Integrating agronomic practices to reduce greenhouse gas emissions while increasing the economic return in a rice-based cropping system

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Abstract:

The effective mitigation of greenhouse gas (GHG) emissions from crop cultivation requires an overall consideration, due to the trade-off relationship between GHGs and the mitigation measures sometimes undermining grain yields and economic returns. To explore the maximum potential of GHG mitigation without sacrificing economic returns, we investigated the GHG emissions, crop yields and economic returns over two years in three rice-cropping systems (rice–wheat, RW; rice–fava bean, RB; and rice–fallow, RF). Different N fertilization levels and straw incorporation methods were employed in each of the rice-cropping systems at the study site, in the Taihu Lake region (TLR) of China. Bean straw produced in the RB system was fermented aerobically before incorporation, while straws produced in the RW and RF systems were directly incorporated into the soils. Relative to the traditional RW system, methane emissions during the rice seasons were significantly reduced by 29–44% in the RB system; annual N inputs were reduced by 43%; annual nitrous oxide emissions for N fertilization treatments were decreased by 56–69%; and the average rice yield was increased by 5.2%. As a result, the GHG intensity (GHGI) was significantly reduced by 11–41%, and the annual net economic benefit (NEB) was increased by 22–94%. Compared with the RW system, the GHGI in the RF system was increased by 8–30% and the NEB was decreased by 3–33%. Considering the current rice/wheat production practices employed in the TLR, the GHGI could be reduced by 26% while the NEB could be improved by 23%, with an annual reduction in the N application rate (currently 525 kg N ha⁻¹) by 20% and the conversion of the RW systems into RB systems that use straw fermentation. The results of this study suggest that high economic return and GHG mitigation could be simultaneously

achieved by the integrated adoption of reasonable reduction in N application rates, the use of appropriate rice-cropping systems and by employing eco-friendly straw management.

Key words: rice-cropping system; straw fermentation; greenhouse gas intensity; N reduction; net economic benefit

1 Introduction

While rice is a staple food worldwide, rice paddies are important anthropogenic sources of greenhouse gas (GHG) emissions (Cai, 2012). It was estimated that about 7.4 Tg methane (CH_4) and 32 Gg N_2O -N nitrous oxide (N_2O) were emitted annually from rice cultivation in China (the largest rice producer in the world), contributing up to 22% of the total GHG emissions from cropland in the country (Yan et al., 2009; Zou et al., 2009). Increases in the soil organic carbon (SOC) of the topsoil in the rice paddies from the 1980s to 2000s only offset the annual emissions of CH_4 and N_2O by 5% (Yan et al., 2011); this highlights the need to reduce CH_4 and N_2O emissions and increase the SOC stocks of paddy soils. The emissions of CH_4 and N_2O from rice paddies, and the SOC stock, are strongly affected by agronomic practices, such as straw managements, N fertilization and cropping rotation practices (Zou et al., 2005; Khosa et al., 2010; Shang et al., 2011).

The Taihu Lake region (TLR), located in the center of the Yangtze River Delta, in southern China, is one of the most important rice production areas in China (Zhao et al., 2012). The dominant cropping system in the TLR comprises a summer rice (*Oryza sativa* L.) and winter wheat (*Triticum aestivum* L.) rotation. Rice and wheat straw are directly incorporated into the soils when the grains are gathered, due to a high degree of mechanization in the TLR (Xia et al., 2014). The direct incorporation of wheat straw before

rice transplanting triggers substantial CH₄ emissions, by providing methanogens with abundant carbon sources (Ma et al., 2009; Shang et al., 2011). Straw incorporation is credited with promoting the accumulation of SOC stocks (Shang et al., 2011; Yan et al., 2011). In considering the balance of these, Xia et al. (2014) found that the global warming potential (GWP) of straw-induced CH₄ emission was 3–4 times that of straw-induced SOC sequestration rate (SOCSR), when wheat straw was directly incorporated before rice transplanting in the TLR. Their study highlights the need to develop an appropriate straw incorporation method.

Aside from substantial CH₄ emissions from the rice paddies, in the TLR, there are also high N₂O emissions that result from the overuse of N fertilizers (240–300 kg N ha⁻¹ for rice and 200–250 kg N ha⁻¹ for wheat) (Zhao et al., 2009). Low N recovery efficiencies (30–35%) demonstrate that the majority of N fertilizer applied is released as reactive N (Nr), which causes a cascade of damaging effects on environmental and human health (Galloway et al., 2008; Ju et al., 2009; Sutton et al., 2011). In addition to the direct CH₄ and N₂O emissions from the rice paddies, the indirect emissions of GHG that result from the production of various agricultural materials cannot be ignored, due to their high application rates in the TLR.

Efforts to reduce the GHG emissions from rice–wheat systems in the TLR have been effective, including the adoption of eco-friendly straw managements (e.g., applying straw during the off-rice season or fermenting straw aerobically before its incorporation) to reduce CH₄ emissions (Yan et al., 2009; Khosa et al., 2010); decreasing the N application rates, to reduce N₂O emissions (Zou et al., 2005; Ma et al., 2013); and cultivating rice with legume

plants, to reduce the annual N fertilizer inputs, N₂O emissions and the GHG emissions associated with N fertilizer production (Zhao et al., 2015b). However, while each of these practices reduces one type of GHG emissions, they may produce favorable conditions for the emissions of other types of GHG; this is called the trade-off relationship between the GHGs (Huang et al., 2013). Moreover, the adoption of individual mitigation practice can sometimes reduce the crop yields and economic returns (Xia et al., 2016). Few studies have explored the combined effects of multiple mitigation measures on GHG emissions and economic returns, in rice-based cropping systems.

