1 **Greenhouse gas emissions and reactive nitrogen releases from rice production with**

2 **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 GHGI with proportion of 70.2–88.6% due to direct straw incorporation, while ammonia (NH$_3$) volatilization dominated 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

45   suggests that GHG and Nr releases, especially CH_4 emission and NH_3 volatilization, from rice

46   production in the TLR could be further reduced, considering the current incorporation pattern of

47   wheat straw and N fertilizer.

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

49   incorporation

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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).

89   Besides such substantial releases of Nr and GHG in a direct way, indirect releases during the

90   production of various agricultural materials used for farming operations, are also not ignorable,

91   due to higher input rates of these materials caused by intensive cultivation (Zhang et al., 2013;

92   Cheng et al., 2014). This warrants the need for life-cycle assessment (LCA) of GHG emissions

93   and Nr releases with respect to rice production in this region.

94   Considerable environmental costs can be caused by the direct and indirect releases of GHG

95   and Nr from rice production in the TLR, for instance, in the form of global warming, water

96   eutrophication, or soil acidification (Ju et al., 2009; Xia and Yan, 2011; Xia and Yan, 2012).

97   Previous studies have proven that environmental costs assessment could provide guidance for

98   emerging policy priorities in mitigating certain GHG or Nr species, after quantifying both their

99   release amounts and damage costs to ecosystems (Gu et al., 2012). However, few studies have

100   attempted to evaluate the total GHG and Nr releases and the associated environmental costs from

101   rice production, with high inputs of N fertilizer and crop straw.

102   In the present study, we conducted two years of simultaneous measurements of CH_4 and

103   nitrous oxide (N_2O) emissions from a rice-wheat cropping system in the TLR to evaluate the

104   impacts of simultaneous inputs of crop straw and N fertilizer on (1) net global warming potential

105   (NGWP) and GHG intensity (GHGI), (2) total Nr losses and Nr intensity (NrI), (3) environmental

106   costs incurred by these GHG and Nr releases associated with rice production, from the perspective

107   of LCA.

108   **2 Materials and methods**

109   **2.1 Experimental site**

110   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 ℃ 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

133   evenly to the soil surface. According to local practices, the water regime of 'flooding-midseason

134   drainage-flooding-moist but non-waterlogged by intermittent irrigation' was adopted. Details of

135   the specific agricultural management practices for rice production are provided in Table 1.

136   **2.3 Gas fluxes and topsoil organic carbon sequestration rate**

137   The CH_4 and N_2O fluxes during the rice-growing seasons of 2013 and 2014 were measured

138   using a static chamber and gas chromatography technique. Details of the procedures used for

139   sampling and analysis the gases were described in Xia et al. (2014).

140   Generally, it takes long-term observations over years to decades before the soil organic

141   carbon (SOC) change is detectable (Yan et al., 2011). The SOC content changes of short-term field

142   experiment couldn't be correctly measured, due to the high variability of SOC during the

143   preliminary several years of the experiment. Therefore, we used the following relationship

144   between the straw input rate (kg C ha^{-1} yr^{-1}) and SOC sequestration rate (SOCSR, kg C ha^{-1} yr^{-1}),

145   obtained through an on-going long-term straw application experiment in the same region, to

146   calculate the SOCSR in this study (Xia et al., 2014):

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

148   This on-gonging long-term field experiment is also taking place at the Changshu

149   Agroecological Experimental Station (since 1990), which includes three straw application levels:

150   0, 4.5 t, and 9.0 t dry-weight ha^{-1} yr^{-1}. The equation (1) was established based on the results of

151   22-year observation (Xia et al., 2014). Same agricultural management practices were applied to

152   the on-going long-term experiment and the experiment of this study.

153   **2.4 Net global warming potential and greenhouse gas intensity**

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

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

156   equations:

157    $$NGWP = \sum_{i=1}^{m} AI_{ico_2} + CH_4 \times 25 + N_2O \times 44/28 \times 298 - SOCSR \times 44/12; \tag{2}$$

158    $$GHGI = NGWP/rice\ yield; \tag{3}$$

159   Here, AI_{ico_2} denotes the GHG emissions from the production and transportation of agricultural

160   inputs, which are calculated by multiplying their application rates by their individual GHG

161   emission factors, such as synthetic fertilizers, diesel oil, electricity and pesticides (Liang, 2009;

162   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

163   the CH_4 and N_2O emissions from rice production, and the SOC sequestration rate, respectively.

