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Assessment of greenhouse gases emissions, global warming potential and net ecosystem economic benefits from wheat field with reduced irrigation and nitrogen management in an arid region of China

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A R T I C L E I N F O

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A B S T R A C T

In the context of global warming, water scarcity, and high fertilizer costs, identifying proper irrigation and nitrogen (N) management is essential for balancing emissions of greenhouse gases (GHGs), crop production, and economic benefits in arid regions. A two-year field study was conducted to determine the interactive effects of irrigation regimes (I_{100}, 750 mm; $I_{80,}$ 600 mm and $I_{60,}$ 450 mm) and N application rates (N_{100}, 300 kg ha^{-1}; N_{75}, 225 kg ha^{-1}; and $N_{50,}$ 150 kg ha^{-1}) on GHGs emissions, global warming potential (GWP), greenhouse gas intensity (GHGI), and net ecosystem economic benefits (NEEB) from spring wheat field in Northwest China. Our results depicted the lowest cumulative GHGs emissions and GWP with low irrigation and low N application rates but at the cost of a significant decrease in grain yield and NEEB compared to full irrigation and N application. Instead, moderate irrigation and N application ($I_{80}N_{75}$) showed the most promising effects by significantly reducing GHGs emissions while improving the NEEB and yield benefits. Compared to $I_{100}N_{100}$, the $I_{80}N_{75}$ treatment decreased cumulative N_2O emissions by 43.2 % and 33.1 %, CO_2 by 28.8 % and 26.2 %, GWP by 46.8 % and 35.6 %, and GHGI by 55.0 % and 43.5 % in 2015 and 2016, respectively. In addition, the $I_{80}N_{75}$ treatment improved the NEEB by 79.3 % and 78.6 % and grain yield by 15.2 % and 16.1 % in 2015 and 2016 compared to $I_{100}N_{100,}$ respectively. The wheat field acted as a CH_4 sink regardless of all the treatments. Soil moisture and inorganic N contents were the primary driving factors that influenced the GHGs emissions from the wheat field, manifested by their significant correlations. Overall, 600 mm irrigation and 225 kg N ha^{-1} fertilization is an effective strategy with a good trade-off between economic benefits and environmental performance from spring wheat fields in the arid region of Northwest China.

1. Introduction

In recent decades, global warming as a result of increased greenhouse gases (GHGs) emissions has become a serious environmental concern (Alhajj Ali et al., 2017; Zhong et al., 2021). Globally, agriculture is considered the largest anthropogenic contributor of GHGs (Feng et al., 2021, 2018), with annual emissions reaching 5.0–5.8 Gt CO_2-eq in 2000–2010 (Shi et al., 2021). The agricultural sources account for 84 % and 52 % of the global anthropogenic nitrous oxide (N_2O) and methane (CH_4) emissions, respectively (Zheng et al., 2021). Given the limited land resources, agriculture intensification is an effective strategy for ensuring food security for the world's growing population (Kamran et al., 2018), but is intimately linked to significant GHGs emissions (Tan et al., 2017). Nevertheless, agricultural practices have the potential to mitigate their impact on climate change (Meijide et al., 2017). Therefore, understanding their impacts and quantification of proper

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agronomic activities for balancing crop production and soil GHGs emissions is essential for sustainable agricultural production.

North China is a major cereal production region, accounting for 25 % of the country's total agricultural area but providing 66 % of the total wheat production (Zhang et al., 2020). In this region, frequent irrigation and high nitrogen (N) fertilization is widely adopted to increase wheat production (Tan et al., 2017; Wang et al., 2016). However, these intensive practices have triggered increased emissions of GHGs (Liu et al., 2011; Zhang et al., 2020). Irrigation amounts regulate the spatial and temporal distribution of soil moisture, a major driver of GHGs emissions because it governs carbon (C) and N mineralization, and microbial activities (Li et al., 2020; Oertel et al., 2016; Scheer et al., 2013). Over-irrigation restricts soil aeration and increases N_2O and CH_4 emissions via denitrification and anaerobic organic matter decomposition (Mateo-Marín et al., 2020). In addition, irrigation volumes regulate CO_2 fluxes by promoting plant biomass and soil microbial activities (Scheer et al., 2013). The irrigation events are likely to result in greater soil CO_2 emissions if the soil is less frequently irrigated or receives less precipitation (Sapkota et al., 2020; Zornoza et al., 2016). Recent studies have suggested an overall shift toward reduced irrigation strategies in water-deficit regions to mitigate GHGs emissions by optimizing soil N and carbon turnover (Kuang et al., 2018; Li et al., 2018; Wang et al., 2016; Zhang et al., 2020). For instance, lower CO_2 and N_2O emissions were reported from wheat fields with deficit irrigations regimes but resulted in varying degrees of yield reduction (Wang et al., 2019) and monsoon climatic conditions (Hou et al., 2019; Zhong et al., 2021). However, it is uncertain how reduced irrigation will affect GHGs emissions and wheat production in arid regions of North China where irrigation meets about 90 % of crop water demands, therefore, finding an appropriate site-specific irrigation strategy is essential.

In modern agriculture, N fertilizers are critical for increasing crop yield and productivity (Feng et al., 2016; Liu et al., 2011). The average N consumption in North China reaches 300 kg ha^{-1} for cereal crops (Li et al., 2020). Unfortunately, crop N use efficiency is very low (~35 %), resulting in up to 70 % fertilizer losses to the environment (Li et al., 2020; Zhang et al., 2020). Nitrogen fertilization enriches soil NO_3-N and NH_4-N pool, subject to enhance nitrification and denitrification processes and contributes to the increase in soil GHGs emissions (Abalos et al., 2014; Yang et al., 2019). Recent studies have indicated that N fertilization contributes 75 % of the total GHGs emissions in wheat production (Alhajj Ali et al., 2017; Zhang et al., 2021). A positive relation between N_2O emissions and N application was generally observed when the input exceeded the optimal rates (Millar et al., 2018; Shcherbak et al., 2014). On the contrary, reasonable N application is supposed with no significant effects on soil N_2O and other GHGs emissions (Li et al., 2020; Yu et al., 2021). The increase in CH_4 emissions with N fertilization is associated with the inhibition of methane monooxygenase enzyme activity and osmotic pressure caused by high NO_3^- and NH_4^+ concentrations (Geng et al., 2017). Whereas the response of soil CO_2 emissions to N fertilization is dependent on soil organic matter, N application promotes soil respiration and increases CO_2 emissions under sufficient C source availability but inhibits it when C is deficient (Pareja-Sánchez et al., 2019; Yang et al., 2019). Recently, a 30 % reduction in farmers' traditional N application has been proposed for reducing 17 % of China's total GHGs emissions from cereals without affecting crop yields (Tan et al., 2017; Wang et al., 2015). However, diverse climatic conditions across different farming locations can cause significant variations in soil N and biological activities and may lead to great discrepancies in GHGs emissions and yields (Yang et al., 2019; Zhang et al., 2020). Therefore, identifying appropriate N amounts conducive to both reducing soil GHGs emissions and improving yield benefits is imperative for sustainable agricultural production in arid regions of China.

Appropriate irrigation and fertilization are conducive to increasing water and nutrient utilization (Li et al., 2020), and their interactive effects are essential for achieving acceptable yields and net profit (Badr et al., 2012; Zhang et al., 2020). In the context of climate change, water shortages, and high fertilizer costs, researchers are increasingly focusing on the coupling effects of reduced water and N and recognized as an effective measure for improving crop economic benefits in arid and semiarid regions (Hou et al., 2019; Ning et al., 2019; Zhong et al., 2021). Also, the synergistic effect of reduced irrigation and fertilizer amounts can mitigate GHGs emissions by increasing N use efficiency and minimizing the risk of N losses compared to conventional management (Li et al., 2020; Yu et al., 2021; Zhang et al., 2020). However, the effects of reduced fertilization on productivity and GHGs emissions greatly vary for different irrigation levels, crops, soil types, and climatic conditions (Badr et al., 2012). Therefore, matching the fertilizer rates and irrigation amounts should be considered simultaneously to improve wheat productivity and mitigate GHGs emissions.

Besides GHGs emissions, another challenge with intensive wheat farming is increasing economic profits. Net ecosystem economic benefit (NEEB) is often used to evaluate agricultural practices for their economic feasibility and environmental sustainability (Li et al., 2015). Despite extensive research on the effects of irrigation and fertilization on crops, the studies mainly focused on comparing irrigation methods and fertilizer types for yield improvement and/or GHGs mitigations. In the extreme arid regions of Northern China where conventional border irrigation and high N fertilization are prevalent, the comprehensive effects of deficit irrigation and N application on global warming potential, greenhouse gas intensity, yield, and economic benefits in spring wheat are not fully explored. Therefore, the present study aimed to better understand the effectiveness of different reduced irrigation and N regimes for reconciling low GHGs emissions and high wheat productivity. A better understanding of the wheat crop response to deficit irrigation and N regimes under arid conditions will help in reducing the inputs, improving crop profitability, and minimizing GHGs emissions for sustainable environmental health.

2. Materials and methods

2.1. Research site description

Field experiments during two wheat growing seasons (2015–2016) were conducted at the experimental station of Lanzhou University (103° 05 'E, 38° 38' N), located in the Hexi Corridor, Gansu Province, Northwest China. The research site is an irrigation-dependent oasis with a typically arid continental climate, cold winters, and hot dry summers. The region enjoys a frost-free period of approximately 175 days. The annual mean air temperature of the region is 7.8 °C. The annual mean precipitation is 110.7 mm (20-year scale, 1996–2016) and more than 60 % is concentrated from June to September (Fig. S1). The annual sunshine accounts for 3000 h and annual evapotranspiration reaches 2644 mm in the region. The soil at the experimental site is sandy loam, classified as Aridisol. Before sowing, soil (0–20 cm) had a pH of 8.5, organic matter of 9.3 g kg^{-1}, available nitrogen 38.7 mg kg^{-1}, phosphorus 20.3 mg kg^{-1}, and potassium 54.5 mg kg^{-1}.

2.2. Experimental design and treatments management

The spring wheat variety "Yongliang-4", a widely grown cultivar in the region was used in the experiment. The experiment was arranged in a factorial design with three irrigation regimes and three N fertilizer levels. The irrigation regimes included 750 (I_{100}), 600 (I_{80}), and 450 mm (I_{60}), representing 100 %, 80 %, and 60 % of the total amount of border irrigation applied by local farmers during the spring wheat growing seasons, respectively. The irrigation intervals were adopted according to the local farmer's practices (Table S1). Nitrogen application rates included 300 (N_{100}), 225 (N_{75}), and 150 kg ha^{-1} (N_{50}), representing 100 %, 75 %, and 50 % of the N management by the local farmers during the spring wheat growing seasons, respectively. Urea (46 % N) was used as the N source and the fertilizers were applied in split doses (Table S1).

