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Air quality, nitrogen use efficiency and food security in China are improved by cost-effective agricultural nitrogen management

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China's gains in food production over the past four decades have been associated with substantial agricultural nitrogen losses, which contribute to air and water pollution, greenhouse gas emissions and damage to human health. Here, we explore the potential to improve agricultural production practices that simultaneously increase yields while addressing these environmental challenges. We link agronomic research with air quality modelling for an integrated assessment of four improved nitrogen management strategies: improved farm management practices with nitrogen use reductions; machine deep placement of fertilizer; enhanced-efficiency fertilizer use; and improved manure management. We find that simultaneous implementation of the four strategies provides the largest benefits, which include: reductions in $PM_{2.5}$ concentrations and associated premature deaths; increases in grain yields and grain nitrogen use efficiency; reductions in NO_3^- leaching and runoff and greenhouse gas emissions. Total benefits of US$30 billion per year exceed the US$18 billion per year in costs. Our findings indicate that policies that improve farmers' agricultural nitrogen management in China will improve both food security and public health while addressing multiple environmental challenges. Similar increases in attention on agricultural policy around the world are likely to provide large benefits in food security, environmental integrity and public health.

Nitrogen (N) use has increased food security in China while simultaneously posing risks to public health, damaging ecosystems and increasing greenhouse gas (GHG) emissions[1–3]. China's grain production increased by 74% between 1982 and 2017, while chemical fertilizer use tripled over the same period[4]. The average nitrogen use efficiency (NUE) for crop production in China is approximately 0.25 (ref. [5]), indicating substantial reactive nitrogen (Nr) losses to the environment. The losses include ammonia (NH_3) and nitrous oxide (N_2O) emissions, which contribute to health-damaging $PM_{2.5}$ (fine particles with aerodynamic diameters of <2.5 μm) air pollution and global warming, respectively, as well as NO_3^- and organic N leaching and runoff, which contaminate drinking water and cause eutrophication[5,6]. As the world's largest meat and egg producer[7], China also contributes approximately 25% of global manure N production[8,9]. Only one-third of manure N annually produced in China is utilized by crops or stored in the soil, with two-thirds released as environmental pollution[8].

Chinese policymakers increasingly recognize the interconnections between air and water pollution, climate change, low NUE and China's poor agricultural N management practices. Facing severe $PM_{2.5}$ air pollution[10–12], previous policies in China have focused on controlling point sources, without attention paid to dispersed agricultural sources. However, it is now known that controlling agricultural NH_3 emissions is effective for $PM_{2.5}$ mitigation when gaseous NH_3 limits the formation of secondary inorganic aerosols (SIAs) ($[SIA] = [NH_4^+] + [NO_3^-] + [SO_4^{2-}]$), which has been reported in winter in the eastern United States[13], Europe[14,15] and China's Pearl River Delta[16], and in summer in Europe[15]. China's Three-Year Action Plan to Win the Battle for a Blue Sky[17], enacted in 2018 for the first time, encourages NH_3 emission reductions, although no quantitative target has been specified. NH_3 emission reductions must be achieved through improved synchronization between N supply and crop demand and improved manure N utilization. Furthermore, such improvements increase crop yields and NUE, as well as decreasing N_2O and NO_3^- leaching and runoff[18]. The government has set several policy measures for improving food security and NUE, which are viewed as separate efforts from air pollution policies, including: zero N fertilizer use increase by 2020[19]; an action plan for converting animal manure to nutrients (2017–2020)[20]; a plan for 75% of livestock and poultry waste to be disposed of with no harm to the environment[17]; and a tightened regulatory standard for the discharge of animal waste to rivers that has been drafted for public review[21].

Air and water pollution, climate change, low NUE and food security in China were addressed separately in previous reports, with foci on either the implications of agricultural NH_3 emission reductions for air pollution[13,16,22], N deposition and acid rain[23], or impacts of N management on water quality[3], GHG emission mitigation and crop yields[18,24]. Worldwide, policies have been largely

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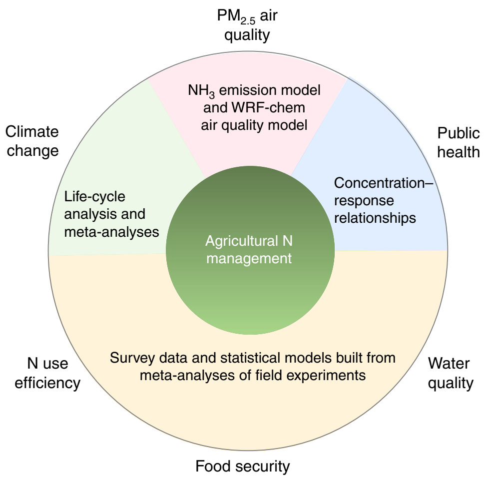

Fig. 1 | An integrated assessment framework for evaluating the impacts of improved N management in China. Impacts on $PM_{2.5}$ were evaluated using the NH_3 emission model from Zhang et al.[26] and the WRF-Chem air quality model. Concentration–response relationships for impacts of fine particulates on premature mortality were analysed using methods from Burnet et al.[11]. Other impacts were estimated using life-cycle GHG analysis (as in Zhang et al.[51]), statistical models of N application rates and Nr emissions, farm surveys of crop yields and N use, and several meta-analyses of field experiments that evaluate impacts of N management on Nr emissions and yields[7,9,18,70,71].

unsuccessful in reducing agricultural Nr pollution, particularly outside of Europe[25].

We identify four promising agricultural N management opportunities for China that are all technically feasible today and that, with the appropriate policy drivers, could be rapidly implemented. We design five scenarios that describe the implementation of these opportunities: improved management practices (moderate farm management improvements and reductions in N application rates); machine application (machine deep placement of fertilizers); enhanced-efficiency fertilizers (the use of controlled-release fertilizers and urease inhibitors); manure management (manure acidification, aerobic composting, cropland injection and improved animal feed); and combined (all of the improvements above combined). These scenarios are detailed in the Methods, Supplementary Methods and Supplementary Tables 1 and 2. Combining agronomic research/data and atmospheric chemistry modelling, we comprehensively evaluate their effects on NH_3 emissions and $PM_{2.5}$ formation, NUE, crop yields, GHG emissions, water pollution, N deposition and acid rain, and compare the benefits and costs (Fig. 1). We utilize a recently improved process-based NH_3 emission model for China[26], which accounts for the dependence of NH_3 emissions on seasonally and geographically varied climate and soil conditions and agricultural practices, which previous inventories excluded. Our comprehensive scientific and economic analyses show that cost-effective agricultural strategies exist in China that simultaneously reduce severe air pollution and massive agricultural N losses while improving food security. Our work provides guidance for policy-making that will facilitate a sustainable food future for China. In addition, our findings are applicable to other countries that seek to simultaneously improve air quality and food security while reducing water pollution and emissions of GHGs.

Results

Impacts of improved agricultural management on air quality.

NH_3 emission reductions. The most ambitious management scenario (all strategies combined) reduces NH_3 emissions by up to 34% annually, with up to 30% in January and 50% in July across much of China (Table 1 and Supplementary Fig. 1). Three scenarios (enhanced-efficiency fertilizer, machine application and manure management) all result in similar annual reductions of ~11%. Wintertime reduction of NH_3 emissions relies mainly on improved manure management, which provides 15% reductions locally. Summertime reductions of NH_3 emissions are seen under all scenarios, with the largest reductions achieved in northeastern and southern China and the North China Plain (Fig. 2).

$PM_{2.5}$ air quality benefits. As NH_3 emissions primarily affect the formation of the inorganic fraction of $PM_{2.5}$, which are SIAs, we hereby present surface SIA concentration reductions (Fig. 3). Under the combined scenario, up to $8\,\mu g\,m^{-3}$ (28%) SIAs reduction can be achieved in both January and July (Fig. 3 and Supplementary Fig. 2). The manure management and machine application scenarios in January and all individual management scenarios in July can reduce SIAs by up to $4\,\mu g\,m^{-3}$. Central and southern China in January and the North China Plain in July have the largest reductions in SIA ($5–8\,\mu g\,m^{-3}$ reduction for both regions). For annual population-weighted $PM_{2.5}$ concentrations, the combined scenario is capable of achieving up to $4.4\,\mu g\,m^{-3}$ reduction in Chongqing and $1–3\,\mu g\,m^{-3}$ reduction across much of China (Supplementary Fig. 3). Other scenarios are capable of achieving $0.5–2\,\mu g\,m^{-3}$ reduction in central China.

Reductions of SIAs are driven mainly by reductions of nitrate (NO_3^-) and ammonium (NH_4^+) aerosols, with sulfate (SO_4^{2-}) concentrations remaining nearly unchanged in both January and July (Supplementary Figs. 4 and 5). This is because gaseous NH_3 will always neutralize H_2SO_4 (the acid form of SO_2) first, forming NH_4HSO_4 and $(NH_4)_2SO_4$, before reacting with HNO_3 (the acid form of NO_x) and forming NH_4NO_3[27]. NH_3 emission reductions will thus limit the formation of nitrate, while having little impact on the formation of sulfate from the oxidation of SO_2 by OH, O_3, H_2O_2 and other oxidants. Some areas have moderate changes in sulfate concentrations, reflecting changes in other aerosols (ammonium and nitrate) that can slightly affect the simulated oxidants of SO_2 (for example, O_3 and H_2O_2) and thus the formation of sulfate.

Furthermore, as evidenced by previous findings from Liu et al.[23], reduced NH_3 emissions will increase the acidity of precipitation in China, mainly by favouring the production of NH_4HSO_4 and shifting the $NH_3(g)–NH_3(aq)–NH_4^+(aq)$ equilibrium towards $NH_3(g)$, which releases more $H^+(aq)$[23]. Reduced NH_3 emissions will also reduce N deposition, which is a recognized threat to plant diversity[28].