The main objective of this study was to evaluate the responses of GHG emissions, crop yields and economic returns in three rice-based cropping systems (rice–wheat, RW; rice–fava bean, RB; and rice–fallow, RF), to different N fertilization levels and straw incorporation methods. We considered that the maximum GHG mitigation levels, without reducing the economic returns, would result from the integrated adoption of appropriate reductions in N application rates, the adoption of a suitable rice-cropping system and the use of eco-friendly straw management practices. In the present study, the CH₄ and N₂O emissions, SOC changes and crop yields were simultaneously measured for two years (2013–2015) in the three rice-based cropping systems to achieve the above mentioned goal.

2 Materials and methods

2.1 Study site

The continuous experiment took place at the Changshu Agroecological Experimental Station (31°32'N, 120°41'E) of the Chinese Academy of Sciences in Jiangsu province, China. The station is located in the TLR, with the dominant cropping system of annual summer rice

and winter wheat rotation. The soil in the experimental field had developed from lacustrine sediments, and was classified as Anthrosol. The basic properties of the topsoil (0–20 cm) were as follows: bulk density, 1.2 g cm⁻³; pH (H₂O), 7.7; total N content, 2.0 g kg⁻¹; organic carbon content, 20.1 g kg⁻¹; available P content, 0.012 g kg⁻¹; and available K content, 0.126 g kg⁻¹.

2.2 Experimental design

The RB and RF system were compared with the traditional RW system due to their increasing implementation in the TLR. This is because social economic factors have affected the implementation of traditional RW cropping systems in the TLR. Some local farmers have started to substitute the winter wheat crop with fava bean, due to the lower yields and quality of wheat grains, and the higher market prices of fava beans (Zhao et al., 2015b). In addition, a large number of farmlands in the TLR now remain fallow during the winter season, because laborers leave the rural area for the city during that period. The winter crop area in Jiangsu province was reduced by 37% from 1999 to 2009 (Zhang et al., 2015).

The continuous field experiment, initiated in June 2012, was arranged in a randomized block design with three replicates of each treatment. The three rice-based cropping systems (RW, RB and RF rotation) were subdivided into different N fertilization treatments. Five treatments were applied to the RW system, RW0, RW120, RW180, RW240 and RW300, with 0/0, 120/90, 180/135, 240/180, and 300/225 kg N ha⁻¹ applied for the rice/wheat cultivation period, respectively (Table A.1). RW300 represents the local rice/wheat production pattern. The RB and RF cropping systems were each subdivided to four treatments (RB0, RB120, RB180 and RB240; and BF0, RF120, RF180 and RF240) which the N fertilization rates during rice cultivation were 0, 120, 180 and 240 kg N ha⁻¹; during the fava bean cultivation

(RB) and fallow (RF) seasons, no N fertilizer was applied.

N fertilizer (urea) was applied at three stages during the rice (wheat)–growing seasons: 40% was applied as basal fertilizer, 30% as tiller fertilizer (elongation and booting fertilizer), and 30% was applied as panicle fertilizer. The application times of the N fertilizer were shown in the Fig.3. P (calcium superphosphate) and K (potassium chloride) fertilizers were applied as basal fertilizers at rates of 30 kg P_2O_5 ha⁻¹ and 60 kg K_2O ha⁻¹, respectively, in each of the rice, wheat and bean seasons. The basal fertilizers, applied before crop transplanting or sowing, were incorporated into the soil by plowing; while top-dressings, were evenly broadcast over the soil surface.

Rice and wheat straw, as well as the weeds that grew during the fallow season of the RF system, were harvested and chopped *in situ* and plowed into the soils prior to the start of the next season. To reduce the content of labile carbon in the fava bean straw, which is easily used by methanogens, the straw was fermented aerobically in a metal fermenter before its incorporation. For this, fava bean straw was chopped and then mixed evenly with fermentative bacteria and sucrose before being put into the metal fermenter. The fermentative bacteria were screened by Peng et al. (2010), who demonstrated that it degraded the straw quickly under suitable conditions. The straw was continuously stirred to minimize the occurrence of anaerobic fermentation, once the metal fermenter was fully activated. A ventilating fan installed in the fermenter was activated every two hours during the fermentation, to release the gases produced and bring some fresh air into the fermenter simultaneously. After fermentation, the straw was applied to the soil evenly before rice transplanting.