Overall, the experiment consisted of nine different treatments combinations; (i) $I_{100}N_{100}$ (ii) $I_{100}N_{75}$ (iii) $I_{100}N_{50}$ (iv) $I_{80}N_{100}$ (v) $I_{80}N_{75}$ (vi) $I_{80}N_{50}$ (vii) $I_{60}N_{100}$ (viii) $I_{60}N_{75}$, and (ix) $I_{60}N_{50}$. Each treatment was replicated four times. The size of each treatment plot was 10 m × 10 m with a 1.5 m buffer between adjacent plots. The partitioning ridges of each subplot were isolated with embedded impervious membranes to eliminate the influence of lateral water and N movement. Wheat seeds were sown at a density of 185 kg ha^{-1} with a manual planter and a row spacing of 20 cm was maintained. Wheat seeds were planted on March 18th and 20th and harvested on July 15th and 19th in 2015 and 2016, respectively. The recommended 120 kg P_2O_5 ha^{-1} and 150 kg K_2O ha^{-1} were uniformly applied to all plots before sowing. No herbicides were applied in both crop growing seasons.

2.3. Sampling and measurements

2.3.1. Precipitation, temperature, and water-filled pore space

Data for air temperature and precipitation were obtained from a local meteorological station. Soil temperature (0–10 cm depth) was recorded in each treatment plot with JM 624 digital thermometers (Jinming Instrument Co. Ltd., Tianjin, China). Parallel to GHG sampling, three soil samples (0–20 cm) were taken from each plot with an auger (3 cm diameter) to determine soil moisture. The moisture contents were evaluated gravimetrically by oven-drying the samples and the results were expressed as soil water-filled pore space (WFPS) using the equation (Lyu et al., 2019):

$$\mathrm{WFPS}(\%) = \frac{Gravimetric\quad water\quad content\quad (\%)\quad \times\quad Soil\quad bulk\quad density}{1 - \quad soil\quad bulk\quad density/2.65} \times 100 \tag{1}$$

Where 2.65 (g cm^{-3}) is the theoretical soil particle density.

2.3.2. Determination of soil inorganic nitrogen

Corresponding to the GHGs samplings, three soil samples (0–20 cm) in each plot were collected near the chamber using an auger (3 cm diameter) for the measurements of soil mineral N (NO_3^- and NH_4^+) contents. Soil samples were transported immediately to the laboratory and extracted with 1 mol L^{-1} KCl solution (1:10 soil to solution ratio). The extracts were examined for soil available NH_4^+ and NO_3^- by colorimetric analysis using a continuous flow analyzer (TRAACS 2000, Bran and Luebbe, Norderstedt, Germany) (Zhang et al., 2021a,b).

2.3.3. Measurement of greenhouse gas fluxes

Soil-borne GHGs fluxes were determined using the static chamber-gas chromatography (GC) method as previously described (Lyu et al., 2019). The chamber consists of a rectangular stainless-steel base frame (50 cm long × 50 cm wide × 10 cm height) permanently interleaved into the soil (10 cm) and a removable top cover box (50 cm long × 50 cm wide × 50 cm height). In general, gas samples were collected once a week between 09:00 am and 11:30 am. The samplings were intensified to once every three days after irrigation and N fertilizer application (irrigation alone did not show an obvious effect on GHGs emissions and once-a-week data is provided). Four gas samples for flux measurements were collected within 30 min at a time interval of ten min (0, 10, 20, and 30 min) using a polypropylene syringe (50 mL) fitted with a nylon stopcock. Gas samples were instantly injected into 300-mL pre-evacuated aluminum foil bags (LB-101, Delin, Dalian China) and transported immediately to the laboratory for analysis. The N_2O concentration was measured by LGR gas Analyzer (908–0015–0000, Los Gatos Research, USA), while CH_4 and CO_2 concentrations were simultaneously analyzed by LGR CH_4/CO_2 Analyzer (908–0011–0001, Los Gatos Research, USA). The GHG fluxes were calculated based on the linear regression slope between concentration and time (Li et al., 2020):

$$J = \frac{dc}{dt} \bullet \frac{M}{V_0} \bullet \frac{P}{P_0} \bullet \frac{T_0}{T} \bullet H \tag{2}$$

Where J is the measured flux of N_2O (μg $m^{-2}h^{-1}$), CH_4 (μg $m^{-2}h^{-1}$), and CO_2 (mg $m^{-2}h^{-1}$), dc/dt is the change rate of the gas concentration in the chamber, M is the molar mass of the measured gas (g mol^{-1}), T is the absolute temperature (K), P is the atmospheric pressure (Pa), V_0, P_0, and T_0 are the volume (mL), absolute temperature (K) and pressure (Pa) at standard conditions, and H is the chamber height (cm).

The cumulative N_2O, CH_4, and CO_2 emissions during the wheat-growing seasons were estimated by using the equation (Yeboah et al., 2016):

$$Y = \sum \frac{(F_i + F_{i+1}) \times (t_{i+1} - t_i)}{2 \times 100} \times 24 \tag{3}$$

Where Y is the cumulative GHG flux (N_2O, CH_4, and CO_2) for the entire wheat growing period (kg ha^{-1}), Fi is the current GHG emissions flux for CO_2, N_2O, or CH_4 (mg m^{-2} h^{-1}), F_{i+1} is the previously measured sampling fluxes, and (t_{i+1}-t_i) is the number of days between two adjacent measurements.

2.3.4. Estimation of global warming potential and greenhouse gas intensity

To estimate GWP, CO_2 was used as a reference gas, and therefore N_2O and CH_4 emissions were converted into "CO_2 equivalents'' (kg CO_2 eq. ha^{-1}). The default GWP values per unit mass of CH_4 and N_2O measured over a 100-year time frame are 27.9 and 273 times that of CO_2 emissions, respectively (IPCC, 2022).

$$GWP = 27.9 \times Y_{CH_4} + 273 \times Y_{N_2O} \tag{4}$$

Where Y_{CH4} and Y_{N2O} represent the cumulative CH_4 and N_2O emissions (kg ha^{-1}), respectively.

The greenhouse gas intensity (GHGI) is a comprehensive measure of economic and environmental gains and represents the GHG balance per unit of crop productivity. GHGI was calculated using the following equation (Lyu et al., 2019):

$$GHGI = \frac{GWP}{wheat\quad yield} \tag{5}$$

2.3.5. Estimation of net ecosystem economic budget

The net ecosystem economic budget (NEEB) was used to assess the environmental and economic benefits of irrigation and N treatments using the following equation (Li et al., 2015):

$$NEEB = \text{Yield gains} - \text{Agriculture activity costs} - \text{GWP costs} \tag{6}$$

The yield gains were calculated from the local market price of wheat yield. The agriculture activity costs comprise the expenses from irrigation, fertilization, crop seeds, and harvesting. The GWP costs were calculated from the product of GWP and carbon trade price (Li et al., 2015). More details on the agriculture activity costs, yield gains, and GWP costs are provided in Table S2 of the Supporting Information (SI).

2.3.6. Determination of wheat grain yield

At physiological maturity, the wheat crop was harvested manually from a 4 × 4 m (16 m^2) area at two random locations within each treatment plot to determine biomass and grain yield (their average was considered as a single replicate). After threshing the spikes, the grain samples were cleaned, sun-dried, and grain yield was determined at 14 % moisture content.

2.4. Statistical analysis

Data in tables and figures are presented as the mean of four replicates ± SD (n = 4). All statistical analyses were performed using the SPSS 20.0 software package (IBM Corp., USA). A three-way factorial ANOVA (analysis of variance) was used to analyze the significance of the year, irrigation, N, and their interactions on GHGs emissions, GWP, GHGI, NEEB, and wheat yield. In addition, to analyze the combined effects of irrigation and N treatments in each crop growing season, a one-way ANOVA was used. Treatment means were compared using Tukey's significant difference test at $P < 0.05$. Before the statistical analysis, the normality of the datasets was examined (Shapiro-Wilk's Normality Test). The relationship of cumulative GHGs emissions with soil moisture and inorganic N contents (NH_4^+ and NO_3^-) was examined following linear-mixed models. Figures were constructed with Excel 2010 (Microsoft Corp., USA) and Origin 9.1 (Origin Lab Corp., USA).

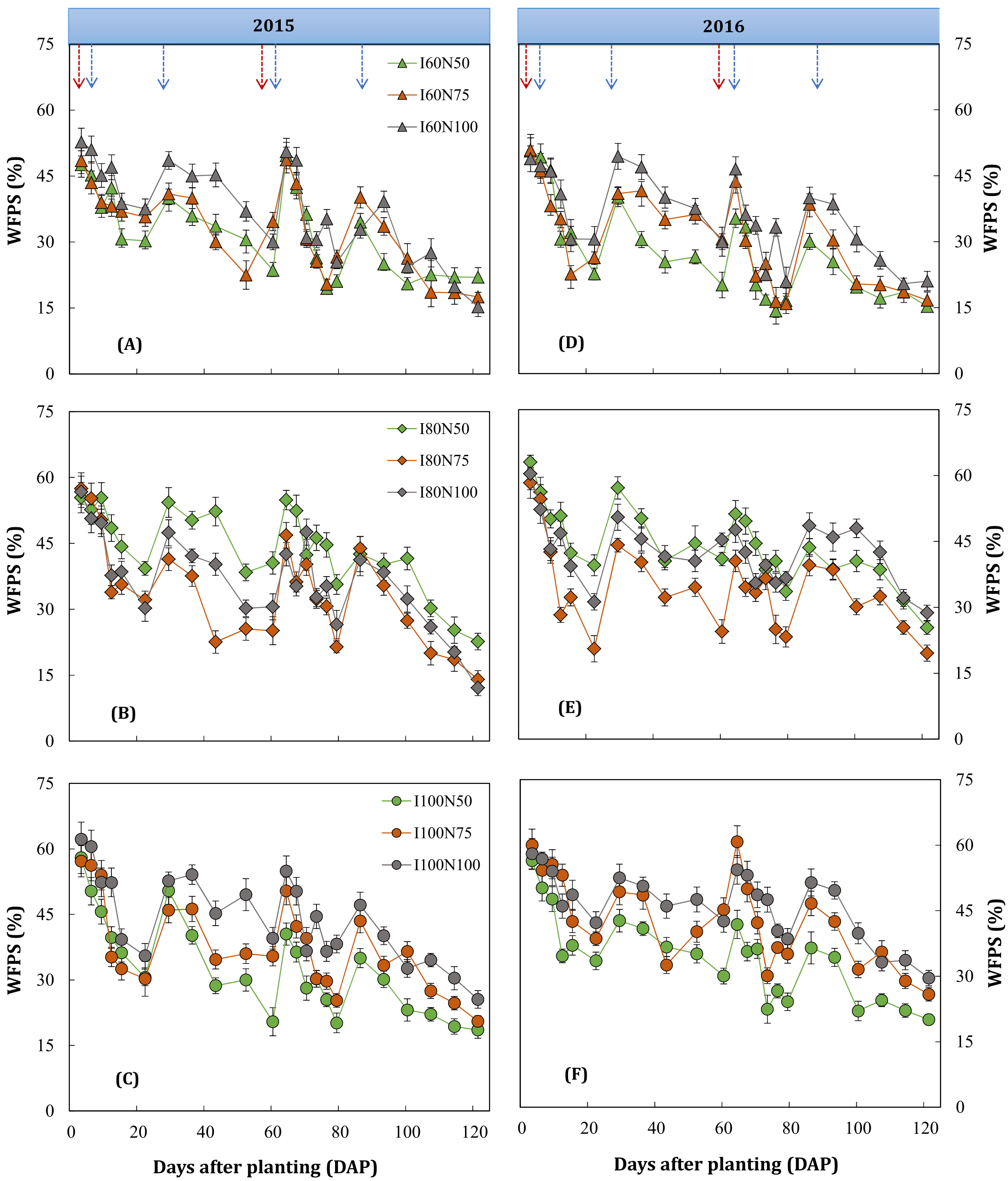

Fig. 1. Effects of irrigation (I) and nitrogen (N) treatments on the seasonal dynamic of soil water-filled pore space (WFPS) at 0–20 cm soil depth in 2015 (A–C) and 2016 (D–F) spring-wheat growing period. Data presented are the means of four replicates ± SD (n = 4). I_{100}, I_{80}, and I_{60} represent irrigation amounts of 750, 600, and 450 mm, while N_{100}, N_{75}, and N_{50} represent nitrogen application rates of 300, 225, and 150 kg ha^{-1}, respectively.