Mortality effects of reduced $PM_{2.5}$. Improved N management has public health benefits by reducing concentrations of $PM_{2.5}$[11]. The combined scenario avoids ~30,500 premature mortalities in China (Supplementary Table 3), which is nearly 3% of the 1.3 million premature deaths resulting from exposure to $PM_{2.5}$ in China reported in 2012. The improved management practices, enhanced-efficiency fertilizer, machine application and manure management scenarios reduce premature mortality by ~5,600, 8,700, 5,300 and 8,800 persons, respectively.

Uncertainties of avoided mortalities as a result of uncertainties in parameters in dose–response relationships are presented in Supplementary Table 3. Factoring in uncertainties of total NH_3 emission estimates for China, the public health benefits presented above are still valid. Using a low (high) estimate for total NH_3 emissions in China of 10 Tg (17 Tg), as seen in other inventories[29–31], would result in public health benefits achieved under the combined scenario that are 0.7% higher (10% lower) than our current estimate (Supplementary Table 4 and Supplementary Fig. 6). The total NH_3 emission used in this study is 14 Tg.

Table 1 | China's national total NH_3 emissions at baseline and with the implementation of five agricultural N management scenarios

Scenario	NH_3 emissions (Tg)			NH_3 emission reduction (%)		
	Annual	January	July	Annual	January	July
Baseline	14.0	0.66	1.66	–	–	–
Improved management practices	13.1	0.64	1.54	6.4	3	7
Enhanced-efficiency fertilizer	12.5	0.65	1.45	10.7	2	13
Machine application	12.5	0.62	1.43	10.7	6	14
Manure management	12.5	0.58	1.50	10.7	12	10
Combined	9.3	0.51	1.01	33.9	23	39

Annual NH_3 emission levels are provided along with values for January and July. Baseline refers to the year 2012. NH_3 emission reductions achieved through the implementation of each N management scenario are shown as a percentage of baseline NH_3 emissions.

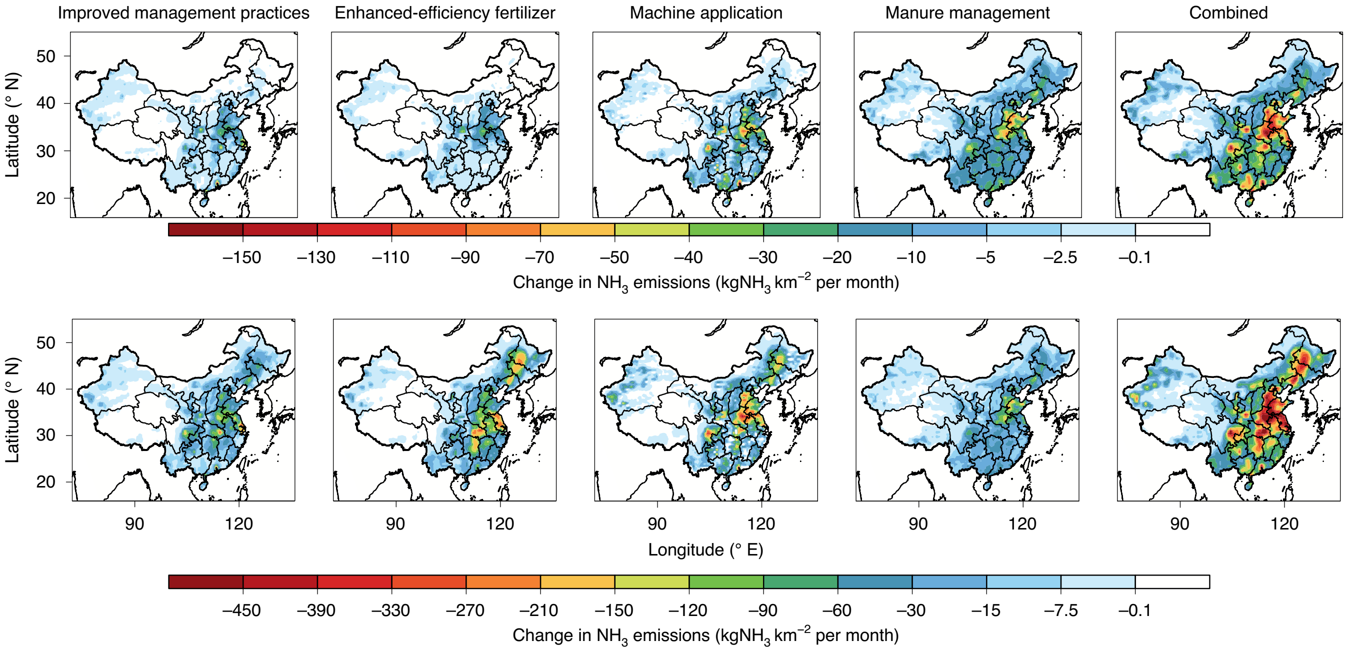

Fig. 2 | NH_3 emission reductions achieved through the implementation of N management scenarios. Changes of NH_3 emissions (where negative values represent a reduction) under the five N management scenarios compared with baseline NH_3 emissions for January (top) and July (bottom) of the year 2012. The geographical maps used in this figure are based on NCL Earth 4 (https://www.ncl.ucar.edu/Document/HLUs/Classes/MapPlotData4_1_earth_4.shtml), with China's national and provincial boundaries based on China's National Catalogue Service for Geographic Information from the National Geomatics Center of China (http://www.webmap.cn/commres.do?method=result100W).

Impacts of improved agricultural management scenarios on yields and other environmental indicators. NH_3 -mitigating strategies also positively affect crop yields, NUE and mitigation of both GHG emissions and water pollution. Here, we evaluate these factors comprehensively at the agro-ecological region level (see the Methods, Supplementary Methods and Supplementary Tables 5 and 6 for more information).

Yield effects. The improved management practices, machine application, enhanced-efficiency fertilizer and combined scenarios increase the national crop yields of the major grains (maize, wheat and rice) by 52, 43, 36 and 52 Mt (9, 8, 7 and 9% from the baseline crop yields), respectively (Fig. 4a and Supplementary Tables 7–9). Yield gains under the combined scenario are conservatively estimated to be the same as under the improved management practices scenario, given the lack of field experiments on the cumulative effects of combining multiple N management tools. Under both improved management practices and combined scenarios, maize contributes to 50% of the total grain yield increases (26 Mt), while wheat and rice contribute 28% (15 Mt) and 22% (11 Mt), respectively (Fig. 4a and Supplementary Table 7). Under the improved management practices scenario, moderate improvements in general management (such as changes in phosphorus and potassium use and N application timing) contribute to the majority of yield gains. The modest reductions in N use under this scenario lead to increased yields in some regions and decreased yields in others (Methods, Supplementary Methods and Fig. 2 of Wu et al. [32]).

Change in NUE. The average cropping system NUE for grain crops increases from the current 38% to 48% under the improved management practices scenario, 41% under the machine application and enhanced-efficiency fertilizer scenarios and up to 48% under the combined scenario (Fig. 4b and Supplementary Tables 8 and 9).

GHG emissions. The improved management practices and combined scenarios both reduce N application rates modestly and thus

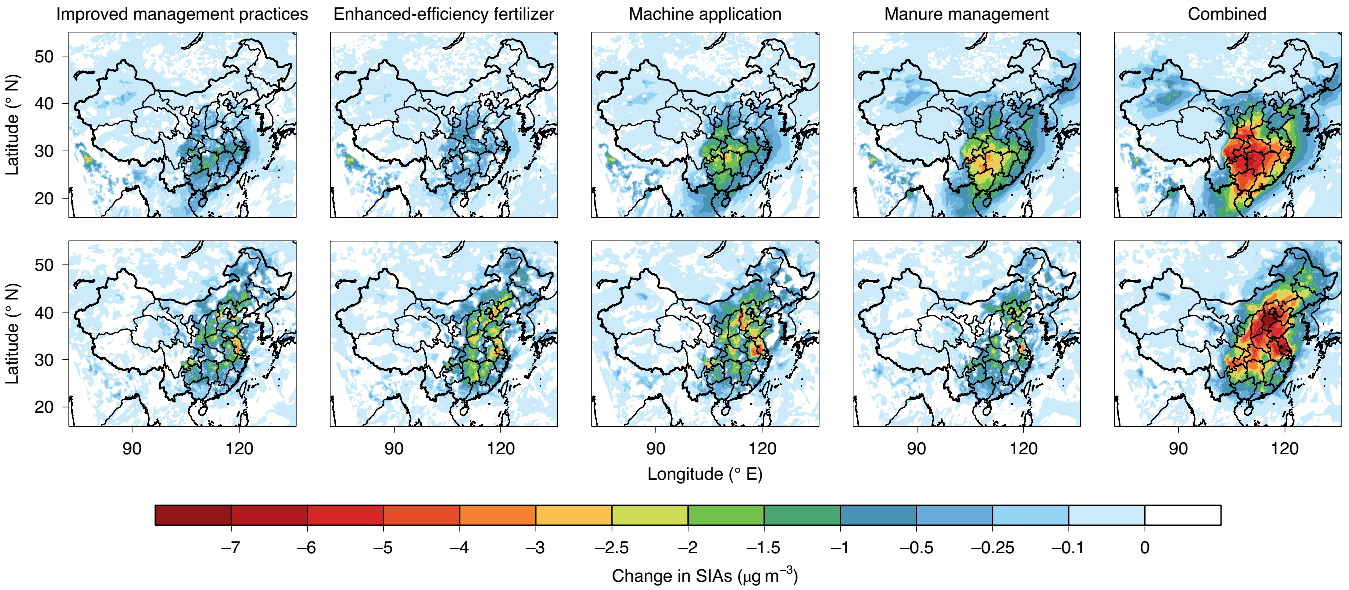

Fig. 3 | $PM_{2.5}$ air quality impacts of the implementation of N management scenarios. Changes in ground-level (a 28-m-thick surface layer in WRF-Chem) concentrations (where negative values represent a reduction) of SIAs (the sum of ammonium, nitrate and sulfate aerosols) under the five management scenarios compared with the baseline simulation, in January (top) and July (bottom) of 2012. The geographical maps used in this figure are based on NCL Earth 4 (https://www.ncl.ucar.edu/Document/HLUs/Classes/MapPlotData4_1_earth_4.shtml), with China's national and provincial boundaries based on China's National Catalogue Service for Geographic Information from the National Geomatics Center of China (http://www.webmap.cn/commres.do?method=result100W).

both each reduce life-cycle GHG emissions by 38 $MtCO_2e$ (Fig. 4e and Supplementary Tables 8 and 9), which represents a reduction in China's total agricultural direct GHG emissions of ~6%[33] (excluding GHGs related to land use change). The machine application and enhanced-efficiency fertilizer scenarios reduce N_2O emissions by 4.7 and 18.8 Mt CO_2e, respectively (Fig. 4c), representing 9 and 36% reductions in China's total grain crop N_2O emissions.

