LETTER

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Producing more grain with lower environmental costs

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Agriculture faces great challenges to ensure global food security by increasing yields while reducing environmental costs[1,2]. Here we address this challenge by conducting a total of 153 site-year field experiments covering the main agro-ecological areas for rice, wheat and maize production in China. A set of integrated soil–crop system management practices based on a modern understanding of crop ecophysiology and soil biogeochemistry increases average yields for rice, wheat and maize from 7.2 million grams per hectare (Mg ha^{-1}), 7.2 Mg ha^{-1} and 10.5 Mg ha^{-1} to 8.5 Mg ha^{-1}, 8.9 Mg ha^{-1} and 14.2 Mg ha^{-1}, respectively, without any increase in nitrogen fertilizer. Model simulation and life-cycle assessment[3] show that reactive nitrogen losses and greenhouse gas emissions are reduced substantially by integrated soil–crop system management. If farmers in China could achieve average grain yields equivalent to 80% of this treatment by 2030, over the same planting area as in 2012, total production of rice, wheat and maize in China would be more than enough to meet the demand for direct human consumption and a substantially increased demand for animal feed, while decreasing the environmental costs of intensive agriculture.

Global agriculture is facing unprecedented challenges and risks. Rates of yield growth have slowed since the 1980s (ref. 4), and even stagnated in many areas[5–7]. Meanwhile, agriculture incurs substantial environmental costs, including emissions of greenhouse gases[8], loss of biodiversity[9], and degradation of land and freshwater[10,11]. These challenges may grow in the future, because global food demand is likely to double by 2050 (reflecting both population growth and increased consumption of animal protein) against a backdrop of a changing climate and growing competition for land, water, labour and energy[2]. The human and environmental costs of expanding agricultural lands are such that most of the necessary production gains must be achieved on existing farmland[1]. Can the necessary increase in yields be accomplished? If so, can the environmental costs of intensive agriculture be mitigated?

We addressed these questions through quantitative field experiments under large-scale agro-ecological conditions. Our experiments included the three main staple crops (rice, wheat and maize), which together account for most global cereal production[12,13], in the main agro-ecological areas of China. We focus on China in part because the yields of these crops already are relatively high there, thanks to 'green revolution' technologies, and in part because China must address the joint challenges of production and environmental degradation expeditiously[14].

From 2009 to 2012, we conducted a total of 153 site-year field experiments (Extended Data Fig. 1). In each experiment four treatments were employed: (1) current practice (the farmers' practice in the region but conducted in experimental plots); (2) improved practice (which modified current practice to offset the major limitations to crop growth); (3) high-yielding (which maximized yields without regard to costs); and (4) integrated soil–crop system management (ISSM, which used advanced crop and nutrient management). ISSM redesigned the whole production system based on the local environment, drawing upon appropriate crop varieties, sowing dates, densities and advanced nutrient management. The ISSM concept had been developed for maize systems[15]; we applied it to a broad range of field situations for wheat and rice in addition to maize. The challenge of increasing yield while reducing environmental costs is greater for tiller crops such as rice and wheat because they change in population structure within crop growing seasons (Supplementary Discussion). In addition to our experiments, we determined yields and nitrogen use in 18,938 farmers' fields in the main cereal production areas of China (Extended Data Table 1).

Our highest yields were achieved in high-yielding treatments, with 8.8, 9.2 and 14.4 Mg ha^{-1} for rice, wheat and maize, respectively (Table 1). These yields are comparable to yield potentials in the areas with the most favourable conditions and intensive agronomic management globally: rice in California (USA) (~9 Mg ha^{-1}) (ref. 5), wheat in Germany (9.5 Mg ha^{-1}), and rainfed and irrigated maize in the USA (13.2 and 15.1 Mg ha^{-1}) (ref. 16). The ISSM treatment achieved 97–99% of the yields in the high-yielding treatments, and the improved practice treatments achieved 88–92% of high-yielding yields; all increased yields (Table 1) and nitrogen uptakes were significantly greater than the current practice treatment (Extended Data Table 2).

Nitrogen fertilizer application rates were greatest in the high-yielding treatment, and decreased in the order current practice then ISSM then improved practice (Table 1). High nitrogen surplus (nitrogen fertilizer applied in excess of uptake by crops) and low nitrogen use efficiency (PFP_N, nitrogen partial factor productivity, in kilograms of grain per kilogram of nitrogen applied) occurred in high-yielding (owing to high nitrogen application) and current practice (owing to low grain yield) treatments, indicating the inefficiency and environmental damage associated with both conventional practices and with attempts to increase yields simply by increasing inputs[17]. Compared with current practice, the nitrogen rates in improved practice and ISSM decreased slightly even as yields increased substantially.

In the improved practice and ISSM treatments, nitrogen surplus was around zero with only a small range from −9 to 16 kg N ha^{-1}, and PFP_N reached 54–57, 41–44 and 56–59 kg N kg^{-1} for rice, wheat and maize, respectively (Table 1). These nitrogen use efficiencies are comparable to those of most 'ecologically intensive' systems worldwide[18].