3. Results

3.1. Environmental factors, soil moisture, and mineral N content

Total rainfall during the first wheat growing season (120 days) was 62.9 mm while it was only 36.9 mm in the following season (122 days), accounting for 52.2 % and 48.9 % of the total annual rainfall in 2015 (120.4 mm) and 2016 (75.4 mm), respectively (Fig. S1). About 70 % of rainfall was less than 5 mm, too little to be effectively utilized by the crops, and the annual total precipitation was mainly dependent on a few large rainfall events with over 10 mm. The daily mean air temperatures during the wheat-growing seasons ranged from 2.8 °C to 26.2 °C in 2015 and 3.1–20.2 °C in 2016, while mean soil temperature ranged from − 1.5 to 20.2 °C in 2015 and − 1.8 to 15.5 °C in 2016 (Fig. S1).

Soil moisture expressed as water-filled pore space (WFPS) showed several drying–wetting cycles and various fluxes were perceived after irrigation events (Fig. 1). Following the first irrigation, the WFPS values remained higher over a week (44.2–62.2 %) and then tended to decline gradually until 22 days after planting (DAP; 30.2–35.6 %), but increased again at 32 DAP (40.0–57.2 %) following the second irrigation event. Thereafter, the WFPS values showed two more fluctuations and increased to relatively higher values at 62 DAP (36.3–60.8 %) and 92 DAP (30.1–51.5 %) in different treatments, corresponding to the third and fourth irrigation events (Fig. 1).

During both wheat growing seasons, all treatments followed comparable seasonal dynamics in soil inorganic N (NO_3^- and NH_4^+) contents, which was mainly dependent on fertilization followed by irrigation. Throughout the study period, the NO_3^- contents of the top 20 cm soil profile ranged from 3.6 to 16.0 mg kg^{-1}, and NH_4^+ contents from 6.1 to 23.6 mg kg^{-1} in all treatments (Figs. 2–3). After the application of fertilizer and irrigation at sowing, soil NO_3^- and NH_4^+ contents increased rapidly and reached peak values (17.2–22.4 and 10.8–22.4 mg kg^{-1}) at 6–9 DAP, thereafter remaining lower until 57 DAP (Figs. 2–3). However, following the third irrigation combined with fertilizer application at 58 DAP, the inorganic N contents increased significantly and reached the maximum values (11.4–14.6 and 35.2–38.1 mg kg^{-1}) at 64–67 DAP, and later declined gradually in all treatments until the end of the wheat-growing season. In general, increased irrigation amounts tended to maintain relatively high NO_3^- and NH_4^+ concentrations in the N addition treatments for a longer period throughout the wheat-growing seasons.

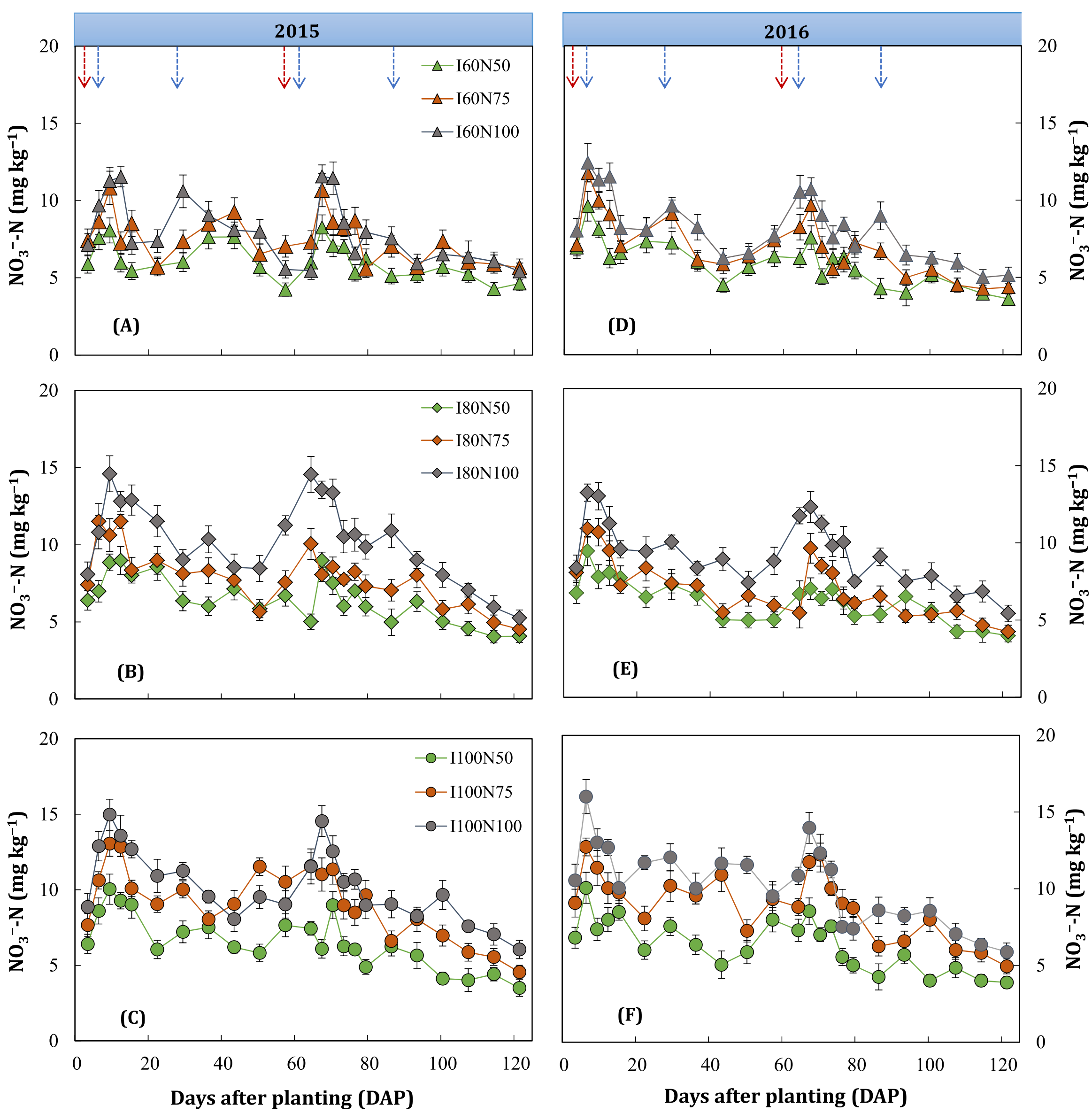

Fig. 2. Effects of irrigation (I) and nitrogen (N) treatments on dynamic of soil NO_3^--N concentration at 0–20 cm soil depth during 2015 (A–C) and 2016 (D–F) spring-wheat growing period. Data presented are the means of four replicates ± SD (n = 4). The abbreviations for treatment names are the same as those described in Fig. 1.

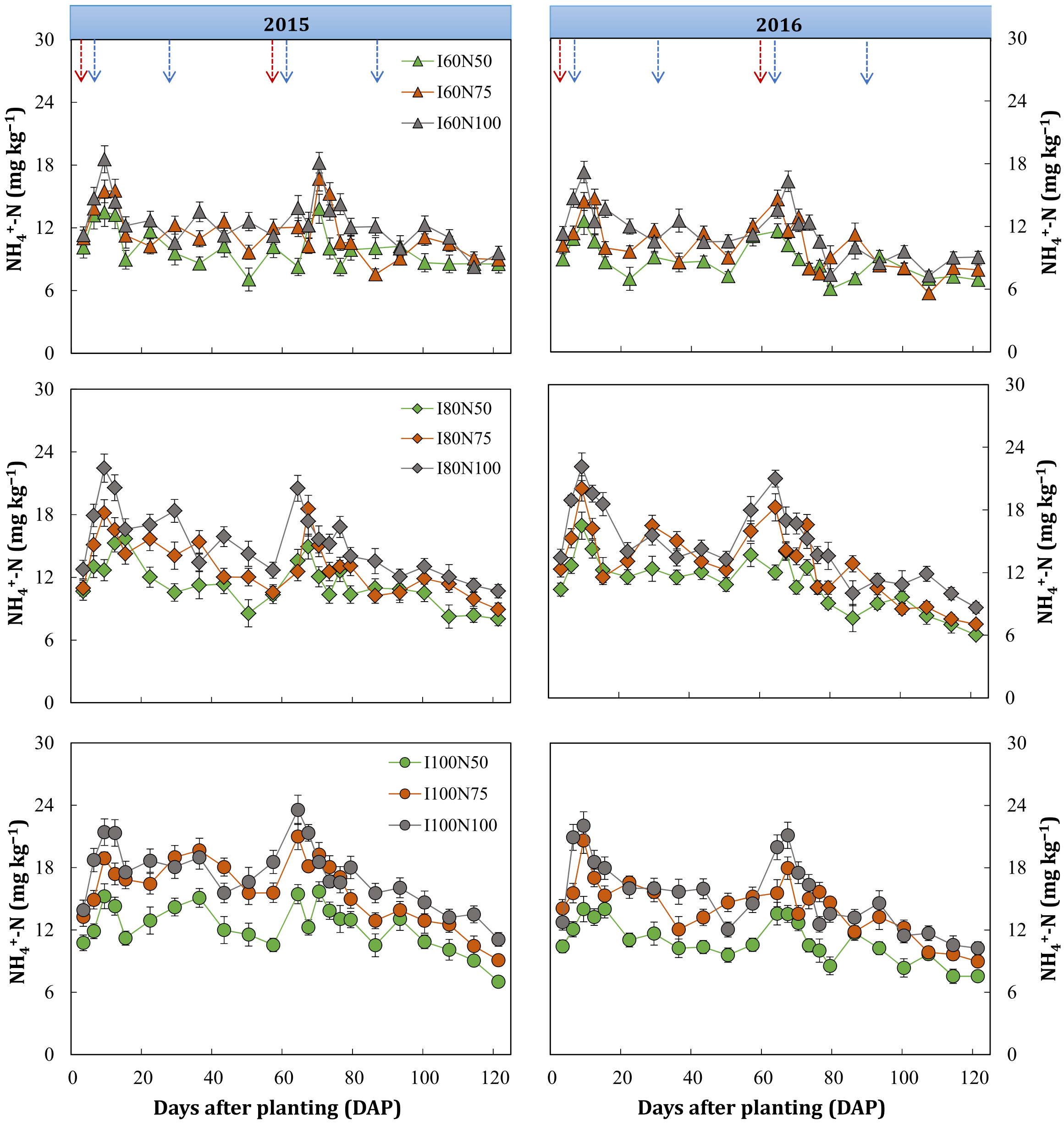

Fig. 3. Effects of irrigation (I) and nitrogen (N) treatments on dynamic of soil NH_4^+-N concentration at 0–20 cm soil depth during 2015 (A–C) and 2016 (D–F) spring-wheat growing period. Data presented are the means of four replicates ± SD (n = 4). The abbreviations for treatment names are the same as those described in Fig. 1.