2.3 Agricultural management practices

The field observation was started in June 2013, at the beginning of the rice season, and continued for two consecutive RW, RB and RF cycles. In 2013–2014, rice was transplanted on June 16–21 at a density of 2.5×10^4 plant ha^{-1} and harvested on October 25–November 3. Wheat was directly sown on November 8–13 at a rate of 150 kg ha^{-1} and harvested on May 25–28. For the RB treatments, fava bean seeds were placed in 2 cm depth planting holes at a rate of 260 kg ha^{-1} on the same day that wheat sown, but the fava beans were harvested a fortnight before harvesting the wheat. During the rice-growing seasons, the field was kept flooded with a water level of 3–5 cm for one month after transplanting; then, it was drained for one week and subsequently intermittently irrigated, until the rice was ready for harvest. During the winter seasons, the field site received water by precipitation only.

2.4 GHG measurements and changes of SOC estimates

The CH_4 and N_2O fluxes were continuously measured from June 2013 to May 2015, covering the cropping periods of the four consecutive seasons, using a static chamber and gas chromatography technique (Xia et al., 2014). The CH_4 and N_2O fluxes were measured once or twice per week during the rice seasons and once per week during the winter seasons. Gas samples were collected more frequently after N fertilizer application, precipitation events, and during the midseason drainage and subsequent re-flooding periods. A detailed description of the gas sampling and analyses employed can be found in Xia et al. (2014).

Change in SOC density (SOC_D, t ha^{-1}) was calculated using the following equation (Zhou et al., 2016):

$$\text{SOC}_D = \text{SOC} \times \rho \times H \times 10^{-1}, \quad (1)$$

where SOC refers to the soil organic carbon content (g kg^{-1}), ρ is soil bulk density (g cm^{-3}) and H is the sampling depth (20 cm).

The SOCSR ($\text{t ha}^{-1} \text{ yr}^{-1}$) was calculated based on the changes of SOCD, using the following equation (Xia et al., 2014):

$$\text{SOC SR} = (\text{SOCD}_{2015} - \text{SOCD}_{2012}) / T, \quad (2)$$

where SOCD_{2015} and SOCD_{2012} refer to the SOCD in 2015 and the initial year (g kg^{-1}), respectively, and T refers to the time period between the initial and final measurements (3 years).

2.5 Net GWP (NGWP), GHGI and net economic benefit (NEB) estimations

The NGWP ($\text{kg CO}_2\text{-equivalent (eq) ha}^{-1}$) for each plot was calculated from the perspective of a life-cycle assessment, using the following equation (Chen et al., 2014):

$$\text{NGWP} = \sum_{i=1}^m \text{AI}_{\text{ico}_2} + \text{CH}_4 \times 25 + \text{N}_2\text{O} \times 44/28 \times 298 - \text{SOC SR} \times 44/12 \times 1000, \quad (3)$$

where AI_{ico_2} denotes the total GHG emissions resulting from the production and transportation of the agricultural inputs, such as fertilizers, diesel, and electricity, among others. CH_4 ($\text{kg CH}_4 \text{ ha}^{-1}$), N_2O (kg N ha^{-1}) and SOCSR ($\text{t C ha}^{-1} \text{ yr}^{-1}$) represent the annual CH_4 emissions, N_2O emissions and the SOC sequestration rate for each plot, respectively. In addition, the GHG emissions associated with the fermentation process in the RB system were included in the calculation of NGWP (Table A.2).

The GHG intensity (GHGI; $\text{kg CO}_2\text{-eq kg}^{-1}$ grain) was calculated by dividing the NGWP by the crop grain yield (Shang et al., 2011; Ma et al., 2013):

$$\text{GHGI} = \text{NGWP} / \text{grain yield}, \quad (4)$$

where the grain yield denotes the annual grain yield of each rice-cropping system.

The net economic benefit (NEB; ¥ ha⁻¹) for each plot was calculated using the following equation:

$$NEB = \sum_{a=1}^b (Y_a \times P_a) - \sum_{m=1}^n (R_m \times P_m), \quad (5)$$

where Y_a (kg ha⁻¹) denotes the yield of crop 'a' (rice, wheat or fava bean); P_a (¥ kg⁻¹) denotes the market price of crop 'a'; R_m represents the application rate of certain agricultural materials (m), such as N fertilizer, diesel oil and electricity; and P_m (¥ kg⁻¹) denotes the market price of (m). The market prices of the crops and various agricultural materials are provided in the Appendix A (Table A.3). The economic inputs associated with the fermentation process in the RB system were included in the calculation of NEB (Table A.2). In addition, the economic and ecological benefit curves of rice production in the three cropping systems were implemented (Fig. B.1), using the method of Xia and Yan (2012).

2.6 Auxiliary data collection and analysis

The daily mean air temperature and precipitation during the experiment were collected by an automatic weather station at the experimental station (Fig.1). The total carbon contents of straw and soil samples were analyzed using a wet oxidation procedure with K₂Cr₂O₇ (Mandal et al., 2008). The direct N₂O emission factor (EF_d%) of the applied N fertilizer (kg N ha⁻¹) was calculated using the following equation (Ma et al., 2013):

$$EF_d\% = 100(EM - E0)/N \quad (6)$$

where, EM (kg N ha⁻¹) denotes the cumulative N₂O emissions from the particular treatment in which N was applied and E0 denotes the cumulative N₂O emissions from the corresponding treatment in which no N was added (i.e., the control).

