Field Crops Research 146 (2013) 66–74

Contents lists available at SciVerse ScienceDirect

Field Crops Research

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

Nitrogen management to reduce yield-scaled global warming potential in rice

X.Q. Liang[a,*], H. Li[b], S.X. Wang[c], Y.S. Ye[a], Y.J. Ji[a], G.M. Tian[a], C. van Kessel[d], B.A. Linquist[d]

[a] Institute of Environmental Science and Technology, College of Environmental and Resource Sciences, Zhejiang University, Hangzhou 310058, China
[b] Institute of Environment, Resource, Soil and Fertilizer, Zhejiang Academy of Agricultural Sciences, Hangzhou 310021, China
[c] Jiangxi Academy of Agricultural Sciences, Nanchang 330200, China
[d] Department of Plant Sciences, University of California, Davis 95616, USA

ARTICLE INFO

Article history:
Received 28 December 2012
Received in revised form 28 February 2013
Accepted 6 March 2013

Keywords:
CH_4 and N_2O
Global warming potential
Manure
Rice
Urea
Yield-scaled emissions

ABSTRACT

Fertilizer N is usually required to achieve optimal yields but when applied in excess there is increased risk of pollution, including higher greenhouse gas (GHG) emissions. Thus, optimal N management must consider both yields and environmental effects. Yield-scaled GWP (Global Warming Potential), which is the GWP (in CO_2 equivalents) per Mg of grain yield, is a useful metric for evaluating management options where the goal is to achieve both high yields with minimal environmental burden. A 6-year field study was conducted to test the hypothesis that the lowest yield-scaled GHG emissions for rice occur when N is applied at optimal N rates for maximum yields, independent of the source of N applied. We tested this hypothesis for organic (manure) and inorganic (urea) N sources. The N rates and sources in each growing season were: 0, 90, 180 and 270 kg N ha^{-1} applied as either urea alone or pig manure combined with urea (where N was added as manure and supplied 60% of the total N rate). The N rates to achieve maximum yields (90 to 180 kg N ha^{-1} depending on year) were similar for both N sources. Seasonal CH_4 and N_2O emissions varied significantly between years but the magnitude of emissions was determined largely by N source. Across N rates, application of manure increased GWP by almost 60% relative to the urea treatments due to higher CH_4 and N_2O emissions. When urea was used as the sole N source, yield-scaled GWP (87 kg $CO_{2\ eq.}$ Mg^{-1} grain) was lowest at optimal N rates for maximum yields. In contrast, when manure was used, yield-scaled GWP was higher than for urea and increased with increasing manure-N rates (from 104 to 171 kg $CO_{2\ eq.}$ Mg^{-1} grain). The lowest yield-scaled GWP for manure was when no manure was applied – despite the low yields. Thus, when manure is used as an N source in flooded rice systems, over application should be avoided.

1. Introduction

Flooded rice (*Oryza sativa* L.) systems emit both CH_4 and N_2O, however they are the largest agricultural source of CH_4 emissions and account for 5–19% of the annual global CH_4 emissions (IPCC, 2007). Despite N_2O being a greenhouse gas (GHG) that is 12 times more potent than CH_4 (IPCC, 2001), results from a recent meta-analysis (Linquist et al., 2012b) reported that CH_4 emissions accounted for almost 90% of total global warming potential (GWP) from rice systems and concluded that efforts to reduce GWP of rice systems should focus on CH_4.

China grows rice on approximately 30 million ha per year and is the world's leading producer (FAO, 2012). It is estimated that 7.7–8.0 Tg CH_4 and 138–154 Gg N_2O are emitted from Chinese rice fields (Yan et al., 2003; Zheng et al., 2004), with a GWP between 219 and 229 Tg $CO_{2\ eq}$. In recent years rapid urbanization in China has increased pressure to intensify agricultural production in places such as the Taihu Lake region; and as a result, rice growers are applying more N (Ju et al., 2009). As the recovery of applied fertilizer N is usually less than 20% (Wang et al., 2003) it can contribute to the high GWP from the rice systems (Huang and Tang, 2010), along with other environmental and economic concerns. Therefore, it is necessary to establish management practices that optimize N use to achieve high yields with minimal environmental cost – the basis of sustainable intensification (Godfray et al., 2011).

Whether in this region or globally, rice production must increase to meet food demand. Mueller et al. (2012) reported that closing the rice grain yield gap to 100% of attainable yields would require a 47% increase in global rice production. Increasing rice production will inevitably increase CH_4 and N_2O along with its associated GWP, however attempts must be made to increase production with the lowest GWP. Linking grain yield with GWP provides a metric to

* Corresponding author. Tel.: +86 57188981721.
E-mail address: liang410@zju.edu.cn (X.Q. Liang).

0378-4290/$ – see front matter © 2013 Elsevier B.V. All rights reserved.
http://dx.doi.org/10.1016/j.fcr.2013.03.002

quantify economic viability (yields) with environmental concerns. This metric has been termed GHG intensity (Mosier et al., 2006) or yield-scaled GHG emissions (van Groenigen et al., 2010) and is reflected in the GWP in CO_2 equivalents per unit of grain yield. Although soil CO_2 fluxes also represent a source of GHG emissions, on a global scale they are largely offset by high rates of net primary productivity and atmospheric CO_2 fixation by crop plants, and are estimated to contribute less than 1% to the GWP of agriculture (Smith et al., 2007). Therefore, CO_2 as a contributor to GWP was not included in our analysis. van Groenigen et al. (2010) reported that yield-scaled GHG emissions were lowest where N was applied at optimal rates and excessive N rates led to high yield-scaled emissions.

Yield-scaled emissions are affected by both N input and source (Shang et al., 2011). In many regions of the world farmers have a choice between organic and inorganic sources of N. Urea and pig manure are the two main N sources applied to rice systems in this region of China. The effect of N source on rice productivity is conflicting with some claiming manure to be more effective than urea at improving yields and N use efficiency (Pan et al., 2009; Bi et al., 2009; Duan et al., 2011) while others urea (de Ponti et al., 2012).

Other studies have reported that manure N sources increase GWP particularly because of higher CH_4 emissions in flooded rice systems (Yang et al., 2010; Shang et al., 2011); and based on a meta-analysis, Linquist et al. (2012a) reported that manure applied at similar N rates as urea resulted in 26% higher CH_4 emissions but no significant difference in N_2O emissions. In addition, some studies have also evaluated the effect of varying rates of mineral fertilizer N on CH_4, N_2O and GWP (i.e. Yao et al., 2012). However, no study has determined how N source (mineral vs. organic) affects yield-scaled GWP. Such studies require an evaluation across a range of N rates and sources to adequately identify optimal N rates for each source and determining GHG emissions from these N treatments. Therefore, we initiated a 6-year field study to test the hypothesis that the lowest yield-scaled GWP for rice is when N is applied at optimal N rates for maximum yields, independent of the source of N applied. Our objective was to determine yield-scaled GWP when different rates of inorganic (urea) or organic (pig manure) N are applied. It is anticipated that the outcome of this research will provide a basis to develop efficient management strategies that address the need for high yields while also addressing environmental concerns.