3.2. Seasonal variation in N_2O emissions

The seasonal soil N_2O fluxes showed characteristics of multiple spikes and large fluctuations, essentially following a similar pattern during both crop growing seasons (Fig. 4). The time of peaked N_2O fluxes was closely related to irrigation and fertilization events, and two major peaks were detected each growing season. The first N_2O emissions peak (48.9–51.79 μg m^{-1} h^{-1}) was perceived at 9 DAP following the effect of first irrigation and N fertilization (Fig. 4A–F). The N_2O fluxes did not change significantly after the second irrigation and tended to decline significantly in a shorter period. However, the N_2O emissions reached the second highest peak (50.03–52.45 μg m^{-1} h^{-1}) at 67 DAP closely after the third irrigation coupled with N fertilization (Fig. 4A–F). Thereafter, the N_2O emissions remained lower in all treatments in the later wheat growing season. After the supplementation of first and third irrigation coupled with N fertilization, the emissions were intensively monitored over two weeks. The two years average data indicated the daily N_2O fluxes reached peak values at 6–8 days and then decreased to relatively lower levels at 9–16 days after treatment application (Fig. 4G–J). During this period, N_2O emissions appeared to be higher at high irrigation and N application rates.

The cumulative N_2O emissions from the wheat field under different years, irrigation, and N treatments are presented in Table 1. The ANOVA results showed significant effects of irrigation, N, and their interaction on cumulative N_2O emissions. However, the year effect was not significant (Table S3). Under the same irrigation level, the GHG emissions were the highest with full N application, while the emissions were markedly decreased with moderate and low N application rates (Table 1 and S4). Among all the treatments, the highest N_2O emission values were perceived with $I_{100}N_{100}$ (1.39 and 1.21 kg ha^{-1}) and $I_{100}N_{75}$ (1.13 and 1.09 kg ha^{-1}) treatments, while the lowest emissions were perceived with $I_{60}N_{50}$ (0.52 and 0.56 kg ha^{-1}) and $I_{60}N_{75}$ (0.54 and 0.64 kg ha^{-1}), followed by $I_{80}N_{50}$ (0.71 and 0.72 kg ha^{-1}), and $I_{80}N_{75}$ (0.79 and 0.81 kg ha^{-1}) treatments in 2015 and 2016, respectively (Table 1). When compared to $I_{100}N_{100}$, the N_2O emissions for $I_{80}N_{75}$, $I_{80}N_{50}$, $I_{60}N_{75}$, and $I_{60}N_{50}$ treatments were decreased by 43.2 % and 33.1

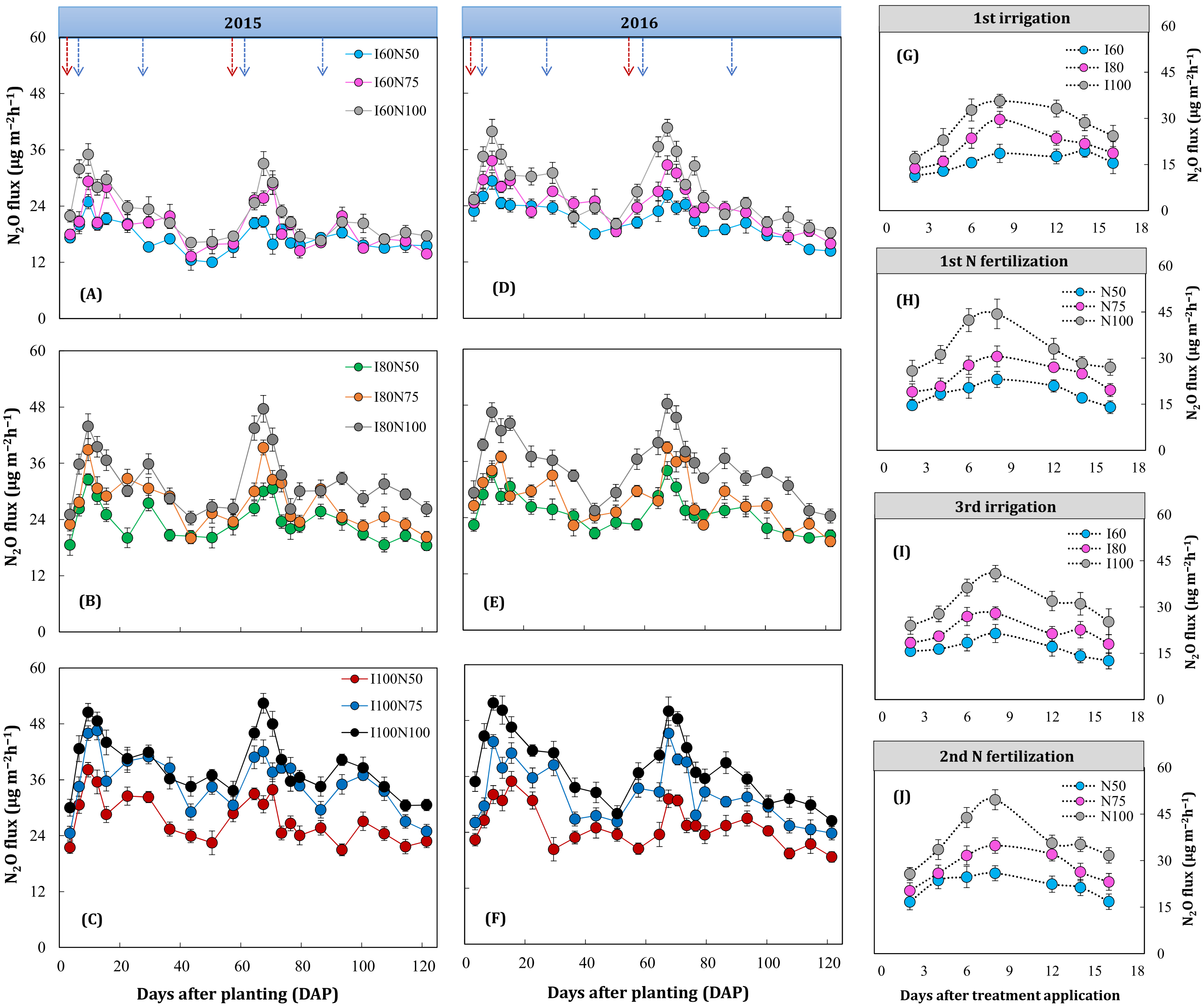

Fig. 4. Effects of irrigation (I) and nitrogen (N) treatments on soil N_2O fluxes during 2015 (A-C) and 2016 (D-F) spring-wheat growing period. Data presented in sub-figures (G-J) are the average of two years. Data presented are the means of four replicates ± SD (n = 4). The abbreviations for treatment names are the same as those described in Fig. 1. The red and blue arrows represent N fertilizer and irrigation supplementation, respectively. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

%, 48.9 % and 41.0 %, 61.0 % and 47.1 %, 61.2 % and 53.7 % in 2015 and 2016, respectively.

3.3. Seasonal variation in CH_4 uptake

Negative CH_4 fluxes were perceived during the experimental periods, indicating higher soil CH_4 uptake than the emissions (Fig. 5). The CH_4 uptake followed a similar pattern in both crop growing seasons, except for minor fluctuations at different intervals. These differences were relatively in magnitude rather than direction. Two distinct CH_4 fluxes were detected over the two growing seasons, the first peak (−35.27 to −40.71 μg m^{-1} h^{-1}) at 12 DAP, presenting the effects of irrigation and nitrogen application at sowing. The CH_4 fluxes reached the second peak at 67 DAP (−33.23 to −37.90 μg m^{-1} h^{-1}) afterwards the third irrigation and second-time N fertilization (Fig. 5A–G). Moreover, when intensively examined over two consecutive weeks after treatment application, the CH_4 uptake reached peak values from 6 to 8 days after the irrigation events (6 G and 6I). However, CH_4 uptake exhibited a gradual declining trend following the N fertilization, and the values appeared to be less negative from 6 to 12 days in all N treatments (Fig. 5H and J).

The ANOVA results depicted significant effects of the year, irrigation and N treatments, and their interactions (except for Y × I × N) on CH_4 uptake (Table S3). The mean CH_4 uptake of all the treatments in 2015 was greater by 11.2 % compared to that in 2016. The decrease in irrigation regimes significantly reduced the CH_4 uptake. In contrast, decreasing the N application rate under the same irrigation regimes markedly increased the CH_4 uptake and the values were less negative at high N rates (Table 1 and S4). Among all the irrigation and N treatments, the lowest CH_4 uptake (−0.45 and −0.44 kg ha^{-1}) was achieved with the conventional full irrigation and N application ($I_{100}N_{100}$), followed by $I_{80}N_{100}$ treatment (−0.44 and −0.39 kg ha^{-1}) in 2015 and 2016 (Table 1). On the other hand, the highest CH_4 uptake was achieved with low N application under high irrigation ($I_{100}N_{50}$; −0.84 and −0.76 kg ha^{-1}) and low N application under medium irrigation regime ($I_{80}N_{50}$; −0.77 and −0.67 kg ha^{-1}) (Table 1).

3.4. Seasonal variation in CO_2 emissions

The seasonal variations in CO_2 fluxes followed similar trends during the two-year field study. After sowing, the CO_2 emissions tended to

Table 1
Effects of irrigation (I) and nitrogen (N) regimes on grain yield of spring wheat, greenhouse gases (N_2O, CH_4, CO_2) emissions, global warming potential (GWP), and greenhouse gas intensity (GHGI) in 2015 and 2016.

