Greenhouse gas emissions and reactive nitrogen releases from rice production with simultaneous incorporation of wheat straw and nitrogen fertilizer

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Abstract

Impacts of simultaneous inputs of crop straw and nitrogen (N) fertilizer on greenhouse gas (GHG) emissions and N losses from rice production are not well understood. A two-year field experiment was established in a rice-wheat cropping system in the Taihu Lake region (TLR) of China to evaluate the GHG intensity (GHGI), reactive N intensity (NrI) of rice production with inputs of wheat straw and N fertilizer. The field experiment included five treatments of different N fertilization rates for rice production: 0 (RN0), 120 (RN120), 180 (RN180), 240 (RN240) and 300 kg N ha^{-1} (RN300, traditional N applied rate in the TLR). Wheat straws were fully incorporated into soil before rice transplantation. The meta-analytic technique was employed to evaluate various Nr losses. Results showed that the response of rice yield to N rate successfully fitted a quadratic model, while N fertilization promoted Nr discharges exponentially (nitrous oxide emission, N leaching and runoff) or linearly (ammonia volatilization). The GHGI of rice production ranged from 1.20 (RN240) to 1.61 (RN0) kg CO_2 equivalent (CO_2 eq) kg^{-1}, while NrI varied from 2.14 (RN0) to 10.92 (RN300) g N kg^{-1}. Methane (CH_4) emission dominated the GHGI with proportion of 70.2–88.6% due to direct straw incorporation, while ammonia (NH_3) volatilization dominated the NrI with proportion of 53.5–57.4%. Damage costs to environment incurred by GHG and Nr releases from current rice production (RN300) accounted for 8.8% and 4.9% of farmers' incomes, respectively. Cutting N application rate from 300 (traditional N rate) to 240 kg N ha^{-1} could improve rice yield and nitrogen use efficiency by 2.14% and 10.30%, respectively, whilst simultaneously reduced GHGI by 13%, NrI by 23% and total environmental costs by 16%. Moreover, the reduction of 60 kg N ha^{-1} improved farmers' income by 639 ¥ ha^{-1}, which would provide them with an incentive to change the current N application rate. Our study

suggests that GHG and Nr releases, especially for CH_4 emission and NH_3 volatilization, from rice production in the TLR could be further reduced, considering the current incorporation pattern of wheat straw and N fertilizer.

Key words: Taihu Lake region, greenhouse gas intensity, Nr intensity, rice production, straw incorporation

1 Introduction

Rice is the staple food for the majority of the world's population. However, while industriously feeding the global population, rice production is an important source of greenhouse gas (GHG) emissions and reactive nitrogen (Nr) releases (Yan et al., 2009; Chen et al., 2014). Rice production in China involves heavy methane (CH_4) emissions due to the water regime managements (e.g., continuous flooding in some regions) and straw incorporation practices (e.g., direct incorporation without any pretreatments) (Yan et al., 2009). Besides, lower nitrogen use efficiency for rice cultivation in China (approximately 31%) aggravates the release of various Nr species, thus threatening ecosystem functions (Galloway et al., 2008; Zhang et al., 2012). Such a dilemma highlights the need for the simultaneous evaluation of GHG emissions and Nr losses for rice production in China. Rice cultivation in intensive agricultural regions, characterized by high inputs of N fertilizer and crop residues, should be prioritized for the implementation of such evaluation (Ju et al., 2009; Chen et al., 2014).

Taihu Lake region (TLR) is one of the most productive areas for rice production in China, largely owing to the popularity of intensive cultivation (Zhao et al., 2012a; Zhao et al., 2012b). Currently, rice yield of this region in some fields can reach up to 8000 kg ha^{-1} or even higher (Ma et al., 2013; Zhao et al., 2015). However, these grain yields are achieved with a cost to environment (Ju et al., 2009). TLR generally receives 550–600 kg N ha^{-1} yr^{-1}, with the rice-growing season accounting for nearly 300 kg N ha^{-1} (Zhao et al., 2012b). Asides from these excessive N inputs, TLR also experiences high amounts of crop residue incorporation, which is highly encouraged by local governments (Xia et al., 2014). However, direct straw incorporation before rice transplantation triggers substantial CH_4 emissions (Ma et al., 2009; Ma et al., 2013).

Besides such substantial releases of Nr and GHG in a direct way, indirect releases during the production of various agricultural materials used for farming operations, are also not ignorable, due to higher input rates of these materials caused by intensive cultivation (Zhang et al., 2013; Cheng et al., 2014). This warrants the need for life-cycle assessment (LCA) of GHG emissions and Nr releases with respect to rice production in this region.

Considerable environmental costs can be caused by the direct and indirect releases of GHG and Nr from rice production in the TLR, for instance, in the form of global warming, water eutrophication, or soil acidification (Ju et al., 2009; Xia and Yan, 2011; Xia and Yan, 2012). Previous studies have proven that environmental costs assessment could provide guidance for emerging policy priorities in mitigating certain GHG or Nr species, after quantifying both their release amounts and damage costs to ecosystems (Gu et al., 2012). However, few studies have attempted to evaluate the total GHG and Nr releases and the associated environmental costs from rice production, with high inputs of N fertilizer and crop straw.

In the present study, we conducted two years of simultaneous measurements of CH_4 and nitrous oxide (N_2O) emissions from a rice-wheat cropping system in the TLR to evaluate the impacts of simultaneous inputs of crop straw and N fertilizer on (1) net global warming potential (NGWP) and GHG intensity (GHGI), (2) total Nr losses and Nr intensity (NrI), (3) environmental costs incurred by these GHG and Nr releases associated with rice production, from the perspective of LCA.

2 Materials and methods

2.1 Experimental site

The field experiment was conducted in a paddy rice field at Changshu Agroecological

Experimental Station (31°32′93″N, 120°41′88″E) in Jiangsu province, which is located in the TLR of China where the cropping system is primarily dominated by summer rice (*Oryza sativa* L.) and winter wheat (*Triticum aestivum* L.) rotation. The climate of the study area is subtropical monsoon, with a mean air temperature of 16.1°C and mean annual precipitation of 990 mm, of which 60–70% occurs during the rice-growing season. The daily mean temperature and precipitation during two rice-growing seasons from 2013 to 2014 are shown in Fig.1. The paddy soil is classified as Anthrosol, which develops from lacustrine sediments. The topsoil (0–20cm) has a pH of 7.68 (H_2O). The bulk density is 1.16 g cm^{-3}, the organic C content is 20.1 g C kg^{-1}, the total N is 1.98 g kg^{-1}, the available P is 11.83 mg kg^{-1} and the available K is 126 mg kg^{-1}.

2.2 Experimental design and field management

The field experiment included five treatments of different N fertilization rates for rice production: 0 (RN0), 120 (RN120), 180 (RN180), 240 (RN240) and 300 kg N ha^{-1} (RN300, traditional N applied rate in the TLR). Consistent with local practices, wheat straws were harvested, chopped and fully incorporated into soil before rice transplantation in all treatments (Table 1). All of the treatments are laid out in a randomized block design with three replicates, and each plot covered an area of 3 m × 11 m (33 m^2).