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Table 1 | Grain yield, nitrogen application rate, PFP_N and nitrogen surplus for rice ($n = 57$), wheat ($n = 40$) and maize systems ($n = 56$) for the four management treatments in field experiments compared with farmers' practice from a total of 18,938 farmers

Crops	Treatment	Yield (Mg ha^{-1})	N rate (kg N ha^{-1})	PFP_N (kg kg^{-1})	N surplus (kg N ha^{-1})
Rice					
	Current practice	7.2 ± 1.1‡	181*	41§	58*
	Improved practice	8.1 ± 1.1†	146‡	57*	7‡
	High-yielding system	8.8 ± 1.2*	192*	47‡	38†
	ISSM	8.5 ± 1.2*†	162†	54†	16‡
	Farmers' practice (n = 6,592)	7.0 ± 1.5	209	41	82
Wheat					
	Current practice	7.2 ± 1.4‡	257†	28‡	74*
	Improved practice	8.3 ± 1.7†	192§	44*	−9†
	High-yielding system	9.2 ± 1.9*	283*	33†	50*
	ISSM	8.9 ± 1.7*†	220‡	41*	2†
	Farmers' practice (n = 6,940)	5.7 ± 1.3	210	33	74
Maize					
	Current practice	10.5 ± 1.6‡	266†	40†	72†
	Improved practice	12.6 ± 2.2†	214‡	59*	−8‡
	High-yielding system	14.4 ± 2.4*	402*	37†	140*
	ISSM	14.2 ± 2.6*	256†	56*	8‡
	Farmers' practice (n = 5,406)	7.6 ± 1.5	220	43	72

Means ± s.d. for yield. Least significant difference testing was performed among the four experimental treatments for each crop, the same footnote symbol(s) within each column are not significantly different at $P < 0.05$.

Reactive nitrogen losses and greenhouse gas (GHG) emissions from agriculture contribute substantially to atmospheric and water pollution in China and elsewhere. Using established empirical models[19,20] (Extended Data Figs 2–4 and Supplementary Discussion) and life-cycle assessment methods[3,19], we evaluated total reactive nitrogen losses and GHG emissions per unit area (expressed as kilograms of nitrogen per hectare or kilograms of carbon dioxide equivalents per hectare), and their intensity per unit grain yield (expressed as kilograms of nitrogen losses or carbon dioxide equivalents per million grams). Total reactive nitrogen losses and GHG emissions both for improved practice and ISSM treatments decreased compared with current practice, while high-yielding significantly increased total reactive nitrogen losses and GHG emissions (except GHG emissions in rice systems) (Fig. 1).

Reactive nitrogen losses in maize systems were higher than those in wheat and rice systems (Fig. 1), mainly because of high nitrate leaching and ammonia volatilization in maize's summer growing season. The total GHG emissions from rice were highest because of high methane (CH_4) emissions (Fig. 1). Nitrogen fertilizer production, transportation and application contributed substantially to the difference in total GHG emissions among treatments, especially for wheat and maize.

These large gains in grain yield for maize, wheat and rice demonstrate a substantial potential to meet food demand on existing farmland. While previous calculations based on historic yield trends suggested strong constraints—for example, suggesting that Chinese rice yields reached a plateau of ~6.4 Mg ha^{-1} by the mid-1990s (ref. 5)—our results demonstrate that this plateau does not represent a biophysical yield limitation (yield ceiling). We suggest that socio-economic factors—particularly extremely small farm sizes and urbanization leading to an increase in the proportion of part-time farmers—contribute to the observed yield plateau (Extended Data Fig. 5). These socio-economic factors could diminish with economic development and changes in land tenure or management arrangements (Supplementary Discussion). More generally, while previous calculations based on historic yield trends suggested that crop yields have reached a plateau in much of the world[5–7], our large-scale experimental results demonstrate that this suggestion requires careful testing.

Equally importantly, our experiments demonstrate that substantially increased yields can be produced with lower inputs of nitrogen fertilizer, and so lower human and environmental costs (Fig. 2). Our survey of farmers also shows that it is possible to achieve high yields in practice, because we found that about 20% and 5% of rice and wheat farmers, respectively, report yields already close to ISSM yields without using excessive fertilizer (Extended Data Table 3). Even so, there is room for further improvement: GHG emission intensities in ISSM (our best treatment) are still higher than published results in other intensive agricultural regions, such as maize in the USA (231 kg CO_2 eq Mg^{-1} of grain, and 13.2 Mg grain ha^{-1})[20], mainly because of coal-based nitrogen fertilizer