Year	Treatments		Grain Yield ($t\ ha^{-1}$)	Cumulative GHGs emissions			GWP ($kg\ ha^{-1}$)	GHGI ($kg\ t^{-1}$)
				N_2O ($kg\ ha^{-1}$)	CH_4 ($kg\ ha^{-1}$)	CO_2 ($Mg\ ha^{-1}$)		
2015	I_{100}	N_{100}	6.13 ± 0.23b	1.39 ± 0.08a	-0.45 ± 0.03a	13.81 ± 0.41a	367.87 ± 21.09a	59.93 ± 1.32a
		N_{75}	6.75 ± 0.29a	1.13 ± 0.06b	-0.59 ± 0.01bc	12.03 ± 0.50b	292.70 ± 15.22b	43.39 ± 2.26b
		N_{50}	5.37 ± 0.17c	0.81 ± 0.03d	-0.84 ± 0.06d	10.64 ± 0.24c	198.98 ± 9.86c	37.09 ± 1.59c
	I_{80}	N_{100}	6.13 ± 0.15b	1.04 ± 0.04c	-0.44 ± 0.03a	11.85 ± 0.37b	272.38 ± 15.11b	44.45 ± 0.89b
		N_{75}	7.26 ± 0.25a	0.79 ± 0.03d	-0.58 ± 0.03c	9.84 ± 0.21cd	195.84 ± 7.61c	26.95 ± 1.09e
		N_{50}	5.55 ± 0.20c	0.71 ± 0.04e	-0.77 ± 0.05d	8.67 ± 0.33ef	172.71 ± 7.05cd	31.12 ± 1.12d
	I_{60}	N_{100}	5.04 ± 0.26 cd	0.77 ± 0.03de	-0.47 ± 0.03b	9.38 ± 0.27de	196.65 ± 7.91c	39.17 ± 3.63c
		N_{75}	4.62 ± 0.23d	0.54 ± 0.04f	-0.54 ± 0.03bc	8.37 ± 0.29f	133.02 ± 15.63de	28.78 ± 1.93de
		N_{50}	4.53 ± 0.16d	0.52 ± 0.02f	-0.63 ± 0.04c	8.13 ± 0.31f	129.62 ± 17.60e	28.60 ± 1.06de
2016	I_{100}	N_{100}	6.09 ± 0.11b	1.21 ± 0.06a	-0.44 ± 0.04ab	15.18 ± 0.52a	319.12 ± 12.02a	52.40 ± 2.13a
		N_{75}	6.37 ± 0.26b	1.09 ± 0.04b	-0.59 ± 0.03cd	14.11 ± 0.26ab	281.19 ± 12.66b	44.17 ± 0.62c
		N_{50}	5.55 ± 0.14c	0.86 ± 0.03c	-0.76 ± 0.06e	11.64 ± 0.31c	212.36 ± 11.19c	38.23 ± 1.12d
	I_{80}	N_{100}	6.00 ± 0.13b	1.09 ± 0.05b	-0.39 ± 0.04a	13.36 ± 0.32b	285.93 ± 9.81b	47.63 ± 2.11b
		N_{75}	6.89 ± 0.31a	0.81 ± 0.03d	-0.57 ± 0.03c	11.21 ± 0.47c	205.48 ± 8.91d	29.60 ± 1.12f
		N_{50}	5.49 ± 0.19c	0.72 ± 0.04e	-0.67 ± 0.04d	9.89 ± 0.22d	178.25 ± 8.64e	32.44 ± 0.89e
	I_{60}	N_{100}	4.90 ± 0.14d	0.71 ± 0.03e	-0.36 ± 0.02a	9.80 ± 0.47d	182.59 ± 8.11e	37.28 ± 0.86d
		N_{75}	4.39 ± 0.21e	0.64 ± 0.02f	-0.46 ± 0.03ab	9.16 ± 0.45de	161.80 ± 7.45f	36.89 ± 2.26d
		N_{50}	4.28 ± 0.13e	0.56 ± 0.03g	-0.51 ± 0.03b	8.37 ± 0.31e	138.91 ± 8.44g	32.45 ± 2.03e
Treatment means								
	I_{100}		6.22 ± 0.12a	1.08 ± 0.04a	-0.62 ± 0.04c	12.90 ± 0.58a	278.70 ± 11.45a	45.87 ± 2.26a
	I_{80}		6.04 ± 0.18a	0.86 ± 0.02b	-0.56 ± 0.02b	10.57 ± 0.40b	219.26 ± 8.62b	35.51 ± 1.11b
	I_{60}		4.63 ± 0.11b	0.63 ± 0.03c	-0.49 ± 0.03a	8.87 ± 0.53c	157.10 ± 10.63c	33.86 ± 1.78b
	N_{100}		5.71 ± 0.08b	1.04 ± 0.04a	-0.43 ± 0.03a	12.51 ± 0.32a	270.75 ± 11.43a	46.81 ± 3.33a
	N_{75}		6.05 ± 0.12a	0.83 ± 0.03b	-0.55 ± 0.05b	10.36 ± 0.35b	212.51 ± 8.79b	35.11 ± 1.17b
	N_{50}		5.13 ± 0.16c	0.70 ± 0.03c	-0.71 ± 0.05c	9.76 ± 0.39b	171.80 ± 8.65c	33.32 ± 2.09b

The abbreviations for treatment names are the same as described in Fig. 1. Data are the means ± SD ($n = 4$). Different lowercase letters indicate significant differences among treatment means based on Tukey's significant difference test ($P < 0.05$).

increase dramatically and reached peak values at 9–12 DAP (577.73 –641.57 $mg\ m^{-2}h^{-1}$). Thereafter, CO_2 emissions followed a gradually declining trend until 58 DAP, but elevated again and reached the second seasonal peak curve at 62 DAP (625.37 –656.01 $mg\ m^{-2}h^{-1}$) (Fig. 6A–F). Both these peak fluxes were associated with the timing of irrigation coupled with N fertilizer application (Fig. 6A–F). When analyzed for two consecutive weeks after treatment application, the highest mean values of daily CO_2 were detected at 6–8 days after irrigation and fertilization and then decreased to relatively low levels at 10–16 days (Fig. 6G–J). During this period, the CO_2 fluxes were always higher at high irrigation and N rates than that at other treatment levels.

The year, irrigation, and N treatments showed significant effects on the cumulative CO_2 emissions (Table S3). The interactive effects of year and treatments were also significant except for Y × N and Y × I × N (Table S3 and S4). The cumulative emissions in 2016 were greater by 10.78 % compared to that in 2015. Reducing the irrigation and N application rates linearly decreased the CO_2 emissions compared to full irrigation and N application (Table 1 and S4). The highest cumulative CO_2 emissions were achieved with $I_{100}N_{100}$ (13.81 and 15.18 $Mg\ ha^{-1}$) and $I_{100}N_{75}$ (12.03 and 14.11 $Mg\ ha^{-1}$) treatments in 2015 and 2016, respectively. On the other hand, the lowest emissions were perceived with $I_{60}N_{50}$ (8.13 and 8.37 $Mg\ ha^{-1}$) and $I_{60}N_{75}$ (8.37 and 9.16 $Mg\ ha^{-1}$) treatments (Table 1). When compared with the $I_{100}N_{100}$ treatment, CO_2 emissions of $I_{60}N_{50}$ and $I_{60}N_{75}$ treatments were reduced by 41.2 % and 39.4 % in 2015 and by 39.7 % and 44.9 % in 2016.

3.5. Global warming potential and greenhouse gas intensity

The GWP was significantly affected by irrigation regimes, N rates, and their interaction, but no significant interaction existed in two years (Table S3). In general, the higher the irrigation and N amounts, the greater GWP values were obtained in wheat plots (Table S4). The conventional $I_{100}N_{100}$ irrigation and N management resulted in the highest GWP values of 367.87 and 319.12 $kg\ ha^{-1}$ in 2015 and 2016, respectively (Table 1). Whereas reducing irrigation and N application rates markedly decreased the GWP values and the least values were evident with $I_{60}N_{75}$ (133.02 and 161.80 $kg\ ha^{-1}$) and $I_{60}N_{50}$ (129.62 and 138.91 $kg\ ha^{-1}$) treatments in 2015 and 2016, respectively (Table 1).

On the other hand, GHGI values decreased with reducing the irrigation levels but showed a different response to N fertilizer rates under different irrigation regimes (Table 1 and S4). At I_{100} and I_{60} irrigation levels, reducing N application rates linearly decreased the GHGI values. However, under the I_{80} irrigation level, the GHGI tended to be reduced significantly by decreasing the N rate from N_{100} to N_{75}, but the GHGI values were elevated again with N_{50} treatment (Table 1 and S4). Among all the irrigation and N combined treatments, the highest GHGI values of 59.93 and 52.40 $kg\ t^{-1}$ were obtained with $I_{100}N_{100}$ treatment, while the lowest values of 26.95 and 29.60 $kg\ t^{-1}$ were achieved with $I_{80}N_{75}$ treatment in 2015 and 2016, respectively (Table 1). The $I_{80}N_{75}$ treatment decreased the GHGI of the wheat crop by 55.0 % and 43.5 % in 2015 and 2016 when compared with the $I_{100}N_{100}$ treatment, respectively.

3.6. Wheat yield and net ecosystem economic benefit

The wheat grain yield was significantly affected by irrigation, N, year, and their interactions (except Y × I and Y × I × N) (Table S3 and S4). The grain yield in 2015 was comparatively greater than that in 2016. Remarkably, a moderate decrease in irrigation and N application rates ($I_{80}N_{75}$) improved the grain yield compared to full irrigation-N management in both crop growing seasons (Table 1). The grain yield achieved with the $I_{80}N_{75}$ treatment was 7.26 and 6.89 $t\ ha^{-1}$, greater by 18.4 % and 13.1 % compared to $I_{100}N_{10}$ treatment in 2015 and 2016, respectively. On the other hand, too much reduction in irrigation and N application rates exhibited negative impacts and the lowest yields were achieved with $I_{60}N_{50}$ treatment (4.53 and 4.28 $t\ ha^{-1}$), which were lower by 26.1 % and 29.7 % compared with that of $I_{100}N_{100}$ treatment in 2015 and 2016, respectively (Table 1).