2.7 Statistical analyses

All statistical analyses were performed with SPSS (version 19.0, USA). Differences in seasonal CH₄ and N₂O emissions, grain yields, SOCSR, GHGI, NGWP and NEB among treatments over 2013–2015 were tested by analysis of variance (ANOVA), during which the mean values were compared by least significant difference (LSD) at the 5% level. Figures were prepared, using Origin (version 9.0, USA).

3 Results and discussion

3.1 Effects of rice-cropping systems and N application rates on crop yield

Across two seasons, the average rice yield of the traditional RW system (8576 kg ha⁻¹) was 2.5% and 8.7% higher than that in the RB (8364 kg ha⁻¹) and RF (7890 kg ha⁻¹), respectively, when the N was applied at the rate of 240 kg N ha⁻¹; but this effect was not statistically significant (Table 1). The RB system significantly increased the rice yield compared with the other systems, when the N application rate was lower than 240 kg N ha⁻¹, except for the treatments that received the N rate of 180 kg N ha⁻¹. The average rice yield of the treatments in the RB system (7627 kg ha⁻¹) was 5.2–6.6% higher than that of the RW (7250 kg ha⁻¹) and RF (7152 kg ha⁻¹) systems. However, these differences were not statistically significant (Table A.4).

Overall, the RB system tended to enhance the rice yields in this study, which confirmed that cropping systems that use legume crops had the potential to produce higher grain yields than those without them (Adviento-Brobe et al., 2007; Gao et al., 2014). Similarly, Zhao et al. (2015b) reported that substituting winter legumes for wheat increased the subsequent rice

yields by 1–3% in the TLR. Including legume plants in cropping systems is proven to increase the soil C and N pools, likely due to the high quality root exudates of legumes that promote the microbial growth efficiency and aggregation (Drinkwater et al., 1998; Cong et al., 2014; McDaniel et al., 2014). Guong et al. (2012) reported that a rice–mungbean (RMB) system had a significantly distinct bacterial community structure, and higher bacterial diversity and richness, in the soils compared with a rice–maize system, which promoted higher rice yields in the RMB system. Moreover, the legume rhizobia can naturally colonize the interior of rice roots and then eventually develop a continuum of root-associated microorganisms; these have the potential to promote rice growth and productivity (Yanni et al., 1997).

The rice yield gradually improved as the N application rates increased from 0 to 240 kg N ha⁻¹ in all of the cropping systems (Table 1). In the RW system, further increases of the N application rate from 240 to 300 kg ha⁻¹ N (the traditional N application rate; not tested in the RB or RF systems) reduced the rice yield by 2.1%. Regarding the other cropping systems, the increases in rice yield with N application rate were not significant above 120 kg N ha⁻¹ (RF system) and 180 kg N ha⁻¹ (RB system). The wheat yield increased significantly with the N application rate, from 0 to 180 kg N ha⁻¹ (Table 1); but a further increase in the N rate (to 225 kg N ha⁻¹) did not significantly increase the yield. In addition, treatments in the RB system produced similar yield of fava bean, while no grain was produced in the fallow season of the RF system.

It is not surprising that rice yields failed to improve significantly as the N input rates increased above a threshold for each system. Previous studies have reported that grain yields

respond to increase in the application N rate according to either quadratic or linear plateau model, instead of a linear model (Xia and Yan, 2012; Cui et al., 2013). These studies provide sound scientific evidence for reducing N application rate, without sacrificing yield. Compared with the current N application rates generally employed in the TLR (RW300: 300/225 kg N ha⁻¹), a reduction of 20% for both rice and wheat seasons (RW240: 240/180 kg N ha⁻¹), saving 105 kg N ha⁻¹ each year, slightly increased the rice yield and did not significantly affected the wheat yield (Table 1). A further 180 kg N ha⁻¹ would be saved by changing from a RW system (RW240) into a RB system (RB240), without significantly affecting the rice yield. However, the long-term effects of reducing the N application rates and converting the cropping systems on the grain yields beyond two years need to be evaluated. This is because past applications of N fertilizer can be retained in soils, leaving a long-term legacy (Sebilo et al., 2013), which may have affected the cropping systems in this study. In addition, the continuous cultivation of fava beans in a field can increase the presence of soil diseases that may damage bean productivity over time (Zhao et al., 2015b).

3.2 Effects of rice-cropping systems on CH₄ emissions

The CH₄ flux patterns were similar for different N treatments in the same cropping system, albeit with some inter-annual variation. When the paddies were kept flooded, the CH₄ fluxes steadily increased over time, peaking 23–30 days after the rice transplanting, up to 17.32–33.33 mg CH₄ m⁻² h⁻¹, mainly depending on the cropping system (Fig.2). Thereafter, the fluxes declined to nearly zero, as the field was gradually drained. Then, the fluxes fluctuated slightly, but remained at negligible rates. Midseason aeration in the 2014 rice

season did not result in a considerable decrease in the CH₄ fluxes due to heavy precipitation. All of the treatments showed a negligible flux rate of CH₄ in the winter seasons, as a consequence of aeration cultivation. Across the two rice seasons, the average CH₄ emissions from each N treatment in the RB system were 29–42% and 6–33% lower than the corresponding N treatments in the RW and RF systems, respectively (Table 1). The CH₄ emissions from each N treatment in the RW and RF systems were not statistically different, except for the RW240 and RF240 treatments. During the winter seasons, all plots served as weak sources of CH₄ emissions (Table 1).