Rice is transplanted in the middle of June and harvested at the beginning of November. N fertilizer (in the form of urea) was split into three parts during the rice-growing season: 40% as basal fertilizer, 30% as tiller fertilizer, and 30% as panicle fertilizer. Phosphorus (in the form of calcium superphosphate) and potassium (in the form of potassium chloride) were applied as basal fertilizer at rates of 30 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}, respectively. All basal fertilizers were thoroughly incorporated into the soil through plowing, while topdressing fertilizers were applied evenly to the soil surface. According to local practices, the water regime of 'flooding-midseason drainage-flooding-moist but non-waterlogged by intermittent irrigation' was adopted. Details of the specific agricultural management practices for rice production are provided in Table 1.

2.3 Gas fluxes and topsoil organic carbon sequestration rate

The CH_4 and N_2O fluxes during the rice-growing seasons of 2013 and 2014 were measured using a static chamber and gas chromatography technique. Details of the procedures used for sampling and analysis the gases were described in Xia et al. (2014).

Generally, it takes long-term observations over years to decades before the soil organic carbon (SOC) change is detectable (Yan et al., 2011). The SOC content changes of short-term field experiment couldn't be correctly measured, due to the high variability of SOC during the preliminary several years of the experiment. Therefore, we used the following relationship between the straw input rate (kg C ha^{-1} yr^{-1}) and SOC sequestration rate (SOCSR, kg C ha^{-1} yr^{-1}), obtained through an on-going long-term straw application experiment in the same region, to calculate the SOCSR in this study (Xia et al., 2014):

$$\text{SOCSR} = \text{Straw input rate} \times 0.0603 + 31.39 \ (R^2 = 0.92); \qquad (1)$$

This on-gonging long-term field experiment is also taking place at the Changshu Agroecological Experimental Station (since 1990), which includes three straw application levels: 0, 4.5 t, and 9.0 t dry-weight ha^{-1} yr^{-1}. The equation (1) was established based on the results of 22-year observation (Xia et al., 2014). Same agricultural management practices were applied to the on-going long-term experiment and the experiment of this study.

2.4 Net global warming potential and greenhouse gas intensity

The net global warming potential (NGWP, kg CO_2 eq ha^{-1}) and greenhouse gas intensity

(GHGI, kg CO_2 eq kg^{-1}) of rice production in the TLR was calculated using the following equations:

$$\mathrm{NGWP} = \sum_{i=1}^{m} \mathrm{AI}_{\mathrm{ico_2}} + \mathrm{CH_4} \times 25 + \mathrm{N_2O} \times 44/28 \times 298 - \mathrm{SOCSR} \times 44/12; \quad (2)$$

$$\mathrm{GHGI} = \mathrm{NGWP}/\text{rice yield}; \quad (3)$$

Here, $\mathrm{AI}_{\mathrm{ico_2}}$ denotes the GHG emissions from the production and transportation of agricultural inputs, which are calculated by multiplying their application rates by their individual GHG emission factors, such as synthetic fertilizers, diesel oil, electricity and pesticides (Liang, 2009; Zhang et al., 2013). CH_4 (kg CH_4 ha^{-1}), N_2O (kg N ha^{-1}) and SOCSR (kg C ha^{-1} yr^{-1}) represent the CH_4 and N_2O emissions from rice production, and the SOC sequestration rate, respectively.

2.5 Total Nr losses and Nr intensity

The total Nr losses (kg N ha^{-1}) and Nr intensity (NrI, g N kg^{-1}) were calculated using the following equations:

$$\text{Total Nr losses} = \sum_{i=1}^{m} \mathrm{AI}_{\mathrm{iN_r}} + (\mathrm{NH_3} + \mathrm{N_2O} + \mathrm{N_{Leaching}} + \mathrm{N_{Runoff}})_{\mathrm{rice}}; \quad (4)$$

$$\mathrm{NH_3}\ \text{volatilization} = 0.17 \times \mathrm{N_{rate}} + 0.64; \quad (5)$$

$$\text{N runoff} = 5.39 \times \mathrm{Exp}\ (0.0054 \times \mathrm{N_{rate}}); \quad (6)$$

$$\text{N leaching} = 1.44 \times \mathrm{Exp}\ (0.0037 \times \mathrm{N_{rate}}); \quad (7)$$

$$\mathrm{NrI} = (1000 \times \text{Total Nr losses})\ /\text{rice yield}; \quad (8)$$

Here, $\mathrm{AI}_{\mathrm{i_{Nr}}}$denotes the Nr lost (mainly through N_2O and NO_X emissions) from the production and transportation of agricultural inputs (Liang, 2009; Zhang et al., 2013), while '$(NH_3+N_2O+N_{Leaching}+N_{Runoff})_{rice}$' represents the NH_3 volatilization, N_2O emissions, N leaching and runoff during the rice-growing season. N_{rate} represents the N fertilizer application rate. Nr empirical models (Equation 5, 6, 7) derived from a meta-analysis of published literature concerning Nr losses from rice production in the TLR. Specific details regarding this literature survey are provided in Appendix A.

2.6 Total environmental costs incurred by GHG and Nr releases and farmer's income

The total environmental costs (¥ ha^{-1}) incurred by GHG and Nr releases and farmer's income from rice production in the TLR were calculated based on the following equations:

$$\text{Environmental costs} = \sum_{i=1}^{n} (Nr_i A \times DC_i) + CO_2 A \times DC_{CO2}; \qquad (9)$$

$$\text{Farmer's income} = \text{rice yield} \times \text{rice price} - \text{input costs}; \qquad (10)$$

$Nr_i A$ (kg N) represents the release amounts of certain Nr species (i), and DC_i (¥ kg^{-1} N) denotes the damage cost (DC) per kg of certain Nr (i). CO_2A (ton) and DC_{CO2} (¥ ton^{-1}) represent the CO_2 emissions amount and global warming cost of CO_2, respectively. N_2O is both a GHG and Nr species, but its environmental cost was calculated as a GHG here. Because the cost of N_2O emission as Nr species is to damage human health (Gu et al., 2012), but the effects of Nr losses on the damage costs of human health were not included in this study. The environmental costs mainly refer to the global warming incurred by GHG emissions, soil acidification incurred by NH_3 and NO_X emissions, and aquatic eutrophication caused by NH_3 emissions, N leaching and runoff (Xia and Yan, 2012).

2.7 Nitrogen use efficiency and N_2O emission factor

Nitrogen use efficiency (NUE) and N_2O emission factor (EF_d%) were respectively calculated by the following equations (Ma et al., 2013; Yan et al., 2014):

$$NUE = (U_N - U_0)/F_N; \qquad (11)$$

$$EF_d\% = (E_N - E_0)/F_N; \qquad (12)$$

Here, U_N is the plant N uptake (kg ha^{-1}) measured in grain at physiological maturity in the N fertilization treatments, while U_0 is the N uptake measured in grain in the treatment without N fertilizer addition (RN0). E_N denotes the cumulative N_2O emissions in the N fertilization treatments, while E_0 denotes the N_2O emissions in the RN0. F_N represents the application rate of N fertilizer. The N uptake in straw and grain was analysed via concentrated sulfuric acid digestion and the Kjeldahl method (Zhao et al., 2015).