2. Materials and methods

2.1. Site description

The field experimental site was located in the Taihu Lake region, Jiaxing City, Zhejiang Province, China (120°40′E, 30°50′N). The climate is a subtropical monsoon climate with mean annual temperature and precipitation of 15.7 °C and 1200 mm, respectively. Prior to the experiment the field had been continuously cropped with rice for more than 700 years (Cao, 2008). The soil was classified as a gleyed paddy soil (clay loam, mixed, mesic Mollic Endoaquepts) derived from the riverine-lucustrine sediments. The initial soil properties of the plow layer (0–15 cm) were: pH, 6.9; organic C, 18.2 g kg^{-1}; total N, 2.65 g kg^{-1}; total P, 1.51 g kg^{-1}; CEC, 8.12 cmol kg^{-1}; sand content,12.1%; clay content, 51.7%; and bulk density, 1.33 g cm^{-3}.

2.2. Fertilization treatments and management

The experiment was established in 2005 with treatments arranged in a randomized complete block design with three replications. The cropping system was a rice-rape (*Brassica napus*) system with rice being grown in the wet season and rape in the dry season. Only results related to rice are reported here. Each plot was 4 m × 5 m. Throughout the six-year experiment, the N treatments remained the same for each plot. To reduce edge effects, non-experimental guard plots planted with rice were established around the entire experiment. In each rice season the following were identical: the variety (cv. JIA 9321), transplanting 25 day old seedlings on July 1, drainage (October 3), and harvest (October 30).

There were seven N treatments: a control (no N) and three rates of N (90, 180, and 270 kg N ha^{-1}) applied either as urea (90U, 180U and 270U) or a combination of manure and urea (90 M, 180 M and 270 M). For urea treatments the N was applied in three doses (basal fertilizer/1st topdressing/2nd topdressing) at a ratio of 3:1:1. For manure treatments, pig manure (C: 8.5%; N: 0.56%; P: 0.43%; K: 0.40%) was applied as the basal N fertilizer and the remaining 40% of the N rate was applied as urea in two equal topdressing doses-similar to the urea treatments. Thus, manure provided 60% of 90, 180 and 270 kg N ha^{-1}, or 54, 108, and 162 kg N ha^{-1}, respectively. To ensure that P and K were not limiting, the control and all urea treatments received 40 kg P_2O_5 ha^{-1} (superphosphate) and 150 kg K_2O ha^{-1} (KCl) and the 90 M treatment received 73 kg K_2O ha^{-1}. These fertilizers were broadcast onto soil surface by hand and incorporated with the 0–5 cm soil layer by puddling at the same time as the basal N fertilizer was applied. The urea topdress application was broadcast onto the flooded field 2 and 4 weeks after transplanting (WAT) each year to all treatments except the control. The irrigation water level was maintained at 5–8 cm from transplanting until the drainage which occurred between 13 and 14 WAT. Other field practices, such as field preparation, tillage, puddling, and weed control, were carried out manually according to the local farming practices.

2.3. Sampling and analysis

Flux measurements of N_2O and CH_4 were performed simultaneously using static Plexiglas chambers and gas chromatography (GC) techniques. The Plexiglas chamber, covered on the outside with bubble foil insulation, was modified from Ni and Zhu (2004) with dimensions of 0.6 m × 0.6 m and a height of 0.6 or 1.2 m depending on plant height. A small fan was fixed to the interior top surface to mix the chamber air before sampling. During each rice growing season, gas samples were taken from 08:00 to 10:00 h weekly. For each sampling event gas samples were taken at 0, 15, 30, 45 min from the middle space of each chamber and transferred into 15 mL vacuum vials and analyzed for CH_4 and N_2O using a GC (Shimadzu, GC-14B series). In addition the temperature inside the chamber was measured for the flux calculation. The gas flux rates were determined from the linear increase in gas concentrations within the chamber over time. The GWP of N_2O and CH_4 emissions was calculated in units of CO_2 equivalents ($CO_{2\ eq.}$) over a 100-yr time horizon. A radiative forcing potential relative to CO_2 of 298 was used for N_2O and 25 for CH_4 (IPCC, 2001).

At harvest, the aboveground biomass and grain yields were determined from a 1 m^2 area within each plot. After oven drying to a constant weight at 60 °C the biomass was weighed to determine yields. The biomass was then ground and analyzed for N content using an Elemental Analyzer (Vario Max, Germany). N recovery efficiency (NRE) was calculated as:

$$\text{NRE} = [\text{N uptake(N treatment)} - \text{N uptake(control)}]/\text{N applied.}$$

2.4. Data analysis

The seasonal CH_4 and N_2O emission totals from all the treatments and rice growing seasons were computed directly from the measured fluxes and were estimated by linear interpolation for days when no measurements were made.

Data were analyzed using EXCEL and SPSS 16 for Windows. Arithmetic means for three replicates were calculated for all parameters. Statistical analysis was accomplished by standard analysis of variance (ANOVA) and mean values compared by least significant difference (LSD) at the 0.01 level of probability. Interactions between N rate, sources, and year were analyzed by General Linear Model in the SPSS. Yield-scaled N_2O, CH_4, and GWP ($N_2O + CH_4$) among N treatments and sources were analyzed using PROC MIXED with least significant difference test at P-value < 0.01 (SAS, 2003).

3. Results and discussion

3.1. Climate conditions

Seasonal precipitation and temperature were collected from a nearby weather station for the entire experimental period. Year to year variation in precipitation and temperature during the rice growing season (July to October), was relatively small (Table 1). Average precipitation was 688 mm with the driest and wettest year between 2009 and 2010, respectively. Mean seasonal temperature averaged 26.4 °C with 2005 being the coolest year and 2008 the warmest.

3.2. Rice yield and N uptake in response to N rate and source

Across years and treatments yields averaged 6430 kg ha^{-1} (Table 2). In general, average yields were similar between years, however 2009 yields were the lowest, averaging 6% lower than the experimental average. The reason for lower yields in 2009 remains unclear and the climate data (Table 1) do not provide any reliable explanation. In the control treatment where no N was applied, yields averaged 5178 kg ha^{-1}. Control yields were initially high in 2005 (6104 kg ha^{-1}) and tended to decline with time to a low of 4167 in 2009. A similar pattern was also observed for N uptake in the control treatment where initially N uptake was high (103 kg N ha^{-1}) but then declined in each year to a low of 58 kg N ha^{-1} in 2010 (Table 2).

There was a yield response to both N sources but no interaction between N rate and year or N source (Table 3). In 2005 through 2007 the optimal N rate was 90 kg N ha^{-1} for both N sources while from 2008 to 2010 it was 180 kg N ha^{-1}. This increase in N requirement over time is most likely due to a diminishing native N supply over time as observed in the control treatment (Table 2). At rates above 180 kg N ha^{-1} yields remained similar or declined for both N sources. Our results concurred well with the findings of Lin et al. (2007) and Pan et al. (2003), who reported that maximum yields could be achieved with N rates of 150 to 227 kg N ha^{-1} for intensively managed paddy fields.

At similar N rates, crop N uptake was higher in the urea than in the manure treatments by 6% on average (Table 2). Nitrogen recovery efficiency (NRE) was also higher in the urea treatment (averaged 47%) than in the manure treatment (averaged 41%) (data not shown), similar to findings of others (Brahmanand et al., 2009; Golden et al., 2006; Wild et al., 2011). However, the NRE reported here for manure is higher than reported for poultry litter which range from 14 to 35% (Bijay-Singh et al., 1997; Takahashi et al., 2004; Golden et al., 2006). The high NRE for manure observed in our study is most likely because the manure treatment included urea in the topdress application.