The RB and RF systems produced less CH₄ emissions than the RW system resulting from less inputs of exogenous carbon (Table A.1). Although they received similar inputs of organic materials, the CH₄ emissions from RB system were significantly lower than those from the RF system (Table 1). This difference can be explained by the aerobic fermentation imposed on the fava bean straw before its incorporation. Previous studies have reported that the addition of fermented straw resulted in considerable decrease of CH₄ emissions, compared with fresh straw (Corton et al., 2000; Linquist et al., 2012). The labile fraction of fresh straw, such as hemicellulose and cellulose, is easily decomposed by microbial organisms during fermentation, which means the fermented straw has a higher lignin content and lower C:N ratio, making it comparatively resistant to methanogens, and resulting in lower CH₄ emissions (Khosa et al., 2010). Linquist et al. (2012) pointed out that assessments of the effects of fermented straw must consider the GHG produced during the fermentation process. In this study, frequent ventilation and continuous stirring of the chopped straw fostered aerobic conditions, and thus low CH₄ emissions were generated during the fermentation process

(Table A.5).

3.3 Effects of rice-cropping systems and N application rate on N₂O emission

Consistent with previous studies, N₂O emissions were less during the rice season in comparison with the wheat season (Ma et al., 2013; Xia et al., 2014). Substantial N₂O emissions occurred only after fertilization and precipitation events, or when the fields were drained (Fig.3). The RB system had the largest N₂O emissions compared with the RW and RF systems, while the RW and RF systems produced similar N₂O emissions, during the rice seasons (Table 1). The higher N₂O emissions from the RB system may have occurred due to a higher accumulation of organic N content in the soil, resulting from the cultivation of rice in rotation with a legume plant (Guong et al., 2012; Cong et al., 2014). The significantly higher N₂O emissions during the cultivation of fava beans compared with leaving the plots to lie fallow further confirmed this (Table 1), even though these treatments received no N additions. Consistent with results derived from meta-analysis of regional to global studies (Xia and Yan, 2012; Chen et al., 2014; Shcherbak et al., 2014), the N₂O emissions increased exponentially as the N application rates increased during the cultivation of rice and wheat (Fig.4).

The EF_d values for wheat cultivation were estimated to be 0.8–1.1%, with an average of 1%, which is identical to the default N₂O emission factor given by the Intergovernmental Panel on Climate Change (IPCC) for fertilized upland soils (IPCC, 2006). The EF_d values were variable for rice cultivation, depending on the different cropping systems (ranging between 0.03% and 0.1%), with an average of 0.07%; this was close to the result of 0.08% reported by Ma et al. (2013) for rice cultivation in the TLR, but much lower than the EF_d

values estimated by other studies (0.3–1.78%) that employed similar water managements (Zou et al., 2005; Zou et al., 2009; Xia et al., 2014).

The lower EF_d values from rice cultivations in this study can partly be explained by the heavy precipitation that occurred during the midseason drainage in the 2014 rice season. Previous studies found that N_2O emissions primarily occurred during the midseason drainage and subsequent re-flooding periods (Yan et al., 2000; Ji et al., 2013). This is because changes in the soil conditions, from anaerobic to aerobic, or vice versa, enhance N-cycle-related microbial processes, such as mineralization, nitrification and denitrification, stimulating substantial N_2O emissions (Xia et al., 2014). However, the heavy precipitation interfered with the midseason aeration in this study and kept the soils at higher levels of water-filled pore space, favoring the further transformation of N_2O produced into N_2 , via denitrification (Yan et al., 2000). Ultimately this resulted in no N_2O emission peaks being observed during midseason drainage (Fig.3), therefore leading to lower EF_d values.

3.4 Effects of rice-cropping systems on SOC

In all of the rice-cropping systems, there was an increase in the SOC stock, as a consequence of straw incorporation (Table 1). The RW system had the largest SOCSR values, which varied between 0.69 and 0.98 t C ha⁻¹ yr⁻¹; these levels which fall within the range of 0.13–2.2 t C ha⁻¹ yr⁻¹, estimated by Pan et al. (2004) for paddy soils in China. The SOCSRs of the RB and RF systems were primarily lower than those in the RW system, due to higher inputs of organic material in the RW system (Table A.1). The SOCSRs of the RB system (0.54–0.87 t C ha⁻¹ yr⁻¹) were obviously higher than those of the RF system (0.40–0.68 kg C

ha⁻¹ yr⁻¹), even although the input rates of exogenous carbon to these systems were comparable (Table A.1). This is because, compared with the fresh straw applied to the RF systems, the fermented bean straw would generally have contained higher contents of lignin and polyphenol; such components would favor the accumulation of lignocelluloses and hemicelluloses that are more resistant to microbial decomposition (Khosa et al., 2010). This would decrease the mineralization rates of SOC, and result in more recalcitrant organic C accumulation (Fan et al., 2014). In addition, Mandal et al. (2008) reported that the conversion efficiency of crop residue C into SOC was doubled after the incorporation of fermented straw, compared with fresh straw. Therefore, appropriate straw management practices in rice paddies would not only reduce CH₄ emissions, but could also benefit the build-up of SOC storage.