2.8 Statistical analysis

Differences in seasonal CH_4, N_2O emissions and rice yield of the two rice-growing seasons from 2013 to 2014 affected by fertilizer treatments, year and their interaction were examined by using a two-way analysis of variance (ANOVA) (Table 2). The grain yield, seasonal CH_4 and N_2O emissions, SOCSR and GHGI of different treatments were tested by analysis of variance, and mean values were compared by least significant difference (LSD) at the 5% level. All these analyses were carried out using the SPSS (Version 19.0, USA).

3 Results and discussion

3.1 Rice yield and NUE

The two-way ANOVA analyses indicated that the rice grain yields were significantly affected by the year and fertilizer treatment (Table 2). The farmer's practice plot (RN300) had an average rice grain yield of 8395 kg ha^{-1}, with an NUE of 31.35%, over the two growing seasons from 2013 to 2014. Compared with RN300, reducing the N fertilizer rate by 20% (RN240) slightly improved the grain yield and NUE to 8576 kg ha^{-1} and 34.58%, respectively. Further N reduction, without additional agricultural managements, could decrease the rice yield by 8.15% (RN180) and 15.18% (RN120) (Table 3). The response of rice yield to the synthetic N application rate in our study successfully fitted a quadratic model (Fig.2), as has been reported in previous studies (Xia and Yan, 2012; Cui et al., 2013a). Reducing N application reasonably, therefore, is considered essential to reduce environmental costs, without sacrificing grain yield (Chen et al., 2014). Our study showed that lowering the N input adopted by local farmer (300 kg N ha^{-1}) by 20% could still enhance the grain yield and NUE. However, a further reduction of N 40% (RN180) would largely undermine the rice yield (Table 3).

Further reduction in N fertilizer may be achieved with improvements of agricultural managements, Ju et al. (2009) reported that, based on knowledge-based N managements, such as optimizing N fertilizer source, rate, timing and place (in accordance with crop demand), rice grain yield in the TLR was not significantly affected by a 30–60% N saving, while various Nr losses would endure a two-fold curbing. Similarly, Zhao et al. (2015) found that the NUE could be improved from 31% to 44%, even under a N reduction of 25% for rice production in the TLR, through the implementation of integrated soil-crop system managements. In the present study, the NUE was improved by 10% via a 20% N reduction, but it still falls behind the NUE values in the studies which received knowledge-based N managements. Previous studies have proven that straw incorporation exerted little impacts on grain yield. For instance, a meta-analysis conducted by Singh et al. (2005) have found that incorporation of crop straw produced no significant trend in improving crop yield in rice-based cropping systems. Moreover, based on a long-term straw incorporation experiment established since 1990 in the TLR, Xia et al. (2014) have reported that long-term incorporation of wheat straw only increased the rice yield by 1%. Therefore, in the present study, the effects of straw incorporation on rice yield were considered as inappreciable.

3.2 CH_4, N_2O emissions and SOSCR

Over the two rice-growing seasons from 2013 to 2014, all treatments showed similar patterns of CH_4 fluxes, albeit with large inter-annual variation (Fig.3a). The seasonal average CH_4 emissions from all plots showed no significant difference, ranging from 289.53 kg CH_4 ha^{-1} in the RN180 plot to 334.61 kg CH_4 ha^{-1} in the RN120 plot (Table 4), much higher than observations conducted in the same region (Zou et al., 2005; Ma et al., 2013). This phenomenon can be attributed to the larger amounts of straw incorporation in this study (Table 1). Relative to the RN300 plot, CH_4 emissions from the RN240 plot decreased by 8% and 10%, during the rice-growing season of 2013 and 2014, respectively, although this effect was not statistically significant (Table 4).

Many studies have shown a clear linear relationship between CH_4 emissions and the amounts of applied organic matter (OM) (Shang et al., 2011; Xia et al., 2014). It is possible that the linear response of CH_4 emissions to OM inputs can become flat or even unobvious (Fig.S1), when the OM applied rates among different treatments were insignificant different (Table S1). It is unsurprising that no obvious relationship between CH_4 emissions and N fertilizer application rates was observed in this study (Fig.S1), because the effects of N fertilization on CH_4 production, transportation and oxidation are complex. For instance, N fertilization can provide methanogens with more carbon substrates in the rhizosphere of plants by stimulating the growth of rice biomass, thus promoting CH_4 production and transportation (Zou et al., 2005; Banger et al., 2012). On the other side, N enrichment could also enhance the activities of methanotrophs, therefore enhancing CH_4 oxidation (Xie et al., 2010; Yao et al., 2012).

The N_2O fluxes were sporadic and pulse-like, and these fluxes showed large variations between different seasons, and the majority of the N_2O peaks occurred after the application of N fertilizer (Fig.3b). The two-way ANOVA analyses indicated that the seasonal N_2O emissions were significantly affected by the year, the fertilizer treatment, and their interactions during the rice-growing seasons (Table 2). The average N_2O emission, during the two rice-growing seasons, ranged from 0.05 kg N ha^{-1} for the RN0 to 0.35 kg N ha^{-1} for the RN300 (Table 4), which increased exponentially as the N fertilizer rate increased; this highlights that the reduction of N fertilizer rate is an effective approach to reduce the N_2O emissions (Zou et al., 2005; Zhang et al., 2012). The average N_2O emission factors varied between 0.03% and 0.1%, with an average of 0.07%, which is comparable with previous studies (0.05–0.1%) conducted in the same region (Ma et al., 2013; Zhao et al., 2015).

The rice paddies have witnessed an increase in the SOC stock as a result of straw incorporation (Table 4). The estimated topsoil (0–20cm) SOCSR varied from 0.13 t C ha^{-1} yr^{-1} for the RN0 plot to 0.197 t C ha^{-1} yr^{-1} for the RN300 plot. The empirical model established through a long-term straw incorporation study in the same region was employed to evaluate the SOCSR in this study, which likely brought uncertainty into the results of this study. Under the same agricultural managements, soil and climatic conditions, cropping systems and straw types, it is reasonable to believe that the rates of straw C stabilizing into SOC (i.e. conversion efficiency of crop residue C into SOC) are similar between these two experiments (Mandal et al., 2008). It is reported that the conversion rates of crop straw to SOC in two main wheat/maize production regions in China, which have similar climatic conditions and agricultural practices, were very close, at 40.524 versus 40.607 kg SOC-C t^{-1} dry-weight straw (Lu et al., 2009). Moreover, the current estimated SOCSR for rice production in the TLR (0.197 t C ha^{-1}), is comparable to the estimation of 0.17 t C ha^{-1} yr^{-1} from Ma et al. (2013) in a study based on a paddy field experiment with OM incorporation in the same region. Therefore, we hold the opinion that the above SOCSR calculation method is appropriate, and the uncertainty incurred by this method unlikely affects the main conclusions of this study.

