MITIGATING ENVIRONMENTAL AND HEALTH DAMAGES: OPPORTUNITIES FROM CHANGES IN AGRICULTURAL PRODUCTION AND FOOD CONSUMPTION PRACTICES IN CHINA

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Abstract

Unsustainable agricultural production and recent dietary shifts in China, have resulted in severe environmental damage including water and air pollution, climate warming and resource depletion as well as dietary health concerns. My dissertation identifies opportunities that changes in agricultural production and diets offer for mitigation of environmental and health damages. It includes three analytical chapters.

Chapter 2 analyzes technological, social and policy options for increasing nitrogen use efficiency (NUE) in Chinese agriculture based on a literature review and data collection. I find that adopting NUE-increasing fertilizer application management brings yield benefit that is much larger than other types of benefit (labor savings, fertilizer purchase savings, reactive nitrogen emission reduction, and greenhouse gas (GHG) emission reduction). We provide policy solutions that incentivize smallholders and large farmers to improve practices.

Chapter 3 analyzes the opportunities of improved production management in agricultural production to simultaneously address China's $PM_{2.5}$ air pollution and low nitrogen use efficiency (NUE). I identify five agricultural nitrogen management technologies and estimate the NH_3 emission reduction and associated $PM_{2.5}$ air quality improvements in 2012. Using additional data, I quantify the yield effects, GHG emission mitigation, nitrogen use efficiency (NUE) improvements, water pollution reduction, and acid rain and nitrogen deposition impacts. Midrange benefits from all strategies of US$30 billion/annum exceed costs of US$18 billion/annum.

Chapter 4 analyzes the implications of four potential future Chinese dietary choices. We find opportunities for improving $PM_{2.5}$ air quality, i.e., shifting from current diets to a *Soy Replace Red Meat Diet* and a *Lancet-EAT Diet* respectively reduces 37% and 18% national NH_3 emissions and up to 12ug/m^3 $PM_{2.5}$ pollution locally. We also find opportunities for improving dietary health, with negative or positive environmental impacts. Shifting from current diet to a *Chinese Dietary Guideline Diet*, *Lancet-EAT Diet* and *Soy Replace Red Meat Diet*, respectively, avoid 1.4, 1.1 and 0.3 million diet-related premature mortalities. However, shifting towards *Soy Replace Red Meat Diet* reduces food-related greenhouse gas emissions, water and land use; yet shifting towards *Chinese Dietary Guideline Diet* and *Lancet-EAT Diet* worsens these impacts.

Chapter 5 summarizes the findings of this dissertation and proposes future research.

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Chapter 1: Introduction

Food production has created multiple environmental challenges including low nitrogen use efficiency (NUE), climate warming, air and water pollution. Many of these challenges result from poor agricultural nitrogen management, i.e. lack of manure management and excess synthetic nitrogen (N) fertilizer application. Poor N management leads to substantial loss of N in reactive forms (Nr), including ammonia (NH_3) emissions to air which contribute to particulate matter air pollution harmful for human health (Burnett et al., 2018), nitrous oxide (N_2O) emissions which contribute to global warming, nitrate (NO_3^-) leaching and runoff to water systems which cause eutrophication (Camargo and Alonso, 2006), acidification and groundwater contamination (Galloway et al., 2003).

Global food production gains have been achieved at costs of worsened Nr pollution. Global synthetic N fertilizer consumption has increased by eight fold between 1960 and 2000 (Tilman et al., 2002). However, only ½ of fertilizer nitrogen actually ends up as crop yield (Cassman et al., 2002; Zhang et al., 2015), with the rest lost to the environment. An European N assessment estimated that total Nr loss from agriculture in Europe cause damage valued at 70-320 billion euro/yr, of which 75% is related to ground water nitrate contamination and air pollution (Sutton et al., 2011). A planetary boundary for N proposed to protect sensitive water systems have been exceeded by almost twice (Steffen et al., 2015).

Agriculture is also responsible for substantial water and land use, and land use change related greenhouse gas emissions (GHGs). Agriculture takes up 37% of global land area. The livestock sector, especially, utilizes 1/3 of global cropland for feed production and 1/3 of freshwater withdrawal. Direct emissions of methane (CH_4) from rice paddies and animal husbandry and nitrous oxide (N_2O) from fertilized soils are responsible for 13% of global GHGs. Accounting for indirect GHGs emissions associated with conversion of forests, savannahs, etc to cropland , agriculture is responsible for 25% of global GHGs emissions (Smith P., 2014).

Environmental damage from the agricultural sector will become increasingly severe, considering future population growth and shifts of diets towards more animal products. At present 800 million people still remain food insecure (FAO, 2016). Demand for crop calories and livestock products are projected to increase by 69% and 115%, respectively, by 2050 compared to the 2000s, accounting for population growth and dietary change (Bouwman, 2013; Searchinger et al., 2014).