164   **2.5 Total Nr losses and Nr intensity**

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

166   following equations:

167    $$Total\ Nr\ losses = \sum_{i=1}^{m} AI_{iN_r} + (NH_3 + N_2O + N_{Leaching} + N_{Runoff})_{rice}; \tag{4}$$

168    $$NH_3\ volatilization = 0.17 \times N_{rate} + 0.64; \tag{5}$$

169    $$N\ runoff = 5.39 \times Exp\ (0.0054 \times N_{rate}); \tag{6}$$

170    $$N\ leaching = 1.44 \times Exp\ (0.0037 \times N_{rate}); \tag{7}$$

171    $$NrI = (1000 \times Total\ Nr\ losses)\ /rice\ yield\ ; \tag{8}$$

172   Here, AI_{iNr} denotes the Nr lost (mainly through N_2O and NO_X emissions) from the production and

173   transportation of agricultural inputs (Liang, 2009; Zhang et al., 2013), while

174   '$(NH_3+N_2O+N_{Leaching}+N_{Runoff})_{rice}$' represents the NH_3 volatilization, N_2O emissions, N leaching

175   and runoff during the rice-growing season. N_{rate} represents the N fertilizer application rate. Nr

176   empirical models (Equation 5, 6, 7) derived from a meta-analysis of published literature

177   concerning Nr losses from rice production in the TLR. Specific details regarding this literature

178   survey are provided in Appendix A.

179   **2.6 Total environmental costs incurred by GHG and Nr releases and farmer's income**

180   The total environmental costs ($¥ha^{-1}$) incurred by GHG and Nr releases and farmer's income

181   from rice production in the TLR were calculated based on the following equations:

182   Environmental costs $= \sum_{i=1}^{n}(Nr_iA \times DC_i) + CO_2A \times DC_{CO2};$ (9)

183   Farmer's income $=$ rice yield $\times$ rice price $-$ input costs;   (10)

184   Nr_iA (kg N) represents the release amounts of certain Nr species (i), and DC_i ($¥kg^{-1}$ N) denotes

185   the damage cost (DC) per kg of certain Nr (i). CO_2A (ton) and DC_{CO2} ($¥ton^{-1}$) represent the CO_2

186   emissions amount and global warming cost of CO_2, respectively. N_2O is both a GHG and Nr

187   species, but its environmental cost was calculated as a GHG here. Because the cost of N_2O

188   emission as Nr species is to damage human health (Gu et al., 2012), but the effects of Nr losses on

189   the damage costs of human health were not included in this study. The environmental costs mainly

190   refer to the global warming incurred by GHG emissions, soil acidification incurred by NH_3 and

191   NO_X emissions, and aquatic eutrophication caused by NH_3 emissions, N leaching and runoff (Xia

192   and Yan, 2012).

193   **2.7 Nitrogen use efficiency and N_2O emission factor**

194   Nitrogen use efficiency (NUE) and N_2O emission factor ($EF_d\%$) were respectively

195   calculated by the following equations (Ma et al., 2013; Yan et al., 2014):

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

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

198   Here, U_N is the plant N uptake (kg ha^{-1}) measured in aboveground biomass at physiological

199   maturity in the N fertilization treatments, while U_0 is the N uptake measured in aboveground

200   biomass in the treatment without N fertilizer addition (RN0). E_N denotes the cumulative N_2O

201   emissions in the N fertilization treatments, while E_0 denotes the N_2O emissions in the RN0. F_N

202   represents the application rate of N fertilizer. The N uptake in straw and grain was analysed via

203   concentrated sulfuric acid digestion and the Kjeldahl method (Zhao et al., 2015).

204   **2.8 Statistical analysis**

205   Differences in seasonal CH_4, N_2O emissions and rice yield of the two rice-growing seasons

206   from 2013 to 2014 affected by fertilizer treatments, year and their interaction were examined by

207   using a two-way analysis of variance (ANOVA) (Table 2). The grain yield, seasonal CH_4 and N_2O

208   emissions, SOCSR and GHGI of different treatments were tested by analysis of variance, and

209   mean values were compared by least significant difference (LSD) at the 5% level. All these

210   analyses were carried out using the SPSS (Version 19.0, USA).

211   **3 Results and discussion**

212   **3.1 Rice yield and NUE**

213   The two-way ANOVA analyses indicated that the rice grain yields were significantly affected

214   by the year and fertilizer treatment (Table 2). The farmer's practice plot (RN300) had an average

215   rice grain yield of 8395 kg ha^{-1}, with an NUE of 31.35%, over the two growing seasons from

216   2013 to 2014. Compared with RN300, reducing the N fertilizer rate by 20% (RN240) slightly

217   improved the grain yield and NUE to 8576 kg ha^{-1} and 34.58%, respectively. Further N reduction,

218   without additional agricultural managements, could decrease the rice yield by 8.15% (RN180) and

219   15.18% (RN120) (Table 3). The response of rice yield to the synthetic N application rate in our

220   study successfully fitted a quadratic model (Fig.2), as has been reported in previous studies (Xia

221   and Yan, 2012; Cui et al., 2013a). Reducing N application reasonably, therefore, is considered

222   essential to reduce environmental costs, without sacrificing grain yield (Chen et al., 2014). Our

223   study showed that lowering the N input adopted by local farmer (300 kg N ha^{-1}) by 20% could

224   still enhance the grain yield and NUE. However, a further reduction of N 40% (RN180) would

225   largely undermine the rice yield (Table 3).