The magnitude of the SOC increase is variable depending on the straw incorporation method, the degree of tillage, the cropping systems and etc. (Yan et al., 2011; Huang et al., 2013). Liu et al. (2014) suggested that straw incorporation in rice-based cropping systems requires an overall consideration, due to the direct incorporation promoting substantial CH_4 emissions. When converting to CO_2 eq, the SOCSR only offsets the CH_4 emissions by 6.2–9.2% in this study (Table 4). This proportion is expected to increase provided that appropriate straw incorporation method (e.g., compost straw before incorporation) and conservative-tillage are adopted. Moreover, previous studies have shown that the combined adoption of conservative-tillage system with straw return had large advantages in increasing SOC stocks while reducing CH_4 emissions (Zhao et al., 2015a; Zhao et al., 2015b).

3.3 NGWP and GHGI

The average NGWP for all treatments varied from 8656 to 11550 kg CO_2 eq ha^{-1} (Table 4). CH_4 emissions dominated the NGWP in all treatments, with the proportion ranging from 70.23% to 88.56%, while synthetic N fertilizer production was the secondary contributor (Table 4). In addition, SOC sequestration offset the positive GWP by 5.18–6.18% in the fertilization treatments. Compared to conventional practice (RN300), the NGWP in the 20% reduction N practice (RN240) decreased by 10.64%. Therein, 6.28% came from CH_4 reduction and 4.31% from N production savings (Table 4). The GHGI of rice production ranged from 1.20 (RN240) to 1.61 (RN0) kg CO_2 eq kg^{-1}, which is higher than previous estimation of 0.24–0.74 kg CO_2 eq kg^{-1} for rice production in other rice-upland crop rotation systems (Qin et al., 2010; Ma et al., 2013). Moreover, the GHGI of current rice production in the TLR (RW300) was estimated to be 1.45 times that of the national average value estimated by Wang et al. (2014a), at 1.38 versus 0.95 kg CO_2 eq kg^{-1}.

Such phenomenon was attributed to the following reasons. First, compared to above studies, current higher amounts of direct straw incorporation (2.9–6.2 Mg dry matter ha^{-1}), before rice transplantation in the TLR, triggered substantial CH_4 emissions (290–335 kg CH_4 ha^{-1}). Crop residue incorporation is regarded as a win-win strategy to benefit food security and mitigate climate change, due to the fact that it possesses a large potential for carbon sequestration (Lu et al., 2009). However, the GWP of straw-induced CH_4 emissions was reported to be 3.2–3.9 times that of the straw-induced SOCSR, which indicates direct straw incorporation in paddy soils worsens rather than mitigates climate changes, in terms of GWP (Xia et al., 2014). The SOC sequestration induced by straw incorporation only offset the positive GWP by 5.2–6.2% in this study. Sensible methods of straw incorporation should therefore be developed to reduce the substantial CH_4 emissions without compromising the build-up of SOC stock in the TLR.

Second, the high N application rate (300kg N ha^{-1}) in the TLR combined with the large emission factor of N fertilizer production, 8.3 kg CO_2 eq kg^{-1} N (Zhang et al., 2013), marked the sector of N fertilizer production as the secondary contributor to the GHGI (Table 4); this sector, however, was not involved in above-mentioned studies. Compared to local farmer’s practices (RN300), reducing the N rate by 20% (RN240) lowered the GHGI by 13%, under the condition of straw incorporation, although this effect was not statistically significant (Table 4). Compared to RN240, however, further reduction of N rate (RN180 or RN120) increased the GHGI, due to the fact that rice yield was considerably reduced under excessive N reduction. Therefore, the joint application of reasonable N reduction and judicious method of straw incorporation would be promising in reducing the GHGI for rice production in the TLR, in consideration of the current situation of simultaneous high inputs of N fertilizer and wheat straw.

3.4 Various Nr losses and NrI

The results of the meta-analysis indicated that N_2O emissions, as well as N leaching and runoff, increased exponentially with an increase in N application rate (Fig.4b-d, $P < 0.01$), while the response of NH_3 volatilization to N rates fitted the linear model best (Fig.4a, $P < 0.01$). The estimated total Nr losses for all treatments varied from 39.3 to 91.7 kg N ha^{-1} in the fertilization treatments (Table 5), accounting for 30.1–32.8% of N application rates. NH_3 volatilization dominated the NrI, with the proportion ranging from 53.5% to 57.4%, mainly because of the current fertilizer application method (soil surface broadcast) and high temperatures in the field (Zhao et al., 2012b; Li et al., 2014). N runoff was the second most important contributor (Table 5). Using ^{15}N micro-plots combined with three-year field measurements, Zhao et al. (2012b) reported that the total Nr losses from rice production in the TLR, under an N rate of 300 kg N ha^{-1}, were 98 kg N ha^{-1}, which is comparable with our estimation of 91.69 kg N ha^{-1} in the RN300 plot. Similarly, Xia and Yan (2011) estimated the Nr losses for life-cycle rice production in this region to be around 90 kg N ha^{-1}. The high proportion (30.1–32.8%) of the applied N fertilizer released as Nr from rice production in the TLR, highlights the need to adopt reasonable N managements to increase the plant N uptake and reduce Nr losses (Ju et al., 2009).

The NrI of rice production in different plots varied between 2.14 g N kg^{-1} (RN0) and 10.92 g N kg^{-1} (RN300), which increased significantly as the N fertilizer rate increased (Table 5). The NrI for rice production in the TLR was estimated to be 10.92 g N kg^{-1} (RN300), which is 68% higher than the national average value estimated by Chen et al. (2014), as a result of higher N fertilizer input in the TLR. Under the condition of straw incorporation, reducing N application rate by 20% pulled the NrI down to 8.42 g N kg^{-1} (RN240) (Table 5). Additional N reduction could further lower the NrI, but the rice yield would be compromised largely (Table 3). Previous studies have proven that direct incorporation of crop straw had insignificant effects on various Nr releases (Xia et al., 2014). Because the majority of N contented in the crop straw is not easily degraded by microorganisms in a short-term period, and can be stabilized in soil in a long-term period, rather than being released as various Nr (Huang et al., 2004; Xia et al., 2014). For instance, a meta-analysis, integrating 112 scientific assessments of the crop residue incorporation on the N_2O emissions, has reported that the practice exerted no statistically significant effect on the N_2O releases (Shan and Yan, 2013). Therefore, the effects of wheat straw incorporation on various Nr losses were considered as negligible in this study.

Extra attention should be paid to the interrelationship between the NrI and GHGI, which could provide hints for the mitigation purpose. For instance, N fertilizer production and application is an intermediate link between the NrI and GHGI (Chen et al., 2014). For the NrI, N fertilization promotes various Nr releases, exponentially or linearly (Fig.4), while N production and application made a secondary contribution to the GHGI (Table 4). Such interrelationships ought to be taken into account fully for any mitigation options pursued, in order to reduce the GHG emissions and Nr discharges from rice production simultaneously (Cui et al., 2013b; Cui et al., 2014).