3.3. Temporal dynamics of CH_4 and N_2O emission fluxes

In general, seasonal CH_4 emissions were low in this study (Table 4). Seasonal CH_4 emissions, averaged across years and N rates, were 19.8, 21.6 and 33.6 CH_4 kg ha^{-1} for the control, urea and manure treatments, respectively. Higher CH_4 emissions from manure are to the result of increased the carbon inputs. On average the use of manure increased CH_4 emissions by 56% relative to urea which is higher than the 26% (95% CI: 12–47%) reported from a meta-analysis by Linquist et al. (2012a).

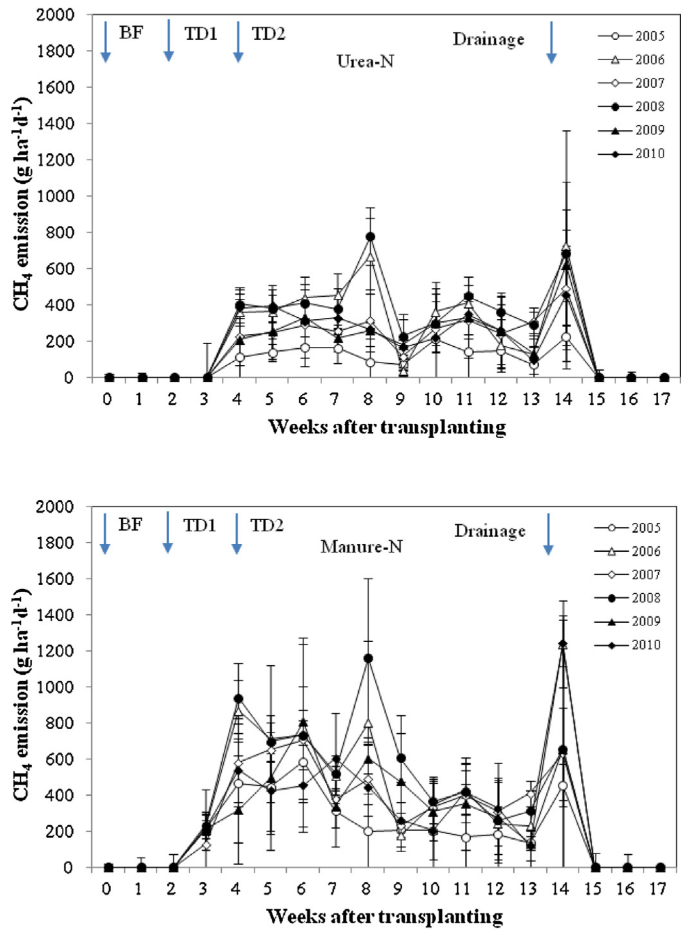

Fig. 1. Temporal dynamics of CH_4 emitted from rice fields which received 180 kg N ha^{-1} of urea and pig manure N fertilizers. BF: basal fertilizer; TD1: the first topdressing; TD2: the second topdressing. Error bars are the standard deviation of the three replicates.

CH_4 emissions followed very similar patterns in all six years of the experiment (Fig. 1). In the first two weeks of the season CH_4 emission rates were near zero for all treatments regardless of fertilizer rate and source (Fig. 2). After two weeks, CH_4 emissions were first observed in the manure treatments while in the urea treatments CH_4 emissions were not observed until week three. The highest CH_4 emissions were observed between the fourth and eighth WAT during tillering and booting stages. During this period CH_4 emissions from the manure treatments were higher than the urea treatments. The mean fluxes during this period for urea (180U) and manure (180 M) were 318 and 583 g CH_4 ha^{-1} d^{-1}, respectively (data not shown). In the urea treatments, CH_4 emissions stayed relatively similar from 8 to 13 WAT after which the field was drained. During this same period, CH_4 emissions from the manure treatments declined to levels similar in magnitude to emissions from urea treatments. The mean fluxes from 8 to 13 WAT for urea (180U) and manure (180 M) were 230 and 294 g CH_4 ha^{-1} d^{-1}, respectively (date not shown). Following drainage at the end of the season a spike in CH_4 emissions was observed followed by a period of no CH_4 emissions. Such spikes in CH_4 emissions following the drain period have been observed in other studies (Yagi et al., 1996; Wassmann

Table 1
Monthly precipitation and temperature during the rice growing seasons from 2005 to 2010.

	2005	2006	2007	2008	2009	2010	Average
	Precipitation (mm)						
July	186	250	179	150	172	212	191
August	202	104	184	200	261	200	192
September	120	255	213	182	145	191	184
October	167	59	116	166	54	160	120
July–October	675	667	692	698	632	763	688
	Temperature (°C)						
July	29.8	29.6	30.2	30.5	29.0	29.3	29.7
August	28.4	30.4	30.0	28.8	28.5	31.3	29.6
September	26.9	23.8	25.2	26.4	25.6	26.5	25.7
October	19.0	22.2	20.3	21.0	21.5	19.1	20.5
July–October	26.0	26.5	26.4	26.7	26.2	26.5	26.4

Table 2
Grain yields and N uptake as a function of N source and rate during six growing seasons.

N treatments	2005	2006	2007	2008	2009	2010	Average
	Grain yield (kg ha^{-1})						
Control[a]	6104b[b]	5942c	5730c	4740d	4167c	4384d	5178d
90U	6517a	6647ab	6513ab	6781bc	6136b	6352bc	6491bc
180U	6627a	6915a	6736a	7140a	6611a	6896a	6821a
270U	6393ab	6770ab	6361b	6907abc	6534a	6818a	6631abc
90 M	6497a	6567b	6449ab	6683c	6074b	6241c	6419c
180 M	6630a	6890ab	6734a	7112a	6554a	6810a	6788ab
270 M	6524a	6815ab	6582ab	7042b	6458a	6675ab	6683abc
	N uptake by plant (kg N ha^{-1})						
Control	103f	92g	84g	72g	63f	58g	79f
90U	135e	135e	126e	126e	105e	101e	121e
180U	170c	178c	168c	178c	151c	143c	165c
270U	203a	218a	212a	224a	189a	183a	205a
90 M	133e	128f	121f	119f	101e	95f	116e
180 M	159d	167d	158d	165d	140d	134d	154d
270 M	190b	199b	195b	209b	179b	172b	191b

[a] Control: 0 N addition; 90U–270U: urea N treatments (90–270 kg N ha^{-1}, 60% urea + 20% urea + 20% urea); 90M–270 M: pig manure treatments (90–270 kg N ha^{-1}, 60% manure + 20% urea + 20% urea).
[b] Values in columns followed by the same letter are not significantly different at the 0.01 level using LSD test.

Table 3
Analysis of variance of rice grain yield, N uptake, GHG emissions and GWP in a 6-yr N rate and source experiment.

