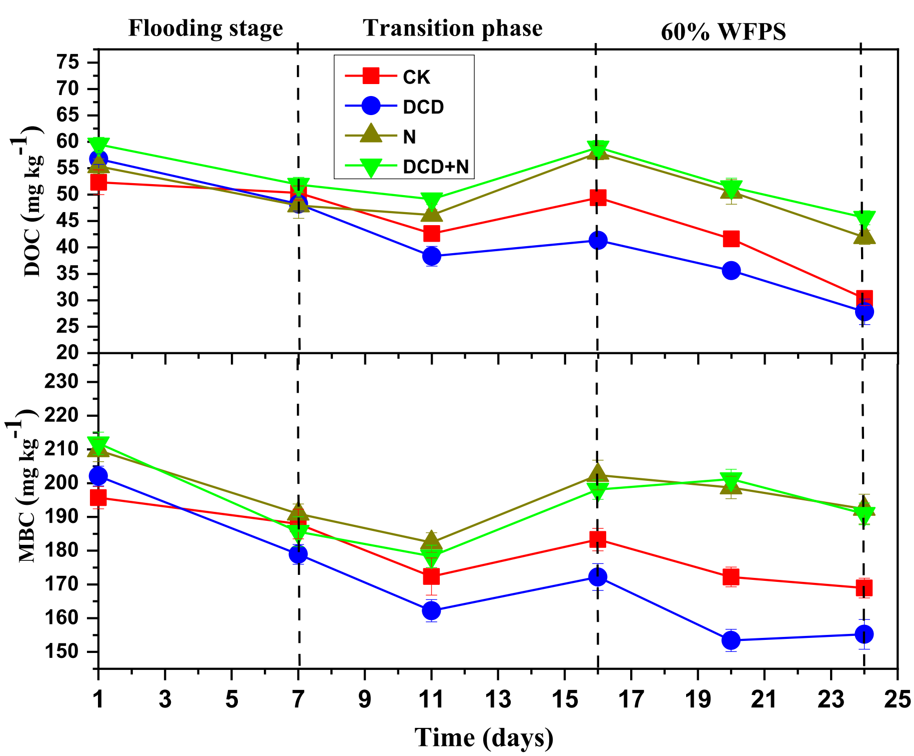

Fig. 9. Soil DOC and MBC contents in different treatments; CK, control; DCD, Dicyandiamide; N nitrogen fertilizer; and Dicyandiamide + N fertilizer, DCD + N. Vertical bars denote standard errors (n = 3).

kg^{-1} in N and DCD + N treatments, respectively.

4. Discussion

4.1. Soil N_2O fluxes

Soil moisture regimes prominently influenced N_2O emissions. At flooding stage, N_2O emissions were relatively lower than 60% WFPS.

Flooding conditions generally induce soil anaerobic environment which consequently influences the biochemical processes limiting mineralization of organic C and N, ultimately less substrates for N_2O production (Li et al., 2022; Neubauer and Megonigal, 2021; Shaaban et al., 2019, 2020). In line to the previous studies reporting low N_2O emissions from flooded paddy fields (Shaaban et al., 2018a; Song et al., 2021; Xia et al., 2020; Xu et al., 2022), N_2O emissions at flooding stage in the present study were eminently lower than 60% WFPS. This resulted from low NO_3^- availability and anaerobic environment due to submerged soil conditions restricting nitrification but supporting denitrification (Chapuis-Lardy et al., 2007). Denitrification is an imperative process which consumes N oxides (such as N_2O) as electron acceptors when oxygen is limited. The most denitrifying bacteria consume oxygen as an electron acceptor, however, they shift on the consumption of N oxides as alternate electron acceptors under limited oxygen conditions (Zumft, 1999). The N_2O-reductase (N_2O-R) is the sole enzyme responsible for the conversion of N_2O to N_2 (Hu et al., 2022). This enzyme is very sensitive to oxygen and thus functionality of this enzyme is accelerated under anoxic soil condition (Jones et al., 2014). Hence, we conjecture that comparatively low N_2O emissions at flooded stage was due to higher functionality of N_2O-R. Higher water contents reinforce anaerobic soil environment favourable for complete denitrification producing N_2 rather than N_2O, and therefore low N_2O emissions (Cai et al., 2013).