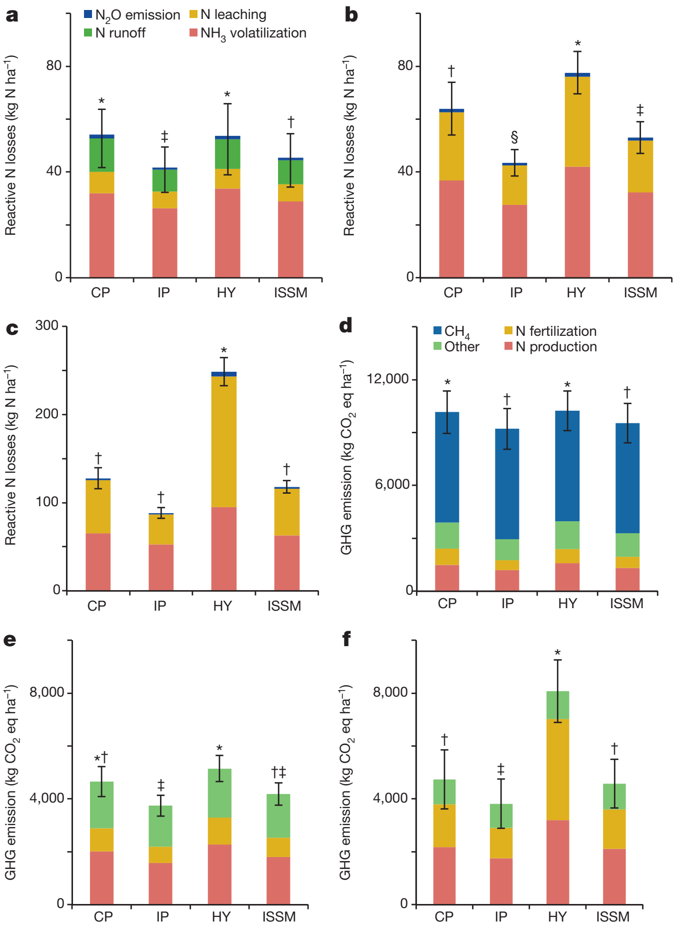

Figure 1 | Reactive nitrogen losses and GHG emissions for four management treatments, based on empirical models of losses and life-cycle assessment. **a–c**, Reactive nitrogen losses (**a**, rice; **b**, wheat; **c**, maize) include N_2O emission, nitrogen leaching, NH_3 volatilization and nitrogen runoff. **d–f**, GHG emissions (**d**, rice; **e**, wheat; **f**, maize) include those from nitrogen fertilizer application, nitrogen fertilizer production and transportation, other sources (phosphorus and potassium fertilizer; crop management) and CH_4 emission (in rice). CP, current practice; IP, improved practice; HY, high-yielding system. Means followed by the same footnote symbol(s) for each crop are not significantly different at $P < 0.05$.

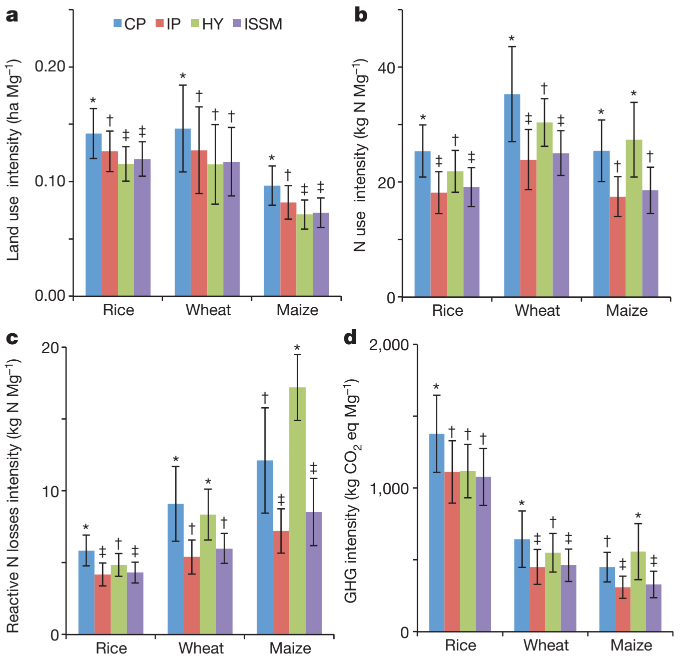

Figure 2 | Substantially increased yields can be produced with lower inputs of nitrogen fertilizer, and so lower human and environmental costs. The intensity of land use (**a**), nitrogen use (**b**), reactive nitrogen losses (**c**) and GHG emissions (**d**) needed to produce 1 Mg of grain, for three crops and four management treatments. Means followed by the same footnote symbol(s) for each crop are not significantly different at $P < 0.05$.

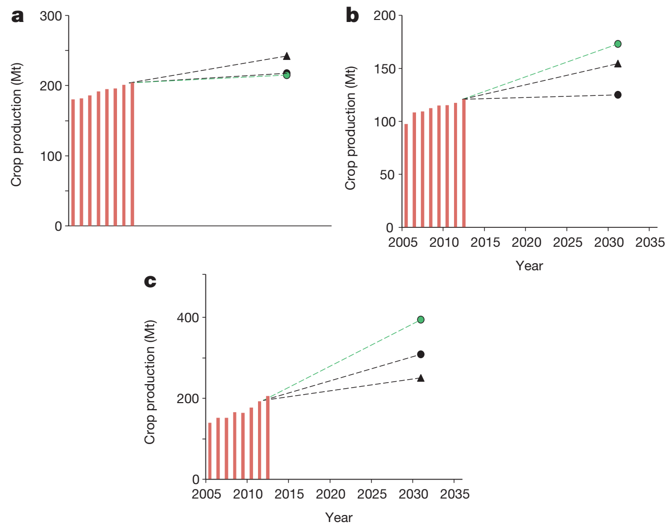

Figure 3 | The projected demand of grain production for 2030 in China. **a**, Rice; **b**, wheat; **c**, maize. Red bars, crop production from 2005 to 2012. Black circles, projected demand in 2030. Black triangles, increasing grain yield by the trend observed from 2005 to 2012, keeping planting area the same as in 2012. Green circles, grain yields that reach 80% of the level observed in our ISSM treatment, over the same planting area as in 2012. Note differences in scale for the different crops.

production in China[21]. Improved fertilizer production technology[21] and innovative fertilizer products[22] could play further roles in mitigating GHG emissions.