3.5 Economic evaluations of GHG emissions and Nr releases and their mitigation potential

The total environmental costs associated with the GHG emissions and Nr releases varied from 1214 ¥ ha^{-1} for the RN0 to 2399 ¥ ha^{-1} for the RN300, which approximately accounted for 10.44–13.47% of the farmer's income and 27.05–32.47% of the input costs, respectively (Table 6). CH_4 emission and NH_3 volatilization were the dominant contributors to the total environmental costs, respectively (Table 4 and Fig.5). The total damage costs to environment accounted for 13.5% of farmer's income under the current rice production in the TLR (RN300). Cutting the N rate from 300 to 240 kg N ha^{-1} slightly improved the farmer's income by 3.64%, while further N reduction would reduce the economic return of farmer's (Table 6).

GHG and Nr releases from rice production in the TLR are expected to possess a large potential for mitigation, due to the current situation of direct straw incorporation and higher N fertilizer inputs. Compared to traditional practice, a reduction of N application rate from 300 to 240 kg N ha^{-1} could alleviate 12.52% for GHGI (Table 4), 22.94% for NrI (Table 5), and 15.76% for environmental costs (Table 6). Further reduction in GHG and Nr releases (especially for CH_4 emissions and NH_3 volatilization) is possible, with the implementation of knowledge-based managements (Chen et al., 2014; Nayak et al., 2015). For the mitigation of Nr releases, switching the N fertilizer application method from surface broadcast to deep incorporation could largely lower the NH_3 volatilization from paddy soils (Zhang et al., 2012; Li et al., 2014). Moreover, other optimum N managements, such as applying controlled-release fertilizers and urease inhibitors, could also effectively increase the NUE and reducing the overall Nr losses (Chen et al., 2014). For the mitigation of GHG emissions, rather than being directly incorporated before rice transplantation, crop residues should be preferentially decomposed under aerobic conditions or used to produce biochar through pyrolysis, which could effectively reduce CH_4 emissions (Linquist et al., 2012; Xie et al., 2013). Moreover, these pre-treatments are also beneficial for carbon sequestration and yield production (Woolf et al., 2010; Linquist et al., 2012).

Most previous studies have merely focused on the quantification of GHG and Nr releases from food production from the perspective of environment assessments (Zhao et al., 2012b; Ma et al., 2013; Zhao et al., 2015). The perspective of economic evaluation is seldom implemented, which goes against encouraging farmer to participate in the abatement of GHG and Nr releases on their own initiative (Xia et al., 2014). The current pattern of rice production in the TLR incurs great costs to the environment, which accounted for 13.47% of the net economic return that farmer ultimately acquire (Table 6). Such an evaluation facilitates the translation of highly specialized scientific conclusions into monetary-based information that is more familiar and accessible for farmers, and therefore likely encouraging them to adopt eco-friendly agricultural managements (Wang et al., 2014b). Profitability is generally considered the main driver for farmer to change their management approach. Compared to traditional N application rate, a reduction of 20% would make environmental costs savings of 14%, whilst simultaneously improving the economic return of farmer’s by 648 ¥ ha^{-1} (Table 6). This represents an incentive for farmers to optimize their N fertilizer application rates, provided that such information is available to them.

Considering the fact that no specific carbon- and Nr-mitigation incentive programs, like the ‘Carbon Farming Initiative’ in Australia (Lam et al., 2013), have been launched in China, an ecological compensation incentive mechanism should be established by governments. This should be a national subsidy program with a special compensation and award fund to cover the extra mitigation costs induced by the adoption of knowledge-based mitigation managements for farmers (Xia et al., 2016). Such a program would provide farmers with a tangible incentive, thus guiding them towards gradually adopting the mitigation managements, which could effectively curb GHG emissions and Nr losses, but likely exert little positive effects on improving their net economic return (Xia et al., 2014). Examples include the composing of crop straws aerobically, or their use to produce biochar before incorporation (Xie et al., 2013), and encouraging the application of deep placement of N fertilizer (Wang et al., 2014b), as well as the application of enhanced-efficiency N fertilizers during the rice-growing season (Akiyama et al., 2010).

4 Conclusions

Our results demonstrated that producing rice yield in the TLR released substantial GHG and Nr, which largely attributed to the current direct straw incorporation and excessive N fertilizer inputs. CH_4 emissions and NH_3 volatilization dominated the GHG and Nr releases, respectively. Reducing N application rate by 20% from the tradition level (300 kg N ha^{-1}) could effectively decrease the GHG emissions, Nr releases and the damage costs to the environment, while increased the rice yield and improved farmer’s income simultaneously. Agricultural managements, such as making straw decompose aerobically before its incorporation and optimizing the application method of N fertilizer, showed large potentials to further reduce the GHG (e.g., CH_4 emission) and Nr releases (e.g., NH_3 volatilization) from rice production in this region. Further studies are needed to evaluate the comprehensive effects of these managements on GHG emissions, Nr releases and farmer’s economic returns.

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Supplementary material

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

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Table 1. Field experimental treatments and agricultural management practices during the rice-growing seasons of 2013 and 2014 in the Taihu Lake region

Treatment[a]	RN0	RN120	RN180	RN240	RN300
Chemical fertilizer application rate ($N:P_2O_5:K_2O$, kg ha^{-1})	0:30:60	120:30:60	180:30:60	240:30:60	300:30:60
Split N application ratio	---	4:3:3	4:3:3	4:3:3	4:3:3
Straw application rate (Mg dry matter ha^{-1})	3.94/2.88[b]	4.49/4.65	4.93/5.18	5.33/5.87	5.81/6.17
Water regime[c]	F-D-F-M	F-D-F-M	F-D-F-M	F-D-F-M	F-D-F-M
Density (10^4 plants ha^{-1})	2.5	2.5	2.5	2.5	2.5

[a]RN0, RN120, RN180, RN240 and RN300 represent N application rates of 0, 120, 180, 240, 300 kg N ha^{-1}, respectively.

[b]3.94/2.88 denote that straw application rates during the rice-growing seasons of 2013 and 2014 are 3.94 and 2.88 Mg dry matter ha^{-1}, respectively.

[c]F, flooding; D, midseason drainage; M, moist but non-waterlogged by intermittent irrigation.

Table 2. Two-way ANOVA for the effects of fertilizer (F) application and year (Y) on CH_4 and N_2O emissions, and rice grain yields in rice paddies.

