Scheer, Clemens, Grace, Peter R., Rowlings, David W., & Payero, Jose (2012) Nitrous oxide emissions from irrigated wheat in Australia : impact of irrigation management. *Plant and Soil*, *359*(1-2), pp. 351-362.

http://dx.doi.org/10.1007/s11104-012-1197-4

Nitrous oxide emissions from irrigated wheat in Australia: Impact of irrigation management

Abstract

Background and Aims

Irrigation management affects soil water dynamics as well as the soil microbial carbon and nitrogen turnover and potentially the biosphere-atmosphere exchange of greenhouse gasses (GHG). We present a study on the effect of three irrigation treatments on the emissions of nitrous oxide (N_2O) from irrigated wheat on black vertisols in South-Eastern Queensland, Australia.

Methods

Soil N_2O fluxes from wheat were monitored over one season with a fully automated system that measured emissions on a sub-daily basis. Measurements were taken from 3 subplots for each treatment within a randomized split-plot design.

Results

Highest N_2O emissions occurred after rainfall or irrigation and the amount of irrigation water applied was found to influence the magnitude of these “emission pulses”. Daily N_2O emissions varied from -0.74 to 20.46 g N_2O-N ha^{-1} day^{-1} resulting in seasonal losses ranging from 0.43 to 0.75 kg N_2O-N ha^{-1} season $^{-1}$ for the different irrigation treatments. Emission factors (EF = proportion of N fertilizer emitted as N_2O) over the wheat cropping season, uncorrected for background emissions, ranged from 0.2 to 0.4% of total N applied for the different treatments. Highest seasonal N_2O emissions were observed in the treatment with the highest irrigation intensity; however, the N_2O intensity (N_2O emission per crop yield) was highest in the treatment with the lowest irrigation intensity.

Conclusions

Our data suggest that timing and amount of irrigation can effectively be used to reduce N_2O losses from irrigated agricultural systems; however, in order to develop sustainable mitigation strategies the N_2O intensity of a cropping system is an important concept that needs to be taken into account.

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Introduction

Irrigated agriculture plays a vital role in meeting the global food demand of a growing population in the context of climate change. It is estimated that nearly two-thirds of future food needs must come from irrigated agriculture (FAO 1996). At the same time agriculture is known to emit significant amounts of greenhouse gases (GHGs) to the atmosphere. Globally agricultural activities (including those on grazing lands) account for 15-20% of total greenhouse gas emissions and the agricultural sector is the largest contributor to non-CO_2 emissions emitting about 50% and 60% of total anthropogenic emissions of CH_4 and N_2O, respectively (Smith et al. 2007). Nitrous oxide emissions from agricultural activities are expected to increase by about 50% by 2020 due to increased use of nitrogen fertilizer and animal manure (US-EPA 2006). Current estimates of N_2O emissions from agriculture still show a wide range of uncertainties due to the scarcity of data for farming systems under different environmental and management conditions (Stehfest and Bouwman 2006). In Australia, there are few detailed studies on N_2O emissions from subtropical farming systems. A study on GHG emissions from sugar cane in north-eastern Australia observed N_2O emissions of 45.9 kg N_2O-N ha^{-1} yr^{-1} and emission factors of 21% (Denmead et al. 2010), in contrast to Barton et al. (2008) who reported low (110 g N_2O-N ha^{-1} yr^{-1}) N_2O emissions from a rain-fed, cropped sandy soil in semi-arid south-western Australia, with an emission factor of only 0.02%. These contrasting results show clearly that more detailed field measurements are required in order to obtain reliable estimates of N_2O emissions from soils and to assess GHG mitigation potential in different agricultural systems.

Soil moisture has been identified as the most sensitive factor to regulate N_2O emissions from croplands since it directly regulates oxygen availability in soil pores, which determines the activity of nitrification and denitrification within the soil profile (Zheng et al. 2000). In irrigated systems soil moisture is considered to be one of the most important factors to mitigate N_2O emissions since it can be easily controlled. Although it has been shown that high soil water levels after irrigation, in combination with high input of mineral N fertilizer, can lead to significantly elevated emissions of N_2O (Liu et al. 2011; Scheer et al. 2008b), there is still only limited information available on N_2O fluxes from irrigated agricultural systems worldwide and no data on irrigation management as a potential abatement strategy

The Murray-Darling River Basin (MBD) is the most productive agricultural region in Australia, covering approximately 90 million hectares and contributing nearly 40 per cent of the country's agricultural gross value product. The total area of irrigated land in the MBD was 1.6 million hectares in 1996/97, increasing by 16% to 1.9 million hectares in 2000/01. Of the total irrigated area in the MDB in 2000/01, 29% was under dairy, 22% under cotton and 13% under cereals (Bryan 2004).

The Darling Downs region of the MDB is especially noted for its deep fertile clay soils, making this region one of the most productive in Australia for grain and cotton. With the availability of irrigation water and the subtropical climate two crops per year can be grown on the same land. This continuous cropping has led to a decline in fertility of many soils and in irrigated crops high rates of nitrogen fertilizer are often needed. The high levels of mineral nitrogen in the soil in combination with high moisture levels after irrigation could possibly lead to significant N losses to the environment and elevated emissions of N_2O can be expected (Scheer et al. 2008b). However, so far the extent of N_2O losses from these irrigated systems is largely unknown due to the lack of field measurements. Moreover, there have been no investigations examining how both the amount and frequency of irrigation affect N_2O emissions from irrigated agricultural systems. Consequently, the aims of our study were to quantify the fluxes of N_2O from irrigated wheat on black vertisol in

South-Eastern Queensland, and at the same time assess the influence of different irrigation regimes on N_2O emissions.

Material and methods

Study site

The field experiment was conducted during the 2009 wheat season at the Agri-Science Queensland, Department of Employment, Economic Development & Innovation (DEEDI) Kingsthorpe research station. The station is located about 20 km north-west of the city of Toowoomba, Queensland, Australia ($27^{o}30'44.5''$ Latitude South, $151^{o}46'54.5''$ Longitude East, 431 m above mean sea level). The climate is sub-tropical with predominantly summer rainfall and mean annual rainfall of 630 mm. The mean daily minimum and maximum temperatures are 16.3 and 27.2°C in the summer, and 5.9 and 17.0°C in winter, respectively. The soil at the site is a haplic, self-mulching, black vertosol (Isbell 2002). It has a heavy clay texture (76% clay) in the 1.5 m root zone profile, with a distinct change in soil color from brownish black (10YR22) in the top 90 cm to dark brown (7.5YR33) deeper in the profile. The soil is of alluvial fan and basalt rock origin, slowly permeable, with a surface slope of about 0.5%. Physical and chemical soil characteristics of the experimental plots are shown in Table1.