In contrast to flooding stage, N_2O emissions were higher at 60% WFPS which is similar to the findings of Qin et al. (2018) and Timilsina et al. (2020) as they revealed that mid-season drainage in paddy field substantially enlarged N_2O emissions. The reason of high N_2O emissions at the initial period of 60% WFPS can be explicated by higher NO_3^--N contents (Heil et al., 2016). Conversion of flooding to 60% WFPS created aerobic environment within the soil and thus conducive to nitrification, producing higher NO_3^- contents and N_2O emissions in all treatments. Moreover, N fertilizer application markedly enlarged N_2O emissions. It is not surprizing pertinent to positive linkage between N fertilizer and N_2O emissions, because higher substrates from N fertilizer (NH_4^+ and NO_3^-) commonly enhance N_2O production (Simon et al., 2018). Therefore, cumulative N_2O emissions were highest in N added soil than all other treatments. Readily available C (such as DOC in the present study) acts as a substrate for soil microbes to perform various functions (Congreves et al., 2018). The larger DOC contents at 60% WFPS increased N_2O emissions. The microbial activity is controlled by readily available C and soil water contents which determine the mineralization of SOM (Wang et al., 2022). Mineralization of native SOM played a key role in N_2O production and emissions by providing substrates to microorganisms in response to changing the flooded condition to 60% WFPS (Congreves et al., 2018; Shaaban et al., 2018b). Microbial biomass is an estimation of growth and proliferation of microbes which increased with the DOC contents signifying that both MBC and DOC were stimulating agents for N_2O fluxes in the present study.

Application of DCD significantly reduced N_2O emissions. DCD is recognized as a good inhibitor for nitrification and has been applied with N fertilizers for several decades (Bronson et al., 1991), and its recommendations for mitigation of N losses are increasing at global level (Adhikari et al., 2021). The plausible explanation of N_2O emission mitigation by DCD application is the retardation of NH_4^+ oxidation (Simon et al., 2020). The decrease in NH_4^+ oxidation rates reduced N_2O emissions in DCD treatments, indicating that it was effective for lowering N_2O emissions. DCD increased the persistence of NH_4^+ and thereby favored minimizing N_2O emissions (Simon et al., 2020).

4.2. Soil CH_4 fluxes

Flooded stage of the experiment produced CH_4 emissions rather than uptake. Submerged paddy fields have demonstrated the increased CH_4 emissions (Liu et al., 2021; Vo et al., 2018). Flooding condition hampers oxygen exchange between soil and atmosphere and creates anaerobic environment within the soil which constraints methanotrophy (oxidation of CH_4) and thus promotes methanogens to produce CH_4 (Meng et al., 2014). We conjecture that anaerobic conditions at flooding stage of the experiment promoted methanogenic activities while suppressed methanotrophic activities leading to CH_4 production.

Moisture affects the solubility of organic carbon and availability to microbes for their metabolism and growth (Kannan et al., 2021). High soil moisture favored the decomposition of native SOM and thus stimulated the methanogens to produce CH_4. DOC and MBC contents were higher at flooding stage than that of 60% WFPS speculating the decomposition of native SOM. Sufficient supply of C enhanced the activities of methanogens which produced CH_4 at flooding stage. Alteration of flooded to 60% WFPS decreased CH_4 emission. Many studies reported this phenomenon that alternating wetting and drying or mid-season drainage markedly lowers CH_4 emission (Codruta Maris, 2015). Our results showed that cumulative CH_4 emissions were reduced by 24% at 60% WFPS. Shi et al. (2017) documented that CH_4 emissions were decreased by 44% during mid-season drainage, while 61% reduction achieved through alternate wetting and drying at an interval of 10 days when compared with the constantly flooded soil. Itoh et al. (2011) stated that mid-season drainage of paddy field lowered CH_4 emission up to 69.5%. This decrease can be attributed to temporary soil aeration created by partial drying of the soil that hampers methanogenic process while stimulates methanotrophy.