Water pollution. NO_3^- leaching and runoff from croplands are lowered by 28, 16 and 40% under the improved management practices, enhanced-efficiency fertilizer and combined scenarios, respectively (Fig. 4d). The manure management and combined scenarios both reduce N loss to water systems from manure by ~20% (2.5 Tg of 12 Tg N loss in 2010 to water from animal manure) (Fig. 4f).

Overall cost–benefit analysis. We use data from China to monetize all environmental and yield impacts and estimate management costs when implementing improved management. Our research includes a full range of impacts and thus provides a comprehensive analysis of benefits and costs that may inform policy-making.

Table 2 provides a summary of economic benefits and costs for agricultural N management scenarios. We find US$9.3–19 billion per annum net monetary benefits across several scenarios, with private benefits that are five to ten times larger than public benefits. The largest net benefit occurs with the improved management practices scenario (benefit/cost ratios of 30–35), followed by the enhanced-efficiency fertilizers (benefit/cost ratios of 2–4) and machine application scenarios (benefit/cost ratios of 2–7) (Supplementary Table 10).

Table 3 provides itemized economic benefits and costs (Supplementary Tables 11–13). Overall, yield benefits provide the largest benefits, valued at US$12.3–15.7 billion per annum across agricultural management scenarios. These are then followed by benefits from NUE, $PM_{2.5}$ air quality improvements, NO_3^- leaching and runoff water pollution mitigation, GHG mitigation and labour savings, each valued at ~US$1–4 billion per annum. The improved management practices and combined scenarios, through reductions in N use for grain crops, show reduced N fertilizer expenditure by US$1.62 billion per annum and reduced life-cycle GHGs related to N fertilizer consumption by US$0.76 billion per annum. Machine application saves US$0.2–0.7 billion per annum of labour compared with conventional hand application. Machine application and enhanced-efficiency fertilizer reduce N_2O emissions valued at US$0.16 and US$0.28 billion per annum, respectively. Total benefits sum to US$7–30 billion (with an estimated range of US$4–50 billion) per annum across scenarios.

The management costs of improving practices vary across scenarios. Machine application can generate management costs that are <60% of manure management. Our cost estimation for the improved management practices scenario excludes the costs to government of implementing education programmes that help farmers improve general management and reduce N use levels, due to the lack of data. In addition, the costs associated with negative environmental impacts (reduced NH_3 emissions, decreases in precipitation pH and hence increases in acid rain damages to crop yields) are valued at US$0.9–3.4 billion per annum across scenarios.

The manure management scenario is beneficial only under high benefit and low/medium cost estimates, due to large fixed costs and price uncertainties. It is always beneficial to use reduced protein content animal feeds, which can reduce animal feed purchase costs with no adverse impacts on animals. The costs of purchasing and operating manure injection machines are more than fully compensated over the machine lifetime by savings from reduced N fertilizer purchases (Supplementary Methods). However, at present, logistical complications, such as matching manure suppliers with crop farmers and testing manure nutrient and pathogen contents, can be an impediment. Using aerobic composting reactors for manure storage also introduces cost uncertainties, because of the large price variations in organic fertilizer, discount rate and machine lifetime (Table 3).

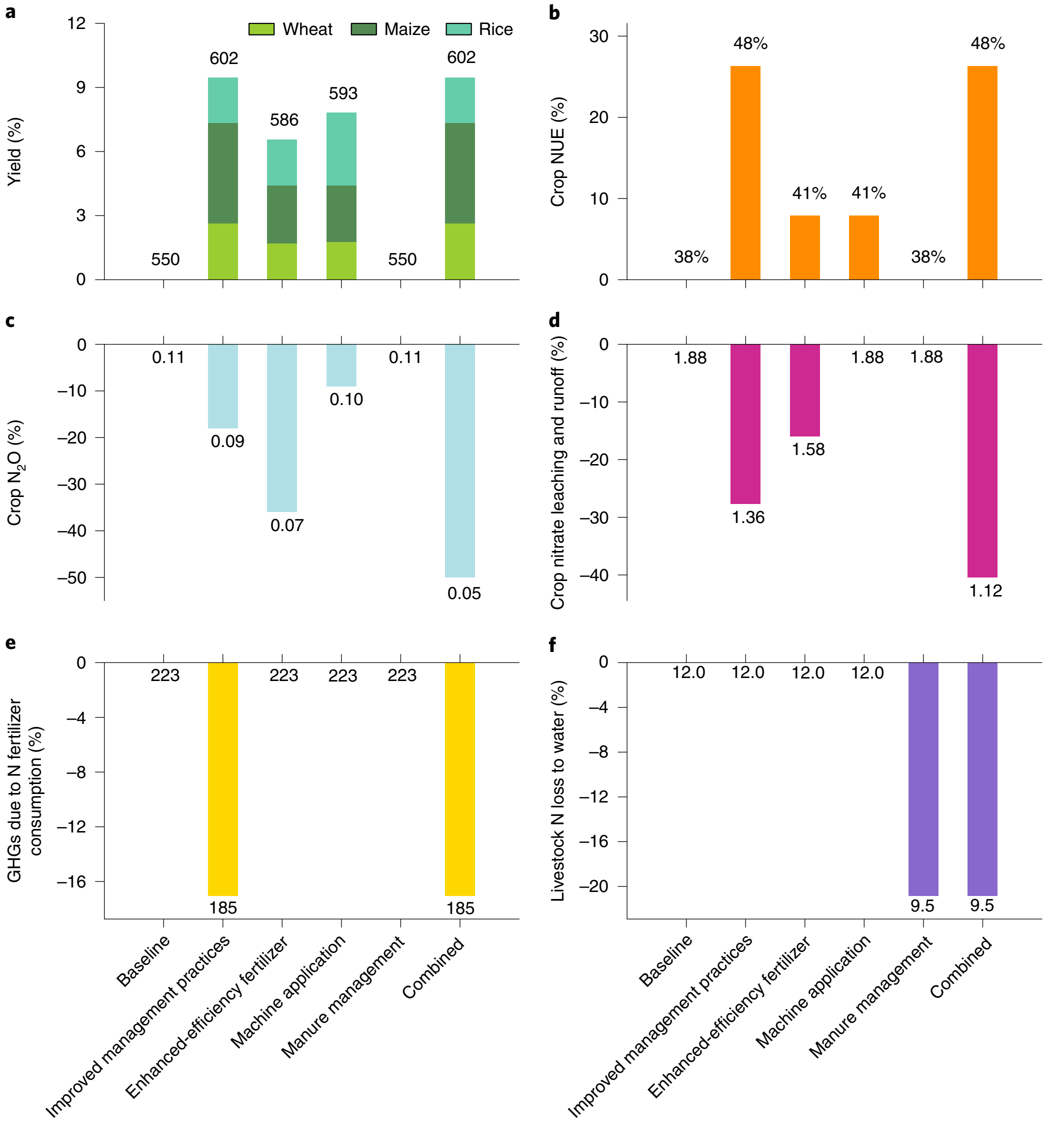

Fig. 4 | Yield, NUE, GHG emissions and water pollution impacts of implementing improved N management scenarios. a, China's national total grain yields (wheat, maize and rice). **b**, Average grain crop system NUE. **c**, Grain crop N$_2$O emissions. **d**, Grain crop NO$_3^-$ leaching and runoff. **e**, Life-cycle GHGs emitted during N fertilizer manufacture, transport and application for grain crops. **f**, Livestock N loss to water. The *y* axes present percentage changes of metrics under each scenario ((scenario − baseline)/baseline). Numbers associated with each bar represent the absolute value of the metric (in Mt (**a**), % (**b**), TgN (**c**, **d** and **f**) and MtCO$_2$e (**e**)). For grain crop production metrics, our calculations are at the level of agro-ecological regions with a total of 33 regions considered (*n* = 33 regions).