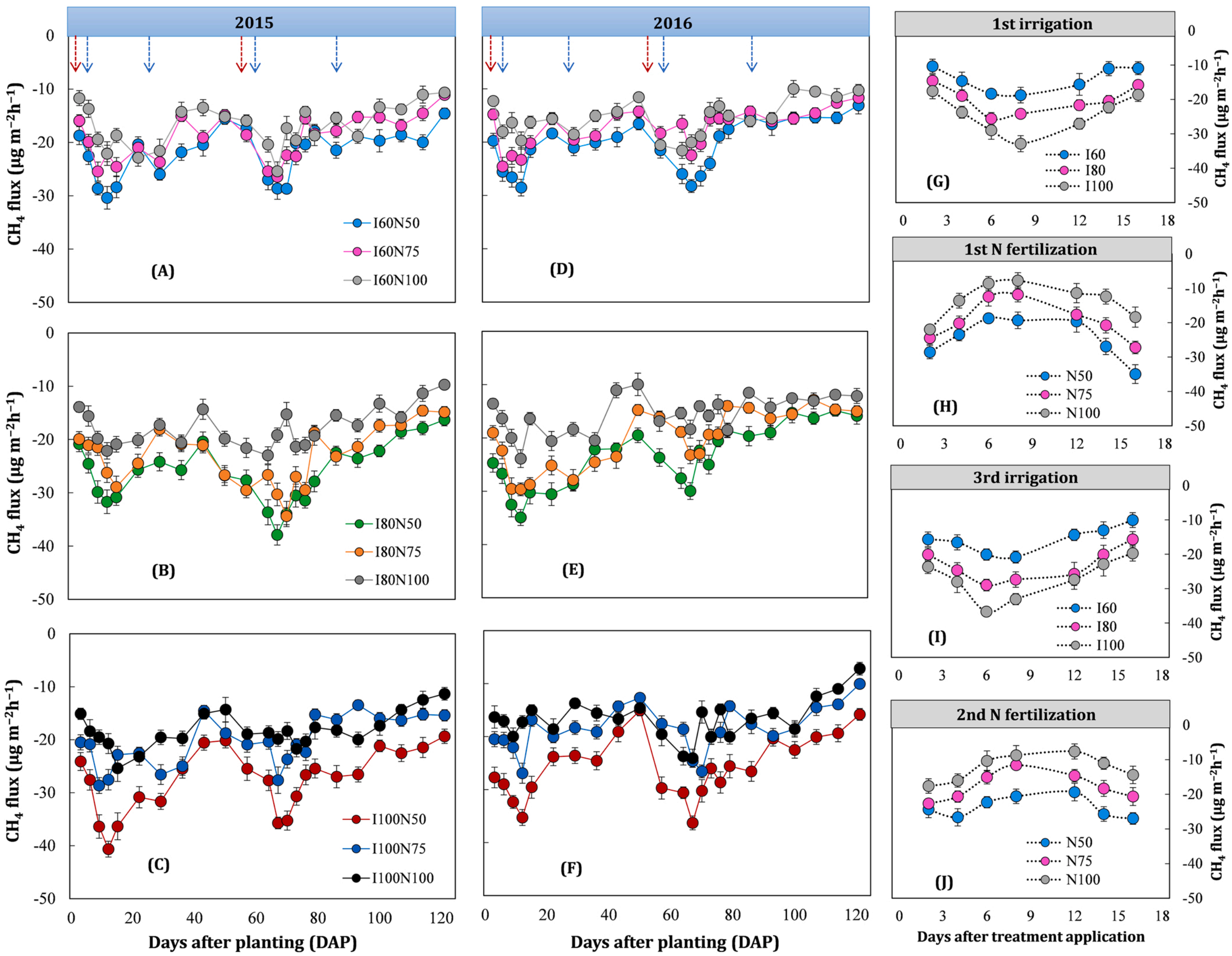

Fig. 5. Effects of irrigation (I) and nitrogen (N) treatments on soil CH_4 fluxes during 2015 (A-C) and 2016 (D-F) spring-wheat growing period. Data presented in sub-figures (G-J) are the average of two years. Data presented are the means of four replicates ± SD (n = 4). The abbreviations for treatment names are the same as those described in Fig. 1. The red and blue arrows represent N fertilizer and irrigation supplementation, respectively. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

The components of NEEB from different irrigation and N treatments are shown in Table 2. No significant difference in NEEB was observed between the two growing seasons (2015 and 2016). A decrease in the irrigation and N rates reduced the agricultural activity costs and the GWP costs, which directly affected the NEEB from the wheat plots among different treatments (Table 2 and S4). Interestingly the highest NEEB values were achieved with $I_{80}N_{75}$ treatment (16,712 and 16,542 CNY ha^{-1}), which were improved by 29.4 % and 25.4 % compared to that of conventional $I_{100}N_{100}$ treatment in 2015 and 2016, respectively (Table 2). Although, too much reduction in irrigation and N application rates resulted in a significant decrease in the agriculture activity and GWP costs and the minimum costs were evident with $I_{60}N_{50}$ and $I_{60}N_{75}$ treatments. Nevertheless, the $I_{60}N_{50}$ and $I_{60}N_{75}$ treatments markedly declined the yield gains, resulting in the lowest NEEB values compared to the rest of the treatments in both crop growing seasons (Table 2). When compared with $I_{100}N_{100}$, the NEEB of spring wheat for $I_{60}N_{75}$ and $I_{60}N_{50}$ treatments decreased by 25.2 % and 27.0 % in 2015, and 27.9 % and 29.8 % in 2016, respectively.

3.7. Relationship between GHGs emissions, soil water content, and soil mineral N

The relationship of N_2O, CH_4, and CO_2 fluxes with soil water content (WFPS) and mineral N (NO_3^- and NH_4^+) content for different treatments are presented in Fig. 7. Soil N_2O fluxes showed strong and significant positive ($P < 0.01$) relations with soil water content, NO_3^-, and NH_4^+ content (Fig. 7). The regression analysis also revealed significant ($P < 0.01$) relationships of CH_4 uptake with WFPS (except for $I_{100}N_{75}$ and $I_{100}N_{100}$ treatments), NO_3^- (except for $I_{60}N_{50}$), and NH_4^+ content (Fig. 7). In addition, significant positive relationships ($P < 0.01$) of CO_2 emissions with WFPS (except for $I_{80}N_{50}$ treatment), NO_3^- and NH_4^+ contents were detected for all the irrigation and N treatments (Fig. 7).

4. Discussion

4.1. Response of N_2O emissions to irrigation and fertilizer application

Soil N_2O is primarily produced as an intermediate through biologically driven autotrophic nitrification and heterotrophic denitrification (Pareja-Sánchez et al., 2020), which are highly dependent on soil water dynamics in crop fields (Sapkota et al., 2020; Shi et al., 2021). Results

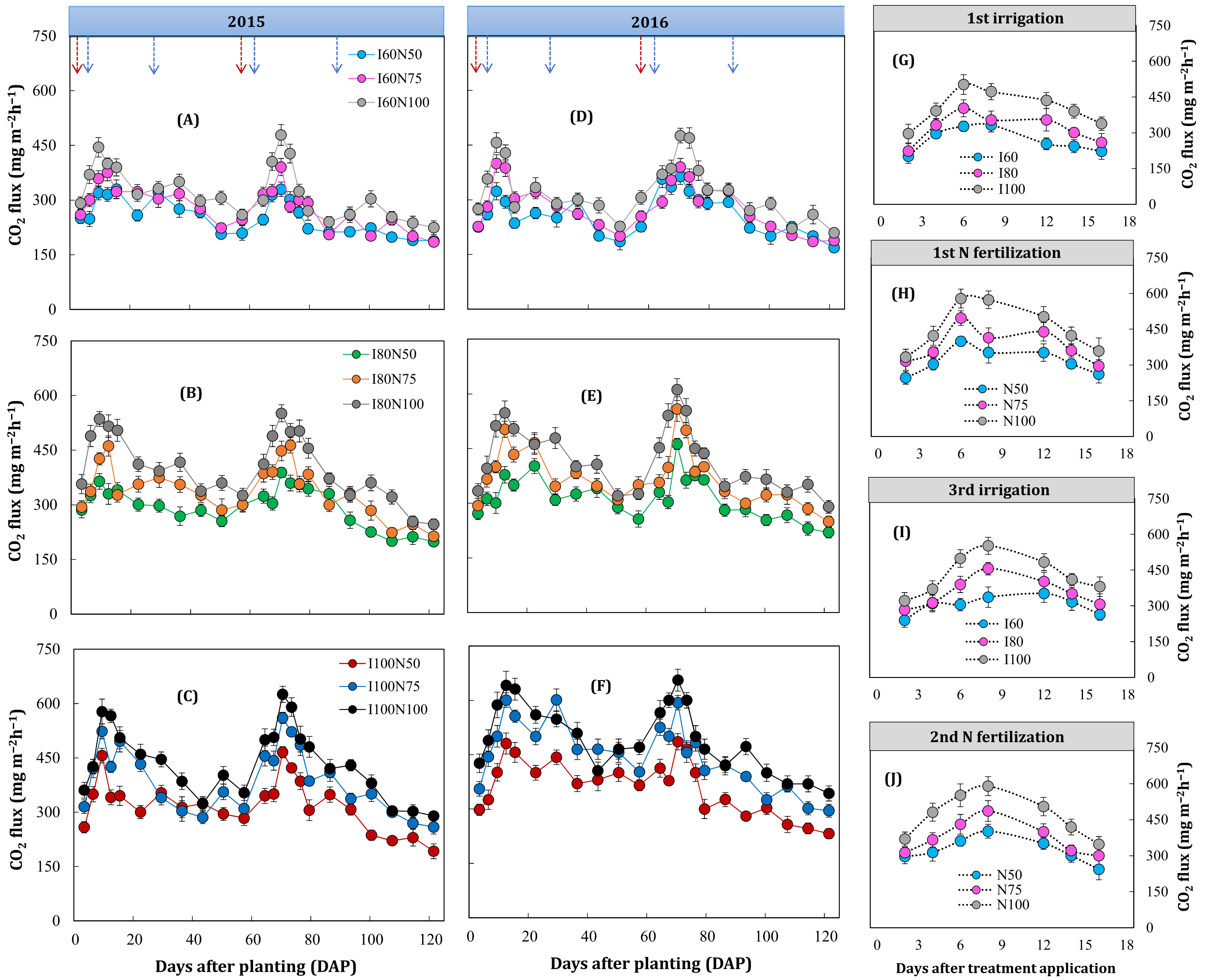

Fig. 6. Effects of irrigation (I) and nitrogen (N) treatments on soil CO_2 fluxes during 2015 (A-C) and 2016 (D-F) spring-wheat growing period. Data presented in sub-figures (G–J) are the average of two years. Data presented are the means of four replicates ± SD (n = 4). The abbreviations for treatment names are the same as those described in Fig. 1. The red and blue arrows represent N fertilizer and irrigation supplementation, respectively. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

from the present study depicted two distinct N_2O emissions fluxes at 9 and 67 DAP. This effect is attributed to the enhanced soil WFPS (38.9–58.6 %) at these stages following the effects of irrigation events, inducing better conditions for biological activity, and boosting net N mineralization (Li et al., 2020). These effects were further validated by a significant positive and linear correlation between N_2O emissions and WFPS in the present study, which is in agreement with findings from previous studies (Scheer et al., 2013; Schwenke et al., 2016; Zhong et al., 2021). Many studies indicated that WFPS from 35 % to 85 % is conducive to N_2O emissions (Hu et al., 2013; IPCC, 2022; Zhang et al., 2020). The critical WFPS for greater N_2O emissions fluxes observed at the above-mentioned growth periods were consistent with these studies. Interestingly, no distinct N_2O fluxes were perceived following the second irrigation (29 DAP) and fourth irrigation (89 DAP) events. This was likely caused by the relatively low soil substrate availability, reducing the rates of microbial activities (Abalos et al., 2014; Ning et al., 2019). These findings suggest that irrigation combined with fertilization led to higher N_2O emissions than irrigation alone, which is in support of the previous studies (Kuang et al., 2018; Zhang et al., 2020).