226   Further reduction in N fertilizer may be achieved with improvements of agricultural

227   managements, Ju et al. (2009) reported that, based on knowledge-based N managements, such as

228   optimizing N fertilizer source, rate, timing and place (in accordance with crop demand), rice grain

229   yield in the TLR was not significantly affected by a 30–60% N saving, while various Nr losses

230   would endure a two-fold curbing. Similarly, Zhao et al. (2015) found that the NUE could be

231   improved from 31% to 44%, even under a N reduction of 25% for rice production in the TLR,

232   through the implementation of integrated soil-crop system managements. In the present study, the

233   NUE was improved by 10% via a 20% N reduction, but it still falls behind the NUE values in the

234   studies which received knowledge-based N managements. Previous studies have proven that straw

235   incorporation exerted little impacts on grain yield. For instance, a meta-analysis conducted by

236   Singh et al. (2005) have found that incorporation of crop straw produced no significant trend in

237   improving crop yield in rice-based cropping systems. Moreover, based on a long-term straw

238   incorporation experiment established since 1990 in the TLR, Xia et al. (2014) have reported that

239   long-term incorporation of wheat straw only increased the rice yield by 1%. Therefore, in the

240   present study, the effects of straw incorporation on rice yield were considered as inappreciable.

241   **3.2 CH$_4$, N$_2$O emissions and SOSCR**

242   Over the two rice-growing seasons from 2013 to 2014, all treatments showed similar patterns

243   of CH$_4$ fluxes, albeit with large inter-annual variation (Fig.3a). The seasonal average CH$_4$

244   emissions from all plots showed no significant difference, ranging from 289.53 kg CH$_4$ ha^{-1} in the

245   RN180 plot to 334.61 kg CH$_4$ ha^{-1} in the RN120 plot (Table 4), much higher than observations

246   conducted in the same region (Zou et al., 2005; Ma et al., 2013). This phenomenon can be

247   attributed to the larger amounts of straw incorporation in this study (Table 1). Relative to the

248   RN300 plot, CH$_4$ emissions from the RN240 plot decreased by 8% and 10%, during the

249   rice-growing season of 2013 and 2014, respectively, although this effect was not statistically

250   significant (Table 4).

251   Many studies have shown a clear linear relationship between CH$_4$ emissions and the amounts

252   of applied organic matter (OM) (Shang et al., 2011; Xia et al., 2014). It is possible that the linear

253   response of CH$_4$ emissions to OM inputs can become flat or even unobvious (Fig.S1), when the

254   OM applied rates among different treatments were insignificant different (Table S1). It is

255   unsurprising that no obvious relationship between CH$_4$ emissions and N fertilizer application rates

256   was observed in this study (Fig.S1), because the effects of N fertilization on CH$_4$ production,

257   transportation and oxidation are complex. For instance, N fertilization can provide methanogens

258   with more carbon substrates in the rhizosphere of plants by stimulating the growth of rice biomass,

259   thus promoting CH$_4$ production and transportation (Zou et al., 2005; Banger et al., 2012). On the

260   other side, N enrichment could also enhance the activities of methanotrophs, therefore enhancing

261   CH$_4$ oxidation (Xie et al., 2010; Yao et al., 2012).

262   The N$_2$O fluxes were sporadic and pulse-like, and these fluxes showed large variations

263   between different seasons, and the majority of the N$_2$O peaks occurred after the application of N

264   fertilizer (Fig.3b). The two-way ANOVA analyses indicated that the seasonal N$_2$O emissions were

265 significantly affected by the year, the fertilizer treatment, and their interactions during the

266 rice-growing seasons (Table 2). The average N_2O emission, during the two rice-growing seasons,

267 ranged from 0.05 kg N ha^{-1} for the RN0 to 0.35 kg N ha^{-1} for the RN300 (Table 4), which

268 increased exponentially as the N fertilizer rate increased; this highlights that the reduction of N

269 fertilizer rate is an effective approach to reduce the N_2O emissions (Zou et al., 2005; Zhang et al.,

270 2012). The average N_2O emission factors varied between 0.03% and 0.1%, with an average of

271 0.07%, which is comparable with previous studies (0.05–0.1%) conducted in the same region (Ma

272 et al., 2013; Zhao et al., 2015).

273 The rice paddies have witnessed an increase in the SOC stock as a result of straw

274 incorporation (Table 4). The estimated topsoil (0–20cm) SOCSR varied from 0.13 t C ha^{-1} yr^{-1} for

275 the RN0 plot to 0.197 t C ha^{-1} yr^{-1} for the RN300 plot. The empirical model established through a

276 long-term straw incorporation study in the same region was employed to evaluate the SOCSR in

277 this study, which likely brought uncertainty into the results of this study. Under the same

278 agricultural managements, soil and climatic conditions, cropping systems and straw types, it is

279 reasonable to believe that the rates of straw C stabilizing into SOC (i.e. conversion efficiency of

280 crop residue C into SOC) are similar between these two experiments (Mandal et al., 2008). It is

281 reported that the conversion rates of crop straw to SOC in two main wheat/maize production

282 regions in China, which have similar climatic conditions and agricultural practices, were very

283 close, at 40.524 versus 40.607 kg SOC-C t^{-1} dry-weight straw (Lu et al., 2009). Moreover, the

284 current estimated SOCSR for rice production in the TLR (0.197 t C ha^{-1}), is comparable to the

285 estimation of 0.17 t C ha^{-1} yr^{-1} from Ma et al. (2013) in a study based on a paddy field experiment

286 with OM incorporation in the same region. Therefore, we hold the opinion that the above SOCSR

287 calculation method is appropriate, and the uncertainty incurred by this method unlikely affects the

288 main conclusions of this study.