Current yields and cropping areas across China combine to produce 204, 121 and 206 Mt of rice, wheat and maize annually[23], with 74% of maize fed to livestock (with 5 Mt of imported maize, and 58 Mt of imported soybean). With population and economic growth, demand for grain in China is expected to reach 218, 125 and 315 Mt of rice, wheat and maize by 2030, by which time China's population is expected to have stabilized. If farmers could achieve grain yields of 80% of the yield level in our ISSM treatment by 2030, using the same planting area as in 2012, total production of rice, wheat and maize would reach 216, 174 and 397 Mt; this is enough to meet the demand for direct human consumption and domestically produced animal feed. Such yields would even suffice to offset imports of animal feed (Fig. 3), while reducing nitrogen use, reactive nitrogen losses and GHG emissions by 21%, 30% and 11% respectively, compared with current levels (scenario 2 in Extended Data Table 4). Further, if we simply reach the projected demand in 2030 with 80% of ISSM yields, then reactive nitrogen losses and GHG emission could be reduced by 48% and 26%, and the land and nitrogen fertilizer used for these three crops could also be reduced by 22% and 33% (scenario 3 in Extended Data Table 4). This change could contribute to the production of other crops and to the protection of natural ecosystems. Also, a relative shift to maize will reduce agricultural demand for water in China[24]. However, if larger quantities of more sustainably produced grain are allocated to an inefficient animal production system, overall benefits will be reduced substantially[25]. Increasing the efficiency and mitigating the environmental/human costs of livestock production systems in China deserves more attention.

The gains in yield and environmental quality that can be achieved through an integrated agronomic approach are striking—especially given that yields in China are already higher than those in most developing countries. The ISSM approach is agronomically robust and relatively easy and inexpensive to adopt (Supplementary Discussion), although the management practices employed for ISSM vary across different crops and different regions. We believe that this approach can be applied elsewhere—and that it should be possible to meet global food demand with more sustainable intensive agriculture on existing cropland, thereby sustaining other natural resources by avoiding the conversion of forest, grassland and marginal lands to agriculture and supporting other ecosystem services such as wetland preservation, wildlife conservation, carbon sequestration, etc. These benefits are achievable if we invest in agronomic research that incorporates an ecosystem perspective, if the effort is pursued across disciplinary and institutional boundaries, and if we provide the technologies, arrangements and incentives that make it viable for farmers to adapt and adopt more knowledge-intensive forms of agriculture.

Online Content Methods, along with any additional Extended Data display items and Source Data, are available in the online version of the paper; references unique to these sections appear only in the online paper.

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Supplementary Information is available in the online version of the paper.

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Author Information Reprints and permissions information is available at www.nature.com/reprints. The authors declare no competing financial interests. Readers are welcome to comment on the online version of the paper. Correspondence and requests for materials should be addressed to F.Z. (zhangfs@cau.edu.cn).

METHODS

Field experiments. A total 153 site-years of field experiments were conducted from 2009 to 2012 within the main agro-ecological areas for rice ($n = 57$), wheat ($n = 40$) and maize ($n = 56$) production in China (Extended Data Fig. 1). Four treatments were designed and compared: (1) current practice, which followed farmers' practices in the region but was conducted in experimental plots; (2) improved practice, which was based on improving farmers' practices beginning with an analysis of limiting factors, followed by implementing key new technologies, mostly through using root zone nutrient management[26] to improve nutrient use efficiency, together with known agronomic management practices (that is, increasing planting density) to increase yield; (3) high-yielding, designed to test yield potential, where crop yields were maximized through inputs so that they made full use of solar radiation and the period with favourable temperatures, without considering the costs of various inputs; (4) ISSM, which redesigned cropping systems using advanced crop and nutrient management to bring yields closer to their biophysical potential, while optimizing various resource inputs (that is, nutrient and water) and minimizing environmental costs, based on an understanding of crop ecophysiology (for example, crop canopy, solar radiation use and dry matter accumulation), physiological nutrient demands by high-yielding crop and the biogeochemical processes relating to nutrient availability and loss[15].

A randomized complete block design with four replications was used for each experiment. At maturity, grain yield and above-ground biomass were sampled and measured in each plot, with $6\,m^2$ for wheat and rice, and $10\,m^2$ for maize. Their nitrogen concentrations were determined using the Kjeldahl procedure. Fertilizer and pesticide use as well as energy use for irrigation and soil tillage in each treatment were recorded.

Survey of farmers. Representative farmers were selected for a face-to-face, questionnaire-based household survey conducted between 2007 and 2009. A total of 6,592 farmers (55 counties in 21 provinces), 6,940 farmers (113 counties in 18 provinces) and 5,406 farmers (66 counties in 22 provinces) were surveyed for rice, wheat and maize in China, respectively. In each province, three to seven counties were randomly selected, and three townships were randomly selected in each county, then two to five villages were randomly selected in each township, and finally about 20 farmers from a village were randomly surveyed to collect information on fertilizer use and grain yield in each farmer's household (Extended Data Table 1). All of these in-house surveys were conducted by professional research staff. Before beginning the survey, an informed consent information sheet was given to each farmer to read (or in some cases was read to the farmer), and verbal informed consent was requested.

Data sources for establishing models of reactive nitrogen losses. An exhaustive literature survey of peer-reviewed publications was undertaken using the ISI Web of Science (Thomson Reuters), Google Scholar (Google) and the China Knowledge Resource Integrated (CNKI) database, to identify articles published before December 2013. The literature survey focused on field measurements of nitrogen losses in the major Chinese agricultural regions, including NH_3 volatilization, nitrogen leaching, N_2O emissions and nitrogen runoff. Studies had to meet specific criteria to be included in the data set. First, nitrogen losses must have been measured both during field operations and throughout the entire growing season. Second, NH_3 volatilization must have been measured using either the micrometeorological method[27] or the wind tunnel method[28] within at least 2 weeks after nitrogen fertilization. The N_2O emissions must have been measured using the static chamber technique[29], daily for 7–10 days after nitrogen fertilization and for 3–10 days after other events that may have triggered N_2O gas emissions such as rainfall, irrigation or tillage, as well as weekly or biweekly during the remaining periods; and nitrogen leaching must have been measured using the suction cap or lysimeter method[30] or the soil sample method[31].Third, only studies that reported crop yields were included. Based on the literature survey, the final data set consisted of 134 published references and 787 observations (Supplementary Information, extended reference list).