Experimental design

The experiment was conducted using three irrigation treatments and three replications arranged in a completely randomized block design. Each experimental plot was 13 m wide x 20 m in length, with the crop planted in the North-South orientation. A 4 m wide buffer zone was planted between plots and a 4 m road was located at the centre of the research area. The irrigation treatments included:

1. High irrigation (HI). Irrigation was applied when 50% of the plant available water capacity (PAWC) was depleted.
2. Medium Irrigation (MI). Irrigation was applied when 60% of the PAWC was depleted.
3. Low irrigation (LI). Irrigation was applied when 85% of the PAWC was depleted.

The plots were irrigated individually with bore water using a hand-shift sprinkler. Results of quality analysis of the bore water are shown in Table 2. Partial-circle sprinkler heads were used to avoid irrigating adjacent plots. Irrigations were applied during times with low wind speeds to assure uniformity of application. Irrigation amounts were measured using a rain gauge installed at the centre of each plot and were scheduled based on neutron probe soil water content measurements. The wheat was planted on June 11 and harvested on October 26, 2009. All treatments received a total N application rate of 200 kg N ha^{-1} applied as urea in three applications. 100 kg N ha^{-1} was applied at sowing, 50 kg N ha^{-1} at first node, and 50 kg N ha^{-1} applied at flag leaf emergence. Amount and timing of fertilizer application and irrigation are shown in Table 3.

Continuous N_2O flux measurement

N_2O fluxes were measured with a mobile fully automated measuring system during the entire cropping season of wheat from June 15 to October 26, 2010. Measurements were taken from 3 subplots for each treatment within a split-plot design. Nine acrylic sampling chambers (50 cm x 50 cm x 15 cm) were fixed on stainless steel frames, when the crop height exceeded about 20 cm a chamber extension of 50 cm height was used. The lids of the chambers were opened and closed automatically with pneumatic pistons. During a normal measurement cycle, three chambers were closed at one time and four air samples taken from each chamber sequentially for 48 min (12 min apart) before the chambers were opened again and the next three chambers closed and sampled. It therefore took 144 min for all chambers to be sampled and up to 10 single flux rates could be determined per chamber and day. The air samples were automatically pumped from the head-space of the chamber into a gas chromatograph SRI 8610C, Torrance/USA) equipped with a ^{63}Ni electron capture detector (ECD) for N_2O analysis. To minimize the interference of moisture vapor and CO_2 on N_2O measurement, an Ascarite (sodium-hydroxide-coated silica) pre-column filled was installed upstream of the ECD and changed at fortnightly intervals.

Sample gas measurements were calibrated automatically by a single point calibration using certified gas standards (Air Liquide, Dallas, TX, USA) of 0.5 ppm N_2O. The detection limit of the system was approximately 0.5 g N_2O-N ha^{-1} day^{-1} without the chamber extensions and 2.0 g N_2O-N ha^{-1} day^{-1} with the chamber extension on. Sample dilution via leakage was considered negligible. Further details on the automated system and analytical conditions applied for gas analyses are found in Breuer et al. (2000) and Kiese and Butterbach-Bahl (2002). N_2O fluxes were calculated from the slope of the linear increase or decrease in N_2O concentration during the chamber lid closure and corrected for air temperature, atmospheric pressure and the ratio of chamber volume to surface area as described in detail by Barton et al. (2008). The Pearson's correlation coefficient (r^2) for the linear regression was calculated and used as a quality check for the measurement. Flux rates were discarded if r^2 was < 0.80.

Auxiliary measurements

Soil temperature (at a depth of 10 cm) and chamber temperature was measured every minute in conjunction with the automatic sampling system using a PT100 probe (IMKO Germany). Soil moisture was measured in each plot at least weekly at 0.10 m depth increments to a depth of 1.5 m with the neutron probe method using a 503DR Hydroprobe (CPN International, Inc., Martinez, CA, USA) that was calibrated for the soil at the research site. The soil water module of the DSSAT model (Jones et al. 2003) was used to simulate the soil moisture content of the upper soil depths on a daily basis. Water-filled pore space (WFPS) was calculated using the measured soil bulk density data (arithmetic means of four samples) using a particle density of 2.65 g cm^{-3}. Additionally, at the beginning and end of the growing season, bulk soil samples were taken from each plot by combining 5–10 soil cores (0–10 cm depth) and analyzed for soil texture (hydrometer method as described by Carter and Gregorich (2008)), total carbon (C %) and total nitrogen (N %) using a Flash EA 1112 NC analyser Thermo Instruments; San Jose, CA. In each plot, grain yield was measured at harvest by collecting 10m from 16 rows of each experimental plot using a plot combine.

Calculations and Statistical Analysis

Statistical analysis was undertaken using SPSS 16.0 (SPSS Inc., USA). Non-normal distribution of N_2O fluxes was shown using the Kolmogorov-Smirnov test. The non-parametric pair-wise Wilcoxon test was used without any data transformation for the comparison of the different irrigation treatments. Daily N_2O losses for each treatment were calculated by averaging hourly losses for that day. Cumulative seasonal N_2O fluxes were calculated by integrating daily N_2O fluxes over the study period. Emission factors were calculated uncorrected for background emission over the cropping season for wheat and expressed as the percentage of the total fertiliser N applied that was emitted as N_2O-N. The N_2O intensity of each treatment was calculated as the ratio of N_2O emissions in relation to crop yield and relates to how much N_2O is emitted per ton of grain produced.

Results

Over the cropping season, a total of 123 mm rain was recorded at the study site. In addition, the HI, MI and LI treatments received 244 mm of irrigation water in 7 applications, 161 mm in 5 applications and 65 mm in 4 applications, respectively (Table 3). Over the season, 200 kg N ha^{-1} was applied as urea to the different treatments. Average N_2O flux (over 137 days) was 5.5 g N_2O-N ha^{-1} day^{-1} in the HI treatment, 3.2 g N_2O-N ha^{-1} day^{-1} in the MI treatment and 3.3 g N_2O-N ha^{-1} day^{-1} in the LI treatment, which corresponded to a total amount of 0.75 kg, 0.43 kg and 0.45 kg of N emitted as N_2O over the season for the different treatments, respectively (Table 4). Statistically, there was no difference in seasonal N_2O emissions from the MI and the LI treatments, while the seasonal emission from the HI treatment was significantly higher. Emission factors, uncorrected for background emissions, varied from 0.2% to 0.4% of the total amount of mineral N applied to the plots (Table 4).