3.5 NGWP and GHGI of different rice-cropping systems

The RW system had the highest annual NGWP (6856–12340 kg CO₂-eq ha⁻¹) while the RB system produced the lowest (5332–7288 kg CO₂-eq ha⁻¹) (Table 2), although extra GHGs were emitted during the fermentation process in the RB system (Table A.2). Similar to the NGWP, the annual GHGI in the RB system varied between 0.62 and 0.67 kg CO₂-eq kg⁻¹ grain, which was lower than that of the RW and RF systems (Fig.5). No significant differences in GHGI were found between the RW and RF systems.

The average annual GHGI for the RW system, varied by seasons, between 0.76 and 1.06 kg CO₂-eq kg⁻¹ grain, which was much higher than previous estimations (0.41–0.74 kg CO₂-eq kg⁻¹ grain) of a RW system with organic matter incorporation (Ma et al., 2013). This was largely due to the consideration of GHG emissions from the production of agricultural

materials in this study. Assessing GHG emissions during the life-cycles of producing grains enables the identification of ‘authentic’ hotspot sources of GHG emissions (Yvon-Durocher et al., 2010; Chen et al., 2014). For example, the production of various agricultural inputs, rather than the field cultivation, has been shown to contribute the most GHG emissions to the carbon footprints of wheat and corn (Xia et al., 2016). For rice production, previous studies mostly focused on the CH₄ emissions from paddy soils only (Yang et al., 2010; Zhang et al., 2015), but the results of our study show that, in addition to CH₄ emissions, N production and fertilization, and irrigation are also large contributors to the GHGI (Fig.5). This highlights the need to adopt knowledge-based N management practices and water regimes (Ju et al., 2009; Ma et al., 2013).

Compared with current production patterns in the TLR (RW300), a 20% reduction in N fertilizer rates (RW240) produced lower GHG emissions from N production and fertilization (20.4%), and higher rice yields (2%) and SOCSR (4%), which ultimately resulted in a lower annual GHGI (17%) (Fig.5). Further reductions in the N application rates (RW180) resulted in an even lower NGWP, but this occurred at the cost of significant losses in rice and wheat yields (Table 1 and Table 2). Without significantly influencing the rice yields, the GHGI was decreased by 12%, with the conversion of the RW240 into the RB240; this was mainly attributed to the considerable decrease in CH₄ emissions (29%) that resulted from straw-fermentation in the RB system. Meanwhile, 180 kg N ha⁻¹ was saved by the cultivation of fava beans in the RB240, avoiding GHG emissions of 1494 kg CO₂-eq (contributing up 20% of the NGWP in RB240). The conversion of a RW (RW240) to a RF system (RF240) would result in significant losses of the annual grain yield (41%), although the NGWP was

decreased by 33%. As a result, the GHGI in the RF240 was 14% higher than that of the RW240 (Fig.5). Overall, it is clear that, in the TLR, security in the rice yields and lower GHG emissions can be achieved simultaneously by adopting appropriate N reduction practices, changing the rice-cropping system and employing eco-friendly straw management practices.

Previous studies have shown that optimizing cereal cropping systems and reducing N fertilizer applications can together produce more grain with lower GHG emissions (Gao et al., 2014; Weller et al., 2016). In an upland cropping system of the North China Plain, Gao et al. (2014) reported that optimizing the double-cropping system and N application rates led to a significant increase in the crop yields and a significant decrease in the annual GHGI. But for rice paddies with straw returns, the advantages of optimizing the rice-cropping system and reducing N application levels (like the RB system in this study), in terms of GHGI mitigation, only be expected when the straw is reasonably applied to the soils (Xia et al., 2014). The direct incorporation of straw before rice transplanting causes the paddy soils to be a source of GHG emissions, due to higher levels of CH₄ emissions compared with SOC accumulation (Liu et al., 2014); this highlights the need to develop an eco-friendly straw incorporation method. Our results showed by fermenting straw aerobically before its incorporation, the CH₄ emissions were significantly reduced, while the effects on the SOC sequestration were insignificant, in comparison with the direct incorporation of wheat straw (Table 1). On the premise of straw fermentation, the RB system was better at reducing the GHGI, compared with the RW and RF systems (Fig.5). Overall, in terms of mitigating the GHG emissions from rice paddies with straw returns, priority should be given to developing eco-friendly straw management practices, such as fermenting straw under aerobic conditions before its

incorporation.

Besides CH₄ emissions, as described above, N fertilization, irrigation and SOCSR affected the GHGI considerably (Fig.5). To further reduce the future GHGI of rice production in the TLR, efforts could be made to adopt knowledge-based N managements, which would improve the current low nitrogen recovery efficiencies; these practices include choosing right N sources and applying them at the right rate, time and place, by considering the dynamics between the N supply from soils and N demand of crops (Ju et al., 2009; Chen et al., 2014). Secondly, a moderate water regime that alternates between dry and wet irrigation could increase the grain-filling rate and enhance rice root growth, which in turn would reduce the GHG emissions that result from power consumption during rice irrigation (Zhao et al., 2015a). Thirdly, although the SOCSRs offset the positive GHGI by 17–34% in this study, these could be further increased by increasing the employment of conservative tillage or no-tillage practices (Lu et al., 2009).