Table 2 | Economic benefits and costs for agricultural N management scenarios in US$ billions per annum

Metrics		Improved management practices	Enhanced-efficiency fertilizer	Machine application	Manure management	Combined
Benefits	Private	17.3 (17.3 to 18.0)	12.3	14.6 (14.3 to 14.8)	5.1 (2.7 to 18.2)	22.9 (20.3 to 38.5)
	Social	2.3 (1.7 to 4.2)	2.2 (1.7 to 3.7)	1.3 (1.0 to 2.2)	1.8 (1.5 to 2.9)	6.5 (5.1 to 11.4)
	Total	19.6 (19.1 to 21.5)	14.5 (14.0 to 16.0)	15.8 (15.3 to 17.0)	7.0 (4.3 to 23.4)	29.4 (25.3 to 50.0)
Total costs		0.64	5.16 (3.82 to 6.49)	4.99 (2.38 to 8.91)	7.42 (5.27 to 11.87)	17.75 (11.65 to 27.45)
Net benefits		19.0 (18.5; 20.8)	9.3 (7.6 to 12.2)	10.8 (6.4 to 14.6)	−0.46 (−7.6 to 18.2)	11.7 (−2.12 to 38.3)

Values in parentheses denote ranges of estimates with economic parameters at low, medium and high levels (provided in the 'Economic analysis' section of the Methods and Supplementary Tables 11–13).

Discussion

In this study, we find that the largest yield and environmental benefits are achieved by moderately improving general farm management while reducing excess N application. The yield benefits provide the majority of total monetized benefits and are achieved mainly through improving general farm management

Table 3 | Itemized economic benefits and costs for agricultural N management scenarios in US$ billions per annum

Metrics	Improved management practices	Enhanced-efficiency fertilizer	Machine application	Manure management	Combined
Private benefits					
Increased crop sales	15.7	12.3	14.1	0	15.7
Reduced N fertilizer purchase expenditure[a]	1.62	0	0	0	1.62
Labour savings	0	0	0.45 (0.2 to 0.7)	0	0.45 (0.2 to 0.7)
Organic fertilizer sales	0	0	0	5.14 (2.74 to 20.5)	5.14 (2.7 to 20.5)
Social benefits					
Value of lives saved from $PM_{2.5}$ air pollution-related deaths	0.70 (0.51 to 1.41)	1.08 (0.78 to 2.18)	0.65 (0.47 to 1.32)	1.08 (0.79 to 2.19)	3.77 (2.74 to 7.61)
Reduced damage from NO_3^- leaching and runoff	0.61	0.41	0.06	0.33	0.65
Reduced social cost of GHGs	0.76 (0.38 to 1.89)	0.28 (0.14 to 0.7)	0.16 (0.08 to 0.4)	0	0.76 (0.38 to 1.89)
Reduced N deposition[b]	0.24	0.41	0.41	0.41	1.29
Costs					
Acid rain damage[b]	0.64	1.07	1.07	1.07	3.39
Technological costs of improving N management	0[c]	4.09 (2.75 to 5.42)	3.92 (1.31 to 7.84)	6.35 (4.2 to 10.8)	14.36 (8.26 to 24.06)

[a] The savings here are for farmers. The savings for society (with urea fertilizer subsidies included) are ~103% of this value. Subsidies were removed in 2015. [b] Values for reduced N deposition and acid rain damage were estimated by scaling the results of Liu et al.[23]. [c] The improved management practices scenario may incur costs for the government to implement education programmes that help farmers reduce N use amounts and improve other general farm management (phosphorus and potassium use, fertilizer application timing and so on). Given the lack of data, this cost is excluded. Ranges of estimates are provided, in the form of a mid-range (low to high estimate), for cost and benefit metrics that are sensitive to the selected values of economic parameters. Values of parameters are provided in the 'Economic analysis' section of the Methods and Supplementary Tables 11–13. A currency conversion rate of US$1 = 6.95 Chinese yuan and prices for the year 2012 are used.

(improving phosphorus and potassium use amounts and the ratio of N applied as starter fertilizers and dressings). Removing excess N application, although substantially reducing Nr losses, provides relatively small yield gains in some regions and yield losses in others. Both reductions in N use and improvements in general farm management might be achieved simultaneously through farmer education programmes. In comparison, machine application and enhanced-efficiency fertilizers provide smaller net benefits than improved management practices, but may be easier to implement in the short term.

Overall, our findings support an increased emphasis on improving agricultural N management to address multiple environmental challenges, including air pollution, food security, low agricultural NUE and climate change. Such improvements are critical for China to achieve the Sustainable Development Goals. Challenges remain in improving N management, for which new policies and programmes are needed, including government education programmes and financial incentives aimed at improved farm management. In addition, technological, service and information support from other stakeholders, including public extension agents, researchers and the private sector, would be highly beneficial.

Our estimated benefits are conservative. For example, recent findings indicate that urease inhibitors can reduce NH_3 emission rates by 84% in China[34]; however, we use the results from a meta-analysis representing a more typical range of 42–58%. Our yield, NUE, GHG and water pollution analysis leaves out potential gains of improved management for Chinese fruits and vegetables, as well as crops other than the three major grain crops. Our yield gain estimation for the combined scenario is conservative and might increase if future field experiments are included to represent the real cumulative effects of N management strategies. Our manure management scenario evaluation does not include other economically beneficial manure management opportunities for animal houses and manure disposal, or damages from manure odour, pathogen content and direct manure discharge to water bodies (which is estimated to have been 30–70% of total manure in the year 2000[35])). Including these benefits and costs could otherwise make manure management more attractive (especially in winter) given its large NH_3 and $PM_{2.5}$ reduction benefits. Our machine application scenario assumes machine deep placement of N, which is less cost effective than hand placement (Supplementary Table 14), mainly because hand placement does not incur large labour costs but avoids machine rental costs. Our improved management practices scenario estimates farmers' N fertilizer purchase savings using N fertilizer prices for 2012, which may underestimate current savings, because N fertilizer prices increased after the removal of subsidies in 2012. Our valuation of avoided $PM_{2.5}$-related premature deaths is based on values of statistical life estimated in studies from 2011 and 2013. Such values are expected to increase as the Chinese economy continues to develop.

Our analysis includes uncertainties in NH_3 emissions and yield estimation (Supplementary Discussion). Our NH_3 emissions are estimated with agricultural data and meteorological conditions for the year 2012. NH_3 emission rates from fertilized soils may increase when future temperature increases[36,37], which may in turn increase the responses of NH_3 emissions to agricultural management. In addition, the surface–air bi-directional exchange of NH_3 is not considered in our study, but is rather treated as separated unidirectional emission and dry deposition in the Weather Research and Forecasting—Chemistry (WRF-Chem) air quality model. Previous studies have shown that bi-directional NH_3 exchange may effectively enhance the atmospheric lifetime of NH_3[38] and affect emission estimates[39,40]. However, NH_3 exchange parameterization in China lacks evaluable measurements and remains highly uncertain. Our yield estimates exclude the effects of climate change, farmers' planting techniques and local soil conditions on yields. Climate change with the CO_2 fertilization effect included is projected to increase Chinese grain crop yields by 3–15% (and with the CO_2 fertilization effect excluded will decrease Chinese grain crop yields by 37%) in the 2020s[41,42]. Research that evaluated the impacts of climate change on crop NUE remains limited and uncertain (Supplementary Discussion). In addition, crop yields are also affected by planting techniques (crop variety, planting density and so on) and soil conditions (pH, total N content, organic carbon matter content and so on). In reality, farmers may encounter a wide range of yield responses to improved management depending on these factors. Optimization of farmers' planting techniques and nutrient, water and pest management, based on local-specific climate and soil conditions, can increase yields and NUE even more than the strategies examined here[24,43]. However, such highly sophisticated precision farming requires detailed local-specific crop modelling and is less likely to be realized nationwide in the next decade.

Our analysis excludes the evaluation of impacts of N management on soil NO_x emissions; it also simulates various Nr flows in a decoupled way by utilizing field experiments, statistical models and process-based models, due to the limited capabilities of current models. Excluding the impacts of N management on soil NO_x emissions, $PM_{2.5}$ formation or subsequent yield-reducing ozone production leads to an underestimation of air quality benefits. Reductions in fertilizer use, as proposed under our N management scenarios, may lower soil NO_x emissions, particularly in high-temperature and irrigated cropland[44]. For Chinese cropland, such effects remain under-researched. Currently, soil NO_x emissions are a minor source of total NO_x emissions in China, compared with energy, transportation and residential sectors[36,37]. However, its importance needs to be further quantified with more field experiments. Our estimation of N_2O emissions, NUE and nitrate pollution impacts is based on statistical models at agro-ecological region levels, instead of process-based models. Ideally, models that comprehensively represent all Nr emission processes at high geographical resolution, as well as the dependence of all Nr emissions on soil/climate conditions and management practice, may be incorporated in the future. However, currently, only limited progress has been made with process-based models for N_2O at national scales and for NH_3 and nitrate leaching and runoff at global scales[45]. The incorporation of accurate crop yield estimation into Earth System Models (ESMs) provides an even greater challenge.

Despite our findings that agricultural NH_3-mitigating strategies are cost effective for addressing air quality, food security and other environmental challenges, future research is needed to compare the cost effectiveness of mitigating emissions of NH_3 compared with other $PM_{2.5}$ precursors (SO_2 and NO_x) mostly from non-agricultural sources. Our estimation of technological adoption costs incurred by NH_3 emission reductions ranges from US\$0 to ~US\$4,000 per tonne of NH_3, which is lower or comparable to that of SO_2 emission reduction (ranging from US\$2,500–5,000 per tonne SO_2 across Chinese mega cities[46]). However, the same mass reduction of SO_2, NO_x and NH_3 results in different levels of $PM_{2.5}$ reduction, and NO_x and SO_2 emission reductions are associated with different externalities (impacts on O_3 pollution and acid rain) from NH_3 emission reductions. Future research on unexploited SO_2 and NO_x mitigating technologies is needed to inform policymakers with the relative cost effectiveness of regulating non-agricultural sectors compared with the agriculture sector in China.