Average yield was highest in the HI treatment with 3.1 t ha^{-1} and significantly lower in the MI (1.9 t/ha) and LI (1.6 t ha^{-1}) treatments, which is at the lower end of irrigated wheat yields recorded in the area in 2009.

The temporal course of the measured N_2O fluxes is displayed in Figure 1. Fertilizer was initially applied at wheat planting and irrigated four days later with 19 mm of irrigation water in all treatments. Subsequently, elevated N_2O emissions were observed in all treatments and increased further after rainfall on June 25-26 (20 mm). This pattern was observed in all three treatments; however absolute emissions were significantly higher in the HI treatment despite the fact that there was no difference in management at the onset of the experiment. Overall, there was high temporal and spatial variation in N_2O fluxes in all treatments and the highest losses occurred after rainfall or irrigation, often in combination with fertilizer application. Highest daily fluxes measured in the HI, MI and LI treatments were 20.5, 13.9 and 13.3 g N_2O-N ha^{-1} day^{-1}, respectively. The magnitude of these "emission pulses" was significantly different for the treatments and influenced by the amount of irrigation water that had been applied.

The temporal course of N_2O fluxes under different irrigation intensities after application of fertilizer for two emission pulses is depicted in Figure 2. For both events the plots were fertilized with 50 kg-N of urea and subsequently irrigated. A higher amount of irrigation in the HI treatment resulted in significantly higher emissions of N_2O for both events. Mean daily emission for the period from 25 August until 2 September was 7.1, 2.6 and 1.7 g N_2O-N ha^{-1} day^{-1} in the HI, MI and LI treatments, respectively; for the period from 22 September until 1 October emissions were 5.8, 4.0 and 3.9 g N_2O-N ha^{-1} day^{-1} for the respective treatments.

A significant diurnal effect of soil temperature (at 10 cm depth) on N_2O fluxes could be observed for certain periods when soil moisture conditions or soil mineral N content were non-limiting and a representative example (28 – 30 June) is shown in Figure 3. Soil temperature variation during this period ranged from 8.4 °C to 15.8 °C with maximum soil temperatures occurring between 15:00 and 16:00 and minimums between 7:00 and 8:00. The diurnal variation of N_2O fluxes was greater than 10-fold for some chambers with maximum emissions between 18:00 and 24:00 and minimums between 8:00 and 14:00. The highest amplitude was observed in one chamber of the HI treatment and ranged from 3.5 to 24.0 g N_2O-N ha^{-1} day^{-1}. Fluxes were generally increasing during daytime and decreasing during the night with daily emission maxima in the late evening and minima in the early morning. Figure 3 shows also a high spatial variability of N_2O fluxes and mean emissions over this three day period ranged from 2.1 g N_2O-N ha^{-1} day^{-1} in one chamber of the LI treatment to 13.2 g N_2O-N ha^{-1} day^{-1} in one chamber of the HI treatment.

Discussion

N_2O emissions from irrigated wheat

Our measurements represent the first field data set in Australia on the influence of irrigation intensities on N_2O emissions. Moreover it is one of the few studies that report on N_2O fluxes from irrigated agricultural systems using a high resolution, fully automated monitoring system. Daily N_2O emissions observed in this study ranged from -0.7 to 20.5 g N_2O-N ha^{-1} day^{-1} and cumulative emissions from 0.42 kg N_2O ha^{-1} to 0.75 kg N_2O-N ha^{-1} over the cropping cycle for wheat. These values would rank in the lower range of emissions reported for other irrigated cereal crops (Liu et al. 2011; Pathak et al. 2002; Scheer et al. 2008a), however, little data is available for irrigated systems in Australia. Rochester (2003) used ^{15}N fertiliser balance studies where fertiliser losses had been measured to estimate N_2O losses from different Australian soil types. He reported N_2O emissions from alkaline grey clay soils to be in the range of 1.6-2.6 kg N_2O-N ha^{-1} but estimated N_2O losses from wheat on acidic soil to be substantially larger (> 10 kg N_2O-N ha^{-1}). Matson et al. (1998) reported seasonal fluxes of up to 6 kg N_2O-N ha^{-1} and flux rates of up to 1500 g N_2O-N ha^{-1} day^{-1} for an irrigated wheat production system in Mexico fertilised with 250kg N ha^{-1} of anhydrous ammonia. We presume that the comparable low N_2O losses at our site even after application of 200 kg N ha^{-1} and intensive irrigation are mainly attributed to the neutral soil pH and the low SOC content at our study site. It has been demonstrated that soil pH can have a strong effect upon the composition of N gases emitted and the activity of the N_2O reductase enzyme is generally thought to increase with increasing pH values (Chapuis-Lardy et al. 2007). Hence at our site the soil pH of 7.2 should have resulted in dinitrogen (N_2) being the major gas emitted following denitrification, however further research is needed on denitrification rates and the N_2/N_2O emissions ratio in such cropping systems to confirm this hypothesis. Moreover, denitrification rates and N_2O production in soils are tightly linked to the SOC status and various studies found a positive correlation between N_2O emissions and SOC in field measurements and laboratory experiments (Bouwman et al. 2002; Weier et al. 1993). In the study region, current practice is to produce two crops per year under irrigation and remove the residues of both crops. This cropping practice has led to a significant decline in SOC. We hypothesize that the low SOC of the soil at our site limited the denitrification activity and consequently the losses of N_2O.

Previous research on N_2O emissions from irrigated agriculture has reported a strong stimulation of N_2O emissions by irrigation and fertilisation (Liu et al. 2011; Scheer et al. 2008a). The results of the present study corroborate these findings since highest emissions were generally found after irrigation events, often in combination with fertilization (Figure 1). During these emission pulses elevated soil moisture contents (WFPS > 70%) coincided with a high availability of mineral N in the soil following fertilisation, although we do not have detailed field data on soil mineral N content to corroborate this finding. This effect was most pronounced immediately after planting in the second half of June when plant growth was still limited and consequently not competing with soil microbial processes for available N in the soil.