3.6 Economic assessment of different rice-cropping systems

As profitability is the main driver for farmers to change their agricultural management practices, we assessed the economic effects of different rice-cropping systems. Although additional labour and energy were required to implement the fermentation process (Table A.2), the net economic benefit (NEB) of the RB system was still 22–94% higher than the RW system (Table 2); this was mainly attributed to the higher price of the fava bean crop, despite its lower grain yields in comparison with wheat (Table 1). Similarly, substituting the fava bean crop for wheat in a rice/wheat cropping system increased the farmer's net economic

return by 23% (Zhao et al., 2015b). As there was no grain produced in the fallow season, the RF system produced the lowest economic returns. However this does not consider the fact that laborers are able to source other income during the fallow season. Compared with the treatment (RW300) that represented current local practices, the NEB was 23% higher in the RB240 treatment, which would be realized by a reasonable N reduction (20%) and the conversion of the cropping system from a RW to a RB system.

The RB system not only effectively alleviated the GHG emissions and improved the NEB, but could also reduce various Nr losses and the subsequent environmental costs, although this is beyond the focus of this study. Compared with the cultivation of wheat, cultivating beans usually results in lower releases of Nr (Zhao et al., 2015b), because no N fertilization is required, thus causing less damage costs to the environment and human health (Xia and Yan, 2012; Gu et al., 2015). Zhao et al. (2015b) reported that their rice/legume systems resulted in significantly lower ammonia volatilization (31–38%) and N runoff (82–86%) levels, reducing the total environment costs by 49–50%, in comparison with a traditional RW system.

After taking into consideration the environmental costs incurred by the emissions of GHGs and Nr losses (Xia and Yan, 2012; Xia et al., 2016), we also preliminarily evaluated the net (economic and ecological) benefit of rice production in the three cropping systems (Appendix B). The economic response curves demonstrated that the maximum net (economic and ecological) benefit appeared at the N application rate of 246 (RW), 230 (RB) and 245 (RF) kg N ha⁻¹ (Fig.B.1); this further highlighted the need to optimize the current N application

rate (300 kg N ha⁻¹) for rice production in the TLR, for the sake of economic return pursuing and environment concerning (Xia and Yan, 2012). The average net (economic and ecological) benefit of rice production in the RB system was 12% higher than the RW system, but the total damage costs to environment were 14% lower. Therefore, employing the RB system with straw fermentation and a reduction in N application rates might be highly attractive for rice farmers in the future, due to the lower damage costs to environment and higher economic returns.

4 Conclusions

We have demonstrated a case study that reduced GHG emissions while increasing the economic returns of a rice-based cropping system in China. In comparison to the current rice/wheat production practices employed in the TLR, the GHGI was 17% lower when the respective N application rates for the rice and wheat seasons were reduced by 20%. However, further reductions of the N application rates resulted in an increase in the GHGI. Based on a 20% reduction in N application rates, the use of a RB system (including straw fermentation) instead of a RW, produced an even lower GHGI, by 12%, and higher NEB, by 22%. However, the RF system had a higher GHGI and a lower NEB compared with the RW system. In order to simultaneously achieve a high NEB and low GHGI, we proposed the integrated adoption of some agronomic practices in the rice/wheat cropping system, including: appropriate levels of N reduction, employing suitable rice-cropping system and the use of eco-friendly straw management practices. Future research should assess the long-term effects of N reduction and the conversion of the RW cropping systems on crop yields, GHG emissions and NEB.

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Supplementary data

Supplementary data (Appendix A and B) associated with this article can be found, in the online version.

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Table 1 Seasonal average CH₄ emissions, N₂O emissions, SOC sequestration rate (SOCSR) and grain yields for the three rice-cropping systems during the two-cycles from 2013 to 2015

Treatment	Rice growing season			Non-rice growing season			Annual
	CH ₄	N ₂ O	Yield	CH ₄	N ₂ O	Yield	SOCSR
	kg CH ₄ ha ⁻¹	kg N ha ⁻¹	kg ha ⁻¹	kg CH ₄ ha ⁻¹	kg N ha ⁻¹	kg ha ⁻¹	t C ha ⁻¹ yr ⁻¹
RW0	^a 306.7(39)a ^{A1}	0.05(0.05)c ^{A1}	5374(617)d ^{B1}	0.35(0.68)a	0.17(0.04)cd ^{A1}	1137(215)d	0.69(0.10) ^{A1}
RW120	334.6(64)a ^{A2}	0.09(0.02)bc ^{A2}	7339(843)c ^{A2}	1.42(1.91)a	0.93(0.56)bc ^{A2}	3394(435)c	0.77(0.19) ^{A2}
RW180	289.5(14)a ^{A3}	0.18(0.07)bc ^{B3}	7711(527)bc ^{B3}	0.86(1.07)a	1.33(0.74)bc ^{A3}	4305(350)b	0.82(0.14) ^{A3}
RW240	295.5(38)a ^{A4}	0.24(0.14)ab ^{B4}	8576(562)a ^{A4}	0.61(0.31)a	2.09(1.13)ab ^{A4}	4840(237)a	0.98(0.13) ^{A4}
RW300	324.5(72)a	0.35(0.25)a	8395(468)abc	1.81(2.18)a	2.71(1.62)a	5121(345)a	0.94(0.18)
RB0	178.5(46)b ^{B1}	0.08(0.05)bc ^{A1}	6409(487)c ^{A1}	0.54(1.64)a	0.21(0.08)a ^{A1}	2144(474)a	0.54(0.05) ^{AB1}
RB120	186.5(50)a ^{B2}	0.14(0.07)bc ^{A2}	7536(670)b ^{A2}	0.63(1.07)a	0.31(0.20)b ^{B2}	2136(167)a	0.64(0.09) ^{A2}