Despite our finding of net benefits, other factors, such as financing and socioeconomic barriers and uneven distribution of costs and benefits, prevent Chinese farmers and livestock ranchers from readily improving their practices. Educating the 224 million smallholders (cultivating land less than two-thirds of a hectare) who cultivate half of the available cropland has high transaction costs. For example, one rural research station in the North China Plain has helped local farmers increase yields from 68% of the attainable level to 97% through knowledge transfer[47]. Scaling up such educational programmes to nationwide smallholders would cost US\$60 billion[47]. The largest opportunities exist with the 15% of large farms that cultivate approximately half of the cropland and industrialized animal production, because improving practices on large farms will probably incur lower transaction costs than those of smallholders.

Opportunities exist for improving the management practices of both large farms and smallholders. Improved management can be facilitated by the development of policies that strengthen knowledge and technological support to farmers. Government programmes can provide agricultural management training to managers of large farms. Burgeoning private service contractors now provide affordable fertilizer application services to large farms[48]. Farmers' professional cooperatives can provide low-cost machinery rental options and knowledge sharing opportunities for smallholders. The government could also consider policy instruments for all farmers, such as agro-environmental payments that reward farmers for delivering positive public goods or mitigating negative environmental externalities. Such policies, widely adopted in Europe and the United States[49], could transform Chinese subsistence farming to multi-functional sustainable agriculture, supporting food security while providing attractive landscapes and driving rural development.

Methods

NUE. NUE represents the ratio of N in crops to total N inputs (natural and human sources). NUE is estimated as $N_{\mathrm{crop}}/\left(N_{\mathrm{dep}}+N_{\mathrm{bfix}}+N_{\mathrm{irri}}+N_{\mathrm{fert}}+N_{\mathrm{manu}}\right)$, where N_{crop}, N_{dep}, N_{bfix}, N_{irri}, N_{fert} and N_{manu} represent, respectively, the N content in harvested crop, atmospheric deposition, biological fixation (microbial conversion of atmospheric N_2 to ammonia that plants can use), irrigation, fertilizer application and manure used as fertilizer.

Agricultural N management scenarios. *Improved management practices.* Chinese agriculture is characterized by heavy N fertilizer use, as well as poor nutrient (N, phosphorus and potassium) management (application timing and amounts)[51–53]. This scenario describes modest reductions in N application quantities and improvements in the general management of China's grain crops (Supplementary Methods and Supplementary Table 2), as well as modest reductions in N application for vegetable and fruit crops. Below, we describe in detail current farm management in China, N reduction levels that help farmers maximize their profits, and impacts of management improvements on crop yields and Nr losses.

China's grain farmers currently apply excess N (for example, at levels that are ~50 and ~150% higher than in the United States for maize and wheat, respectively). Farmers' substantial engagement in off-farm jobs and limited nutrient management knowledge have prevented them from carefully following the 4R principles (the right time, right amount, right form and right method) for nutrient management. Farmers in China over-apply N as a form of yield insurance. They perceive little cost for large over-application because fertilizer prices were low before 2012 due to substantial governmental subsidies that have been gradually reduced since 2012. The ratios of N applied as dressing and base fertilizers also remain sub-optimal. The application of other nutrients, such as phosphorus and potassium, varies substantially across individual farms[54,55].

Overuse of N increases all types of Nr losses[32,55–57] as crops' demands for N vary across growing stages and dumping fertilizers into fields prevents N supply from meeting crops' demands. In addition, when excess N is provided, crop yields can decrease due to unproductive tiller, crop lodging and increases in the incidence of diseases and crop senescence[58] (Supplementary Methods). According to field experiments, when N use rates are reduced from the current levels to levels that optimize farmers' net return (crop sales minus seed and fertilizer input costs) (Supplementary Methods and Supplementary Table 2), yields increase in some regions but decrease in others (for example, in northeastern China, northwestern China and parts of the North China Plain). This is because local-specific general farm management, crop variety, climate and soil conditions (Fig. 2 of Wu et al. [32]) have also affected grain crop yield responses to N input levels (Supplementary Methods).

However, when general farm management is also improved along with N reductions, field experiments find large overall yield gains of ~10% compared with yields under farmers' conventional N application levels and management practices. These have been demonstrated by Wu et al.[32] and Wu et al.[57] in meta-analyses of nationwide field experiments (~2,000 for wheat, ~2,000 for maize and ~1,500 for rice) conducted under the 3141 Experiment Project by the Soil Testing and Fertilizer Prescription Program of the Chinese Ministry of Agriculture during 2005–2010[56]. These field experiments test the grain crop yields under various N application levels. Statistical analyses were conducted to identify for each crop production region the N application levels that would help maximize farmers' profits. Management of these field experiments provides moderate farm management improvements to farmers compared with conventional practices (for example, one-third of the N is applied as base fertilizers and two-thirds are applied as dressings, and phosphorus and potassium fertilizers are applied following local technicians' recommendations[32,55–57]). More details are provided in the Supplementary Methods.

We use current N use and yield levels by grain crop production region from surveys. We use yield levels under improved farm management and reduced N application rates from field experiments[32,56,57]. In addition, N application rates for vegetables and fruits are reduced following recommendations derived from Chinese field experiments (Supplementary Methods and Supplementary Table 1).

Enhanced-efficiency fertilizer. Enhanced-efficiency fertilizers can increase yields and NUE while decreasing Nr losses, but are not yet widely utilized in China. China's farmers typically apply rapid-release fertilizers (for example, for grain crops, ~50% of N is from urea and the rest is from compound fertilizers[51]).

This scenario describes the use of controlled-release fertilizers (which improve the synchronization of N supply with crops' demands) and urease inhibitor fertilizers (which reduce the hydrolysis rate of urea and thus the rapid NH_3 volatilization post-application). Controlled-release fertilizers also reduce N_2O and NO_3^- leaching and runoff by 15–60%[59]. Urease inhibitor reduces NH_3 emission rates by 40–70% depending on crop type and N application rates[59]. Enhanced-efficiency fertilizers can easily be promoted with governmental subsidies and product marketing.

Machine application. In China, currently only 30% of crop fields are fertilized using machinery. The Chinese government has set a number of policies to promote machinery use (for example, increasing the percentage of machine fertilized areas from the current 30% of cropland to 40% by 2020[19]).

This scenario describes the replacement of hand fertilizer application with deep placement of N by machines (5–7 cm next to and 5–8 cm below crop seeds) in all grain crop regions except in hilly regions. This makes N more available to crops and reduces Nr losses. We also discuss two variations of this scenario: deep placement of N by hand; and machine broadcasting.

Deep placement of N near crop roots can substantially reduce NH_3 emission (that is, by 35% for wheat and rice systems and by 70% for maize), as well as N_2O emission and NO_3^- runoff[60]. Reductions in N losses thus make N more available to fulfil crop demand, thus increasing yields or sustaining current yields with less N input. A meta-analysis of field experiments has found that deep placement can increase grain yields by 5–9% in China compared with hand broadcasting (Supplementary Table 6).

Switching from hand application to deep placement does not require change of fertilizer products. In all cases, Chinese farmers typically use compound fertilizers or bulk blending fertilizers as starter fertilizers and urea as N dressing. The ratio of urea N to other types of N is roughly 1:1.

Despite existing policy drivers, machine application provides management improvements that are perceived to be less risky to farmers compared with reductions in chemical inputs. It is also feasible since village-level machine rental centres have appeared across China and serve the needs of all farmers in one village. A literature review shows that machinery for the application of starter fertilizers and dressings for major crops and vegetables is available in China (that is, seed and starter fertilizer application machines, hand-push liquid fertilizer application machines, high ground-clearance dressing machines and so on[61]).

Manure management. Improved management of manure can reduce Nr pollution, since each stage of manure handling (in animal houses and during collection, storage and treatment) is subject to Nr losses. In China, major issues include direct discharge of manure to water bodies/landfill and lack of manure management in animal houses and during storage stages[9]. This scenario describes using acidification and aerobic composting (for indoor animal manure storage), cropland injection (for manure spreading) and improved animal feed (which reduces unnecessary N supply)[62].

During manure storage, acidification of manure and anaerobic composting reactors are used for all indoor animals, reducing NH_3 emission rates during storage by 60%[62]. During manure spreading, 25% of stored manure from indoor animals is applied deep below the surface to cropland using specialized machines, reducing NH_3 emissions by 80% per hectare compared with surface application[62]. Low-protein animal feed replaces conventional animal feed, which includes more protein N than chicken, cattle and pigs utilize. This results in a 10% decrease in the N content of manure from these animals[62]. More details are provided in the Supplementary Methods.

Improved manure management is easier to implement than all fertilizer management practices. This is because animal production in China is more industrialized than crop production. Livestock monopolies exist, which contract with small animal farmers for animal product collection. The government faces lower transaction costs when imposing regulations on livestock monopolies than on millions of small crop farmers.

Combined. This scenario simultaneously implements all of the strategies mentioned above. N fertilizer is applied at reduced rates, in the form of enhanced-efficiency fertilizers and with machine deep placement. Manure handling during storage and spreading processes, as well as animal feed, are also improved.

NH_3 emissions in the baseline and N management scenarios. We utilize an NH_3 emission model to obtain baseline NH_3 emissions for the year 2012 (Supplementary Methods and Supplementary Fig. 7) and NH_3 emissions in the five N management scenarios. The NH_3 emission model we use was published by Zhang et al.[26] and is an updated bottom-up high-resolution NH_3 emission estimation tool for China.