A clear diurnal N_2O response to daily temperature fluctuations could be observed when other soil parameters (e.g. WFPS) were not overriding the temperature effect (Figure 3). Highest fluxes generally occurred in the evening/night and lowest fluxes during the morning/early afternoon following the soil temperature at 10 cm depth with a time lag of 2–5 h. This contrast to other studies who reported highest fluxes in the afternoon (Livesley et al. 2008; Scheer et al. 2008b) or observed no significant diurnal variation in N_2O emissions (Smith and Dobbie 2001). It is not clear what caused N_2O fluxes to peak at night, the time lag of several hours compared to the soil temperature peak could indicate that emissions may have originated from a greater depth than 10 cm (Smith et al. 1998). Moreover, we presume that oxygen and carbon availability within the soil profile could have been affected by diurnal patterns of plant/root activity which in turn will effect soil microbial activity and N_2O production. The significant diurnal variation shows clearly that single point measurements made during the day do not adequately represent the true N_2O daily flux rates. Taking only one sample per day between 9-11am as it is commonly practiced by many manual sampling campaigns would have resulted in a 37 to 43% underestimation of the mean seasonal N_2O flux of the different treatments. These findings demonstrate the need to check the diurnal N_2O emission pattern and emphasize the value of automated trace gas measurements with repeated sub-daily measurements for obtaining reliable absolute flux estimates.

N_2O emission factors and N_2O emissions in relation to crop yield.

In the current study, N_2O-N losses over the cropping season for wheat (uncorrected for background fluxes) ranged from 0.2 to 0.4% of total fertiliser N applied for the different treatments. These values are within the range of emission factors reported for other irrigated wheat systems. Scheer et al. (2008a) estimated seasonal losses of 0.3 to 0.5% from an irrigated winter wheat in Uzbekistan, while Pathak et al. (2002) reported seasonal emissions of 0.2 to 0.4% from winter wheat in the Indo-Gangetic plains in India. The IPCC (2006) recommends the use of an emission factor (1% of N applied) to calculate fertiliser induced emissions from cropped soils for inventory purposes. We did not measure the background N_2O emissions and present data for the wheat cropping season only, therefore these emission factors need to be treated with caution when comparing them to other long term studies. Nonetheless the data suggests that the proportion of fertiliser N lost as N_2O from irrigated cropping systems on black vertisol in Australia is likely to be considerably lower than the IPCC default value and that the IPCC value may not be suitable for such a system. However, the uncorrected emission factors found in this study are in good agreement with the emission factor for irrigated cotton (0.5%) used by the Australian government for their national GHG Inventory report (ANGA 2010). This demonstrates the need to use the local emission factors to reliably estimate emissions for irrigated cropping systems on black vertisols in Australia (Galbally et al. 2005). Whilst we have used a single season study to estimate N_2O emissions, the uncertainty associated with this value is minimal when you consider we have used irrigation to significantly reduce the impacts of water limitations on crop growth. More long term field measurements including different fertilizer management strategies would ideally be required in order to understand the influence of management, climate and soil on N_2O emissions and to validate emission factors for specific agricultural systems in Australia.

The ratio of N_2O emissions in relation to crop yield shows how much N_2O is emitted per ton of grain produced and can be used as an indicator on how effective the cropping system is in terms of maximizing crop yield and reducing N_2O emissions. In our study this “N_2O intensity” ranged from 0.23 to 0.28 kg N_2O-N per ton of yield (Table 4). While the highest absolute N_2O emissions were found in the HI treatment, this N_2O intensity was actually highest in the LI treatment, due to a significantly reduced crop yield (Table 4). These data demonstrate clearly that for the identification of mitigation options of N_2O emissions and best management practices the N_2O intensity of cropping systems is an important concept that needs to be taken into account. We think that for sustainable farming strategies it is more important to increase fertilizer use efficiency and minimize the N_2O intensity rather than reducing absolute N_2O emissions alone.

Influence of irrigation management on N_2O emissions and implications for mitigating N_2O emissions

Highest seasonal N_2O emissions were observed in the treatment with the highest irrigation intensity and periods of high N_2O fluxes were generally observed after irrigation or rainfall events. Furthermore we saw a clear impact of the amount of irrigation water on the magnitude of N_2O during the irrigation events following fertiliser application in August and September (Figure 3). Irrigation events greater than 50 mm in the HI treatment significantly increased the magnitude of N_2O emissions (up to 4 fold) during the following days compared to the MI and LI treatment where only 20 mm to 35 mm of irrigation water had been applied. These observations are in line with other studies where a clear relationship between amount of rainfall/irrigation and seasonal N_2O emissions has been found (Dobbie and Smith 2003; Liu et al. 2011). Moreover, several studies identified soil moisture and soil mineral N concentration as key factors regulating N_2O emission from croplands and the reported optimum soil water content for elevated N_2O emissions was found in the range from 50%-99% WFPS (del Prado et al. 2006; Dobbie et al. 1999; Liu et al. 2010; Zheng et al. 2000). In our study it seems that the high irrigation amounts in the HI treatment increased the soil water content to such an extent that the optimum water content was reached and sustained for a longer period compared to the lower irrigation levels. Unfortunately we do not have precise field data for soil moisture during these irrigation events and the modelled soil moisture values are not accurate enough to show those differences. However, apart from these single events where the differences of N_2O emissions can clearly be attributed to the irrigation intensity we also observed significant difference in N_2O fluxes between the different treatments at the onset of the study when there was no difference in the treatment. These differences are most likely caused by observed high spatial variability within the different treatments as shown in Figure 3 which partly overrode the irrigation effect of this study. It remains unclear what caused this high spatial variability, but we assume that it was caused by natural soil heterogeneity and differences in soil carbon, nitrogen and water content due to the management history of the plots.

The high emissions of N_2O at the onset of the study also suggest that the timing of the fertilizer applications can be an important factor influencing the magnitude of N_2O emissions. A high fertilizer application rate before planting when there is no competition from plant uptake with soil microbial processes for available N in the soil is likely to increase N_2O emissions and should generally be avoided. This finding is in accordance with a study by Ortiz-Monasterio et al. (1996) who reported a reduction of N_2O emissions of more than 50% by applying fertiliser later in the season for irrigated wheat in Mexico (Schulze et al. 2009). Splitting the fertiliser into several applications compared to a large single initial dose could be another option to reduce N_2O emissions by decreasing the availability of mineral N in the soil during irrigation events. However, the effect of splitting fertiliser application is not entirely clear in the literature (Allen et al. 2010; Burton et al. 2008; Weier 1999) and needs to be tested for this specific cropping system. Overall the study shows that timing and amount of irrigation can effectively be used to reduce N_2O losses from irrigated agricultural systems in South-Eastern Queensland. Large irrigation amounts following fertilization or when the soil mineral N content is high and large application of fertilizer N before planting should be avoided wherever possible. However, the ratio of N_2O emissions in relation to crop yield from the different treatments clearly demonstrates that the irrigation and fertilization management needs to be matched with ‘best management practice’ in order to mitigate N_2O emissions and reduce the N_2O intensity of the cropping system at the same time.