Reactive nitrogen losses and GHG emission calculations. Using extensive and localized databases, reactive nitrogen loss models were developed based on the relationships between N_2O emission, nitrogen leaching, runoff or NH_3 volatilization, and nitrogen application rate or nitrogen surplus[19,32]. These relationships were subjected to linear or exponential regression analysis to identify the best-fit curves. Corrected R^2 values were used for model selection in addition to visual inspection of each response curve type. The results revealed an exponential relationship between the nitrogen surplus and direct N_2O emissions, nitrogen leaching and runoff, while NH_3 volatilization was linearly correlated with the rate of nitrogen fertilizer application (Extended Data Figs 2–4).

Nitrogen surplus was defined as nitrogen application minus above-ground nitrogen uptake. Nitrogen uptake in field experiments was calculated by measured nitrogen concentration multiplied by measured biomass, and in the survey of farmers it was calculated by reported yield multiplied by the parameters of nitrogen required to produce a unit of grain[33,34]. Based on the established reactive nitrogen loss models, we calculated the amount of reactive nitrogen lost to the environment, expressed as kilograms of nitrogen per hectare, and the reactive nitrogen loss intensity (reactive nitrogen losses per unit grain yield), expressed as kilograms of nitrogen per million grams.

The total GHG emissions, including CO_2, CH_4 and N_2O during the whole life cycle of crop production, consisted of three components: (1) those during nitrogen fertilizer application, including direct and indirect N_2O emissions, which can be calculated based on the empirical reactive nitrogen losses model mentioned above; (2) those during nitrogen fertilizer production and transportation; and (3) those during the production and transportation of phosphorus and potassium fertilizer and pesticides to the farm gate, and diesel fuel use in farming operations such as sowing, tillage and harvesting.

Total N_2O emissions resulting from anthropogenic nitrogen inputs to agricultural soils occur through a direct pathway (that is, directly from the soils to which the nitrogen is added), and through two indirect pathways, via the volatilization of compounds such as NH_3 and NO_x with subsequent re-deposition downwind, and N_2O emission there, and through leaching and runoff and subsequent N_2O emission downstream. Indirect N_2O emissions can be estimated following the IPCC methodology[35], whereby 1% and 0.75% of the volatilized NH_3-N and leached NO_3-N are lost as N_2O-N, respectively. For rice, the impact of the CH_4 emissions was calculated as carbon dioxide equivalents with 209 and 65 kg ha^{-1} of emissions for single rice in south and northeast China, respectively, and 245 and 323 kg ha^{-1} for early rice and late rice in the double rice system, respectively[36]. The 100-year global warming potentials of CH_4 and N_2O are 25 and 298 times the intensity of CO_2 on a mass basis, respectively[19]. The soil CO_2 flux was not included as a contribution to global warming potential in our analysis: the net flux is much less than gross CO_2 emissions that can be measured, and net fluxes have been estimated to contribute less than 1% to the global warming potential of agriculture on a global scale[37]. The change of soil organic carbon content was also not included in our analysis, because it was difficult to detect small changes in the short time our experiments were in place.

System boundaries were set as the periods of the life cycle from the production of inputs (such as fertilizers and pesticides), delivery of the inputs to the farm gates, farming operations and the crop harvesting period. Using the emission factors for all agricultural inputs given in Supplementary Table 1, we calculated total global warming potential per unit area, expressed as kilograms of carbon dioxide equivalents per hectare, and the GHG intensity, expressed as kilograms of carbon dioxide equivalents per million grams of grain.

Projection of food demand for China in 2030. The human population of China is projected to reach a peak of 1.47 billion around 2030, and the diet structure will change to more animal-derived protein with the development of urbanization (urbanization is projected to reach 80% in 2030)[25]. Using a nutrient flows in food chains, environment and resource (NUFERNUFER) model[25], we project that the demand for rice, wheat and maize in 2030 for China will be 218, 125 and 315 Mt, respectively, for a total of 658 Mt for the three crops. Demand for animal feed is expected to include 308 Mt of maize and another 50 Mt of soybean.

Data analysis. For all field experiments, data analysis used one-way analysis of variance in SAS[38]. The means of management treatments were compared using least significant difference at a 0.05 level of significance for grain yield, nitrogen application, PFP_N, nitrogen surplus, reactive nitrogen losses and GHG emissions.