Factor	df	CH_4 (kg ha^{-1})			N_2O (kg N ha^{-1})			Yield (kg ha^{-1})		
		SS	F	P	SS	F	P	SS	F	P
F	4	8739	0.79	0.55	0.33	12.46	< 0.01	39297547	32.96	< 0.01
Y	1	4492	1.62	0.22	0.11	16.41	< 0.01	2810414	9.43	< 0.01
F×Y	4	2532	0.23	0.92	0.18	7.1	< 0.01	750639	0.63	0.65
Model	9	15763	0.63	0.77	0.62	10.52	< 0.01	42858600	15.97	< 0.01
Error	16	20			0.13			5962260		

Table 3. Rice yield and nitrogen use efficiency (NUE) for the two rice-growing seasons from 2013 to 2014 in the Taihu Lake region

Year	Treatment[a]	Yield (kg ha^{-1})	NUE (%)
2013	RN0	4829 ± 207	---
	RN120	7079 ± 645	23.40
	RN180	7655 ± 601	28.12
	RN240	8273 ± 569	33.61
	RN300	8029 ± 101	30.63
2014	RN0	5919 ± 131	---
	RN120	7598 ± 1077	23.86
	RN180	7768 ± 570	21.19
	RN240	8880 ± 435	35.54
	RN300	8761 ± 369	32.07
Two-year average	RN0	5374 ± 617d[b]	---
	RN120	7339 ± 843c	23.63
	RN180	7711 ± 527bc	24.66
	RN240	8576 ± 562a	34.58
	RN300	8395 ± 468ab	31.35

[a]Definitions of the treatment codes are given in the footnotes of Table 1.

[b]Mean±SD; different letters within the same column indicate a significant difference at p<0.05.

Table 4. The net global warming potential (NGWP) and greenhouse gas intensity (GHGI) for the two rice-growing seasons from 2013 to 2014 in the Taihu Lake region

Year	Treatment[a]	CH_4 emission	N_2O emission	SOCSR	Irrigation	N fertilizer production	Others	NGWP	GHGI
		kg CH_4 ha^{-1}	kg N ha^{-1}	kg C ha^{-1} yr^{-1}	kg CO_2 eq ha^{-1}				kg CO_2 eq kg^{-1}
2013	RN0	306.07 ± 41[b]	0.08 ± 0.01	129.58	1170	0	217	8601	1.78
	RN120	317.26 ± 92	0.10 ± 0.01	154.07	1170	996	265	9845	1.39
	RN180	287.8 ± 12	0.13 ± 0.01	171.54	1170	1494	277	9568	1.25
	RN240	273.27 ± 36	0.14 ± 0.06	185.50	1170	1992	291	9670	1.17
	RN300	305.13 ± 90	0.16 ± 0.03	196.87	1170	2490	285	10927	1.36
2014	RN0	307.22 ± 47	0.02 ± 0.05	129.58	1256	0	240	8711	1.47
	RN120	351.96 ± 28	0.09 ± 0.02	154.07	1256	996	276	10805	1.42

	RN180	291.25 ± 18	0.24 ± 0.04	171.54	1256	1494	280	9795	1.26
	RN240	317.65 ± 28	0.34 ± 0.12	185.50	1256	1992	303	10972	1.24
	RN300	343.8 ± 61	0.53 ± 0.21	196.87	1256	2490	301	12169	1.39
Two-year	RN0	306.65 ± 39a	0.05 ± 0.05b	129.58c	1213	0	229	8656	1.61 ± 0.25a
average	RN120	334.61± 64a	0.09 ± 0.02b	154.07bc	1213	996	271	10322	1.40 ± 0.16b
	RN180	289.53 ± 14a	0.18 ± 0.07ab	171.54ab	1213	1494	279	9679	1.25 ± 0.09bc
	RN240	295.46 ± 38a	0.24 ± 0.14ab	185.50ab	1213	1992	297	10321	1.20 ± 0.08cd
	RN300	324.47 ± 72a	0.35 ± 0.25a	196.87a	1213	2490	293	11550	1.38 ± 0.21bc

[a]Definitions of treatment codes are given in the footnotes of Table 1.

[b]Mean±SD; different letters within same column indicate a significant difference at $p<0.05$.

Table 5. The seasonal average reactive N (Nr) losses and reactive N intensity (NrI) for the two rice-growing seasons from 2013 to 2014 in the Taihu Lake region

Treatment[a]	NH_3 volatilization	N runoff	N leaching	N_2O emission	NO_X emission	Total Nr losses	NrI
	kg N ha^{-1}						g N kg^{-1}
RN0	0.64	5.39	1.44	0.07	3.96	11.50	2.14
RN120	21.04	10.30	2.24	0.12	5.62	39.32	5.36
RN180	31.24	14.25	2.80	0.21	6.44	54.93	7.12
RN240	41.44	19.70	3.50	0.27	7.26	72.17	8.42
RN300	51.64	27.24	4.37	0.38	8.07	91.69	10.92

[a]Definitions of treatment codes are given in the footnotes of Table 1.

Table 6. The economic indicators (two-season average) for rice production of the growing seasons from 2013 to 2014 in the Taihu Lake region (unit: ¥ ha^{-1})

Treatment[a]	Yield income[b]	Input costs[c]	Farmer's income[d]	Environmental costs[e]	
				GHG emissions	Nr releases
RN0	16125	4493	11632	1143	71
RN120	22020	6104	15916	1363	376
RN180	23130	6542	16588	1278	535
RN240	25725	7277	18448	1362	700
RN300	25185	7385	17800	1525	874

[a]Definitions of treatment codes are given in the footnotes of Table 1.

[b]Yield income = rice yield × rice price.

[c]Input costs denote the economic input of purchasing various agricultural materials and hiring labours.

[d]Farmer's income = Yield income – Input costs.

[e]Environmental costs denoted the sum of the acidification costs, eutrophication costs and global warming costs incurred by GHG emissions and Nr releases. The cost prices of GHG and Nr releases are as followed: GHG emission, 132 ¥ t^{-1} CO_2 eq (Xia et al., 2014); NH_3 volatilization, 13.12 ¥ kg^{-1} N; N leaching, 6.12 ¥ kg^{-1} N; N runoff, 3.64 ¥ kg^{-1} N; NO_X emission, 8.7 ¥ kg^{-1} N (Xia and Yan, 2011).

Figure captions

Fig. 1. Seasonal variations in the daily precipitation and the temperature during the two rice-growing seasons of (a) 2013 and (b) 2014.

Fig.2. Relationship between N fertilizer application rate and average rice yield over the two rice-growing seasons of 2013 and 2014 in the Taihu Lake region. The vertical bars represent standard errors.

Fig.3. Seasonal variations in (a) CH_4 and (b) N_2O fluxes during the two rice-growing seasons from 2013 to 2014 in the Taihu Lake region. The arrow indicates N fertilizer application. The vertical bars represent standard errors.

Fig.4. Relationship between N fertilizer application rate and (a) NH_3 volatilization, (b) N runoff, (c) N leaching and (d) N_2O emissions for rice production in the Taihu Lake region. These relationships were obtained through a meta-analysis.

Fig.5. Seasonal average total environmental costs incurred by greenhouse gas (GHG) emissions and reactive N (Nr) losses for rice production in Taihu Lake region.