Conclusion

To our knowledge this is the first study in Australia on how the amount and frequency of irrigation is affecting N_2O emissions from irrigated agricultural systems. Daily N_2O emissions, ranged from -

0.7 to 20.5 g N_2O-N ha^{-1} day^{-1}, and were at the lower end of N_2O fluxes reported from irrigated agricultural systems. The amount of fertilizer N lost as N_2O ranged from 0.2 to 0.4% of total fertiliser applied. These emission factors, although uncorrected for background N_2O emissions and measured over the wheat cropping season only, are considerably lower than the IPCC default value of 1%, suggesting that this default value may not be appropriate for soil C-deficient irrigated systems. A large diurnal variation of N_2O fluxes during some periods clearly demonstrates that daily point measurements are often insufficient to produce representative N_2O flux rates. These results highlight the need for more long term studies based on automated trace gas measurements with sub-daily resolution in order to reliably estimate emissions for specific agricultural systems.

The amount of irrigation water was found to influence the magnitude of N_2O fluxes, in particular after the application of N fertilizer. This indicates that timing and amount of irrigation can potentially be used to reduce N_2O losses from these cropping systems by avoiding large irrigation amounts following fertilization or when the soil mineral N content is high. In addition, this study demonstrates that the N_2O intensity concept should be taken into account when developing sustainable mitigation options for N_2O emissions from cropping systems. It is more important to minimize the N_2O intensity by increasing fertilizer use efficiency and optimize yield performance rather than reducing absolute N_2O emissions alone.

Acknowledgments

We thank Geoff Robinson for his valuable help in the field measuring campaign. The Department of Employment, Economic Development & Innovation (DEEDI) for providing the study site and the farm staff for planting and harvesting the experimental plots. This research was undertaken as part of the national Nitrous Oxide Research Program (NORP) funded by the Grains Research and Development Corporation (GRDC) and Department of Agriculture, Fishery and Forestry (DAFF). We also thank two anonymous reviewers for valuable comments on an earlier version of the manuscript.

References

Allen DE, Kingston G, Rennenberg H, Dalal RC, Schmidt S (2010) Effect of nitrogen fertilizer management and waterlogging on nitrous oxide emission from subtropical sugarcane soils. Agric. Ecosyst. Environ. 136, 209-217.

ANGA (2010) Australian National Greenhouse Accounts In National Inventory Report 2008. Department of Climate Change and Energy Efficiency.

Barton L, Kiese R, Gatter D, Butterbach-Bahl K, Buck R, Hinz C, Murphy DV (2008) Nitrous oxide emissions from a cropped soil in a semi-arid climate. Glob. Change Biol. 14, 177-192.

Bouwman AF, Boumans LJM, Batjes NH (2002) Modeling global annual N_2O and NO emissions from fertilized fields. Global Biogeochemical Cycles 16.

Breuer L, Papen H, Butterbach-Bahl K (2000) N_2O emission from tropical forest soils of Australia. Journal of Geophysical Research-Atmospheres 105, 26353-26367.

Bryan B, Marvanek, S. (2004) Quantifying and Valuing Land Use Change for Integrated Catchment Management Evaluation in the Murray-Darling Basin 1996/97–2000/01. . CSIRO Land and Water, Adelaide.

Burton DL, Zebarth BJ, Gillarn KM, MacLeod JA (2008) Effect of split application of fertilizer nitrogen on N(2)O emissions from potatoes. Can. J. Soil Sci. 88, 229-239.

Carter MR, Gregorich EG (2008) Soil sampling and methods of analysis. Canadian Society of Soil Science.

Chapuis-Lardy L, Wrage N, Metay A, Chotte JL, Bernoux M (2007) Soils, a sink for N2O? A review. Glob. Change Biol. 13, 1-17.

del Prado A, Merino P, Estavillo JM, Pinto M, Gonzalez-Murua C (2006) N_2O and NO emissions from different N sources and under a range of soil water contents. Nutr. Cycl. Agroecosyst. 74, 229-243.

Denmead OT, Macdonald BCT, Bryant G, Naylor T, Wilson S, Griffith DWT, Wang WJ, Salter B, White I, Moody PW (2010) Emissions of methane and nitrous oxide from Australian sugarcane soils. Agric. For. Meteorol. 150, 748-756.

Dobbie KE, McTaggart IP, Smith KA (1999) Nitrous oxide emissions from intensive agricultural systems: Variations between crops and seasons, key driving variables, and mean emission factors. Journal of Geophysical Research-Atmospheres 104, 26891-26899.

Dobbie KE, Smith AK (2003) Nitrous oxide emission factors for agricultural soils in Great Britain: the impact of soil water-filled pore space and other controlling factors. Glob. Change Biol. 9, 204-218.

FAO (1996) Rome declaration on world food security and worldfood summit plan of action. Food and Agriculture Organization of the United Nations, Rome.

Galbally I, Meyer M, Bentley S, Weeks I, Leuning R, Kelly K, Phillips F, Barker-Reid F, Gates W, Baigent R, Eckard R, Grace P (2005) A study of environmental and management drivers of non-CO_2 greenhouse gas emissions in Australian agro-ecosystems. Environmental Sciences 2, 133-142.

IPPC (2006) Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme. Ed. BL Eggleston H.S., Miwa K., Ngara T. and Tanabe K, IGES, Japan.

Isbell RF (2002) The Australian soil classification CSIRO Publishing, Melbourne.

Jones JW, Hoogenboom G, Porter CH, Boote KJ, Batchelor WD, Hunt LA, Wilkens PW, Singh U, Gijsman AJ, Ritchie JT (2003) The DSSAT cropping system model. Eur. J. Agron. 18, 235-265.

Kiese R, Butterbach-Bahl, K. (2002) N_2O and CO_2 emissions from three different tropical forest sites in the wet tropics of Queensland, Australia. Soil Biology and Biochemistry 34, 975-987.

Liu CY, Wang K, Meng SX, Zheng XH, Zhou ZX, Han SH, Chen DL, Yang ZP (2011) Effects of irrigation, fertilization and crop straw management on nitrous oxide and nitric oxide emissions from a wheat-maize rotation field in northern China. Agric. Ecosyst. Environ. 140, 226-233.