Addition of N fertilizer enlarged CH_4 emissions. Nitrogen fertilizers provide substrates for microbes (such as methanogens) to produce CH_4. Nevertheless, it is not confirm regarding methanotrophic process in NH_4^+-fertilizer amended soils. Application of N fertilizer have shown diverse effects on CH_4 production; inhibitory, stimulatory and no effects (Dan et al., 2001; Sun et al., 2016). Nitrogen fertilizer augmented NH_4^+ contents both at flooding and 60% WFPS stages and produced high emissions of CH_4 in the current study. Hawthorne et al. (2017) reported that the growth and activity of methanotrophs were stimulated by NH_4^+ contents which increased CH_4 by the relief of N limitation for CH_4 oxidizing bacteria under N-limiting environments. Obviously, NH_4^+ and NO_3^- concentrations were ominously high in N fertilized soil which probably favored growth and activities of methanogens leading to larger CH_4 emissions in the fertilized soil. However, DCD application decreased cumulative CH_4 emissions when compared to the N and control treatments. Bharati et al. (2000) conducted a laboratory study and reported that DCD potentially reduced CH_4 emissions with up to 47%. Another study revealed that application of DCD reduced CH_4 emissions up to 30% in a rice-wheat field (Bayer et al., 2015). Moreover, nitrifiers can concurrently consume and oxidize CH_4 and NH_4^+, but the DCD application reduced conversion of NH_4^+ and, therefore, CH_4 was consumed rather than NH_4^+ by nitrifiers in DCD applied soil. This was possibly because NH_4^+ and CH_4 molecules bear similar size that leads a competing inhibition between NH_4^+ and CH_4 oxidation.

4.3. Soil CO_2 fluxes

Soil CO_2 emissions were high at flooding condition. Similar results have been reported by earlier studies that CO_2 enlarged with an increase in soil moisture in croplands (Beare et al., 2009), grazing pastures (Blodau and Moore, 2003), peat lands (Goldhammer and Blodau, 2008) and deserts (Sponseller and Fisher, 2008). The higher soil respiration rates generally occur at flooded soil rather than dry soil. Microbial activities are increased with the increase of soil water content (Marzaioli et al., 2022). Increased soil moisture augmented DOC and consequently microbes in soil consumed DOC as substrates and produced high CO_2 emissions in flooded soil. Similar results have also been proposed in an earlier study where increasing soil moisture increased DOC contents (Kalbitz et al., 2000). The processes and mechanisms involved in increased CO_2 emissions at high soil moisture are: microbial decomposition of organic matter, and availability and use of mineralized C and N by microbes (Xiao et al., 2019).

Soil CO_2 emissions gradually decreased at flooding stage and were lowest at the mid of transition phase. At the end of transition phase but at the initiation of 60% WFPS, CO_2 emissions steadily increased and reached at the peak in all experimental treatments. In line to these results, Sánchez-Andrés et al. (2010) also found that CO_2 emissions enlarged when flooding soil changed to wet soil. High CO_2 emissions following the conversion of flooding to 60% WFPS can be explained by simultaneous higher DOC contents. Dynamics of moisture regimes definitely decompose native SOM and consequently increase DOC which could possibly enlarge CO_2 emissions (Yan et al., 2022; Yang et al., 2019). An experiment conducted by Masyagina et al. (2017) revealed that drainage of water triggered microbial activity and increased soil CO_2 emissions. Microbial biomass carbon and DOC contents substantially increased when soil moisture changed from flooding to 60% WFPS. The emissions of CO_2 from soils have been identified to be dependent on readily available C. Thus, it seemed that change of flooding to 60% WFPS considerably increased soil CO_2 emissions because of higher DOC contents.