The difference in N_2O emissions magnitude among irrigation treatments in the present study is directly attributed to the differences in the soil water distribution patterns. The increasing availability of soil moisture with high irrigation regimes might have facilitated the formation of anoxic microsites, which promotes N_2O emissions, particularly in the presence of high soil nitrate contents (Pareja-Sánchez et al., 2020; Peyron et al., 2016). High irrigation regimes would completely and simultaneously cause a large volume of soil water-filled pores, resulting in large pulses of N_2O release from wetted soil (Wang et al., 2016; Zhong et al., 2021). Low irrigation regimes, on the other hand, leave a large volume of unfilled or partially filled pores which lead to more sporadic and generally less intense N_2O emissions pulses (Sapkota et al., 2020). Our results corroborate the findings of previous studies that indicated positive relations of N_2O emissions pulses with the increase in soil moisture contents in arid and semiarid environments (Abalos et al., 2014; Zhang et al., 2020). Wang et al. (2016) and Li et al. (2020) demonstrated that proper irrigation amount has the prospective of mitigating N_2O emissions by regulating soil aeration and inhibiting denitrification. However, according to Sapkota et al. (2020), high irrigation volumes in arid regions result in increased N_2O production, but the emissions in such regions are particularly sensitive to fertilizer

Table 2
Changes in components of the net ecosystem economic budget (NEEB) under different irrigation and nitrogen treatments in spring wheat growing seasons (2015–2016).

Year	Treatments		Yield gains (CNY ha^{-1})	Agriculture activity costs (CNY ha^{-1})	GWP costs (CNY ha^{-1})	NEEB (CNY ha^{-1})
2015	I_{100}	N_{100}	19,441 ± 493.4c	6484	38.15 ± 2.25a	12,919 ± 516.6c
		N_{75}	20,965 ± 367.8bc	6214	30.35 ± 1.64b	14,720 ± 496.3b
		N_{50}	17,998 ± 600.7d	5944	20.63 ± 1.34c	12,033 ± 428.3c
	I_{80}	N_{100}	20,729 ± 412.1bc	6284	28.25 ± 2.09b	14,417 ± 382.1b
		N_{75}	22,747 ± 539.9a	6014	20.31 ± 1.54c	16,712 ± 608.7a
		N_{50}	17,555 ± 581.1de	5744	17.91 ± 1.34cd	11,793 ± 296.2b
	I_{60}	N_{100}	16,316 ± 329.4ef	6084	20.39 ± 1.44c	10,212 ± 426.0d
		N_{75}	15,493 ± 442.5fg	5814	13.79 ± 1.71d	9665 ± 308.8d
		N_{50}	14,878 ± 272.5g	5544	13.44 ± 1.37d	9320 ± 366.9d
2016	I_{100}	N_{100}	19,706 ± 556.6cd	6484	33.09 ± 1.85a	13,189 ± 369.3cd
		N_{75}	21,021 ± 327.7b	6214	29.16 ± 2.16a	14,777 ± 587.1b
		N_{50}	18,766 ± 458.4d	5944	22.02 ± 2.04b	12,800 ± 313.5d
	I_{80}	N_{100}	20,491 ± 472.3bc	6284	29.65 ± 1.71a	14,177 ± 327.0 bc
		N_{75}	22,577 ± 624.0a	6014	21.31 ± 1.24b	16,542 ± 473.2a
		N_{50}	18,448 ± 421.1d	5744	18.48 ± 2.01 bc	12,686 ± 237.9d
	I_{60}	N_{100}	16,659 ± 323.9e	6084	18.93 ± 1.38 bc	10,556 ± 271.4e
		N_{75}	15,462 ± 295.3ef	5814	16.78 ± 1.15cd	9632 ± 379.4ef
		N_{50}	14,821 ± 354.2g	5544	14.40 ± 1.14d	9263 ± 251.8f
Treatment means						
	I_{100}		19,650 ± 383.8b	6214	28.90 ± 1.79a	13,407 ± 242.9b
	I_{80}		20,425 ± 419.2a	6014	22.65 ± 0.88b	14,388 ± 251.6a
	I_{60}		15,605 ± 195.5c	5814	16.29 ± 1.25c	9774 ± 316.4c
	N_{100}		18,890 ± 259.1b	6284	28.08 ± 1.31a	12,578 ± 143.5b
	N_{75}		19,711 ± 211.9a	6014	21.92 ± 1.66b	13,675 ± 259.1a
	N_{50}		17,078 ± 175.8c	5744	17.81 ± 1.09b	11,316 ± 239.8c

The abbreviations for treatment names are the same as described in Fig. 1. Data are the means ± SD (n = 4). Different lowercase letters indicate significant differences among treatment means based on Tukey's significant difference test ($P < 0.05$).

inputs. Therefore, these findings highlight the significance of managing both irrigation and fertilizer to effectively mitigate N_2O emissions.

Soil inorganic N serves as a substrate for the activities of nitrifiers and denitrifiers microbes (Ning et al., 2019), hence, N fertilizer management is decisive for controlling N_2O emissions from crop fields (Millar et al., 2018). Our results validated these findings as the correlation analysis revealed a significant positive association of seasonal N_2O emissions with soil NO_3^- and NH_4^+ contents at the experimental site. The N_2O emission fluxes were markedly intensified closely after N fertilization during the wheat growth stages, as it directly improved the soil NO_3^- and NH_4^+ pool. Each time, these effects lasted for about two weeks after fertilization. A similar increase in N_2O emissions with the availability of a greater specific substrate has also been confirmed across various N-managed soils (Li et al., 2020; Ning et al., 2022; Pareja-Sánchez et al., 2019; Zhang et al., 2021). The difference in cumulative N_2O emissions was comparatively lower between N_{50} and N_{75} treatments. The underlying mechanism is mainly through the efficient consumption of soil N by the wheat crop that reduces accessible substrate pools for the microbial process of N_2O production (Feng et al., 2016; Lyu et al., 2019). Nevertheless, the N_2O emissions were intensified with the N_{100} treatment because the over-application of N failed to stimulate crop growth. Once applied to crop fields, fertilizers are not fully utilized by plants (Li et al., 2015; Shi et al., 2021), and the surplus N induces direct and indirect emissions of soil N_2O (Millar et al., 2018; Ni et al., 2021). As the growth of spring wheat was markedly lower under N_{100} compared to the N_{75} treatment. Therefore, N uptake and utilization by wheat crops under N_{100} treatments were reduced, while the excess soil N increased the substrate availability for boosting nitrification and denitrification, thereby, elevating N_2O emissions. Consistently, an exponential increase in N_2O emissions was found when N input exceeded crop demand (Millar et al., 2018; Ni et al., 2021; Shcherbak et al., 2014; Yang et al., 2019; Yu et al., 2021). These results emphasize the importance of matching crop fertilizer demands which would reduce soil microbial nitrification and denitrification, and thus N_2O emissions.

4.2. Response of CH_4 emissions to irrigation and fertilizer application

In agricultural croplands, soil CH_4 emission is primarily regulated by ubiquitous methanogens and methanotrophs microorganisms (Oertel et al., 2016; Zhang et al., 2020). Based on the relative activities of these microorganisms, the soil may act as a source or sink of CH_4 (Tan et al., 2017). In the present study, negative CH_4 values in all treatments indicated that wheat fields acted as a net CH_4 sink, which is common in agricultural soils in dryland areas and well-drained sites (Ghani et al., 2022; Lyu et al., 2019; Mateo-Marín et al., 2020; Wang et al., 2016). A relatively low soil organic matter (9.3 g kg^{-1}) at the study site and low temperatures during the wheat growing period may have played a role in reducing CH_4 production by inhibiting methanogenesis, as confirmed by other studies (Li et al., 2020; Wang et al., 2016). Previously, high irrigation regimes have been reported to result in extended waterlogged or strictly anaerobic soil conditions, reducing CH_4 uptake from croplands (Li et al., 2018; Peyron et al., 2016). However, greater CH_4 uptake was observed with high irrigation regimes in the present study, which is likely due to the enhanced methanotrophs' activity stimulated by increased moisture availability during the drier wheat seasons (Pareja-Sánchez et al., 2019). As soil moisture under the arid climatic conditions of the current study does not reach the saturation point most of the time due to the irrigation intervals and high evapotranspiration rate, and would not cause a strict anaerobiosis condition. Instead, an optimum WFPS (20–60 %) with high irrigation regimes will increase the diffusivity, improving the soil porosity and air circulation, which accelerate methanotrophic CH_4 oxidation, as validated by previous studies (Wang et al., 2016; Zhang et al., 2020).

Nitrogen management is considered a key factor influencing CH_4 emissions from crop fields (Geng et al., 2017; Pareja-Sánchez et al., 2019). We observed a decrease in cumulative CH_4 uptake from the wheat field with the increase in N application, and the values were less negative at a high N application. Several factors may have contributed to the decrease in CH_4 uptake with N fertilization, including inhibition of ammonia monooxygenase responsible for CH_4 oxidation (Lyu et al., 2019), toxic inhibition by nitrite and hydroxylamine produced during the nitrification process (Li et al., 2020), and high osmotic pressure caused by excessive mineral-N concentration (Chen et al., 2016). In our study, higher soil nitrate concentration with increasing N fertilizer amounts, and negative correlations of CH_4 emissions with soil ammonium were evident, explaining the decrease in soil CH_4 uptake at high N rates. The stimulation or inhibition of CH_4 oxidation is greatly