Liu CY, Zheng XH, Zhou ZX, Han SH, Wang YH, Wang K, Liang WG, Li M, Chen DL, Yang ZP (2010) Nitrous oxide and nitric oxide emissions from an irrigated cotton field in Northern China. Plant Soil 332, 123-134.

Livesley SJ, Kiese R, Graham J, Weston CJ, Butterbach-Bahl K, Arndt SK (2008) Trace gas flux and the influence of short-term soil water and temperature dynamics in Australian sheep grazed pastures of differing productivity. Plant Soil 309, 89-103.

Matson PA, Naylor R, Ortiz-Monasterio I (1998) Integration of environmental, agronomic, and economic aspects of fertilizer management. Science 280, 112-115.

Ortiz-Monasterio I, Matson PA, Panek J, Naylor RL (1996) Nitrogen fertilizer management for N_2O and NO emissions in Mexican irrigated wheat. In Transactions 9th Nitrogen Workshop. pp 531-534, Braunschweig, Germany.

Pathak H, Bhatia A, Prasad S, Singh S, Kumar S, Jain MC, Kumar U (2002) Emission of nitrous oxide from rice-wheat systems of Indo-Gangetic plains of India. Environ. Monit. Assess. 77, 163-178.

Rochester IJ (2003) Estimating nitrous oxide emissions from flood-irrigated alkaline grey clays. Aust. J. Soil Res. 41, 197-206.

Scheer C, Wassmann R, Kienzler K, Ibragimov N, Lamers JPA, Martius C (2008a) Methane and nitrous oxide fluxes in annual and perennial land-use systems of the irrigated areas in the Aral Sea Basin. Glob. Change Biol. 14, 2454-2468.

Scheer C, Wassmann R, Klenzler K, Lbragimov N, Eschanov R (2008b) Nitrous oxide emissions from fertilized irrigated cotton (Gossypium hirsutum L.) in the Aral Sea Basin, Uzbekistan: Influence of nitrogen applications and irrigation practices. Soil Biology & Biochemistry 40, 290-301.

Schulze ED, Luyssaert S, Ciais P, Freibauer A, Janssens IA, Soussana JF, Smith P, Grace J, Levin I, Thiruchittampalam B, Heimann M, Dolman AJ, Valentini R, Bousquet P, Peylin P, Peters W, Rodenbeck C, Etiope G, Vuichard N, Wattenbach M, Nabuurs GJ, Poussi Z, Nieschulze

J, Gash JH, CarboEurope T (2009) Importance of methane and nitrous oxide for Europe's terrestrial greenhouse-gas balance. Nature Geoscience 2, 842-850.

Smith KA, Dobbie KE (2001) The impact of sampling frequency and sampling times on chamber-based measurements of N2O emissions from fertilized soils. Glob. Change Biol. 7, 933-945.

Smith KA, Thomson PE, Clayton H, McTaggart IP, Conen F (1998) Effects of temperature, water content and nitrogen fertilisation on emissions of nitrous oxide by soils. Atmos. Environ. 32, 3301-3309.

Smith P, Martino D, Cai Z, Gwary D, Janzen H, Kumar P, McCarl B, Ogle S, O’Mara F, Rice C, Scholes B, Sirotenko O (2007) Agriculture. In Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Ed. B Metz, Davidson, O.R.,Bosch, P.R., Dave, R., Meyer, L.A. Cambridge University Press, United Kingdom and New York NY, USA.

Stehfest E, Bouwman L (2006) N_2O and NO emission from agricultural fields and soils under natural vegetation: summarizing available measurement data and modeling of global annual emissions. Nutr. Cycl. Agroecosyst. 74, 207-228.

US-EPA (2006) Global Anthropogenic Non-CO_2 Greenhouse Gas Emissions: 1990-2020. United States Environmental Protection Agency, EPA 430-R-06-003, Washington, D.C.

Weier KL (1999) N_2O and CH_4 emission and CH_4 consumption in a sugarcane soil after variation in nitrogen and water application. Soil Biology & Biochemistry 31, 1931-1941.

Weier KL, Doran JW, Power JF, Walters DT (1993) Denitrification and the dinitrogen/nitrous oxide ratio as affected by soil water, available carbon and nitrate. Soil Science Society of America Journal 57, 66-72.

Zheng X, Wang M, Wang Y, Shen R, Gou J, Li J, Jin J, Li L (2000) Impacts of soil moisture on nitrous oxide emission from croplands: a case study on the rice-based agro-ecosystem in Southeast China. Chemosphere - Global Change Science 2, 207-224.

Table 1. Selected soil properties of the experimental plots at the Kingsthorpe research station, Queensland, Australia.

Soil Property	Irrigation Treatment		
	HI	**MI**	**LI**
Carbon 0-10cm (g kg^{-1})	16.1 ± 1.0	15.0 ± 0.3	15.5 ± 0.6
Nitrogen 0-10cm (g kg^{-1})	1.5 ± 0.15	1.7 ± 0.03	1.8 ± 0.07
pH (H_2O)	7.3 ± 0.4	7.2 ± 0.2	7.2 ± 0.2
Texture (USDA)	Clay	Clay	Clay
Clay (%)	76	76	76
Silt (%)	16	16	16
Sand (%)	7	7	7

Table 2. Results of water quality analysis from bore water sample collected at Kingsthorpe research station during March 2009.

Analysis	Result	Units
pH	8.1	pH units
Electrical Conductivity	2048	µS/cm
Total dissolved ions	1311	mg/L
Bicarbonate Alkalinity	277.5	mg $CaCO_3$/L
Total Alkalinity	281	mg $CaCO_3$/L
Total Hardness	691	mg $CaCO_3$/L
Sodium Absorption Ratio	1.2	
Calcium	102	mg/L
Magnesium	110	mg/L
Sodium	75	mg/L
Phosphorus	<1	mg/L
Potassium	11	mg/L
Sulphur	40	mg/L
Aluminium	<0.1	mg/L
Zinc	<0.01	mg/L
Iron	<0.01	mg/L
Copper	<0.01	mg/L
Manganese	<0.01	mg/L
Boron	0.05	mg/L
Molybdenum	0.05	mg/L
Salt from Chloride	747	mg/L
Chloride	452	mg/L
Corrected SAR	1.6	
Effective Conductivity	1540	µS/cm
Residual Alkali	<0.01	meq/L

Table 3. Amount of irrigation water [mm] and N fertilizer applied to three irrigation treatments (I-50, I-60 and I-85%) at the Kingsthorpe research station, Queensland, Australia in 2009.