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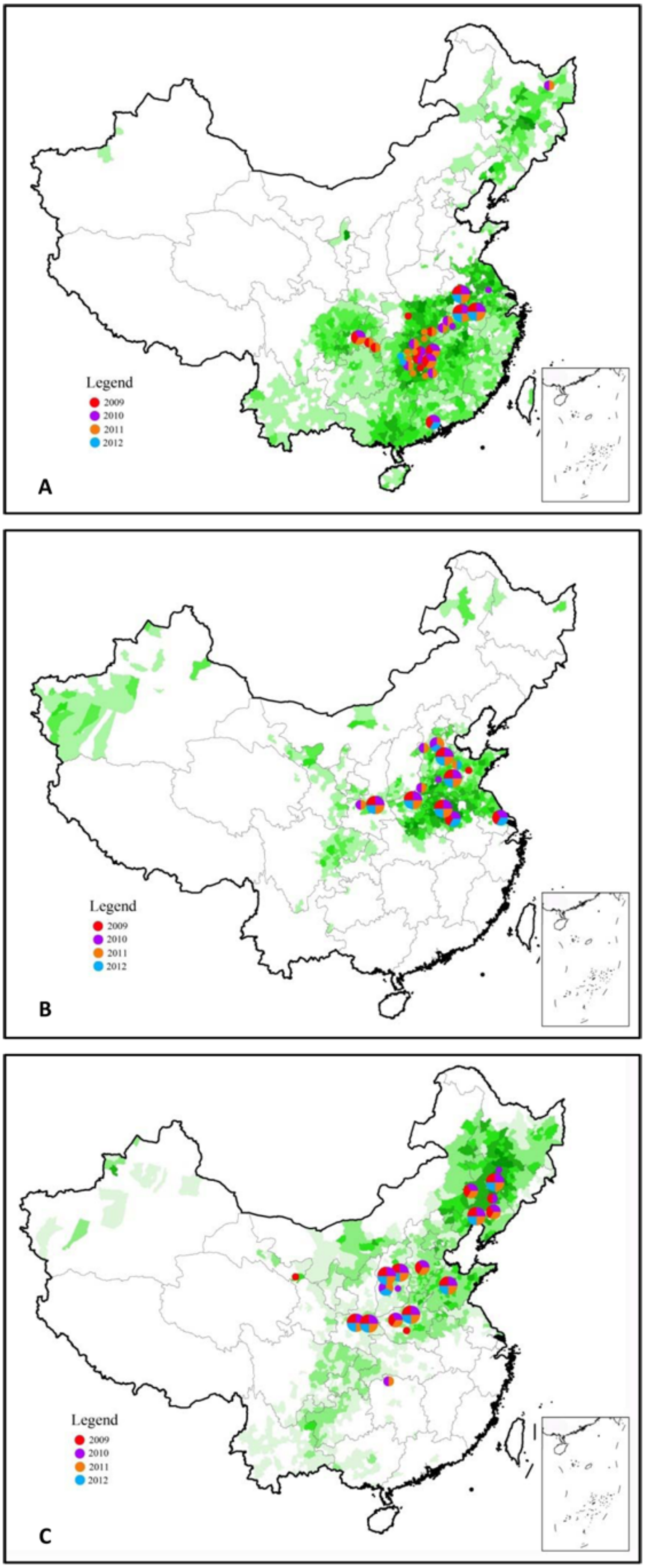
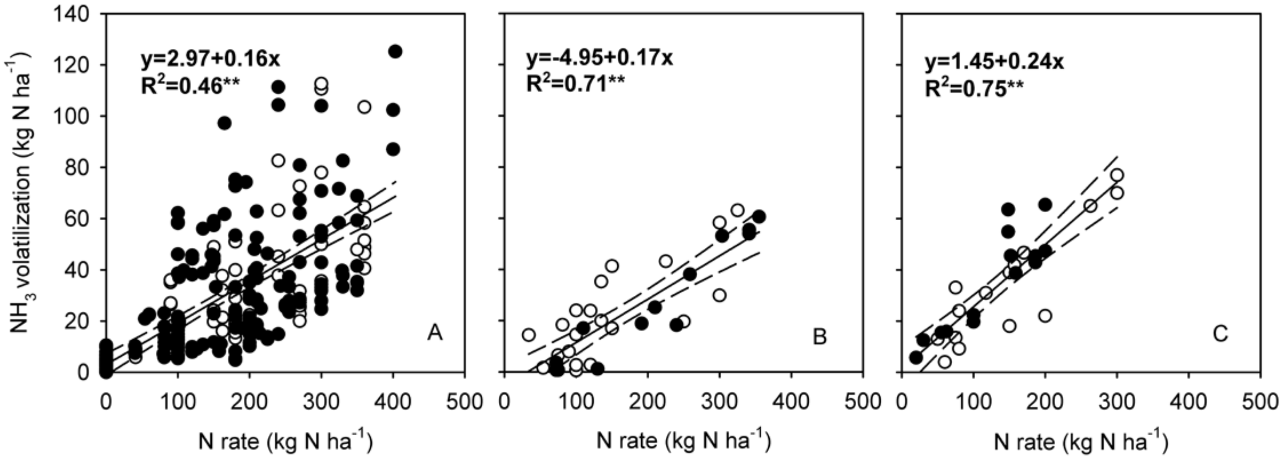
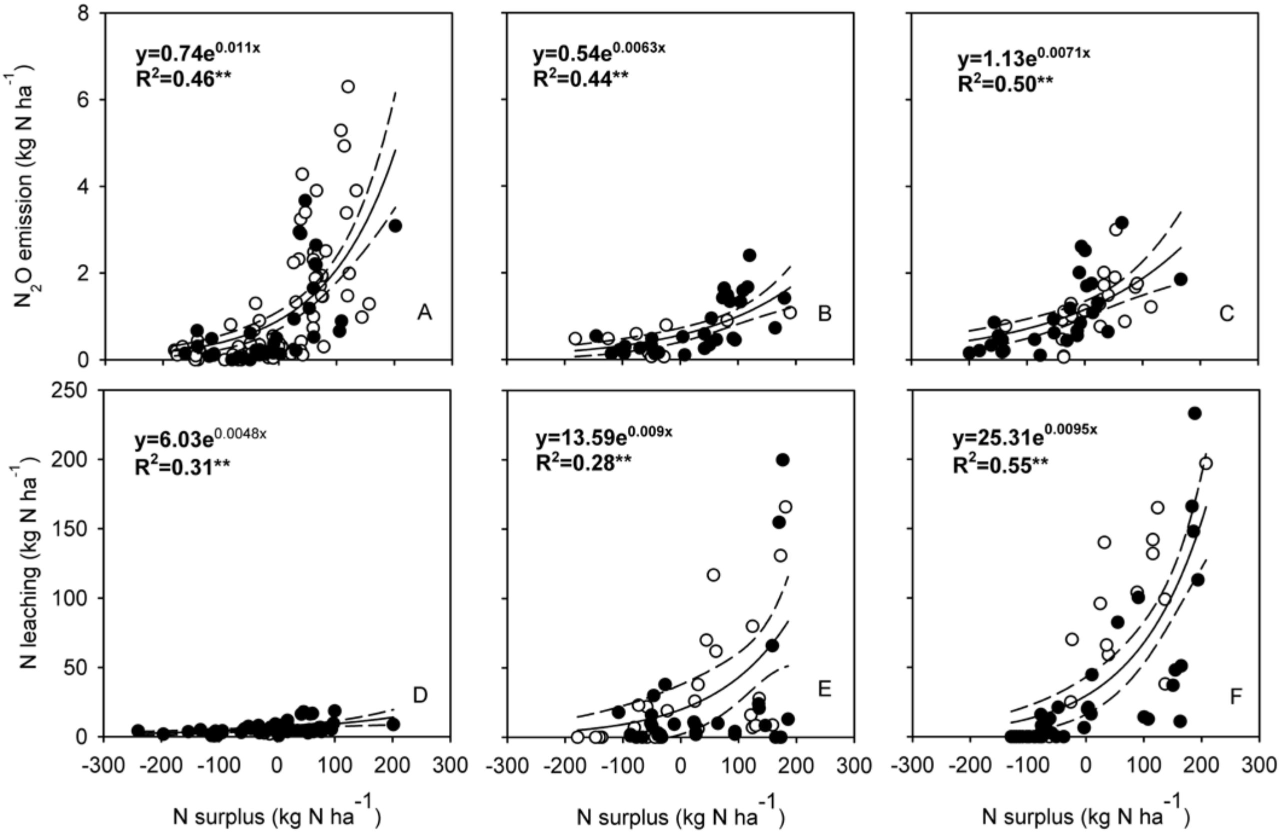