289   The magnitude of the SOC increase is variable depending on the straw incorporation method,

290 the degree of tillage, the cropping systems and etc. (Yan et al., 2011; Huang et al., 2013). Liu et al.

291 (2014) suggested that straw incorporation in rice-based cropping systems requires an overall

292 consideration, due to the direct incorporation promoting substantial CH_4 emissions. When

293 converting to CO_2 eq, the SOCSR only offsets the CH_4 emissions by 6.2–9.2% in this study (Table

294 4). This proportion is expected to increase provided that appropriate straw incorporation method

295 (e.g., compost straw before incorporation) and conservative-tillage are adopted. Moreover,

296 previous studies have shown that the combined adoption of conservative-tillage system with straw

297 return had large advantages in increasing SOC stocks while reducing CH_4 emissions (Zhao et al.,

298 2015a; Zhao et al., 2015b).

299 **3.3 NGWP and GHGI**

300   The average NGWP for all treatments varied from 8656 to 11550 kg CO_2 eq ha^{-1} (Table 4).

301 CH_4 emissions dominated the NGWP in all treatments, with the proportion ranging from 70.23%

302 to 88.56%, while synthetic N fertilizer production was the secondary contributor (Table 4). In

303 addition, SOC sequestration offset the positive GWP by 5.18–6.18% in the fertilization treatments.

304 Compared to conventional practice (RN300), the NGWP in the 20% reduction N practice (RN240)

305 decreased by 10.64%. Therein, 6.28% came from CH_4 reduction and 4.31% from N production

306 savings (Table 4). The GHGI of rice production ranged from 1.20 (RN240) to 1.61 (RN0) kg CO_2

307 eq kg^{-1}, which is higher than previous estimation of 0.24–0.74 kg CO_2 eq kg^{-1} for rice production

308 in other rice-upland crop rotation systems (Qin et al., 2010; Ma et al., 2013). Moreover, the GHGI

309   of current rice production in the TLR (RW300) was estimated to be 1.45 times that of the national

310   average value estimated by Wang et al. (2014a), at 1.38 versus 0.95 kg CO_2 eq kg^{-1}.

311   Such phenomenon was attributed to the following reasons. First, compared to above studies,

312   current higher amounts of direct straw incorporation (2.9–6.2 Mg dry matter ha^{-1}), before rice

313   transplantation in the TLR, triggered substantial CH_4 emissions (290–335 kg CH_4 ha^{-1}). Crop

314   residue incorporation is regarded as a win-win strategy to benefit food security and mitigate

315   climate change, due to the fact that it possesses a large potential for carbon sequestration (Lu et al.,

316   2009). However, the GWP of straw-induced CH_4 emissions was reported to be 3.2–3.9 times that

317   of the straw-induced SOCSR, which indicates direct straw incorporation in paddy soils worsens

318   rather than mitigates climate changes, in terms of GWP (Xia et al., 2014). The SOC sequestration

319   induced by straw incorporation only offset the positive GWP by 5.2–6.2% in this study. Sensible

320   methods of straw incorporation should therefore be developed to reduce the substantial CH_4

321   emissions without compromising the build-up of SOC stock in the TLR.

322   Second, the high N application rate (300kg N ha^{-1}) in the TLR combined with the large

323   emission factor of N fertilizer production, 8.3 kg CO_2 eq kg^{-1} N (Zhang et al., 2013), marked the

324   sector of N fertilizer production as the secondary contributor to the GHGI (Table 4); this sector,

325   however, was not involved in above-mentioned studies. Compared to local farmer's practices

326   (RN300), reducing the N rate by 20% (RN240) lowered the GHGI by 13%, under the condition of

327   straw incorporation, although this effect was not statistically significant (Table 4). Compared to

328   RN240, however, further reduction of N rate (RN180 or RN120) increased the GHGI, due to the

329   fact that rice yield was considerably reduced under excessive N reduction. Therefore, the joint

330   application of reasonable N reduction and judicious method of straw incorporation would be

331 promising in reducing the GHGI for rice production in the TLR, in consideration of the current

332 situation of simultaneous high inputs of N fertilizer and wheat straw.