	Irrigation Treatment			Fertilizer
Date	**HI**	**MI**	**LI**	**[kg-N ha^{-1}]**
12/06/2009				100
16/06/2009	19	19	19	
1/08/2009	51			
11/08/2009		60		
25/08/2009	57			50
26/08/2009		25		
27/08/2009			20	
21/09/2009	69	34		50
23/09/2009			13	
25/09/2009			13	
8/10/2009	27			
9/10/2009	21	23		
Total [mm]	**244**	**161**	**65**	**200**

Table 4. N_2O fluxes, emission factors, irrigation/rain amount, grain yield and yield - N_2O ratio from three irrigation treatments (HI, MI and LI) at the Kingsthorpe research station, Queensland, Australia in 2009. Means denoted by a different letter indicate significant differences between the treatments (Wilcoxon test; $p < 0.05$).

	Irrigation Treatment		
Measurements	**HI**	**MI**	**LI**
Average Flux [g N_2O-N $ha^{-1} day^{-1}$]	5.5^a	3.2^b	3.3^b
Seasonal Flux [kg N_2O-N ha^{-1} $season^{-1}$]	0.75^a	0.43^b	0.45^b
Emission Factor [%]*	0.38	0.22	0.23
Irrigation + rain [mm]	367	284	188
Grain yield [t ha^{-1}]	3.1	1.9	1.6
N_2O intensity [kg-N_2O-N t-$yield^{-1}$]	0.24	0.23	0.28

***uncorrected for background emissions of N_2O-N**

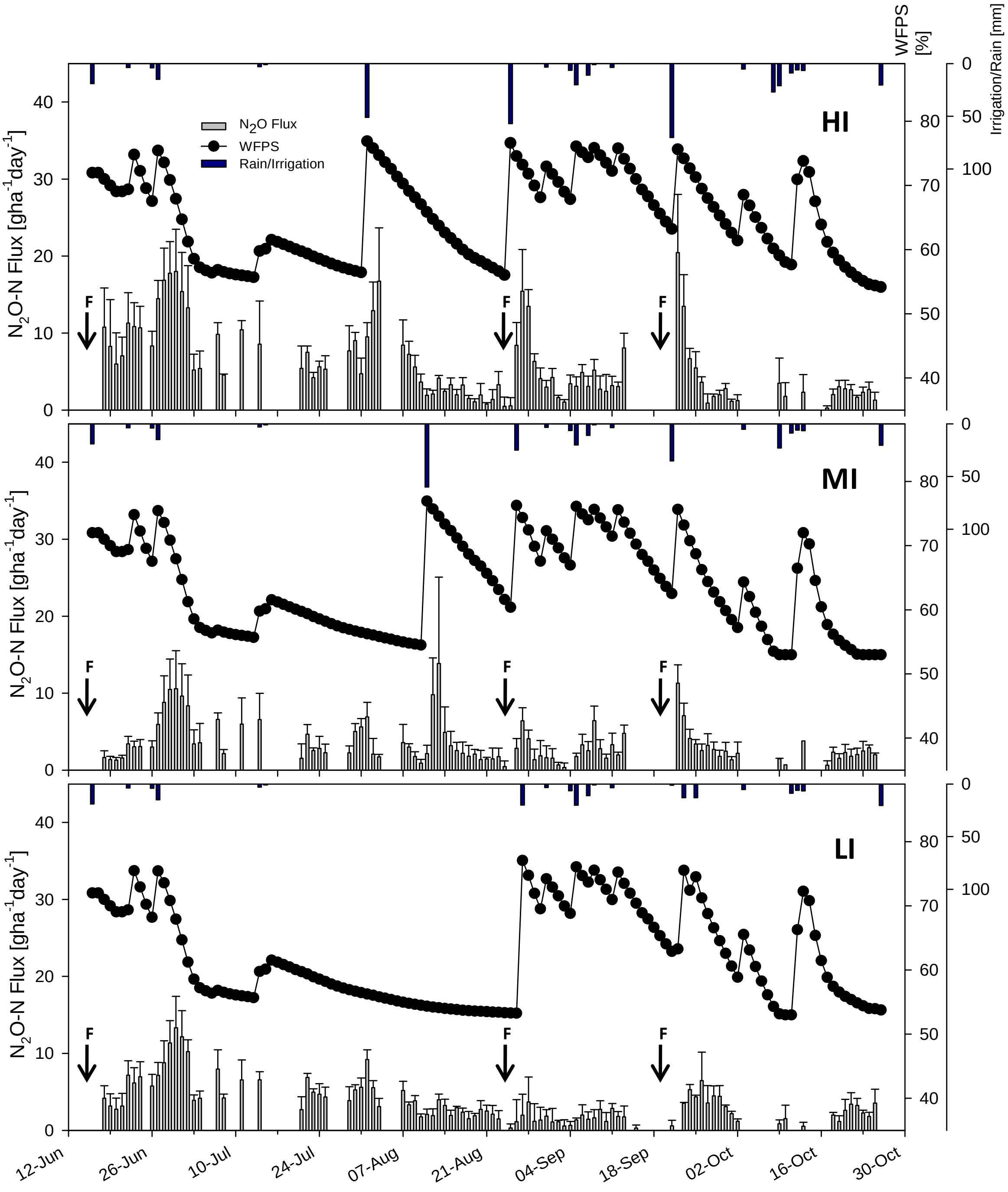

Figure 1. Daily N_2O fluxes, amounts of rainfall and irrigation and water-filled pore space (WFPS) in relation to three irrigation treatments (HI, MI and LI) at the Kingsthorpe research station, Queensland, Australia in 2009. Error bars indicate the standard error of the means (n = 3). Arrows indicate the timing of N fertilizer applications.