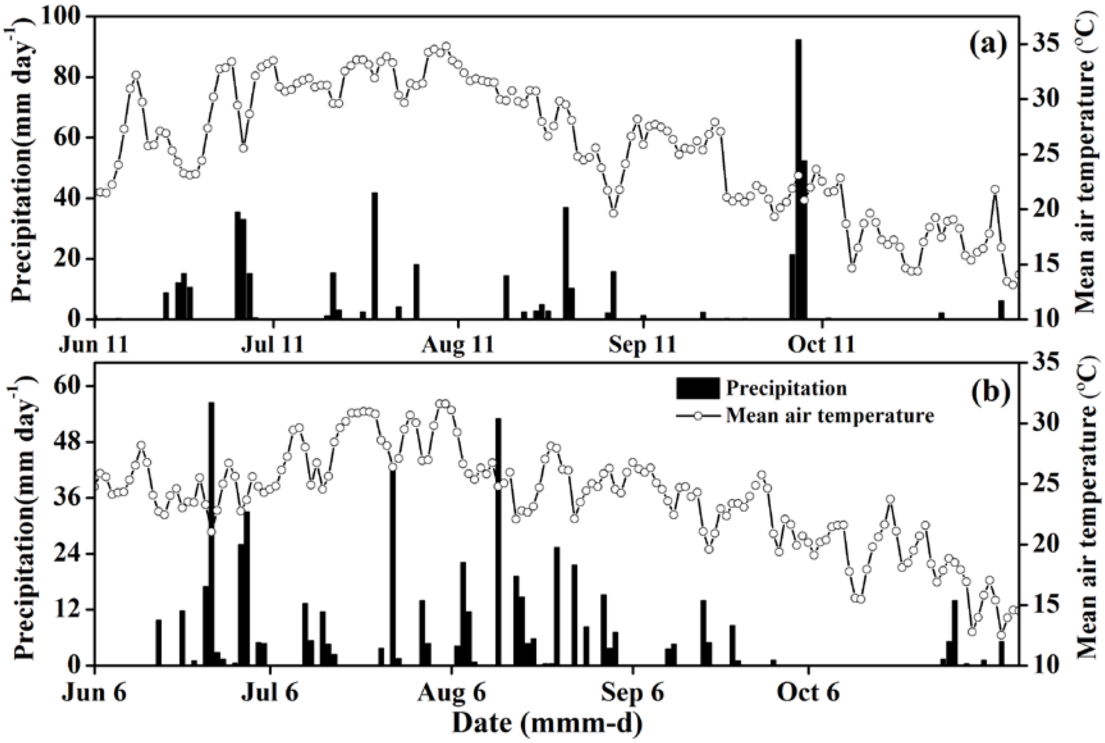

(a)
(b)
Precipitation(mm day^{-1})
Mean air temperature (°C)
Precipitation
Mean air temperature
Jun 11
Jul 11
Aug 11
Sep 11
Oct 11
Jun 6
Jul 6
Aug 6
Sep 6
Oct 6
Date (mmm-d)

Fig.1

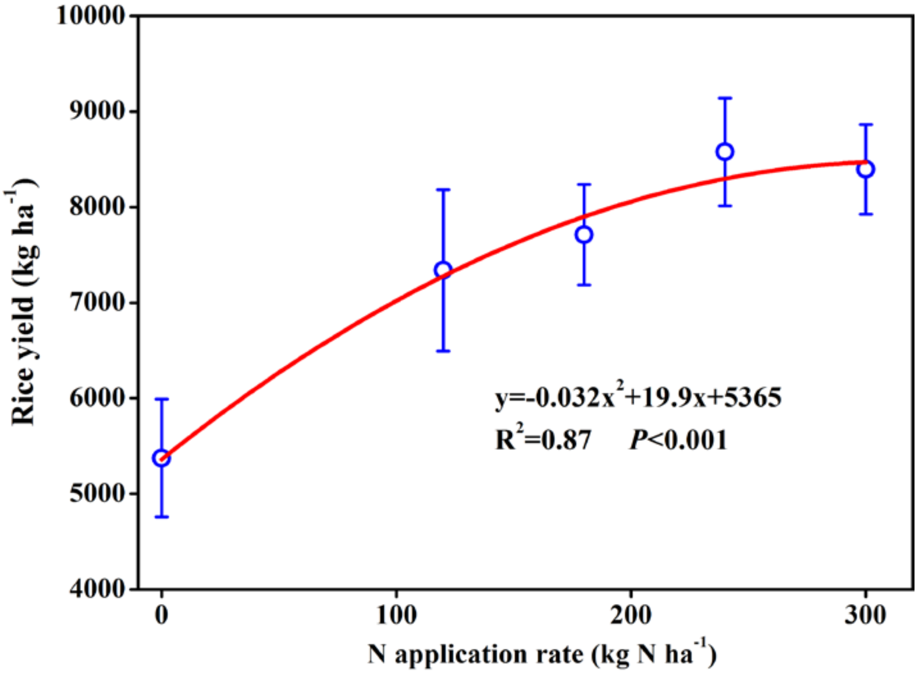

10000
9000
8000
7000
6000
5000
4000
Rice yield (kg ha-1)
0
100
200
300
N application rate (kg N ha-1)
y=-0.032x2+19.9x+5365
R2=0.87 P<0.001