333 **3.4 Various Nr losses and NrI**

334   The results of the meta-analysis indicated that N_2O emissions, as well as N leaching and

335 runoff, increased exponentially with an increase in N application rate (Fig.4b-d, $P < 0.01$), while

336 the response of NH_3 volatilization to N rates fitted the linear model best (Fig.4a, $P < 0.01$). The

337 estimated total Nr losses for all treatments varied from 39.3 to 91.7 kg N ha^{-1} in the fertilization

338 treatments (Table 5), accounting for 30.1–32.8% of N application rates. NH_3 volatilization

339 dominated the NrI, with the proportion ranging from 53.5% to 57.4%, mainly because of the

340 current fertilizer application method (soil surface broadcast) and high temperatures in the field

341 (Zhao et al., 2012b; Li et al., 2014). N runoff was the second most important contributor (Table 5).

342 Using ^{15}N micro-plots combined with three-year field measurements, Zhao et al. (2012b) reported

343 that the total Nr losses from rice production in the TLR, under an N rate of 300 kg N ha^{-1}, were 98

344 kg N ha^{-1}, which is comparable with our estimation of 91.69 kg N ha^{-1} in the RN300 plot.

345 Similarly, Xia and Yan (2011) estimated the Nr losses for life-cycle rice production in this region

346 to be around 90 kg N ha^{-1}. The high proportion (30.1–32.8%) of the applied N fertilizer released

347 as Nr from rice production in the TLR, highlights the need to adopt reasonable N managements to

348 increase the plant N uptake and reduce Nr losses (Ju et al., 2009).

349   The NrI of rice production in different plots varied between 2.14 g N kg^{-1} (RN0) and 10.92 g

350 N kg^{-1} (RN300), which increased significantly as the N fertilizer rate increased (Table 5). The NrI

351 for rice production in the TLR was estimated to be 10.92 g N kg^{-1} (RN300), which is 68% higher

352 than the national average value estimated by Chen et al. (2014), as a result of higher N fertilizer

353   input in the TLR. Under the condition of straw incorporation, reducing N application rate by 20%

354   pulled the NrI down to 8.42 g N kg^{-1} (RN240) (Table 5). Additional N reduction could further

355   lower the NrI, but the rice yield would be compromised largely (Table 3). Previous studies have

356   proven that direct incorporation of crop straw had insignificant effects on various Nr releases (Xia

357   et al., 2014). Because the majority of N contented in the crop straw is not easily degraded by

358   microorganisms in a short-term period, and can be stabilized in soil in a long-term period, rather

359   than being released as various Nr (Huang et al., 2004; Xia et al., 2014). For instance, a

360   meta-analysis, integrating 112 scientific assessments of the crop residue incorporation on the N_2O

361   emissions, has reported that the practice exerted no statistically significant effect on the N_2O

362   releases (Shan and Yan, 2013). Therefore, the effects of wheat straw incorporation on various Nr

363   losses were considered as negligible in this study.

364   Extra attention should be paid to the interrelationship between the NrI and GHGI, which

365   could provide hints for the mitigation purpose. For instance, N fertilizer production and

366   application is an intermediate link between the NrI and GHGI (Chen et al., 2014). For the NrI, N

367   fertilization promotes various Nr releases, exponentially or linearly (Fig.4), while N production

368   and application made a secondary contribution to the GHGI (Table 4). Such interrelationships

369   ought to be taken into account fully for any mitigation options pursued, in order to reduce the

370   GHG emissions and Nr discharges from rice production simultaneously (Cui et al., 2013b; Cui et

371   al., 2014).

372   **3.5 Economic evaluations of GHG emissions and Nr releases and their mitigation**

373   **potential**

374   The total environmental costs associated with the GHG emissions and Nr releases varied

375   from 1214 ¥ha^{-1} for the RN0 to 2399 ¥ha^{-1} for the RN300, which approximately accounted for

376   10.44–13.47% of the farmer's income and 27.05–32.47% of the input costs, respectively (Table 6).

377   CH_4 emission and NH_3 volatilization were the dominant contributors to the total environmental

378   costs, respectively (Table 4 and Fig.5). The total damage costs to environment accounted for 13.5%

379   of farmer's income under the current rice production in the TLR (RN300). Cutting the N rate from

380   300 to 240 kg N ha^{-1} slightly improved the farmer's income by 3.64%, while further N reduction

381   would reduce the economic return of farmer's (Table 6).

382   GHG and Nr releases from rice production in the TLR are expected to possess a large

383   potential for mitigation, due to the current situation of direct straw incorporation and higher N

384   fertilizer inputs. Compared to traditional practice, a reduction of N application rate from 300 to

385   240 kg N ha^{-1} could alleviate 12.52% for GHGI (Table 4), 22.94% for NrI (Table 5), and 15.76%

386   for environmental costs (Table 6). Further reduction in GHG and Nr releases (especially for CH_4

387   emissions and NH_3 volatilization) is possible, with the implementation of knowledge-based

388   managements (Chen et al., 2014; Nayak et al., 2015). For the mitigation of Nr releases, switching

389   the N fertilizer application method from surface broadcast to deep incorporation could largely

390   lower the NH_3 volatilization from paddy soils (Zhang et al., 2012; Li et al., 2014). Moreover, other

391   optimum N managements, such as applying controlled-release fertilizers and urease inhibitors,

392   could also effectively increase the NUE and reducing the overall Nr losses (Chen et al., 2014). For

393   the mitigation of GHG emissions, rather than being directly incorporated before rice

394   transplantation, crop residues should be preferentially decomposed under aerobic conditions or

395   used to produce biochar through pyrolysis, which could effectively reduce CH_4 emissions

396   (Linquist et al., 2012; Xie et al., 2013). Moreover, these pre-treatments are also beneficial for

397 carbon sequestration and yield production (Woolf et al., 2010; Linquist et al., 2012).