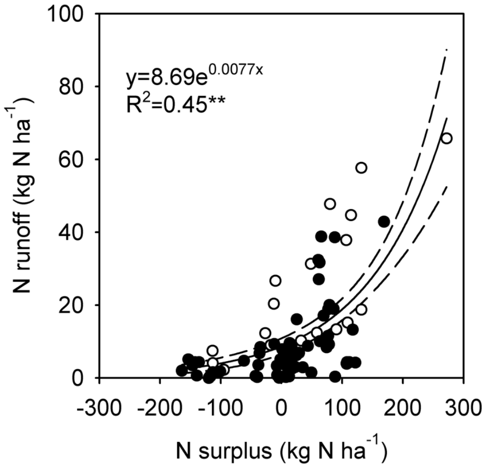
Extended Data Figure 1 | The distribution of experiments for grain from 2009 to 2012 in China. **a**, Rice (*n* = 57); **b**, wheat (*n* = 40); **c**, maize (*n* = 56). The background green colour represents the planting area for each crop; darker green means a larger density of planting area regionally for that crop. The dots represent sites, and each colour in a dot represents a year of measurements.



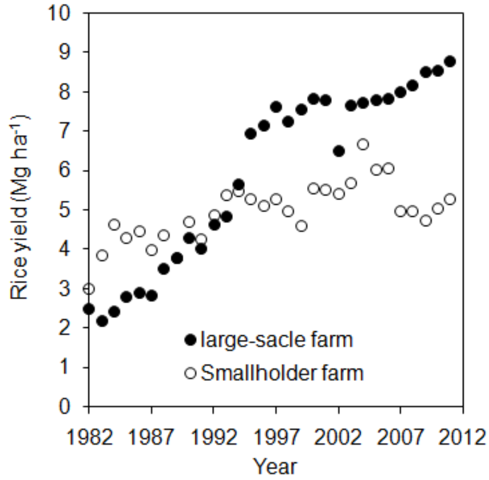
Extended Data Figure 2 | Linear models of NH_3 volatilization based on nitrogen application rate. Rate of nitrogen fertilizer application was plotted against NH_3-N volatilization for (**a**) rice ($n = 265$) (Supplementary Information, extended references 1–36 for rice), (**b**) wheat ($n = 34$) and (**c**) maize ($n = 29$) (Supplementary Information, extended references 37–60 for wheat and maize) growing seasons, respectively. $^{**}P = 0.01$. Filled and hollow circles represent data from Chinese journals (or theses) and ISI journals, respectively.



Extended Data Figure 3 | Exponential models of N_2O emissions and nitrogen leaching based on nitrogen surplus. Nitrogen surplus was plotted against N_2O-N emissions for (**a**) rice ($n = 118$) (Supplementary information, extended references 7, 36, 61–84 for rice), (**b**) wheat ($n = 40$) and (**c**) maize ($n = 48$) growing seasons (Supplementary information, extended references 85–99 for wheat and maize), and against nitrogen leaching for (**d**) rice ($n = 52$) (Supplementary information, extended references 7, 100–113 for rice), (**e**) wheat ($n = 59$) and (**f**) maize ($n = 56$) (Supplementary information, extended references 44, 114–121 for wheat and maize). Nitrogen surplus was defined as nitrogen application rate minus above-ground nitrogen uptake. **Regression significant at $P < 0.01$. Solid and hollow circles represent data from Chinese journals (or theses) and ISI journals, respectively.