Fig.2

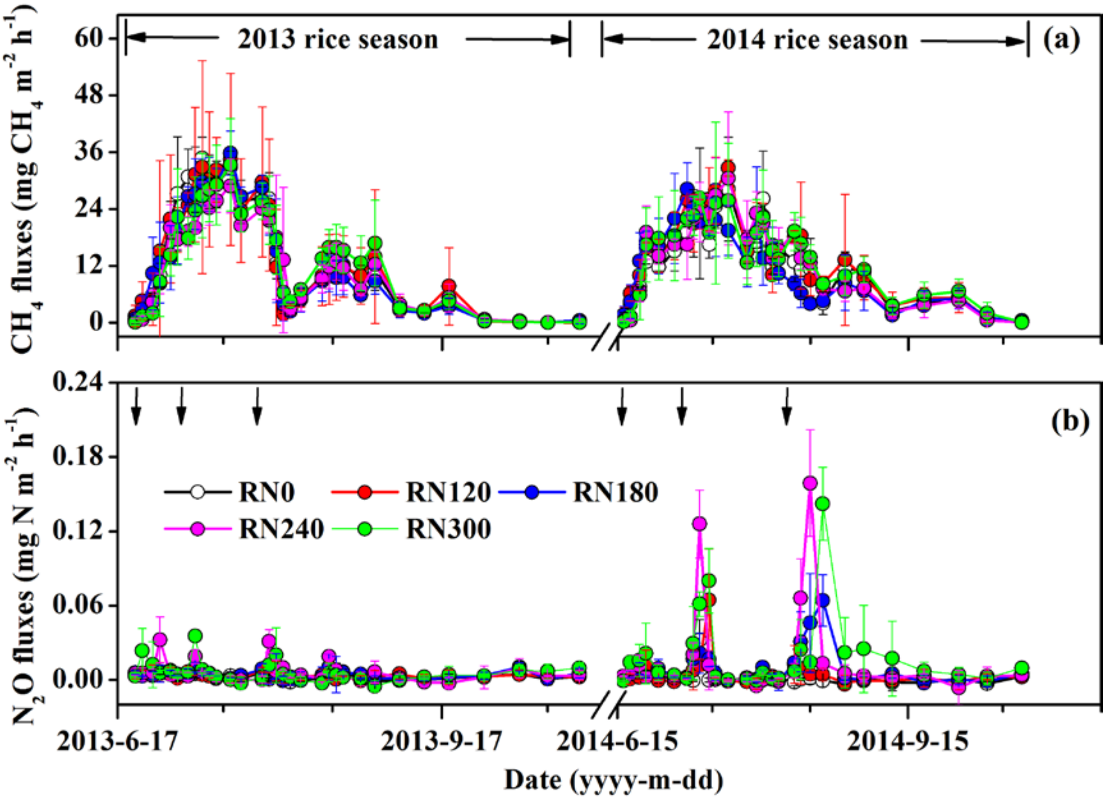

2013 rice season
2014 rice season
(a)
(b)
CH4 fluxes (mg CH4 m-2 h-1)
N2O fluxes (mg N m-2 h-1)
RN0
RN120
RN180
RN240
RN300
2013-6-17
2013-9-17
2014-6-15
2014-9-15
Date (yyyy-m-dd)

Fig.3

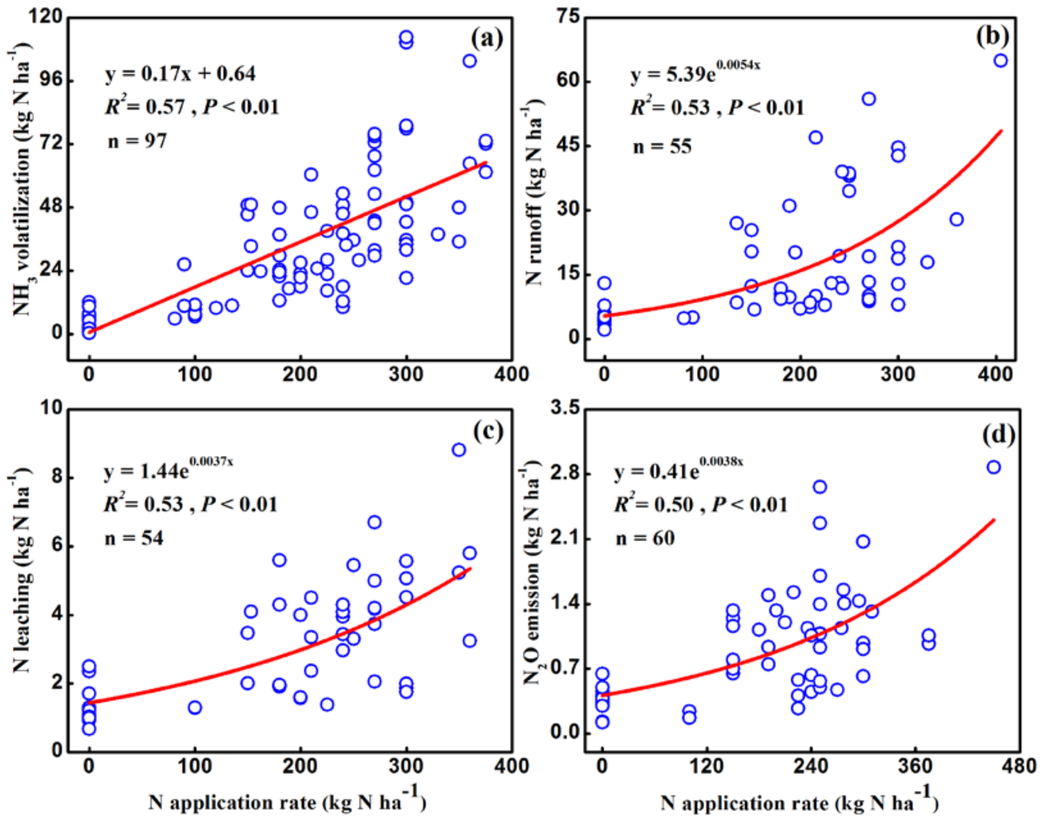
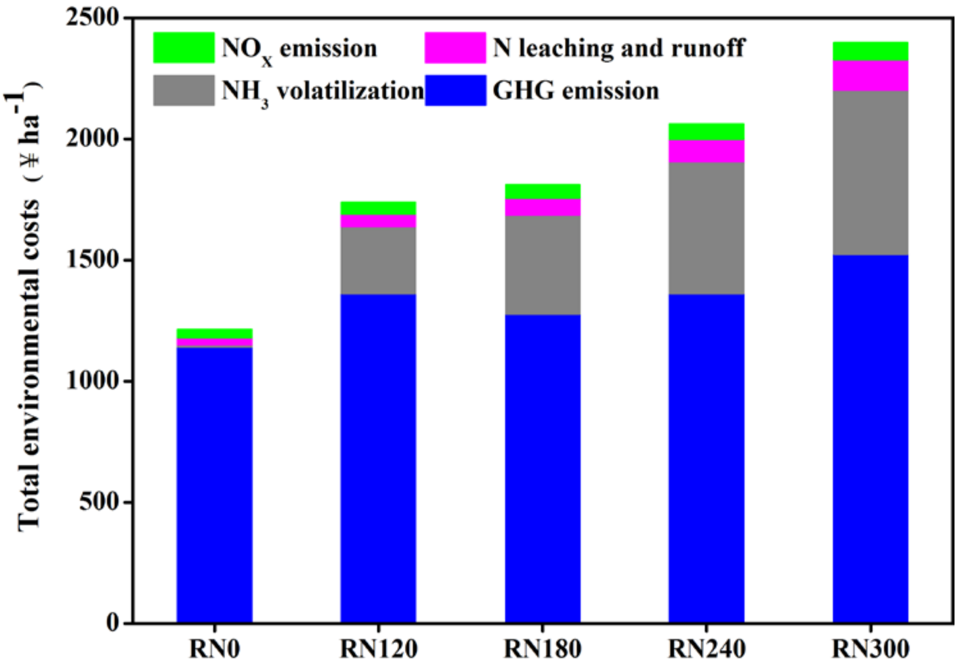

(a)
y = 0.17x + 0.64
R² = 0.57 , P < 0.01
n = 97
NH3 volatilization (kg N ha-1)
(b)
y = 5.39e0.0054x
R² = 0.53 , P < 0.01
n = 55
N runoff (kg N ha-1)
(c)
y = 1.44e0.0037x
R² = 0.53 , P < 0.01
n = 54
N leaching (kg N ha-1)
(d)
y = 0.41e0.0038x
R² = 0.50 , P < 0.01
n = 60
N2O emission (kg N ha-1)
N application rate (kg N ha-1)
N application rate (kg N ha-1)

Fig.4



NO$_X$ emission
N leaching and runoff
NH$_3$ volatilization
GHG emission
Total environmental costs（￥ ha^{-1}）
2500
2000
1500
1000
500
0
RN0
RN120
RN180
RN240
RN300

Fig.5