398 Most previous studies have merely focused on the quantification of GHG and Nr releases

399 from food production from the perspective of environment assessments (Zhao et al., 2012b; Ma et

400 al., 2013; Zhao et al., 2015). The perspective of economic evaluation is seldom implemented,

401 which goes against encouraging farmer to participate in the abatement of GHG and Nr releases on

402 their own initiative (Xia et al., 2014). The current pattern of rice production in the TLR incurs

403 great costs to the environment, which accounted for 13.47% of the net economic return that farmer

404 ultimately acquire (Table 6). Such an evaluation facilitates the translation of highly specialized

405 scientific conclusions into monetary-based information that is more familiar and accessible for

406 farmers, and therefore likely encouraging them to adopt eco-friendly agricultural managements

407 (Wang et al., 2014b). Profitability is generally considered the main driver for farmer to change

408 their management approach. Compared to traditional N application rate, a reduction of 20% would

409 make environmental costs savings of 14%, whilst simultaneously improving the economic return

410 of farmer's by 648 ¥ha^{-1} (Table 6). This represents an incentive for farmers to optimize their N

411 fertilizer application rates, provided that such information is available to them.

412 Considering the fact that no specific carbon- and Nr-mitigation incentive programs, like the

413 'Carbon Farming Initiative' in Australia (Lam et al., 2013), have been launched in China, an

414 ecological compensation incentive mechanism should be established by governments. This should

415 be a national subsidy program with a special compensation and award fund to cover the extra

416 mitigation costs induced by the adoption of knowledge-based mitigation managements for farmers

417 (Xia et al., 2016). Such a program would provide farmers with a tangible incentive, thus guiding

418 them towards gradually adopting the mitigation managements, which could effectively curb GHG

419 emissions and Nr losses, but likely exert little positive effects on improving their net economic

420 return (Xia et al., 2014). Examples include the composing of crop straws aerobically, or their use

421 to produce biochar before incorporation (Xie et al., 2013), and encouraging the application of deep

422 placement of N fertilizer (Wang et al., 2014b), as well as the application of enhanced-efficiency N

423 fertilizers during the rice-growing season (Akiyama et al., 2010).

424 **4 Conclusions**

425   Our results demonstrated that producing rice yield in the TLR released substantial GHG and

426 Nr, which largely attributed to the current direct straw incorporation and excessive N fertilizer

427 inputs. CH_4 emissions and NH_3 volatilization dominated the GHG and Nr releases, respectively.

428 Reducing N application rate by 20% from the tradition level (300 kg N ha^{-1}) could effectively

429 decrease the GHG emissions, Nr releases and the damage costs to the environment, while

430 increased the rice yield and improved farmer's income simultaneously. Agricultural managements,

431 such as making straw decompose aerobically before its incorporation and optimizing the

432 application method of N fertilizer, showed large potentials to further reduce the GHG (e.g., CH_4

433 emission) and Nr releases (e.g., NH_3 volatilization) from rice production in this region. Further

434 studies are needed to evaluate the comprehensive effects of these managements on GHG

435 emissions, Nr releases and farmer's economic returns.

441 **Supplementary material**

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

443 version.

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595 **Table 1.** Field experimental treatments and agricultural management practices during the

596 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

597 [a]RN0, RN120, RN180, RN240 and RN300 represent N application rates of 0, 120, 180, 240, 300

598 kg N ha^{-1}, respectively.

599 [b]3.94/2.88 denote that straw application rates during the rice-growing seasons of 2013 and 2014

600 are 3.94 and 2.88 Mg dry matter ha^{-1}, respectively.

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

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608 **Table 2.** Two-way ANOVA for the effects of fertilizer (F) application and year (Y) on CH_4 and

609 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		

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622 **Table 3.** Rice yield and nitrogen use efficiency (NUE) for the two rice-growing seasons from 2013