Source	Rice grain yield				N uptake by rice			Yield-scaled GWP[a]		
	df	Mean square	F	P < F	Mean square	F	P < F	Mean square	F	P < F
Intercept	1	4534	91,000	0.00	2,210,954	243,200	0.00	1,538,805	16,680	0.00
Year	5	1.38	27.60	0.00	3765	414	0.00	19,210	208	0.00
Nrate	2	1.11	22.35	0.00	56,220	6184	0.00	4038	43.78	0.00
Nsource	1	0.01	0.18	0.67	2706	298	0.00	79,638	863	0.00
Year * Nrate	10	0.07	1.42	0.19	155	17.00	0.00	55.93	0.61	0.80
Year * Nsource	5	0.02	0.35	0.88	12.97	1.43	0.22	99.23	1.08	0.38
Nrate * Nsource	2	0.04	0.73	0.49	191	21.01	0.00	6906	74.87	0.00
Year * Nrate * Nsource	10	0.01	0.14	1.00	6.54	0.72	0.70	68.37	0.74	0.68
Error	84	0.05			9.09			92.25		

Source	Seasonal CH_4 flux				Seasonal N_2O flux			Seasonal GWP		
	df	Mean square	F	P < F	Mean square	F	P < F	Mean square	F	P < F
Intercept	1	79,547	15,310	0.00	6,905,532	2098.00	0.00	61,370,000	18,620	0.00
Year	5	1278	246	0.00	12,920	3.93	0.00	849,961	258	0.00
Nrate	2	143	27.62	0.00	379,922	115.44	0.00	232,180	70.42	0.00
Nsource	1	5028	968	0.00	121,780	37.00	0.00	3,522,317	1068	0.00
Year * Nrate	10	4.55	0.88	0.56	1663	0.51	0.88	3428	1.04	0.42
Year * Nsource	5	4.51	0.87	0.51	5320	1.62	0.17	4113	1.25	0.29
Nrate * Nsource	2	548	105	0.00	535	0.16	0.85	348,907	106	0.00
Year * Nrate * Nsource	10	7.43	1.43	0.18	1441	0.44	0.92	3841	1.17	0.33
Error	84	5.19			3291			3297		

[a] Yield-scaled GWP is the seasonal GWP in CO_2 equivalents per unit of grain yield.

Table 4
Seasonal CH_4 and N_2O fluxes and calculated GWP in rice fields fertilized with urea and manure.

N treatments	2005	2006	2007	2008	2009	2010	Average
	CH_4 (kg ha^{-1})						
Control[a]	12.3c[b]	25.6e	22.1c	32.1b	14.6e	12.3c	19.8
90U	15.0c	33.2c	24.4bc	35.7b	19.6d	15.0c	23.8
180U	11.8c	28.1d	20.9c	31.7b	21.0d	11.8c	20.9
270U	11.7c	29.3d	19.3c	29.5b	19.2d	11.7c	20.1
90 M	21.9b	36.9b	27.9b	38.3b	26.9b	21.9b	29.0
180 M	26.5b	44.4a	35.7a	49.9a	34.0a	26.5a	36.2
270 M	32.8a	46.5a	39.3a	53.7a	38.6a	32.8a	40.6
	N_2O (g ha^{-1})						
Control	94c	146b	137c	177c	99d	95c	82
90U	143c	176b	154c	173c	164cd	139c	158
180U	183bc	266ab	240bc	323ab	238c	190bc	240
270U	332ab	333ab	367a	442a	327ab	332ab	356
90 M	236bc	187b	207bc	263bc	204c	225c	220
180 M	303ab	296ab	321ab	353ab	249bc	298b	303
270 M	454a	410a	426a	440a	398a	462a	432
	GWP (kg $CO_{2\ eq.}$ ha^{-1})						
Control	335d	683e	594c	855b	393e	336d	533
90U	417d	882d	656bc	944b	540d	416d	643
180U	349d	782d	593c	890b	597d	351d	594
270U	393d	833cd	591c	868b	578d	393d	609
90 M	617c	979b	760b	1035b	733c	614c	790
180 M	753b	1197a	988a	1353a	923b	752b	994
270 M	955a	1286a	1110a	1475a	1083a	958a	1145

[a] Control: 0 N addition; 90U–270U: urea N treatments (90–270 kg N ha^{-1}, 60% urea + 20% urea + 20% urea); 90M–270M: pig manure treatments (90–270 kg N ha^{-1}, 60% manure + 20% urea + 20% urea).
[b] Values in columns followed by the same letter are not significantly different at the 0.01 level using LSD test.

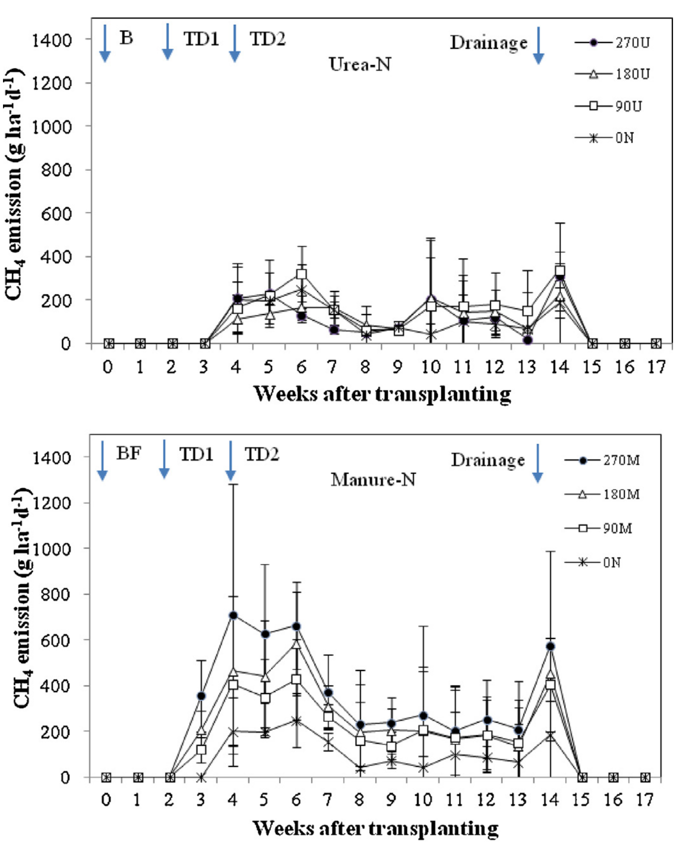

Fig. 2. CH_4 emissions as affected by nitrogen rate from rice fields which received urea or pig manure N fertilizer. Data is presented for 2005 only. BF: basal fertilizer; TD1: the first topdressing; TD2: the second topdressing. Error bars are the standard deviation of the three replicates.

et al., 1996) and it is thought to be the release of entrapped CH_4 gas as soil dries (van der Gon et al., 1996). These spikes highlight the importance of sampling following drainage events to fully account for all seasonal CH_4 emissions.

Nitrous oxide emissions were also followed a similar pattern in each of the six years of the study (Fig. 3). The highest N_2O emission rates were observed at the very beginning of the season and emissions gradually declined to zero by 5 WAT. The N_2O emissions during this early portion of the growing season were likely the result of basal and all topdress N fertilizer applications. Others have also reported that when N fertilizer is broadcasted on a flooded rice field it leads to small but significant N_2O emissions (Chen et al., 1997; Ghosh et al., 2003; Pathak et al., 2002). In contrast, when all of the N fertilizer is applied as a basal application, N_2O emissions are typically low after the field has been flooded (Qin et al., 2010; Wang et al., 2011). Topdressed fertilizers are often recommended to improve N use efficiency (Linquist et al., 2003), however, in cases where topdressed fertilizers do not improve N use efficiency, applying all of the N as a basal application may result in lower N_2O fluxes.