Earlier bottom-up NH_3 emission inventories have low spatial resolution and no seasonality (for example, the NH_3 emission inventory used by the Greenhouse Gas and Air Pollution Interactions and Synergies model) or use NH_3 emission factors primarily developed for other regions[26,63]. Previous inventories of NH_3 emissions in China vary by a factor of two and have discrepancies in spatial distribution and seasonality[26]. Our evaluation of modelled NH_3 against satellite observations indicates that this NH_3 emission model reasonably captures the spatial and seasonal variations of NH_3 levels over China and outperforms other widely used inventories in summer when NH_3 emissions are high (Supplementary Methods, Supplementary Figs. 8–13 and Supplementary Tables 15 and 16).

The NH_3 emission model represents N fertilizer application for 18 crops (including maize, wheat, rice, potato, sweet potato, rapeseed, soybean, groundnut, tobacco, cotton, citrus, banana, grape, apple, pear, other fruits and vegetables). Crop fertilizer application amounts and NH_3 emission factors are parametrized with fertilizer application timing, rates, types (including urea, ammonium bicarbonate, ammonium sulfate and so on), methods (injection versus broadcast), climate variables (temperature, wind and so on) and local soil (pH) conditions. The model includes major animal production (cattle, goats, sheep, pigs and poultry) in grazing, intensive and free-range systems. The total ammonium nitrogen content excreted by outdoor animals is subject to NH_3 volatilization and is without further management. Total ammonium nitrogen excreted by indoor animals goes through several stages of management (that is, animal housing, manure storage and manure spreading), with each stage suspect to NH_3 volatilization. NH_3 emissions are gridded at 1/4° × 1/4° resolution. More details are provided in the Supplementary Methods.

Air quality simulation. We use the WRF-Chem model (version 3.6.1)—an online-coupled meteorology–chemistry model—to simulate $PM_{2.5}$ formation at baseline and under the various scenarios. The physical and chemical schemes used are: Carbon-Bond Mechanism Version Z (CBMZ) for gas-phase chemistry; the four-bin Model for Simulating Aerosol Interactions and Chemistry (MOSAIC) for aerosol chemistry; the Rapid Radiative Transfer Model for General Circulation Models (RRTMG) scheme for short- and longwave radiation; the Morrison scheme for cloud microphysics[64]; the Yonsei University scheme for boundary layer mixing[65]; and the Noah land surface model for land surface[66]. Meteorological boundary conditions come every 6 h from the 2014 National Centers for Environmental Prediction Final Analyses data. Chemical initial and boundary conditions are from a 2014 simulation of the global chemical transport model—the Model for Ozone and Related Chemical Tracers version 4 (MOZART-4).

Anthropogenic emissions of air pollutants are from the Multi-resolution Emission Inventory for China (http://www.meicmodel.org)[67] for the year 2012 and from Hemispheric Transport of Air Pollutants version 2.2 outside China[68] for the year 2010. Biogenic emissions are calculated online using the Model of Emissions of Gases and Aerosols from Nature (MEGAN) scheme[69] and open biomass burning emissions are from the Global Fire Emission Database (GFED) version 4 (www.globalfiredata.org) for the year 2012.

We conduct six sets of air quality simulations: one baseline and five agricultural management scenarios. The only difference between the WRF-Chem air quality simulations for the baseline and management scenarios is modified NH_3 emissions resulting from the implementation of various agricultural N management options. N_2O is not considered in air quality simulations because it is chemically inert in the troposphere. Each simulation set includes 1 month of simulation for January and 1 month of simulation for July (in all cases after 6 d of spin-up) for the year 2012. The model resolution is 27 km × 27 km with the domain covering China and parts of other Asian countries (9° N–58° N latitude; 60° E–156° E longitude). There are 37 vertical layers extending from the surface to 50 hPa with a 28-m-deep surface layer.

Evaluation of air quality modelling. Evaluation of our baseline WRF-Chem-modelled NH_3 concentrations against satellite measurements shows reasonable agreement on the spatial and seasonal variations of atmospheric NH_3 levels (Supplementary Methods and Supplementary Figs. 8–13). Evaluation of WRF-Chem baseline simulation of meteorology and evaluation of WRF-Chem speciated $PM_{2.5}$ with satellite and surface measurements is provided in the Supplementary Methods, Supplementary Figs. 14–17 and Supplementary Tables 17 and 18.

Estimating yield, GHG emissions, NUE and water pollution impacts of improved N management in grain crops. We divide China's grain crop production into seven wheat production regions, 13 maize regions and 13 rice regions. All calculations are conducted at agro-ecological regional levels and aggregated to national estimates using the crop planting areas listed in the China Statistical Yearbook (see the data availability statement for the model we use).

Baseline N fertilizer application amounts and yields are from a study by Wu et al.[32] that summarizes thousands of large-scale household farm surveys (Supplementary Methods and Supplementary Table 2). Baseline emissions of N_2O, NO_3^- leaching and NO_3^- runoff are estimated based on relationships between emission and Nr fertilizer use amount[70,71] (Supplementary Methods and Supplementary Table 5). These statistical relationships are from two studies, i.e. Cui et al.[71] which is a meta-analysis of 205 published studies covering 317 sites in China including 1,332 observations for wheat and rice and Cui et al.[70] which reports 16 on-farm site-years of experiments for maize.

For the improved management practices scenario, we estimate yield gains resulting from both moderate decreases in N use and moderate improvements in general farm management using farm surveys of current N use and yield levels and a meta-analysis of nationwide field experiments (Supplementary Methods). These field experiments test the response of crop yields to N use reductions under improved farm management compared with current management[32,54,55,57]. They also show that, under current farm management, reductions in N use may decrease maize yields in some production regions (NE1, NE4, NCP1, NCP2 and NW3; as defined in Supplementary Table 2) while increasing yields in others[32]. Improved farm management (phosphorus and potassium use and the ratio of N applied as starter fertilizers and dressings) provides the majority of yield gains for many production regions. More details are provided in the Supplementary Methods. We estimate Nr emission relationships between emission and Nr fertilizer use amount[70,71] (Supplementary Methods and Supplementary Table 5). For the machine application and enhanced-efficiency fertilizer scenarios, we estimate yields and Nr emissions using management-specific factors from a meta-analysis of Chinese field experiments (Supplementary Methods and Supplementary Table 6)[18]. This meta-analysis summarized field trials conducted across China from 376 studies, providing 1,166 observations. We estimate reduced N loss to water from animal farms achieved in manure management using total N loss from animals and the contribution by various manure handling stages[7,9] (Supplementary Methods).

We calculate NUE for each scenario using additional data, including N deposition estimated by atmospheric chemistry models, biological N fixation, N content in yields, N in irrigation water and so on[26,72–74]. For scenarios that do not include N fertilizer application reductions, we calculate GHG mitigation using N_2O emission reductions[70,71]. For scenarios that include N fertilizer application reductions, we calculate GHG mitigation using an estimate of 13.5 tCO_2e (including N_2O, CH_4 and CO_2 with the global warming potential at the 100-year time scale) emitted per tonne of N fertilizer consumed in China during N fertilizer manufacture, transport and application and post-application[51] (Supplementary Methods).

Estimating acid rain and N deposition impacts. Monetized impacts of improved N management on acid rain and N deposition are obtained from a recent study[23].

Public health impact of $PM_{2.5}$. We calculate premature mortalities of four diseases due to exposure to $PM_{2.5}$ for adults (≥25 years old) under five agricultural N management scenarios. The four diseases considered are chronic obstructive pulmonary disease, lung cancer, ischaemic heart disease and ischaemic stroke.

For each province in China, we calculate the premature deaths of each disease based on:

$$\mathrm{Mort}_{i,P} = \mathrm{POP}_P \times \mathrm{Mortbase}_{i,P} \times \left(1 - \frac{1}{\mathrm{RR}_{i,P}}\right)$$

where $\mathrm{Mort}_{i,p}$ is the number of premature mortalities in province P from disease i; POP_P is the number of exposed targeted populations in province P considering adults (≥25 years old) in 2012 (from the 2013 China Statistical Yearbook[75]); $\mathrm{Mortbase}_{i,P}$ is the baseline mortality rate in province P for disease i in 2012 (from the Global Burden of Disease study[76]); and $\mathrm{RR}_{i,P}$ is the relative risk factor for one disease i (adopted from Burnett et al.[11]). Relative risk factors for ischaemic heart disease and stroke are by age group. There are 12 age groups considered (that is, 25–29, 30–34, 35–39, 40–44, 45–49, 50–54, 55–59, 60–64, 65–69, 70–74, 75–79 and >80 years). Relative risk factors for lung cancer and chronic obstructive pulmonary disease are the same for all persons ≥25 years old. Uncertainties of avoided mortalities as a result of uncertainties in parameters in dose–response relationships are presented in Supplementary Table 4.

Economic analysis. We calculate costs and benefits when the economic parameters used are at low, medium and high levels. We utilize various economic parameters (with ranges in parentheses), including: value of statistical life (US\$0.124 million per person[77] (US\$0.09–0.25 million per person)[78,79]; urea price (US\$0.6 kg^{-1} N); rice (US\$0.40 kg^{-1} N); wheat (US\$0.34 kg^{-1} N); maize (US\$0.27 kg^{-1} N)[80]; damage cost of NO_3^- (\$1.32 kg^{-1} N (\$0.2 kg^{-1} N from drinking water health impacts and \$1.12 kg^{-1} N from eutrophication))[30,81]; social costs of carbon (\$20.4 tCO_2e^{-1} (\$10–50 tCO_2e^{-1}))[82]; time saved by machine application (4 h ha^{-1} (2.5–7.5 h ha^{-1})); labour costs (US\$7.34–43.2 d^{-1}); organic fertilizer sale price (US\$43.2 t^{-1} (US\$28.8–143.9 t^{-1})); machine rental price (US\$5.8 ha^{-1} (US\$2.9–8.6 ha^{-1})); composting reactor lifetime (30 years (20–50 years)); discount rate (4% (2–15%)); controlled-release fertilizer price (US\$0.9 kg^{-1} N (0.8–1.0 kg^{-1} N)); and NBPT (*N*-(*n*-butyl) thiophosphoric triamide) urease inhibitor price (US\$14.4 kg^{-1} applied at 0.001 $kg\,kg^{-1}$ N)[18]. More details of the economic parameters in use can be found in Supplementary Tables 19–21. We conduct our own cost analysis for implementing each N management scenario (Supplementary Methods).