623 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

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

625 [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$_2$O 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 average	RN0	306.65 ±39a	0.05 ±0.05b	129.58c	1213	0	229	8656	1.61 ±0.25a
	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.
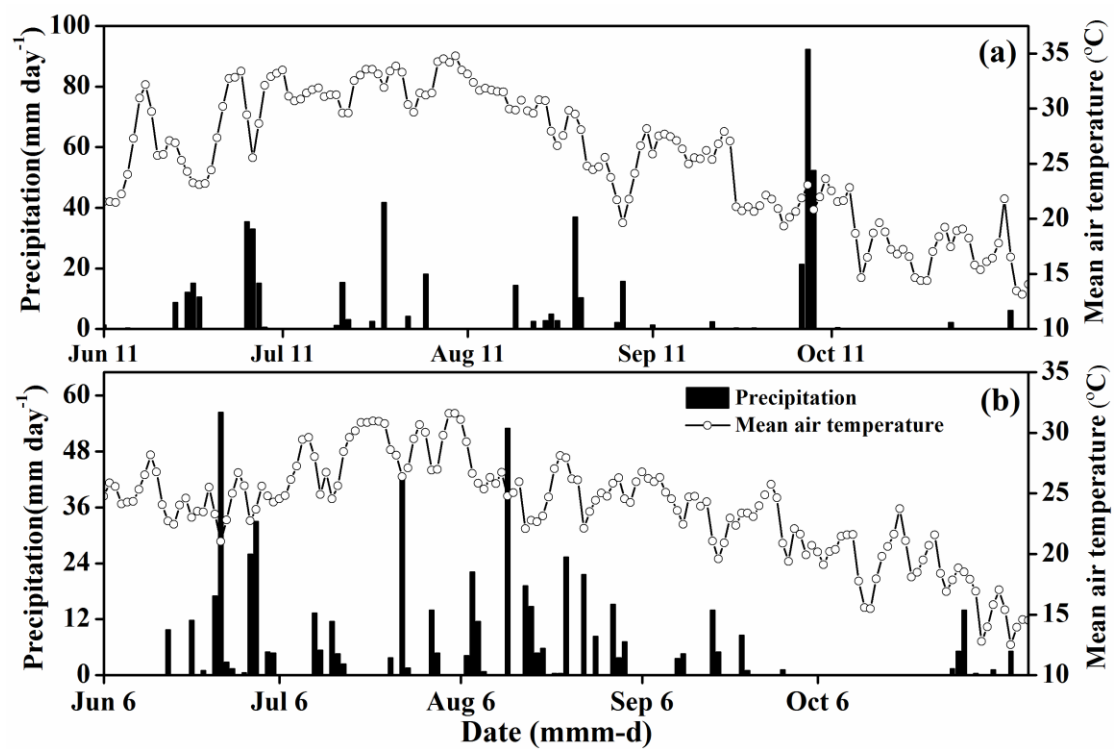
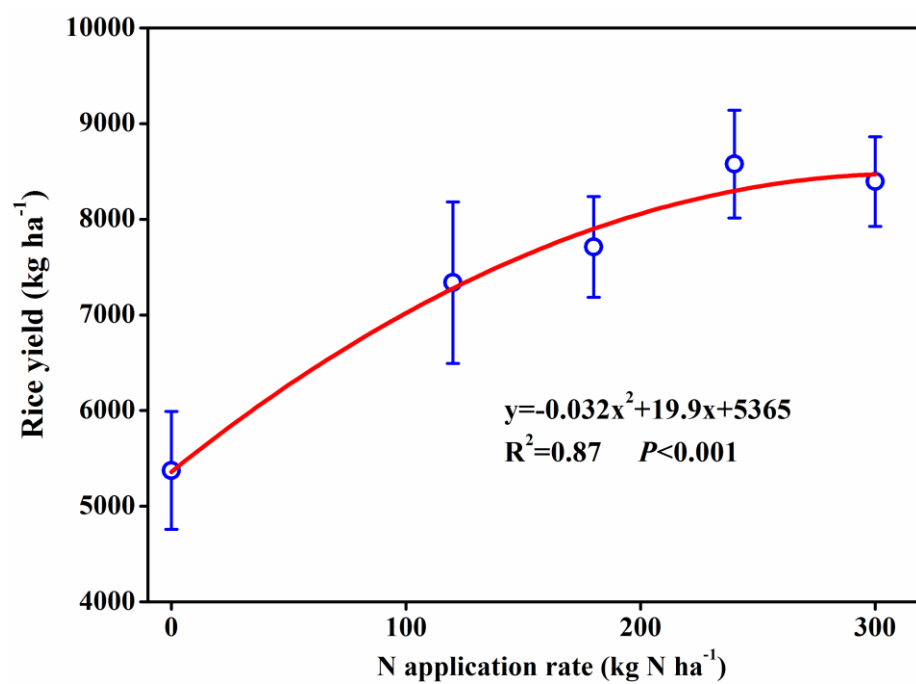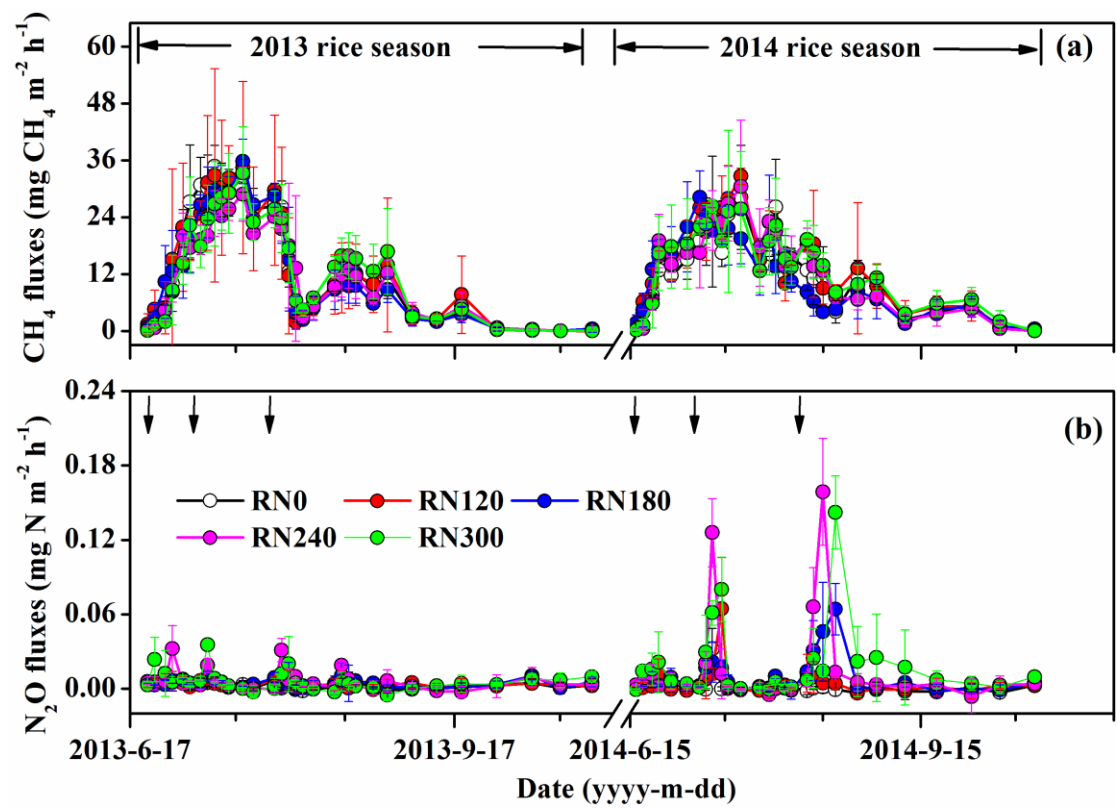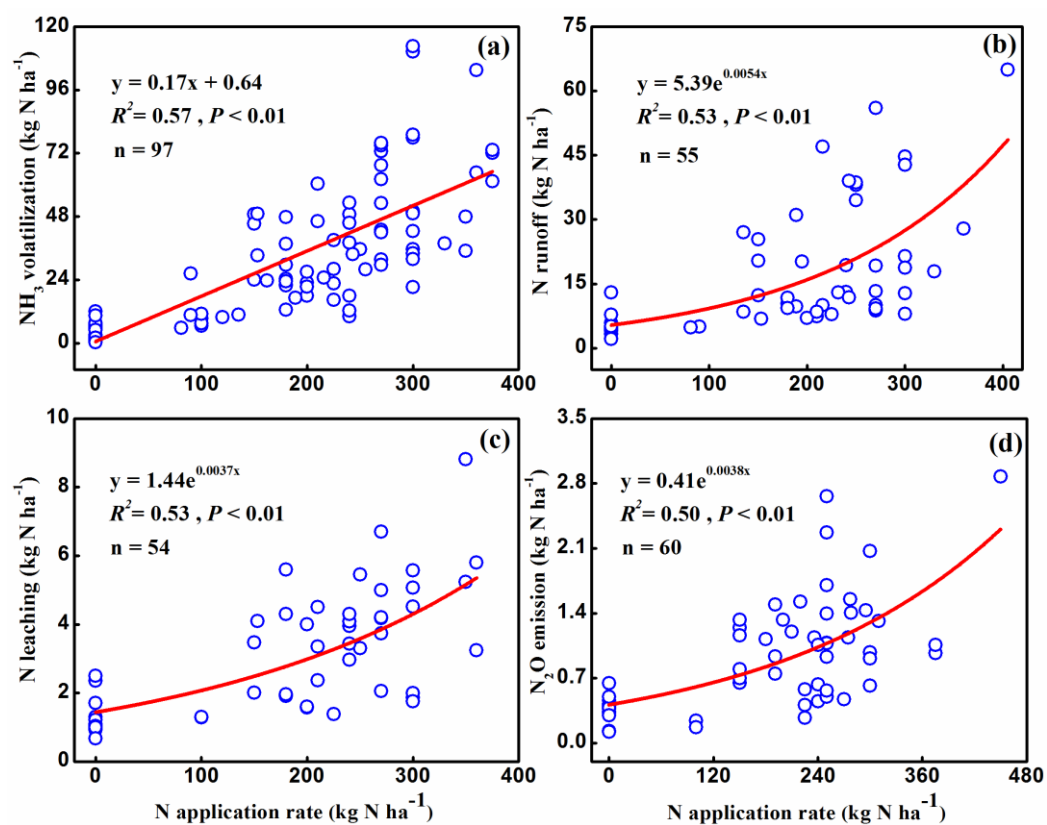**Fig.1**

Fig.2

Fig.3

Fig.4

Fig.5