From 5 to 13 WAT, no N_2O emissions were observed (Fig. 3) and similar results have also been found reported by others (Qin et al., 2010; Zou et al., 2007). Low N_2O emissions when fields are maintained in a flooded condition are due to limited nitrification activity. Furthermore, a large proportion of N_2O produced from denitrification is likely reduced to N_2 caused due to low soil redox potentials (Firestone and Davidson, 1989; Zou et al., 2007). At the end of the growing season, shortly after draining the field for harvest, a large spike in N_2O emissions was observed. This spike in emissions may be due to the shift from anaerobic to aerobic conditions which enhance mineralization of available C and N, thereby favoring N_2O emissions (Zou et al., 2007). Similar N_2O emission patterns from continuously flooded fields have also been reported (Qin et al., 2010; Wang et al., 2011). Again, as with CH_4, a full accounting of all N_2O emissions requires sampling throughout the drain periods at frequent enough intervals to capture the fluxes.

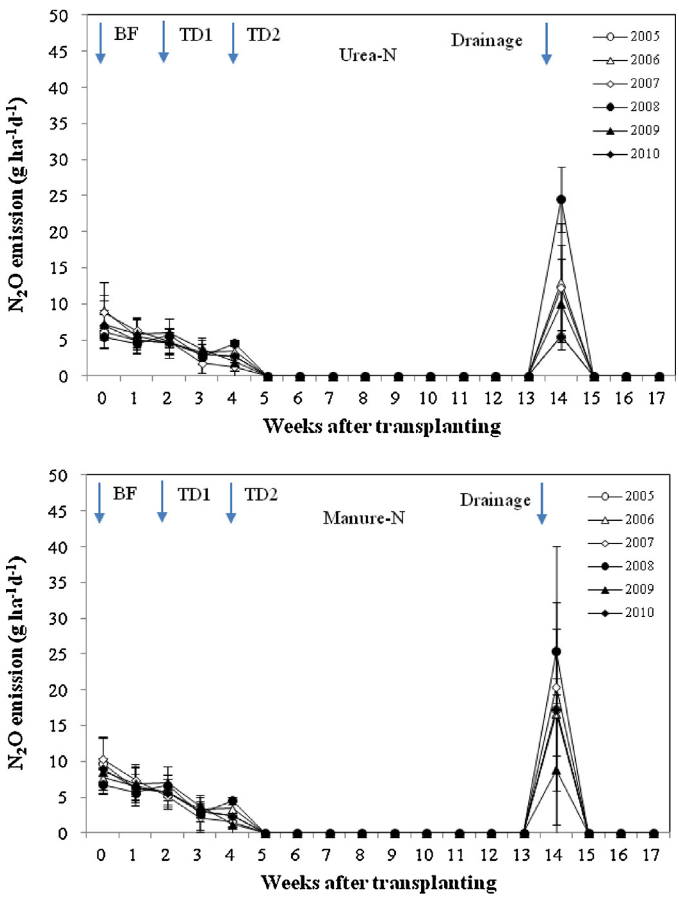

Fig. 3. Temporal dynamics of N_2O emitted from rice fields which received 180 kg N ha^{-1} of urea or pig manure N fertilizers. BF: basal fertilizer; TD1: the first topdressing; TD2: the second topdressing. Error bars are the standard deviation of the three replicates.

Considering both GHG, there was roughly a two-fold difference in GWP between the years when the highest values were found (2006 and 2008) and the years when the lowest values were found (2005 and 2010) (Table 4). While these differences can partly be explained by differences in average seasonal temperatures (2006 and 2008 were the warmest and 2005 was the coolest) it cannot explain all the variations in GWP as 2010 had similar average seasonal temperatures to 2006 (Table 1). The high GWP in 2006 and 2008 is the result of high CH_4 and N_2O emissions and vice versa for the two low GWP years. Such differences in emissions between seasons with similar management practices is not an uncommon phenomenon and remains often difficult to explain (Wassmann et al., 2000).

3.4. N source and rate effects on CH_4, N_2O fluxes and GWP

There was a significant N rate by N source interaction for seasonal CH_4 emissions (Table 3). When manure was the primary N source, CH_4 emissions increased with increasing N rate (Table 4, Fig. 2) due to higher C inputs with increasing N rate. In contrast, when urea was the N source, CH_4 emissions increased for the 90U N rate and then decreased with the highest N rate (270U) having similar CH_4 emissions to the 0N control (Table 4). Such a response to urea-N fertilizer is consistent with the findings of a meta-analysis by Linquist et al. (2012a) where CH_4 emissions where highest in the lower to moderate N rates but declined at the higher N rates. They hypothesized that this was due to the combined effects of N fertilization on CH_4 production, oxidation

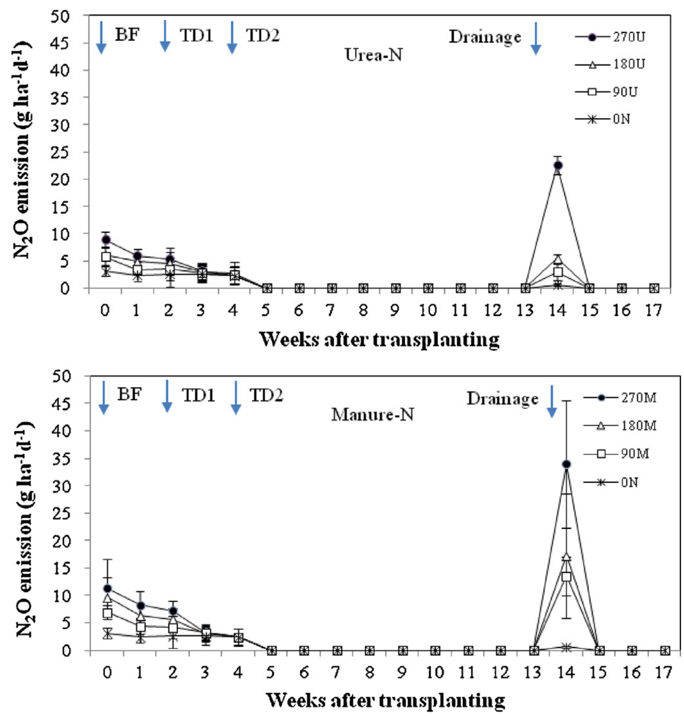

Fig. 4. N_2O emissions as affected by N rate from rice fields which received urea or pig manure N fertilizer. Data is presented for 2005 only. BF: basal fertilizer; TD1: the first topdressing; TD2: the second topdressing. Error bars are the standard deviation of the three replicates.

and transport. Nitrogen, at low to optimal N rates increases plant growth and these plants provide more C substrate for methanogenesis as roots and root exudates serve as a major carbon source for CH_4 production (Lu et al., 2000). As most CH_4 is emitted through the plant, higher tiller numbers (due to an increase in N rate) provide a larger pathway for CH_4 to be transported to the atmosphere. In contrast, at high N rates there is more NH_4^+ in the soil solution which can stimulate CH_4 oxidation (Bodelier and Laanbroek, 2004) and lead to a reduction in the amount of CH_4 emitted. Others have also reported that across a range of N inputs CH_4 emissions declined with increasing fertilizer N rates (Sass et al., 2002; Yao et al., 2012).