Fertilizer application also exerted an increasing trend on soil CO_2 emissions. Huang et al. (2014) found that CO_2 emissions under fertilized treatments were 2-folds as compared to non-fertilized. Nevertheless, application of N fertilizer had also shown contradictory effects on soil CO_2 emissions (Ding et al., 2007). We conjectured that N fertilizer had positive effect on augmenting soil organic C content, which in turn released into the atmosphere in the form of CO_2 (Lin et al., 2021). It is also reported that fertilizer application influences microbial activities that are directly related to CO_2 emissions (Xiao et al., 2018). Soil treated with fertilizer had higher contents of MBC as compared to non-fertilized soil. These indications revealed that microbial growth and activities were accelerated following fertilizer application which resulted in CO_2 production.

Application of DCD did not prominently influence CO_2 emissions. Previous research has documented that DCD had not significant effects on CO_2 emissions (Jagrati et al., 2008). This can be explained by no effects of DCD on soil microbial community (Guo et al., 2013; Zaman and Blennerhassett, 2010). The findings of O'Callaghan et al. (2010) confirmed that DCD did not affect soil microbial communities. Tamir et al. (2013) examined DCD effects on a specific group of the soil microbial community and found no significant effects. Results of a 3-year field study showed that application of DCD decreased CO_2 emissions by 7% (Weiske et al., 2001a). Another study reported that DCD minimizes CO_2 emission from acidic soils (Elrys et al., 2020). Raza et al. (2021) also stated that DCD significantly decreased CO_2 emissions from calcareous soils. No abundant reports of DCD effects on soil respiration and mineralization are found and, thus, results need to be further tested under field conditions.

5. Conclusion

This study demonstrated that the change in water regime had varied effects on GHGs emissions. The differences in GHGs emissions at flooding and 60% WFPS were controlled by availability of moisture, mineral N, DOC and MBC contents. When soil moisture content changed from flooding to 60% WFPS, CH_4 emissions decreased, but N_2O and CO_2 substantially increased. Addition of N fertilizer also stimulated N_2O, CH_4 and CO_2 fluxes but mitigated with application of DCD. In short, water regimes has a strong regulating effects on C and N dynamics and, thus, emissions of GHGs. Further research is suggested to explore the inhibitory effects of nitrification inhibitors on GHGs in the field using different soil types.

Author's contribution

Muhammad Shaaban and Muhammad Salman Khalid conceived and executed research work, Muhammad Shaaban, Ronggui Hu and Minghua Zhou analyzed data and wrote the manuscript.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgement

The present research was funded by the National Natural Science Foundation of China (Grant No. U20A20107), the Regional Key STS Project of the Chinese Academy of Sciences (Grant No. KFJ-STS-QYZD-2021-24-001) and the Chinese Academy of Sciences President's International Fellowship Initiative (Grant No. 2020PC0066).

References

Abbasi, M.K., Adams, W.A., 2000. Gaseous N emission during simultaneous nitrification–denitrification associated with mineral N fertilization to a grassland soil under field conditions. Soil Biol. Biochem. 32, 1251–1259.

Adhikari, K.P., et al., 2021. Management and implications of using nitrification inhibitors to reduce nitrous oxide emissions from urine patches on grazed pasture soils–A review. Sci. Total Environ. 791, 148099.

Bayer, C., et al., 2015. A seven-year study on the effects of fall soil tillage on yield-scaled greenhouse gas emission from flood irrigated rice in a humid subtropical climate. Soil Tillage Res. 145, 118–125.

Beare, M.H., et al., 2009. Compaction effects on CO_2 and N_2O production during drying and rewetting of soil. Soil Biol. Biochem. 41, 611–621.

Bharati, K., et al., 2000. Influence of six nitrification inhibitors on methane production in a flooded alluvial soil. Nutrient Cycl. Agroecosyst. 58, 389–394.

Blodau, C., Moore, T.R., 2003. Micro-scale CO_2 and CH_4 dynamics in a peat soil during a water fluctuation and sulfate pulse. Soil Biol. Biochem. 35, 535–547.

Bronson, K., et al., 1991. Nitrogen-15 recovery in winter wheat as affected by application timing and dicyandiamide. Soil Sci. Soc. Am. J. 55, 130–135.