Reporting Summary. Further information on research design is available in the Nature Research Reporting Summary linked to this article.

Data availability

The datasets generated during this study are available from Princeton University's DataSpace repository (http://arks.princeton.edu/ark:/88435/dsp01pz50gz996). Source data are provided with this paper.

Code availability

NCAR Command Language (NCL) is used for analyses and visualizations in this study[50]. The NCL code is available from Princeton University's DataSpace repository (http://arks.princeton.edu/ark:/88435/dsp01pz50gz996).

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References

1. Shi, Y., Cui, S., Ju, X., Cai, Z. & Zhu, Y.-G. Impacts of reactive nitrogen on climate change in China. *Sci. Rep.* **5**, 8118 (2015).
2. Gu, B., Sutton, M. A., Chang, S. X., Ge, Y. & Chang, J. Agricultural ammonia emissions contribute to China's urban air pollution. *Environ. Sci. Technol.* **45**, 168–174 (2014).
3. Yu, C. et al. Managing nitrogen to restore water quality in China. *Nature* **567**, 516–520 (2019).
4. Cui, K. & Shoemaker, S. P. A look at food security in China. *NPJ Sci. Food* **2**, 4 (2018).
5. Zhang, X. et al. Managing nitrogen for sustainable development. *Nature* **528**, 51–59 (2015).
6. Galloway, J. N. et al. The nitrogen cascade. *Bioscience* **53**, 341–356 (2003).
7. Bai, Z. et al. China's livestock transition: driving forces, impacts, and consequences. *Sci. Adv.* **4**, eaar8534 (2018).
8. Bai, Z. et al. Livestock housing and manure storage need to be improved in China. *Environ. Sci. Technol.* **51**, 8212–8214 (2017).
9. Bai, Z. et al. Nitrogen, phosphorus, and potassium flows through the manure management chain in China. *Environ. Sci. Technol.* **50**, 13409–13418 (2016).
10. Brauer, M. et al. Ambient air pollution exposure estimation for the global burden of disease 2013. *Environ. Sci. Technol.* **50**, 79–88 (2016).
11. Burnett, R. et al. Global estimates of mortality associated with long-term exposure to outdoor fine particulate matter. *Proc. Natl Acad. Sci. USA* https://doi.org/10.1073/pnas.1803222115 (2018).
12. Huang, R. J. et al. High secondary aerosol contribution to particulate pollution during haze events in China. *Nature* **514**, 218–222 (2014).
13. Pinder, R. W., Adams, P. J. & Pandis, S. N. Ammonia emission controls as a cost-effective strategy for reducing atmospheric particulate matter in the Eastern United States. *Environ. Sci. Technol.* **41**, 380–386 (2007).
14. Banzhaf, S. et al. Impact of emission changes on secondary inorganic aerosol episodes across Germany. *Atmos. Chem. Phys.* **13**, 11675–11693 (2013).
15. Megaritis, A. G., Fountoukis, C., Charalampidis, P. E., Pilinis, C. & Pandis, S. N. Response of fine particulate matter concentrations to changes of emissions and temperature in Europe. *Atmos. Chem. Phys.* **13**, 3423–3443 (2013).
16. Wang, S. et al. Impact assessment of ammonia emissions on inorganic aerosols in East China using response surface modeling technique. *Environ. Sci. Technol.* **45**, 9293–9300 (2011).
17. *Three-Year Action Plan to Win the Battle for a Blue Sky* [in Chinese] (The National Development and Reform Commission of the State Council of China, 2018); http://www.gov.cn/zhengce/content/2018-07/03/content_5303158.htm?gs_ws=weixin_636662351573937202&from=timeline&isappinstalled=0
18. Xia, L. et al. Can knowledge-based N management produce more staple grain with lower greenhouse gas emission and reactive nitrogen pollution? A meta-analysis. *Glob. Change Biol.* **23**, 1917–1925 (2017).
19. *Action Plan for Zero Growth of Fertilizer Consumption by 2020* [in Chinese] (Chinese Ministry of Agriculture, 2015); http://jiuban.moa.gov.cn/zwllm/tzgg/tz/201503/t20150318_4444765.htm
20. *Action Plan for Manure Nutrient Usage (2017–2020)* [in Chinese] (Chinese Ministry of Agriculture, 2017); http://www.moa.gov.cn/nybgb/2017/dbq/201801/t20180103_6134011.htm
21. *Standard of Animal Waste Discharge to Waters (Draft Version for Public Review)* [in Chinese] (Ministry of Ecology and Environment of the People's Republic of China, 2011); http://www.mee.gov.cn/gkml/hbb/bgth/201103/W020110328492079276914.pdf
22. Paulot, F. & Jacob, D. J. Hidden cost of US agricultural exports: particulate matter from ammonia emissions. *Environ. Sci. Technol.* **48**, 903–908 (2014).