RB180	212.3(63)a ^{B3}	0.22(0.11)ab ^{A3}	8199(507)a ^{A3}	0.65(1.21)a	0.35(0.25)a ^{B3}	2219(233)a	0.76(0.21) ^{A3}
RB240	210.3(59)a ^{AB4}	0.33(0.25)a ^{A4}	8364(697)a ^{A4}	2.16(3.01)a	0.39(0.17)a ^{B4}	2421(467)a	0.87(0.04) ^{AB4}
RF0	266.5(37)a ^{A1}	0.03(0.04)d ^{A1}	5875(674)b ^{B1}	0.58(0.71)a	0.07(0.02)bc ^{B1}	--	0.40(0.11) ^{B1}
RF120	288.3(65)a ^{A2}	0.11(0.03)c ^{A2}	7185(781)a ^{A2}	0.86(1.01)a	0.11(0.05)ab ^{C2}	--	0.46(0.10) ^{B2}
RF180	241.7(29)a ^{A3}	0.16(0.03)b ^{B3}	7660(709)a ^{B3}	1.33(2.64)a	0.13(0.07)ab ^{C3}	--	0.63(0.08) ^{A3}
RF240	224.2(54)a ^{B4}	0.23(0.04)a ^{B4}	7890(580)a ^{A4}	0.66(3.01)a	0.13(0.07)a ^{C4}	--	0.68(0.15) ^{B4}

^aMean(SD); different lowercase letters indicate significant differences among treatments in the same rice-cropping system, and different uppercase letters with the same Arabic numeral, indicate significant differences among treatments in different rice-cropping systems ($P < 0.05$).

Table 2. The annual average net GWP (NGWP) and net economic benefit (NEB) for different rice-cropping systems during the two-cycles from 2013 to 2015

Treatment ^a	NGWP	NEB
	kg CO ₂ -eq ha ⁻¹ yr ⁻¹	¥ ha ⁻¹ yr ⁻¹
RW0	^a 6856(970)A ¹	12403(1407)B ¹
RB0	5332(1264)B ¹	24057(3003)A ¹
RF0	6703(935)A ¹	12059(1525)B ¹
RW120	9537(2133)A ²	19774(1841)B ²
RB120	6418(1205)B ²	26372(1939)A ²
RF120	8174(1626)AB ²	14912(1769)C ²
RW180	9346(832)A ³	21775(930)B ³
RB180	7028(1419)B ³	28452(844)A ³
RF180	6938(713)B ³	15932(1609)C ³
RW240	10198(1507)A ⁴	24373(1497)B ⁴
RB240	7288(1485)B ⁴	29767(3163)A ⁴
RF240	6850(1806)B ⁴	16393(1318)C ⁴
RW300	12340(2531)	24245(1034)

^aMean(SD); different uppercase letters with the same Arabic numeral, indicate significant differences among the treatments in the different rice-cropping systems ($P < 0.05$).

Figure captions:

Fig.1. Seasonal variation in the daily precipitation and temperature during the two-cycles of three rice-cropping systems from 2013 to 2015.

Fig.2. Seasonal variations in the CH₄ fluxes during the two cycles of the rice–wheat (a), rice–fava bean (b) and rice–fallow (c) system, from 2013 to 2015. The vertical bars denote standard error.

Fig.3. Seasonal variations in the N₂O fluxes during the two cycles of the rice–wheat (a), rice–fava bean (b) and rice–fallow (c) system, from 2013 to 2015. The arrows indicate when the N fertilizer was applied. The vertical bars denote standard error.

Fig.4. Relationships between N fertilizer application rates and seasonal N₂O emissions during the rice-growing seasons in the rice–wheat (a), rice–fava bean (b), rice–fallow (c) systems, and during the wheat-growing season of the rice–wheat system (d). The vertical bars denote standard error.

Fig.5. Annual average greenhouse gas intensity (GHGI) of each N treatment for the three rice-cropping systems from 2013 to 2015. (1) ‘N fertilization’ denotes the production, transportation and application of N fertilizer. (2) The ‘SOCSR’ denotes the SOC sequestration rate. (3) ‘Others’ denotes other sources of GHG emissions, such as the production and transportation of phosphorus, potassium and diesel oils, and their application. The GHG

emissions associated with the fermentation process in the RB system were also included in the 'Others'. (4) Different lowercase letters above the error bars indicate significant differences among treatments in the same rice-cropping system, and different uppercase letters with the same Arabic numeral indicate significant differences among treatments in different rice-cropping systems ($P < 0.05$).

Fig.1

Fig.2

Fig.3

Fig.4

Fig.5