Seasonal N_2O emissions varied significantly between years with the annual magnitude of emissions determined by N rate and source (Table 3). With both fertilizer sources, N_2O emissions increased with increasing N rate (Table 4). On average and across years, N_2O emissions increased from 125 g N_2O ha^{-1} in the control to 356 g N_2O ha^{-1} in the 270U urea treatment and 432 g N_2O ha^{-1} in the 270 M manure treatment. On average, the use of manure as an N source increased seasonal N_2O emissions by 29% compared to the urea treatments (Table 4). Higher emissions from the manure treatments were observed both at the beginning of the growing season and in the post-drain spike at the end of the season (Figs. 3 and 4). This finding is in agreement with Shang et al. (2011) who observed the highest N_2O emissions when combined inorganic/organic fertilizers were applied. In contrast, when organic and inorganic N sources were compared in a meta-analysis no difference in N_2O emissions between these sources was found (Linquist et al., 2012a).

The relationship between N_2O and N rate was not linear but increased with increasing N rate for both N sources. The average N-fertilizer induced emission factor for urea and manure was 0.10% and 0.13%, respectively, which is lower than the average value of 0.22% reported for continuously flooded rice systems (Akiyama et al., 2005).

When the GWP of the two gases is considered the overall pattern is similar to CH_4 because CH_4 emissions represented 85–92% of GWP; similar to what has been reported in other flooded rice systems (Linquist et al., 2012b). Across N rates, the combined application of manure increased GWP by almost 60% relative to the urea treatments. The highest GWP (averaged 1145 kg $CO_{2\ eq.}$ ha^{-1}) was observed with the highest rate of manure N which had the highest CH_4 (40.6 kg ha^{-1}) and N_2O emissions (432 g ha^{-1}) (Table 4).

The GWP values reported here do not reflect the effects of manure relative to urea on net soil C sequestration or soil respiration. As others have reported that manure increases soil C content (Chakraborty et al., 2011; Tirol-Padre et al., 2007) and soil respiration (Chakraborty et al., 2011), a full assessment of GWP would need to be carried out which was outside the scope of this study.

3.5. Yield-scaled GWP

Our objectives were to identify N management practices for rice that result in the lowest yield-scaled GWP in order to addresses both food security and environmental concerns. There was no interaction between year and N rate or year and source for either N_2O, CH_4, GWP or yield-scaled GWP (Table 3) so averaged results across years are discussed. When only N_2O is considered, the yield-scaled N_2O response for urea was similar to what has been observed for other upland crops (van Groenigen et al., 2010; Venterea et al., 2011). The lowest yield-scaled values were at 90 kg N ha^{-1} which was close to the optimal N rate in this study (optimal N rates were between 90 and 180 kg N ha^{-1}; Fig. 5). At N rates above 90 kg N ha^{-1} there was a non-linear increase in N_2O emissions. With the use of manure instead of urea, yield-scaled N_2O emissions increased with increasing manure inputs and were higher than urea at the same N rate. Considering only yield-scaled CH_4, we observed a decline with increasing urea-N rates with the lowest level (76 kg $CO_{2\ eq.}$ Mg^{-1} grain) being at 180 and 270 kg N ha^{-1} (Fig. 5). In contrast, when manure was used as the primary N source yield-scaled emissions increased from 97 to 151 kg $CO_{2\ eq.}$ Mg^{-1} grain, despite an increase in yields across these N rates.

When both N_2O and CH_4 are considered, the effect was generally similar to what was observed for CH_4 only, because CH_4 was the main GHG emitted from this system (Table 4). When urea was used as the sole N source the yield-scaled GWP decreased with increasing N rate and was the lowest (87 kg $CO_{2\ eq.}$ Mg^{-1} grain) at 180 kg N ha^{-1} (Fig. 5). With the manure, despite increasing yield with higher manure rates, yield-scaled GWP increased due to the large increase in CH_4 emissions (and N_2O) from manure applications.

These results indicate that urea-N led to the lowest yield-scaled GWP. In this case, it was lowest at N rates required to achieve maximum yields. However, in flooded rice systems the lowest yield-scaled GWP does not necessarily occur at optimal N rates (Yao et al., 2012; Linquist et al., 2012a) in contrast to upland crops, where N_2O rather than CH_4 is the major GHG (van Groenigen et al., 2010). The primary reason for this difference between flooded and upland crops in GHG is that in flooded systems excess NH_4^+ can promote the oxidation of CH_4 (Bodelier and Laanbroek, 2004). Although an overuse of urea does not have a large effect on yield-scaled GHG emissions, overuse should be avoided as it can lead to other forms of pollutions such as nitrate leaching (Goulding, 2000). The use of manure significantly increased both area (GWP ha^{-1}) (Table 4) and yield-scaled GWP. At 180 kg N ha^{-1}, yield scaled GWP in the manure treatment was 146 kg $CO_{2\ eq.}$ Mg^{-1} grain and 68% higher than when only urea was used. As stated earlier the GWP here only accounts for N_2O and CH_4 and does not account for changes in soil C or respiration that would likely occur with the use of manure (Chakraborty et al., 2011; Tirol-Padre et al., 2007). However, based on these findings, when manure is used, yield-scaled GWP increased and therefore over application of manure needs to be avoided.

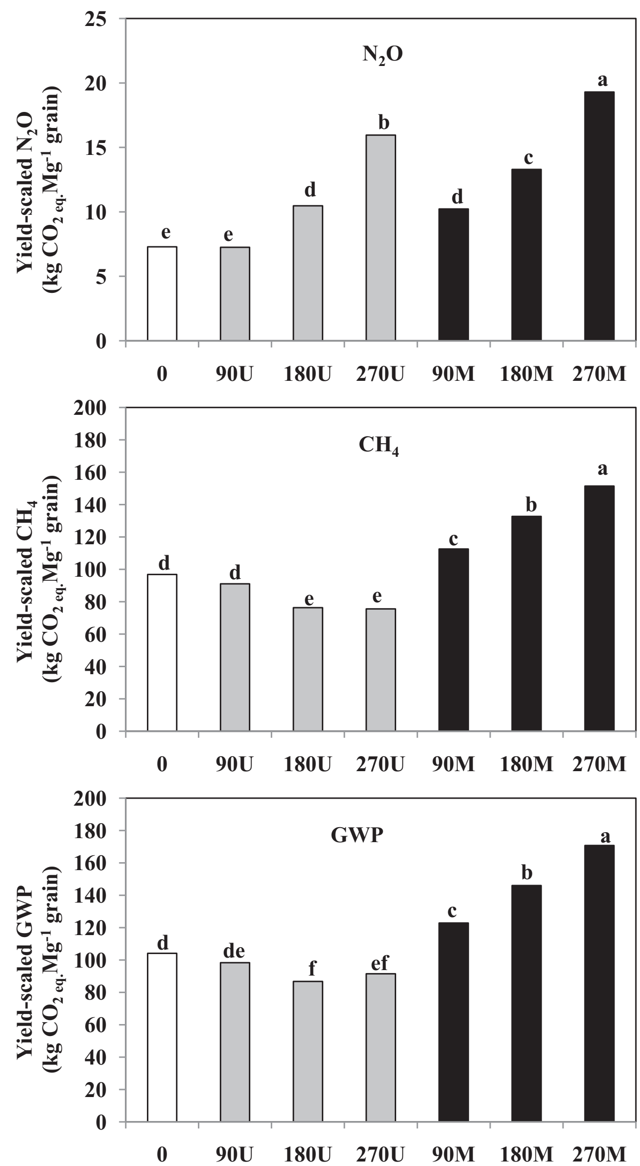

Fig. 5. Relationship between yield-scaled N_2O, CH_4 and GWP and N application rates and source in rice fields receiving urea and manure. Values above each bar with the same letter are not significantly different at the 0.01 level using LSD test.