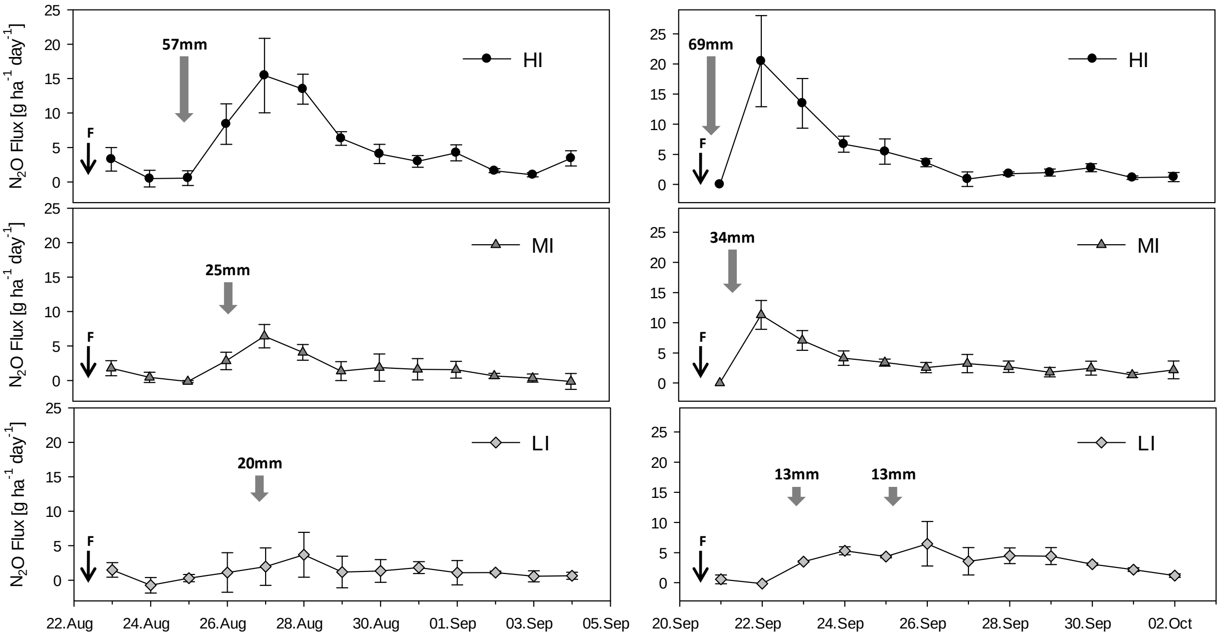

Figure 2. Influence of irrigation and fertilizer application on N_2O emissions for two selected irrigation periods from three irrigation treatments (HI, MI and LI) at the Kingsthorpe research station, Queensland, Australia in 2009. Black arrows mark the timing of fertilization (50kg of Urea N), grey arrows with numbers show timing and amount of irrigation for the different treatments. Error bars indicate the standard error of the means (n = 3).

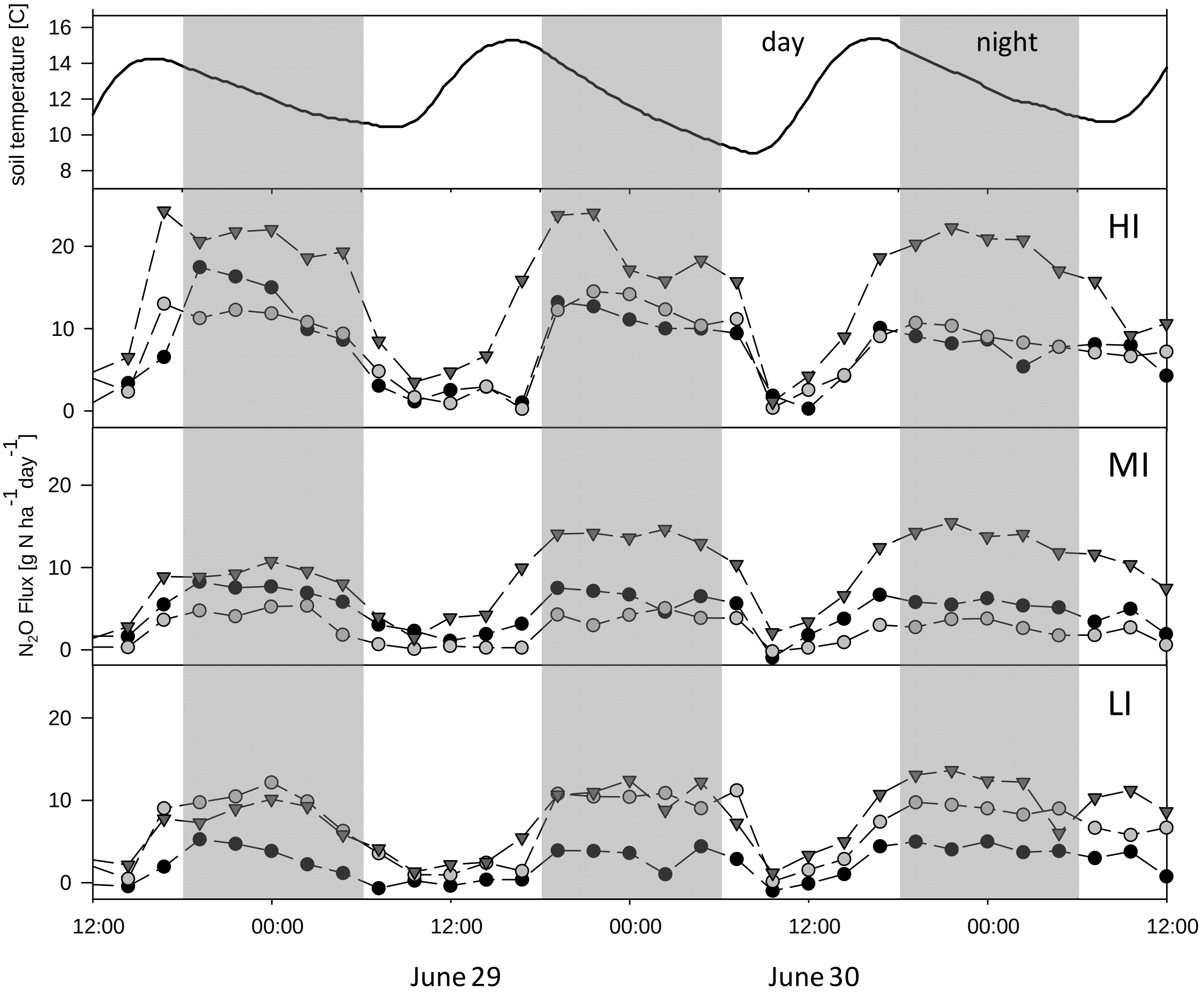

Figure 3. Diurnal pattern of N_2O fluxes and soil temperature (10cm) during three measuring days in June 2009 from three irrigation treatments (HI, MI and LI) at the Kingsthorpe research station, Queensland, Australia in 2009. Fluxes are displayed for each individual measuring chamber (n=3) per treatment.