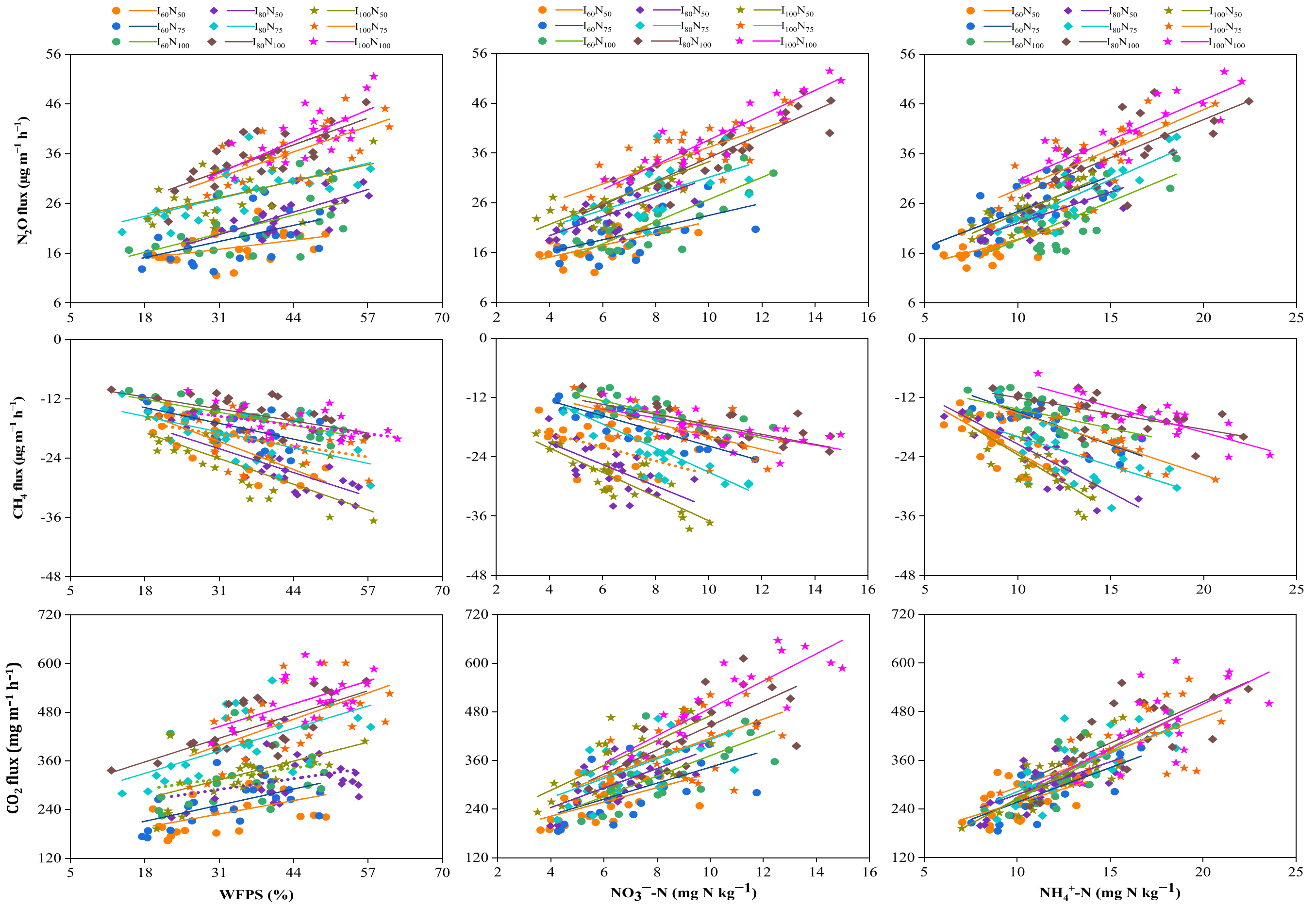

Fig. 7. Relationship of greenhouse gases (N_2O, CH_4, and CO_2) with soil water-filled pore space (WFPS) and inorganic N content (NO_3^--N, and NH_4^+-N) under different irrigation and N treatments. The solid lines represent statistically significant correlations (P < 0.01), while the dashed lines show insignificant relationships (P > 0.05).

dependent on N addition amounts (Zheng et al., 2021). For instance, a meta-analytical study by Peng et al. (2019) reported high soil CH_4 uptake at 40 kg N ha^{-1} (4 g ha^{-1} $year^{-1}$), while Li et al. (2012) observed peak values at 90 N kg ha^{-1}. However, Geng et al. (2017) reported greater CH_4 uptake at a very low N application of 10 kg N ha^{-1}, while a further increase in N rate was associated with a linear decline in CH_4 uptake. The difference in the N threshold for CH_4 uptake in these experiments is likely due to the difference in the soil available NH_4^+-N and NO_3^--N concentration at the experimental sites. As the soil in the arid regions of Northwest China is N-limited, an optimum N rate of 225 kg ha^{-1} might be suitable for wheat demand with fewer losses. Whereas, a high N rate (300 kg ha^{-1}) used by local farmers would result in greater N losses due to the amount and intensity of irrigation in the region, potentially masking the effect of inorganic N on methanotrophy (Lyu et al., 2019).

4.3. Response of CO_2 emissions to irrigation and fertilization

Soil microbial respiration and root respiration are the primary sources of CO_2 emissions (Guardia et al., 2021), which are governed by moisture availability within the crop root zone (Sapkota et al., 2020). Our results advocated that variations in CO_2 emissions fluxes were closely linked with the difference in soil moisture contents (WFPS) during wheat growing seasons. The two distinct peaks of CO_2 emission fluxes perceived in both seasons were essentially associated with higher WFPS (40.1–60.8 %) following the effects of irrigation events. In arid climates, rapid soil rewetting is known to increase CO_2 pulses, if the soil is less frequently irrigated or receives fewer precipitation episodes (Sapkota et al., 2020). The possible mechanisms of CO_2 emissions in the short term following irrigation of dry soils are: (i) the increased accessibility of rendered non-biomass soil organic materials to microbes after the irrigations (Li et al., 2020), and (ii) the wetting of dry soils triggering microbial cells to entirely lyse or responding to the increase in soil potential by releasing microbial carbon (Halverson et al., 2000). The released microbial carbon is later mineralized by soil microbes, instantaneously increasing soil respiration (Fierer and Schimel, 2003; Sapkota et al., 2020). Since soil organic matter was relatively low at the experimental site, microbial C is most likely the primary substrate mineralized to generate significant CO_2 pulses. The cumulative CO_2 emissions from wheat plots were sharply increased by irrigation regimes, with the highest values perceived with I_{100} and I_{80} treatments. This increase in CO_2 emissions can be ascribed to the significant influence of I_{100} and I_{80} treatments by (i) shifting soil moisture regimes to stimulate heterotrophic soil respiration, and (ii) improving crop productivity which stimulate autotrophic root respiration (Abalos et al., 2014). However, the relationship between soil moisture and soil CO_2 emissions is intricate and contradictory conclusions were stated by previous studies. For instance, Hou et al. (2019) and Zhong et al. (2021) reported a positive association between soil CO_2 emissions and soil moisture. A study by Wang et al. (2019) found no differences in CO_2 fluxes of the high-moisture treatment than those of the low and intermediate-moisture treatments. Whereas, Abalos et al. (2014) and Xu et al. (2016) observed that high moisture hindered the diffusion of O_2 and CO_2 in soil pores and inhibit the activities of plant roots and aerobic microorganisms. The primary reason for discrepancies in the relationship between CO_2 emissions and soil moisture in the reported studies was due to the variation in climatic conditions. Since our experimental site has a warmer arid climate and therefore irrigation events are unlikely to cause strict anaerobic conditions. Perhaps, significant positive correlations between soil CO_2 emissions and WFPS were obvious in all treatments that can further explain their relationship in the arid agro-climatic conditions of northwest China.

After fertilizer application, CO_2 is produced during the rapid transformation of urea ($CO(NH_2)_2$) into NH_4^+, bicarbonate, and hydroxyl, facilitated by soil moisture and urease enzymes (Li et al., 2020; Zhang et al., 2021). This could be a possible reason for peak CO_2 emissions fluxes following N fertilization. All of the carbon in fertilizer exceeding the crop demand is assumed to be released as CO_2 (Lyu et al., 2019). As a result, CO_2 emissions fluxes persisted for several days following fertilizer application in our study, supporting the findings from the previous studies (Abalos et al., 2014; Zhang et al., 2021). In general, N application rates increased the cumulative CO_2 emissions, with a significant difference between N_{100} and N_{75}/N_{50} treatments. The difference between N_{50} and N_{75} treatments was not significant, implying that CO_2 emissions are increased only by excessive N application. In explanation, N_{50} and N_{75} treatments meet the crop nutrient demands, leaving comparatively low soil N residuals, and had little impact on the direct emissions of soil CO_2 (Yu et al., 2021). Another possible reason could be the low soil organic contents in the experimental soil. In low organic matter soils, only excessive N applications are reported to increase seasonal CO_2 emissions, but emissions increase markedly at any N rate in soils with sufficient organic matter (Sainju et al., 2012; Zhang et al., 2020). The mechanism of soil CO_2 emissions following fertilization is inextricably linked to the stimulatory effects of N on microbial activities and soil respiration (Tan et al., 2017; Yu et al., 2021). Therefore, applying an appropriate amount of N fertilizer based on crop demands is an effective strategy for reducing both direct and indirect CO_2 emissions.

4.4. Response of GWP, GHGI, and NEEB to irrigation and N application

The determination of GWP and GHGI provided an assessment of the trade-off between wheat production and GHGs emissions under different irrigation and N treatments. Our results portrayed significantly higher GWP values with full irrigation and fertilizer management than that with reduced treatments. Positive GWP values indicated that wheat production systems acted as a net GHGs source and both irrigation and N treatments play a significant role in driving GWP in arid regions. Meanwhile, N_2O was found to be the primary contributor to GWP in each cropping cycle because CH_4 uptake exceeded emissions in wheat fields. These findings are consistent with those of prior studies in which N_2O was identified as the major contributor to GWP in dry environments (Li et al., 2020; Lyu et al., 2019; Zheng et al., 2021). Accordingly, in arid regions where intensive irrigation and fertilization are practised, efforts to reduce GWP should focus more on N_2O rather than CH_4. Moreover, GHGI under different treatments was influenced both by the difference in wheat yield and GWP. On average, the full irrigation and N management ($I_{100}N_{100}$) conceded the highest seasonal GWP and GHGI. Remarkably, the lowest GHGI values were achieved with moderately reduced irrigation and N treatment ($I_{80}N_{75}$), resulting from the simultaneous decrease in N_2O emissions and increase in wheat yield compared to other treatments. Therefore, GWP mitigation measures should emphasize optimizing irrigation and N management to reduce N_2O emissions, and improving grain yield to minimize the GHGI. Several studies on maize (Li et al., 2020; Zhang et al., 2020) and rice production (Chen et al., 2016; Yu et al., 2021) corroborate this hypothesis.

The cropland economic benefit analyses are often confined to yield profits and input costs, but the relationship between economic benefit and environmental impacts is often overlooked (Li et al., 2015, 2020). In this study, NEEB was determined by the wheat yield gains, agricultural inputs, and GWP costs, allowing an accurate and comprehensive assessment of irrigation and N treatments for economic feasibility and environmental sustainability. Remarkably, the NEEB of the $I_{100}N_{100}$ treatment was significantly lower than that of the $I_{80}N_{75}$ treatment, implying that intensive agriculture with high inputs may not always improve economic profits. In the present study, the yield benefits of full irrigation and N application failed to compensate for the higher investment and at the same time increased the GHGI and GWP as a result of significant N_2O emissions during the wheat growing seasons. On the other hand, very low irrigation and N application rates ($I_{60}N_{50}$) may substantially decrease the agriculture activity costs and GWP but would not be effective in improving the NEEB because of no yield gains. Overall, these findings specified the possibility of balancing the economic and environmental benefits for wheat production in arid regions of Northwest China by optimizing irrigation and N fertilizer management.

5. Conclusions

Results from the two years study in the spring wheat fields provided insights into the assessment of wheat yield, net economic benefits, GHGs emissions, GWP, and their relationships with various soil properties as affected by the conventional full and deficit irrigation and N application rates. Full irrigation and N application were intrinsically associated with high GHGs emissions, GWP and GHGI. On the other hand, reduced irrigation and fertilization showed the potential of decreasing GHG emissions and improving GWP and GHGI but with variable effects on the wheat grain yield and NEEB values. A decrease in irrigation amount by 20 % and N fertilizer rate by 25 % ($I_{80}N_{75}$) was the best management that resulted in the highest wheat yield and net economic benefits, concurrently lowering the seasonal fluxes and cumulative GHG emissions compared to full irrigation and nitrogen treatment ($I_{100}N_{100}$). These findings suggest that reducing irrigation and fertilizer rate in an appropriate amount has the potential to protect the agricultural environment by reducing GWP and GHGI, whilst increasing the economic advantages of spring wheat in arid the region of China.

Authors contribution

FH designed and supervised the experiments. MK and ZY performed experiments. MK wrote the manuscript. IA, QJ, MUG, and SC helped in lab analysis. XC and TL helped in the data analysis. KHMS and SF helped in writing and refining the manuscript. All authors have read and approved the final 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.

Data Availability

Data will be made available on request.

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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.2022.108197.

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