4. Conclusions

This is the first study that we are aware of that has evaluated organic and inorganic N sources, across a range of inputs to adequately access optimal N rates, on yield-scaled GWP. We hypothesized that the lowest yield-scaled GWP would occur at optimal N rates required for maximum yields, independent of N source applied. This hypothesis was partially confirmed when urea was used as an N source but not for manure. Based on these findings, manure inputs should not be applied in excess of that required for optimal yields as this increases GHG emissions rapidly. Furthermore, urea N resulted in lower yield-scaled GWP across all N rates.

Although manure resulted in higher GWP, the effect of manure on net soil C sequestration and respiration was not taken into account, an important area for further research.

Acknowledgements

The authors are grateful for funding from the National Natural Science Foundation of China (21077088; 41271314; 0901142), and National Key Basic Research Project of China (2002CB410807).

References

Akiyama, H., Yagi, K., Yan, X.Y., 2005. Direct N_2O emissions from rice paddy fields: summary of available data. Global Biogeochem. Cycles 19, GB1005.
Bi, L.D., Zhang, B., Liu, G.R., Li, Z.Z., Liu, Y.R., Ye, C., Yu, X.C., Lai, T., Zhang, J.G., Yin, J.M., Liang, Y., 2009. Long-term effects of organic amendments on the rice yields for double rice cropping systems in subtropical China. Agric. Ecosyst. Environ. 129, 534–541.
Bijay-Singh, B., Singh, Y., Maskina, M.S., Meelu, O.P., 1997. The value of poultry litter for wetland rice grown in rotation with wheat. Nutr. Cycl. Agroecosyst. 47, 243–250.
Bodelier, P.L.E., Laanbroek, H.J., 2004. Nitrogen as a regulatory factor of methane oxidation in soils and sediments. FEMS Microbiol. Ecol. 47, 265–277.
Brahmanand, P.S., Ghosh, B.C., Sahoo, N., 2009. Effect of organic and inorganic nitrogen on productivity of rice in rice-fish farming system. Arch. Agron. Soil Sci. 55, 663–670.
Cao, Z.H., 2008. Study of prehistoric irrigated paddy and ancient paddy soils in China (in Chinese). Acta Pedol. Sin. 45, 784–791.
Chakraborty, A., Chakrabarti, K., Chakraborty, A., Ghosh, S., 2011. Effect of long-term fertilizers and manure application on microbial biomass and microbial activity of a tropical agricultural soil. Biol. Fertil. Soils 47, 227–233.
Chen, G.X., Huang, G.H., Huang, B., Yu, K.W., Wu, J., Xu, H., 1997. Nitrous oxide and methane emissions from soil-plant systems. Nutr. Cycl. Agroecosyst. 49, 41–45.
de Ponti, T., van Ittersum, M.K., Rijk, B., 2012. The crop yield gap between organic and conventional agriculture. Agric. Syst. 108, 1–9.
Duan, Y., Xu, M., He, X., Li, S., Sun, X., 2011. Long-term pig manure application reduces the requirement of chemical phosphorus and potassium in two rice-wheat sites in subtropical China. Soil Use Manage 27, 427–436.
FAO (Food and Agriculture Organization of the United Nations), 2012. Rice Market Monitor. XV (1):3. Available from: http://www.fao.org/economic/est/publications/rice-publica tions/ricemarket-monitor-rmm/en/
Firestone, M.K., Davidson, E.A., 1989. Microbial basis of NO and N_2O production and consumption in soils. In: Andreae, M.O., Schimel, D.S. (Eds.), Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere. Wiley, New York, pp. 7–21.
Godfray, H.C.J., Pretty, J., Thomas, S.M., Warham, E.J., Beddington, J.R., 2011. Linking policy on climate and food. Science 331, 1013–1014.
Golden, B.R., Slaton, N.A., Norman, R.J., Gbur, E.E., Brye, K.R., DeLong, R.E., 2006. Recovery of nitrogen in fresh and pelletized poultry litter by rice. Soil Sci. Soc. Am. J. 70, 1359–1369.
Ghosh, S., Majumdar, D., Jain, M.C., 2003. Methane and nitrous oxide emissions from an irrigated rice of North India. Chemosphere 51, 181–195.
Goulding, K., 2000. Nitrate leaching from arable and horticultural land. Soil Use Manage. 16, 145–151.
Huang, Y., Tang, Y.H., 2010. An estimate of greenhouse gas (N_2O and CO_2) mitigation potential under various scenarios of nitrogen use efficiency in Chinese croplands. Global Change Biol. 16, 2958–2970.
IPCC (Intergovernmental Panel on Climate Change), 2001. In: Houghton, J.T., Ding, Y., Griggs, D.J., Noguer, M., van der Linden, P.J., Dai, X., Maskell, K., Johnson, C.A. (Eds.), Radiative Forcing of Climate Change. Climate Change 2001: The Scientific Basis. Contribution of Working Group I to The Third Assessment Report of The Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK/New York, NY, USA.
IPCC (Intergovernmental Panel on Climate Change), 2007. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.), Climate Change 2007: The Physical Science Basis, Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom/New York, NY, USA.
Ju, X.T., Xing, G.X., Chen, X.P., Zhang, S.L., Zhang, L.J., Liu, X.J., Cui, Z.L., Yin, B., Christie, P., Zhu, Z.L., Zhang, F.S., 2009. Reducing environmental risk by improving N management in intensive Chinese agricultural systems. Proc. Natl. Acad. Sci. U.S.A. 106, 3041–3046.
Lin, D.X., Fan, X.H., Hu, F., Zhao, H.T., Luo, J.F., 2007. Ammonia volatilization and nitrogen utilization efficiency in response to urea application in rice fields of the Taihu Lake region, China. Pedosphere 17, 639–645.
Linquist, B.A., Sengxua, P., 2003. Efficient and flexible management of nitrogen for rainfed lowland rice. Nutr. Cycl. Agroecosyst. 67, 107–115.
Linquist, B.A., Adviento-Borbe, M.A., Pittelkow, C., van Kessel, C., van Groenigen, K.J., 2012a. Fertilizer management practices and greenhouse gas emissions from rice systems: a quantitative review and analysis. Field Crops Res. 135, 10–21.
Linquist, B.A., van Groenigen, K.J., Adviento-Borbe, M.A., Pittelkow, C., van Kessel, C., 2012b. An agronomic assessment of greenhouse gas emissions from major cereal crops. Global Change Biol. 18, 194–209.
Lu, Y.H., Wassmann, R., Neue, H.U., Huang, C.Y., 2000. Atmospheric pollutants and trace gasses – Dissolved organic carbon and methane emissions from a rice paddy fertilized with ammonium and nitrate. J. Environ. Qual. 29, 1733–1740.