Cahalan, E., et al., 2015. The effect of the nitrification inhibitor dicyandiamide (DCD) on nitrous oxide and methane emissions after cattle slurry application to Irish grassland. Agric. Ecosyst. Environ. 199, 339–349.

Cai, Y., et al., 2013. Nitrous oxide emissions from Chinese maize–wheat rotation systems: a 3-year field measurement. Atmos. Environ. 65, 112–122.

Celik, S., 2020. The Effects of Climate Change on Human Behaviors. Environment, Climate, Plant and Vegetation Growth. Springer, pp. 577–589.

Chapuis-Lardy, L., et al., 2007. Soils, a sink for N_2O? A review. Global Change Biol. 13, 1–17.

Codruta Maris, S., 2015. Effect of Nitrogen Fertilization and Water Management of GHGs (N_2O, CO_2 and CH_4) Emissions from Intensive Mediterranean Agricultural Systems. Universitat de Lleida.

Congreves, K.A., et al., 2018. Nitrous oxide emissions and biogeochemical responses to soil freezing-thawing and drying-wetting. Soil Biol. Biochem. 117, 5–15.

Dan, J., et al., 2001. Effect of a Late Season Urea Fertilization on Methane Emission from a Rice Field in Italy, vol. 83. Agriculture Ecosystems & Environment, pp. 191–199.

Di, H., Cameron, K., 2011. How does the application of different nitrification inhibitors affect nitrous oxide emissions and nitrate leaching from cow urine in grazed pastures? Soil Use Manag. 28, 54–68.

Di, H.J., et al., 2009. Nitrification driven by bacteria and not archaea in nitrogen-rich grassland soils. Nat. Geosci. 2, 621–624.

Ding, W., et al., 2007. CO_2 emission in an intensively cultivated loam as affected by long-term application of organic manure and nitrogen fertilizer. Soil Biol. Biochem. 39, 669–679.

Elrys, A.S., et al., 2020. Do soil property variations affect dicyandiamide efficiency in inhibiting nitrification and minimizing carbon dioxide emissions? Ecotoxicol. Environ. Saf. 202, 110875.

Goldhammer, T., Blodau, C., 2008. Desiccation and product accumulation constrain heterotrophic anaerobic respiration in peats of an ombrotrophic temperate bog. Soil Biol. Biochem. 40, 2007–2015.

Guo, Y.J., et al., 2013. Effect of 7-year application of a nitrification inhibitor, dicyandiamide (DCD), on soil microbial biomass, protease and deaminase activities, and the abundance of bacteria and archaea in pasture soils. J. Soils Sediments 13, 753–759.

Hawthorne, I., et al., 2017. Application of biochar and nitrogen influences fluxes of CO_2, CH_4 and N_2O in a forest soil. J. Environ. Manag. 192, 203–214.

Heil, J., et al., 2016. A review of chemical reactions of nitrification intermediates and their role in nitrogen cycling and nitrogen trace gas formation in soil. Eur. J. Soil Sci. 67, 23–39.

Hu, L., et al., 2022. NosZ gene cloning, reduction performance and structure of Pseudomonas citronellolis WXP-4 nitrous oxide reductase. RSC Adv. 12, 2549–2557.

Huang, J.Y., et al., 2014. Effects of nitrogen fertilization on soil labile carbon fractions of freshwater marsh soil in Northeast China. Int. J. Environ. Sci. Technol. 11, 2009–2014.

Itoh, M., et al., 2011. Mitigation of methane emissions from paddy fields by prolonging midseason drainage. Agric. Ecosyst. Environ. 141, 359–372.