Extended Data Figure 4 | Exponential model of nitrogen runoff based on nitrogen surplus for rice production. Nitrogen surplus was defined as nitrogen application rate minus above-ground nitrogen uptake ($n = 81$) (Supplementary information, extended references 8, 104, 122–134). $^{**}P < 0.01$. Solid and hollow circles represent data from Chinese journals (or theses) and ISI journals, respectively.



Extended Data Figure 5 | Rice yields over time in smallholder and large-scale farms in Heilongjiang province from 1982 to 2011. Large-scale farms were around 20 ha; smallholder farms were less than 2 ha. Data from refs 39, 40.

Extended Data Table 1 | Grain yields, nitrogen application rates, calculated PFP_N, nitrogen surplus, the total and the intensity of reactive nitrogen losses and GHG emissions in farmers' fields for rice (*n* = 6,592), wheat (*n* = 6,940) and maize (*n* = 5,406) in China

Crops	Grain Yield	N rate	PFP_N	N surplus	Nr losses	Nr intensity	GHG emission	GHG emission intensity
	Mg ha^{-1}	kg N ha^{-1}	kg kg^{-1}	kg N ha^{-1}	kg N Mg^{-1}	kg N ha^{-1}	kg CO_2eqha^{-1}	kg CO_2eqMg^{-1}
Rice	7.0	209	41	82	66	9.9	10,343	1,574
	(4.5-9.8)	(86-412)	(16-83)	(-46-280)	(28-142)	(3.7-21)	(4,365-15,465)	(580-2,618)
Wheat	5.7	210	33	74	65	12	3,707	671
	(3.8-7.5)	(81-360)	(15-67)	(-49-223)	(19-147)	(2.3-27)	(2,203-5,766)	(368-1,117)
Maize	7.6	220	43	72	120	17	4,436	621
	(5.4-10.5)	(85-413)	(17-90)	(-66-256)	(36-274)	(4.6-39)	(2,273-8,269)	(287-1,179)

Values are mean and range (from the 5th to 95th percentiles).

Extended Data Table 2 | Above-ground biomass, harvest index (HI) and crop nitrogen uptake for rice (*n* = 57), wheat (*n* = 40) and maize (*n* = 56) in field experiments with four management treatments

Crops	Treatment	Biomass Mg ha^{-1}	HI	Crop N uptake kg N ha^{-1}
Rice	CP	11.7 †	0.52 *	123 ‡
	IP	12.8 *†	0.54 *	138 †
	HY	14.1 *	0.54 *	155 *
	ISSM	13.4 *†	0.54 *	147 *†
Wheat	CP	13.4 ‡	0.46 †	183 †
	IP	14.8 †‡	0.48 *†	201 †
	HY	16.7 *	0.48 *†	234 *
	ISSM	15.8 *†	0.49 *	218 *†
Maize	CP	18.4 ‡	0.49 †	194 ‡
	IP	20.8 †	0.52 *	222 †
	HY	23.7 *	0.52 *	261 *
	ISSM	23.3 *	0.52 *	249 *†

Means followed by the same letter(s) within each column for each crop are not significantly different at $P < 0.05$.

Extended Data Table 3 | Yield and nitrogen rates of farmer average, top farmers and ISSM

Crop		Yield (Mgha^{-1})	N rate (kg N ha^{-1})
Rice	Farmers average	7.0	209
	Top 20% yield of farmers	8.6	159
	ISSM	8.5	162
Wheat	Farmers average	5.7	210
	Top 5% yield of farmers	8.4	234
	ISSM	8.9	220
Maize	Farmers average	7.6	220
	Top 5% yield of farmers	11.3	229
	ISSM	14.2	256

Extended Data Table 4 | Total production, weighted average of grain yield and nitrogen rate, total land use, nitrogen fertilizer use, reactive nitrogen losses, and GHG emissions in 2005, 2012 and projected in 2030 under three scenarios for all three crops (rice, wheat and maize) in China

	Unit	2005	2012	2030		
				S1	S2	S3
Production	Mt	417	531	656	786	658
Grain yield	Mg ha^{-1}	5.4	5.9	7.4	8.8	9.6
N rate	kg N ha^{-1}	213	217	213	172	188
Land use	Million ha	78	89	89	89	69
N use	Mt	16.6	19.4	19.0	15.4	12.9
Nr losses	Mt	6.7	7.9	8.3	5.5	4.1
GHG emission	Mt CO_2eq	500	558	542	498	411

Scenario 1 (S1): 'business as usual', grain yield increased by the trend with the recent 8 years (from 2005 to 2012) and nitrogen application rate the same as 2012 (there was almost no change from 2005 to 2012). Grain yields were 6.8, 5.0 and 5.9 Mg ha^{-1}, planting areas were 30.1, 24.3 and 35.0 Mha (ref. 23), and nitrogen application rates were 209, 210, and 220 kg N ha^{-1} in 2012 (Extended Data Table 1) for rice, wheat and maize, respectively. Scenario 2 (S2): both grain yield and nitrogen application with 80% of ISSM and land use the same as 2012. Scenario 3 (S3): both grain yield and nitrogen application with 80% of ISSM, and crop production just enough to reach projected demand for rice, wheat and maize in 2030 (requiring less cropland).