Mosier, A.R., Halvorson, A.D., Reule, C.A., Liu, X.J.J., 2006. Net global warming potential and greenhouse gas intensity in irrigated cropping systems in northeastern Colorado. J. Environ. Qual. 35, 1584–1598.
Mueller, N.D., Gerber, J.S., Johnston, M., Ray, D.K., Ramankutty, N., Foley, J.A., 2012. Closing yield gaps through nutrient and water management. Nature 490, 254–257.
Ni, W.Z., Zhu, Z.L., 2004. Evidence of N_2O emission and gaseous nitrogen losses through nitrification – denitrification induced by rice plants (*Oryza sativa* L.). Biol. Fertil. Soils 40, 211–214.
Pan, G.X., Zhou, P., Li, Z.P., Smith, P., Li, L.Q., Qiu, D.S., Zhang, X.H., Xu, X.B., Shen, S.Y., Chen, X.M., 2009. Combined inorganic/organic fertilization enhances N efficiency and increases rice productivity through organic carbon accumulation in a rice paddy from the Tai Lake region. China Agric. Ecosyst. Environ. 131, 274–280.
Pan, G.X., Zhu, Q.H., Zhang, Y., Li, L.Q., Huang, X.M., Wang, J.F., Jing, X.Y., 2003. Minimum application rates for high-yielding rice production system in the Taihu Lake region as a field measure for controlling N and P agricultural loading (In Chinese). Huanjing Kexue 24, 96–100.
Pathak, H., Bhatia, A., Prasad, S., Singh, S., Kumar, S., Jain, M.C., Kumar, U., 2002. Emission of nitrous oxide from rice-wheat systems of Indo-Gangetic plains of India. Environ. Monit. Assess. 77, 163–178.
Qin, Y.M., Liu, S.W., Guo, Y.Q., Liu, Q.H., Zou, J.W., 2010. Methane and nitrous oxide emissions from organic and conventional rice cropping systems in Southeast China. Biol. Fertil. Soil 46, 825–834.
Statistical Analysis Systems (SAS) 2003. SAS System Version 8.0 manual, Cary, NC, USA.
Sass, R.L., Andrews, J.A., Ding, A.J., Fisher, F.M., 2002. Spatial and temporal variability in methane emissions from rice paddies: implications for assessing regional methane budgets. Nutr. Cycl. Agroecosyst. 64, 3–7.
Shang, Q.Y., Yang, X.X., Gao, C.M., Wu, P.P., Liu, J.J., Xu, Y.C., Shen, Q.R., Zou, J.W., Guo, S.W., 2011. Net annual global warming potential and greenhouse gas intensity in Chinese double rice-cropping systems: a 3-year field measurement in long-term fertilizer experiments. Global Change Biol. 17, 2196–2210.
Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., Mara, F.O', Rice, C., Scholes, B., Sirotenko, O., 2007. Agriculture. In: Metz, B., Davidson, O.R., Bosch, P.R., Dave, R., Meyer, L.A. (Eds.), Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK/New York, NY, USA, pp. 497–540.
Takahashi, S., Uenosono, S., Nagatomo, M., 2004. Rice uptake of nitrogen from aerobically and anaerobically composted poultry manure. J. Plant Nutr. 27, 731–741.
Tirol-Padre, A., Ladha, J.K., Regmi, A.P., Bhandari, A.L., Inubushi, K., 2007. Organic amendments affect soil parameters in two long-term rice-wheat experiments. Soil Sci. Soc. Am. J. 71, 442–452.
van der Gon, H.A.C.D., van Breemen, N., Neue, H.U., Lantin, R.S., Aduna, J.B., Alberto, M.C.K., 1996. Release of entrapped methane from wetland rice fields upon soil drying. Global Biogeochem. Cycles 10, 1–7.
van Groenigen, J.W., Velthof, G.L., Oenema, O., van Groenigen, K.J., van Kessel, C., 2010. Towards an agronomic assessment of N_2O emissions: a case study for arable crops. Eur. J. Soil Sci. 61, 903–913.
Venterea, R.T., Maharjan, B., Dolan, M.S., 2011. Fertilizer source and tillage effects on yield-scaled nitrous oxide emissions in a corn cropping system. J. Environ. Qual. 40, 1521–1531.
Wang, G.H., Zhang, Q.C., Huang, C.Y., 2003. SSNM-A new approach to increasing fertilizer N use efficiency and reducing N loss from rice fields (in Chinese). J. Zhejiang Univ. (Agr. & Life Sci.) 29, 67–70.
Wang, J.Y., Jia, J.X., Xiong, Z.Q., Khalil, M.A.K., Xing, G.X., 2011. Water regime-nitrogen fertilizer-straw incorporation interaction: field study on nitrous oxide emissions from a rice agroecosystem in Nanjing, China. Agric. Ecosyst. Environ. 141, 437–446.
Wassmann, R., Neue, H.U., Alberto, M.C.R., Lantin, R.S., Bueno, C., Llenaresas, D., Arah, J.R.M., Papen, H., Seiler, W., Rennenberg, H., 1996. Fluxes and pools of methane in wetland rice soils with varying organic inputs. Environ. Monit. Assess. 42, 163–173.
Wassmann, R., Neue, H.U., Lantin, R.S., Buendia, L.V., Rennenberg, H., 2000. Characterization of methane emissions from rice fields in Asia. I. Comparison among field sites in five countries. Nutr. Cycl. Agroecosyst. 58, 1–12.
Wild, P.L., van Kessel, C., Lundberg, J., Linquist, B.A., 2011. Nitrogen availability from poultry litter and pelletized organic amendments for organic rice production. Agron. J. 103, 1284–1291.
Yagi, K., Tsuruta, H., Kanda, K., Minami, K., 1996. Effect of water management on methane emission from a Japanese rice paddy field: automated methane monitoring. Global Biogeochem. Cycles 10, 255–267.
Yan, X., Cai, Z., Ohara, T., Akimoto, H., 2003. Methane emission from rice fields in mainland China: amount and seasonal and spatial distribution. J. Geophys. Res. 108 (D16), ACH10–1–12 10.1029/2002J.

Yang, X., Shang, Q., Wu, P., Liu, J., Shen, Q., Guo, S., Xiong, Z., 2010. Methane emissions from double rice agriculture under long-term fertilizing systems in Hunan. China. Agric. Ecosyst. Environ. 137, 308–316.

Yao, Z.S., Zheng, X.H., Dong, H.B., Wang, R., Mei, B.L., Zhu, J.G., 2012. A 3-year record of N_2O and CH_4 emissions from a sandy loam paddy during rice seasons as affected by different nitrogen application rates. Agric. Ecosyst. Environ. 152, 1–9.

Zheng, X., Han, S., Huang, Y., Wang, Y., Wang, M., 2004. Re-quantifying the emission factors based on field measurements and estimating the direct N_2O emission from Chinese croplands. Global Biogeochem. Cycles 18, GB2018, doi: 10.1029/2003G.

Zou, J.W., Huang, Y., Zheng, X.H., Wang, Y.S., 2007. Quantifying direct N_2O emissions in paddy fields during rice growing season in mainland China: dependence on water regime. Atmos. Environ. 41, 8030–8042.