23. Liu, M. et al. Ammonia emission control in China would mitigate haze pollution and nitrogen deposition, but worsen acid rain. *Proc. Natl Acad. Sci. USA* **116**, 7760–7765 (2019).
24. Chen, X. et al. Producing more grain with lower environmental costs. *Nature* **514**, 486–489 (2014).
25. Velthof, G. L. et al. The impact of the Nitrates Directive on nitrogen emissions from agriculture in the EU-27 during 2000–2008. *Sci. Total Environ.* **468**, 1225–1233 (2014).
26. Zhang, L. et al. Agricultural ammonia emissions in China: reconciling bottom-up and top-down estimates. *Atmos. Chem. Phys.* **18**, 339–355 (2018).
27. Seinfeld, J. H., Pandis, S. N. & Noone, K. *Atmospheric Chemistry and Physics: From Air Pollution to Climate Change* (American Institute of Physics, 1998).
28. Bobbink, R. et al. Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. *Ecol. Appl.* **20**, 30–59 (2010).
29. Huang, X. et al. A high-resolution ammonia emission inventory in China.*Global Biogeochem. Cycles* **26**, GB1030 (2012).
30. Gu, B. J. et al. Atmospheric reactive nitrogen in China: sources, recent trends, and damage costs. *Environ. Sci. Technol.* **46**, 9420–9427 (2012).
31. Kang, Y. et al. High-resolution ammonia emissions inventories in China from 1980 to 2012. *Atmos. Chem. Phys.* **16**, 2043–2058 (2016).
32. Wu, L., Chen, X., Cui, Z., Zhang, W. & Zhang, F. Establishing a regional nitrogen management approach to mitigate greenhouse gas emission intensity from intensive smallholder maize production. *PLoS ONE* **9**, e98481 (2014).
33. *FAOSTAT Database* (Food and Agriculture Organization of the United Nations, accessed 13 March 2018); http://www.fao.org/faostat/en/#data
34. Li, Q. et al. A new urease-inhibiting formulation decreases ammonia volatilization and improves maize nitrogen utilization in North China Plain. *Sci. Rep.* **7**, 43853 (2017).
35. Strokal, M. et al. Alarming nutrient pollution of Chinese rivers as a result of agricultural transitions. *Environ. Res. Lett.* **11**, 024014 (2016).
36. Lu, X. et al. Exploring 2016-2017 surface ozone pollution over China: source contributions and meteorological influences. *Atmos. Chem. Phys.* **19**, 8339–8361 (2019).
37. Lin, J.-T. Satellite constraint for emissions of nitrogen oxides from anthropogenic, lightning and soil sources over East China on a high-resolution grid. *Atmos. Chem. Phys.* **12**, 2881–2898 (2012).
38. Zhu, L. et al. Sources and impacts of atmospheric NH_3: current understanding and frontiers for modeling, measurements, and remote sensing in North America. *Curr. Pollut. Rep.* **1**, 95–116 (2015).
39. Zhu, L. et al. Global evaluation of ammonia bidirectional exchange and livestock diurnal variation schemes. *Atmos. Chem. Phys.* **15**, 12823–12843 (2015).
40. Sutton, M. A. et al. Towards a climate-dependent paradigm of ammonia emission and deposition. *Phil. Trans. R. Soc. B Biol. Sci.* **368**, 20130166 (2013).
41. Piao, S. et al. The impacts of climate change on water resources and agriculture in China. *Nature* **467**, 43–51 (2010).
42. Erda, L. et al. Climate change impacts on crop yield and quality with CO_2 fertilization in China. *Phil. Trans. R. Soc. B Biol. Sci.* **360**, 2149–2154 (2005).
43. Chen, X.-P. et al. Integrated soil–crop system management for food security. *Proc. Natl Acad. Sci. USA* **108**, 6399–6404 (2011).
44. Oikawa, P. et al. Unusually high soil nitrogen oxide emissions influence air quality in a high-temperature agricultural region. *Nat. Commun.* **6**, 8753 (2015).
45. Riddick, S. N. et al. Estimate of changes in agricultural terrestrial nitrogen pathways and ammonia emissions from 1850 to present in the Community Earth System Model. *Biogeosciences* **12**, 3397–3426 (2015).
46. Kanada, M. et al. Regional disparity and cost-effective SO_2 pollution control in China: a case study in 5 mega-cities. *Energy Policy* **61**, 1322–1331 (2013).
47. Zhang, W. et al. Closing yield gaps in China by empowering smallholder farmers. *Nature* **537**, 671–674 (2016).
48. Development Research Center of the State Council of China & Shandong Supply and Marketing Cooperatives *Scale service and modernization of agriculture: supply and marketing cooperatives in Shandong Province Exploration Theory and Practice* [in Chinese] (China Development Press, 2015).
49. Baylis, K., Peplow, S., Rausser, G. & Simon, L. Agri-environmental policies in the EU and United States: a comparison. *Ecol. Econ.* **65**, 753–764 (2008).
50. NCAR Command Language Version 6.3.0 (UCAR, NCAR, CISL & TDD, 2020); http://dx.doi.org/10.5065/D6WD3XH5
51. Zhang, W. et al. New technologies reduce greenhouse gas emissions from nitrogenous fertilizer in China. *Proc. Natl Acad. Sci. USA* **110**, 8375–8380 (2013).
52. Wu, L. et al. Current potassium-management status and grain-yield response of Chinese maize to potassium application. *J. Plant Nutr. Soil Sci.* **176**, 441–449 (2013).
53. Huang, J., Hu, R., Cao, J. & Rozelle, S. Training programs and in-the-field guidance to reduce China's overuse of fertilizer without hurting profitability. *J. Soil Water Conserv.* **63**, 165A–167A (2008).
54. Wu, L. *Fertilizer Recommendations for Three Major Cereal Crops Based on Regional Fertilizer Formula and Site Specific Adjustment in China*. PhD thesis, China Agricultural Univ. (2014).
55. Chen, X. *Fertilizer Use Recommendations for China's Three Major Crops in Their Typical Agri-Ecological Zones* [in Chinese] (China Agricultural Press, 2016).
56. Wu, L. *Nitrogen Fertilizer Demand and Greenhouse Gas Mitigation Potential Under Nitrogen Limiting Conditions for Chinese Agriculture Production*. PhD thesis, China Agricultural Univ. (2014).
57. Wu, L., Chen, X., Cui, Z., Wang, G. & Zhang, W. Improving nitrogen management via a regional management plan for Chinese rice production. *Environ. Res. Lett.* **10**, 095011 (2015).
58. Mi, G. et al. Ideotype root architecture for efficient nitrogen acquisition by maize in intensive cropping systems. *Sci. China Life Sci.* **53**, 1369–1373 (2010).
59. Trenkel, M. E. *Slow- and Controlled-Release and Stabilized Fertilizers: An Option for Enhancing Nutrient Use Efficiency in Agriculture* (International Fertilizer Association, 2010).
60. Xia, L., Ti, C., Li, B., Xia, Y. & Yan, X. Greenhouse gas emissions and reactive nitrogen releases during the life-cycles of staple food production in China and their mitigation potential. *Sci. Total Environ.* **556**, 116–125 (2016).
61. Chen, S., Zhang, S., Sun, X. & Li, Y. Design and experiment of self-propelled high-ground-clearance spreader for paddy variable-rate fertilization [in Chinese with English abstract]. *Trans. Chin. Soc. Agricult. Eng.* **28**, 16–21 (2012).
62. Cao, Y. et al. Review on ammonia emission mitigation techniques of crop-livestock production system [in Chinese]. *Sci. Agricult. Sinica* **51**, 566–580 (2018).
63. Paulot, F. et al. Ammonia emissions in the United States, European Union, and China derived by high-resolution inversion of ammonium wet deposition data: interpretation with a new agricultural emissions inventory (MASAGE_NH3). *J. Geophys. Res. Atmos.* **119**, 4343–4364 (2014).
64. Morrison, H., Curry, J. A. & Khvorostyanov, V. I. A new double-moment microphysics parameterization for application in cloud and climate models. Part I: description. *J. Atmos. Sci.* **62**, 1665–1677 (2005).
65. Hong, S.-Y., Noh, Y. & Dudhia, J. A new vertical diffusion package with an explicit treatment of entrainment processes. *Mon. Weather Rev.* **134**, 2318–2341 (2006).
66. Chen, F. & Dudhia, J. Coupling an advanced land surface–hydrology model with the Penn State–NCAR MM5 modeling system. Part I: model implementation and sensitivity. *Mon. Weather Rev.* **129**, 569–585 (2001).
67. Li, M. et al. MIX: a mosaic Asian anthropogenic emission inventory under the international collaboration framework of the MICS-Asia and HTAP. *Atmos. Chem. Phys.* **17**, 935–963 (2017).
68. Janssens-Maenhout, G. et al. HTAP_v2.2: a mosaic of regional and global emission grid maps for 2008 and 2010 to study hemispheric transport of air pollution. *Atmos. Chem. Phys.* **15**, 11411–11432 (2015).
69. Guenther, A. et al. Estimates of global terrestrial isoprene emissions using MEGAN (Model of Emissions of Gases and Aerosols from Nature). *Atmos. Chem. Phys.* **6**, 3181–3210 (2006).
70. Cui, Z. et al. Closing the yield gap could reduce projected greenhouse gas emissions: a case study of maize production in China. *Glob. Change Biol.* **19**, 2467–2477 (2013).
71. Cui, Z. et al. Closing the N-use efficiency gap to achieve food and environmental security. *Environ. Sci. Technol.* **48**, 5780–5787 (2014).
72. Zhao, Y. et al. Atmospheric nitrogen deposition to China: a model analysis on nitrogen budget and critical load exceedance. *Atmos. Environ.* **153**, 32–40 (2017).
73. Gu, B. J., Ju, X. T., Chang, J., Ge, Y. & Vitousek, P. M. Integrated reactive nitrogen budgets and future trends in China. *Proc. Natl Acad. Sci. USA* **112**, 8792–8797 (2015).
74. Bouwman, L. et al. Exploring global changes in nitrogen and phosphorus cycles in agriculture induced by livestock production over the 1900–2050 period. *Proc. Natl Acad. Sci. USA* **110**, 20882–20887 (2013).
75. All China Marketing Research Co, *China Census Data by County 2000–2010* (2014); https://chinadatacenter.net/Data/ServiceContent.aspx?id=1622
76. Burnett, R. T. et al. An integrated risk function for estimating the global burden of disease attributable to ambient fine particulate matter exposure. *Environ. Health Perspect.* **122**, 397–403 (2014).
77. Xu, X., Chen, R., Kan, H. & Ying, X. Meta-analysis of contingent valuation studies on air pollution-related value of statistical life in China. *Chin. Health Res.* **1**, 64–67 (2013).
78. Xie, X. *The Value of Health: Applications of Choice Experiment Approach and Urban Air Pollution Control Strategy.* PhD thesis, Peking Univ. (2011).
79. Nielsen, C. P. & Ho, M. S. *Clearer Skies Over China: Reconciling Air Quality, Climate, and Economic Goals* (MIT Press, 2013).
80. Chinese National Development and Reform Commission *Information Summary on the Production Costs and Revenues of National Agricultural Products* [in Chinese] (China Statistics Press, 2016).
81. Ying, H., Ye, Y., Cui, Z. & Chen, X. Managing nitrogen for sustainable wheat production. *J. Clean. Prod.* **162**, 1308–1316 (2017).
82. Schiermeier, Q. Prices plummet on carbon market. *Nature* **457**, 365 (2009).

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Author contributions

Y.G., L.Z., T.D.S. and D.L.M. designed the study. Y.G., Y.C., M.Z., D.P. and J.Y. performed the research. L.W., Z.C., W.Z., F.Z., L.M., Y.S. and M.A.Z. contributed data and analytical tools. Y.G., T.D.S., L.Z. and D.L.M. analysed the results and wrote the manuscript.

Competing interests

The authors declare no competing interests.

Additional information

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Data collection	We utilized the official version of WRF-chem air quality model v.3.6.1 which is publicly available. Data collected from published literature include farmers' N application and crop yields at regional levels, effectiveness of N management on yield and reactive nitrogen (Nr) emissions, damage costs, labor costs, machine rental costs, NH3 and PM2.5 observations, etc. These information are all provided in the manuscript and Supplemental Appendix (text and tables) with sources clearly specified. Insights on what are the most promising and easy-to-implement management practices are obtained through a literature review.
Data analysis	All the plotting scripts are written in NCAR command language (NCL). Statistical analysis to evaluate model performance is written in Python and Matlab. Co-benefit anlaysis for yield and Nr emission pollution are conducted within EXCEL. All these code have been made available through the data statement.

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Study description	This manuscript conducts an integrated assessment for agricultural N management technologies in China: We start from scenario design where we identify the technologies to include through a literature review and based on Chinese policy documents which encourage adoption of certain technologies than others. We then evaluate the full-range of benefits associated with improved N management. The NH3 mitigation benefit is evaluated with a published NH3 emission model (properly referenced in our manuscript). The PM2.5 mitigation benefit is evaluated with WRF-chem air quality model. We conduct six sets of simulations, one baseline simulation for the year 2012 and five counter-factual scenarios where N management is implemented. Each set of simulation includes simulation for both January and July. We also evaluate the impacts of N management on crop yield, Nr emissions, greenhouse gas emissions utilizing published data. We monetize all environmental benefits. Costs for implementing each of N management in China are also estimated utilizing China-specific cost information.
Research sample	Our research is for China and thus based on existing knowledge of atmospheric chemistry and agronomy in China. The emission inventories utilized and data sources of yield, N application, etc are also provided in the manuscript.
Sampling strategy	N/A
Data collection	Data needed for designing N management technologies as well as for conducting cost-benefit analysis are conducted by Yixin Guo and Tim Searchinger through a literature review.
Timing and spatial scale	June 2017 - present The literature collected is not limited to when they are published but mostly have been pretty recent, i.e. 2010 and later. All the information collected are specifically for China.
Data exclusions	N/A
Reproducibility	Others will be able to reproduce our results
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