Jagrati, S., et al., 2008. Decomposition of dicyandiamide (DCD) in three contrasting soils and its effect on nitrous oxide emission, soil respiratory activity, and microbial biomass-an incubation study. Aust. J. Soil Res. 46, 517–525.
Jiang, J., et al., 2019. Beneficial influences of pelelith and dicyandiamide on gaseous emissions and the fungal community during sewage sludge composting. Environ. Sci. Pollut. Control Ser. 26, 8928–8938.
Jones, C.M., et al., 2014. Recently identified microbial guild mediates soil N_2O sink capacity. Nat. Clim. Change 4, 801–805.
Kalbitz, K., et al., 2000. Controls on the dynamics of dissolved organic matter in soils: a review. Soil Sci. 165, 277–304.
Kannan, P., et al., 2021. Applying both biochar and phosphobacteria enhances Vigna mungo L. growth and yield in acid soils by increasing soil pH, moisture content, microbial growth and P availability. Agric. Ecosyst. Environ. 308, 107258.
Kelliher, F., et al., 2008. The temperature dependence of dicyandiamide (DCD) degradation in soils: a data synthesis. Soil Biol. Biochem. 40, 1878–1882.
Lam, S.K., et al., 2018. Direct and indirect greenhouse gas emissions from two intensive vegetable farms applied with a nitrification inhibitor. Soil Biol. Biochem. 116, 48–51.
Law, M.M.S., Lai, D.Y.F., 2021. Impacts of wetting-drying cycles on short-term carbon and nitrogen dynamics in Amynthas earthworm casts. Pedosphere 31, 423–432.
Li, Z., et al., 2022. Variations and controlling factors of soil denitrification rate. Global Change Biol. 28, 2133–2145.
Lin, S., et al., 2013. Nitrous oxide emissions from yellow brown soil as affected by incorporation of crop residues with different carbon-to-nitrogen ratios: a case study in central China. Arch. Environ. Contam. Toxicol. 65, 183–192.
Lin, S., et al., 2021. Effects of inorganic and organic fertilizers on CO_2 and CH_4 fluxes from tea plantation soil. Elementa: Science of the Anthropocene 9, 90–99.
Liu, F., et al., 2021. Resilience of methane cycle and microbial functional genes to drought and flood in an alkaline wetland: a metagenomic analysis. Chemosphere 265, 129034.
Liu, Y., et al., 2019. Emission mechanism and reduction countermeasures of agricultural greenhouse gases–a review. Greenhouse Gases: Sci. Technol. 9, 160–174.
Marzaioli, R., et al., 2022. Soil microbial biomass, activities and diversity in Southern Italy areas chronically exposed to trace element input from industrial and agricultural activities. Appl. Soil Ecol. 174, 104392.
Masyagina, O., et al., 2017. Soil CO_2 Emission, Microbial Activity, C and N after Landsliding Disturbance in Permafrost Area of Siberia. World Landslide Forum.
Meng, H.N., et al., 2014. Response of CH_4 emission to moss removal and N addition in boreal peatland of Northeast China. Biogeosciences 11, 4809–4816.
Neubauer, S.C., Megonigal, J.P., 2021. Biogeochemistry of Wetland Carbon Preservation and Flux. Wetland Carbon and Environmental Management, pp. 33–71.
O'Callaghan, M., et al., 2010. Effect of the nitrification inhibitor dicyandiamide (DCD) on microbial communities in a pasture soil amended with bovine urine. Soil Biol. Biochem. 42, 1425–1436.
Ozlu, E., Kumar, S., 2018. Response of surface GHG fluxes to long-term manure and inorganic fertilizer application in corn and soybean rotation. Sci. Total Environ. 626, 817–825.
Qin, H., et al., 2018. Abundance of transcripts of functional gene reflects the inverse relationship between CH_4 and N_2O emissions during mid-season drainage in acidic paddy soil. Biol. Fertil. Soils 54, 885–895.
Ran, Y., et al., 2020. Physicochemical determinants in stabilizing soil aggregates along a hydrological stress gradient on reservoir riparian habitats: implications to soil restoration. Ecol. Eng. 143, 105664.
Raza, S., et al., 2021. Dicyandiamide efficacy of inhibiting nitrification and carbon dioxide emission from calcareous soil depends on temperature and moisture contents. Arch. Agron Soil Sci. 1–17.
Robinson, A., et al., 2014. The effect of soil pH and dicyandiamide (DCD) on N_2O emissions and ammonia oxidiser abundance in a stimulated grazed pasture soil. J. Soils Sediments 14, 1434–1444.
Ruser, R., Schulz, R., 2015. The effect of nitrification inhibitors on the nitrous oxide (N_2O) release from agricultural soils—a review. J. Plant Nutr. Soil Sci. 178, 171–188.
Sánchez-Andrés, R., et al., 2010. Do changes in flood pulse duration disturb soil carbon dioxide emissions in semi-arid floodplains? Biogeochemistry 101, 257–267.
Shaaban, M., et al., 2013. Short term influence of gypsum, farm manure and commercial humic acid on physical properties of salt affected soil in rice paddy system. J. Chem. Soc. Pakistan 35, 1034–1040.
Shaaban, M., et al., 2014. Nitrous oxide emission from two acidic soils as affected by dolomite application. Soil Res. 52, 841–848.
Shaaban, M., et al., 2015. Dolomite application to acidic soils: a promising option for mitigating N2O emissions. Environ. Sci. Pollut. Res. 22, 1–10.
Shaaban, M., et al., 2016. Effects of dicyandiamide and dolomite application on N_2O emission from an acidic soil. Environ. Sci. Pollut. Control Ser. 23, 6334–6342.
Shaaban, M., et al., 2018a. Reduction in soil N_2O emissions by pH manipulation and enhanced nosZ gene transcription under different water regimes. Environ. Pollut. 235, 625–631.
Shaaban, M., et al., 2018b. The interactive effects of dolomite application and straw incorporation on soil N_2O emissions. Eur. J. Soil Sci. 69, 502–511.
Shaaban, M., et al., 2019. Restoring effect of soil acidity and Cu on N_2O emissions from an acidic soil. J. Environ. Manag. 250, 109535.
Shaaban, M., et al., 2020. The effects of pH change through liming on soil N_2O emissions. Processes 8, 702–709.
Shi, H., et al., 2017. Effects of grazing on CO_2, CH_4, and N_2O fluxes in three temperate steppe ecosystems. Ecosphere 8, e01760.
Shurpali, N., et al., 2019. Introduction to Greenhouse Gas Emissions. Greenhouse Gas Emissions. Springer, pp. 1–5.
Simon, P.L., et al., 2018. Nitrous oxide emission factors from cattle urine and dung, and dicyandiamide (DCD) as a mitigation strategy in subtropical pastures. Agric. Ecosyst. Environ. 267, 74–82.
Simon, P.L., et al., 2020. Does Brachiaria humidicola and dicyandiamide reduce nitrous oxide and ammonia emissions from cattle urine patches in the subtropics? Sci. Total Environ. 720, 137692.
Soil-Survey-Staff, 2010. Keys to Soil Taxonomy, eleventh ed. USDA Natural Resources Conservation Service, Washington, DC.
Song, K., et al., 2021. Evaluation of methane and nitrous oxide emissions in a three-year case study on single rice and ratoon rice paddy fields. J. Clean. Prod. 297, 126650.
Sponseller, R.A., Fisher, S.G., 2008. The influence of drainage networks on patterns of soil respiration in a desert catchment. Ecology 89, 1089–1100.
Sun, H., et al., 2016. CH4 emission in response to water-saving and drought-resistance rice (WDR) and common rice varieties under different irrigation managements. Water, Air, Soil Pollut. 227, 1–12.
Tamir, G., et al., 2013. Organic N mineralization and transformations in soils treated with animal waste in relation to carbonate dissolution and precipitation. Geoderma 209, 50–56.
Timilsina, A., et al., 2020. Nitrous oxide emissions from paddies: understanding the role of rice plants. Plants 9, 180.
Vance, E.D., et al., 1987. An extraction method for measuring soil microbial biomass C. Soil Biol. Biochem. 19, 703–707.
Vo, T.B.T., et al., 2018. Methane emission from rice cultivation in different agro-ecological zones of the Mekong river delta: seasonal patterns and emission factors for baseline water management. Soil Sci. Plant Nutr. 64, 47–58.
Wang, X., et al., 2015. Optimizing net greenhouse gas balance of a bioenergy cropping system in southeast China with urease and nitrification inhibitors. Ecol. Eng. 83, 191–198.
Wang, Y., et al., 2022. Risk assessment of rainstorm disasters in the Guangdong–Hong Kong–Macao greater Bay area of China during 1990–2018. Geomatics, Nat. Hazards Risk 13, 267–288.
Weiske, A., et al., 2001a. Influence of the nitrification inhibitor 3,4-dimethylpyrazole phosphate (DMPP) in comparison to dicyandiamide (DCD) on nitrous oxide emissions, carbon dioxide fluxes and methane oxidation during 3 years of repeated application in field experiments. Biol. Fertil. Soils 34, 109–117.
Weiske, A., et al., 2001b. Influence of the nitrification inhibitor 3, 4-dimethylpyrazole phosphate (DMPP) in comparison to dicyandiamide (DCD) on nitrous oxide emissions, carbon dioxide fluxes and methane oxidation during 3 years of repeated application in field experiments. Biol. Fertil. Soils 34, 109–117.
Wu, L., et al., 2017. Conversion from rice to vegetable production increases N_2O emission via increased soil organic matter mineralization. Sci. Total Environ. 583, 190–201.
Xia, L., et al., 2020. Simultaneous quantification of N_2, NH_3 and N_2O emissions from a flooded paddy field under different N fertilization regimes. Global Change Biol. 26, 2292–2303.
Xiao, D., et al., 2019. Effects of tillage on CO_2 fluxes in a typical karst calcareous soil. Geoderma 337, 191–201.
Xiao, Y., et al., 2018. Effects of biochar, N fertilizer, and crop residues on greenhouse gas emissions from acidic soils. Clean 46, 1700346.
Xu, X., et al., 2022. Biochar derived from spent mushroom substrate reduced N_2O emissions with lower water content but increased CH_4 emissions under flooded condition from fertilized soils in Camellia oleifera plantations. Chemosphere 287, 132110.
Yan, W., et al., 2022. Water level regulates the rhizosphere priming effect on SOM decomposition of peatland soil. Rhizosphere 21, 100455.
Yang, X.-D., et al., 2019. Soil moisture and salinity as main drivers of soil respiration across natural xeromorphic vegetation and agricultural lands in an arid desert region. Catena 177, 126–133.
Yuesi, W., Yinghong, W., 2003. Quick measurement of CH_4, CO_2 and N_2O emissions from a short-plant ecosystem. Adv. Atmos. Sci. 20, 842–844.
Zaman, M., Blennerhassett, J.D., 2010. Effects of the different rates of urease and nitrification inhibitors on gaseous emissions of ammonia and nitrous oxide, nitrate leaching and pasture production from urine patches in an intensive grazed pasture system. Agric. Ecosyst. Environ. 136, 236–246.
Zhang, S., et al., 2020. Responses of soil carbon decomposition to drying-rewetting cycles: a meta-analysis. Geoderma 361, 114069.
Zhou, M., et al., 2014. Nitrous oxide emissions during the non-rice growing seasons of two subtropical rice-based rotation systems in southwest China. Plant Soil 383, 401–414.
Zhou, M., et al., 2019a. Effects of afforestation on soil nitrous oxide emissions in a subtropical montane agricultural landscape: a 3-year field experiment. Agric. For. Meteorol. 266, 221–230.
Zhou, M., et al., 2019b. Afforestation and deforestation enhanced soil CH_4 uptake in a subtropical agricultural landscape: evidence from multi-year and multi-site field experiments. Sci. Total Environ. 662, 313–323.
Zumft, W.G., 1999. The denitrifying prokaryotes. In: Dworkin, M., Falkow, S., Rosenberg, E., Schleifer, K.-H., Stackebrandt, E. (Eds.), The Prokaryotes: an Evolving Electronic Resource for the Microbiological Community., third ed. Springer-Verlag, New York.