Agricultural production-side technologies and consumer-side strategies can mitigate environmental damage from agriculture and generate co-benefits for public health. Production-side technologies include improved crop planting technologies, improved seed varieties, 4R principal for N fertilizer application (right source, right amount, right time, right place), and improved manure management in animal houses, during manure storage and disposal. For example, when nitrogen management technologies and planting techniques are simultaneously improved following suggestions provided by crop modeling tools, maize yields can double with no additional chemical inputs needed, according to experiments in China (Chen et al., 2011).

Manure management, such as manure acidification, manure collection in chicken houses with conveyer belts, manure injection into cropland, liquid manure treatment systems, can reduce emissions of reactive nitrogen. Often times these technologies benefit more than one environmental issue while increasing yields (Xia et al., 2017).

Consumer-side strategies include reducing food waste and shifting food consumption towards products with smaller environmental footprints. For example, poultry, pork and goat products have significantly smaller nitrogen use, GHG emissions and water withdrawal than beef; crops have smaller environmental footprints than diary and animal products (Eshel et al., 2014).

Dietary transitions also provide opportunities for improving human dietary health. In 2017, globally 11.7 million premature mortalities were attributed to dietary risks (Institute for Health Metrics and Evaluation (IHME), 2018). The leading dietary risk factors include low intake of fruits, vegetables and legumes and high intake of red meat which contribute to premature mortalities due to heart diseases, stroke, diabetes mellitus type II and site-specific cancers (World Cancer Research Fund/American Institute for Cancer Research, 2018). In particular, production and consumption of red meat neither benefits the environment nor health (Kim et al., 2017).

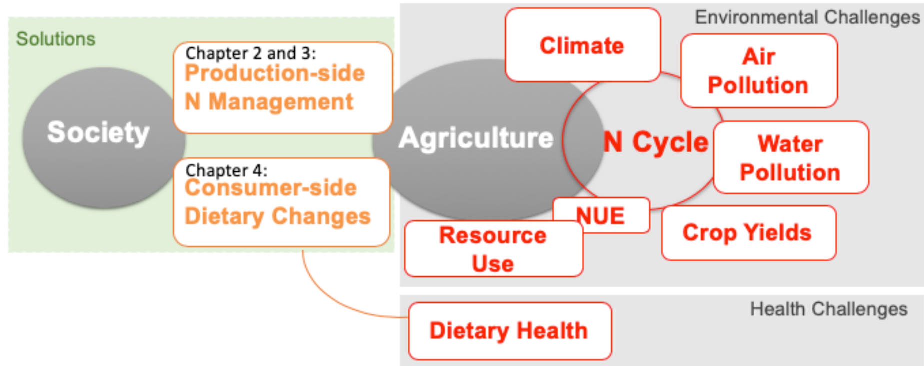

Figure 1: Overarching framework of this dissertation: addressing multiple environmental challenges and health challenges through production-side N management and consumer-side dietary changes. NUE is short for nitrogen use efficiency.

China is one of the world's largest food producer; substantial gains in food production in China has been through intensive nitrogen use. China has successfully produced one fourth of global grain, fed one fifth of global population, with only less than one tenth of global arable land (FAOSTAT, 2015). Grain production in China has increased by 74% during 1982-2017 time period (Cui et al., 2018). However, chemical fertilizers use has increased by 200% and per hectare N fertilizer input in China is four times of the global average. Intensive N input results in very low crop NUE of only 25-28% in China, compared to a world average of 42-47% and very high NUE of 68% achieved in U.S. and Canada (Lassaletta et al., 2014; Zhang et al., 2015). China is also the world largest milk and egg producer, contributing to 18-30% global manure N production (Bai et al., 2018). However, only 30% of manure N has been disposed with no harm to the environment.

Poor nitrogen management in China results in many environmental challenges. China remains a major hotspot for NH_3 emission, health-damaging $PM_{2.5}$ (fine particulate matter with aerodynamic diameters less than 2.5 μm) air pollution and eutrophication. China's very high NH_3 emissions, e.g. 11.7Tg in 2010, are ~5 times of that in the U.S., contributing to roughly 10% of $PM_{2.5}$ concentrations in the North China Plain, Yangtze River Delta and the Pearl River Delta overall (Wang et al., 2011), and even more (~18%) during bad haze events.

Chinese diets in the past decade have evolved in ways that are increasingly problematic for dietary health and the environment. Supply of animal protein has increased by 50% during 1998-2011 and share of calories from starchy food has decreased by 17% (He et al., 2019). This results in decreased undernourishment rates yet rapidly increasing obesity rates and all-cause mortality rates from dietary risks. China's all-cause premature mortality rates from dietary risks have increased from 0.14% in 1990 to 0.22% in 2017, higher than the global rate which stabilized around 0.14% (Global Burden of Disease Study, 2017). Environmental footprints of food production also increased. China's agricultural water footprint has tripled from 1961 to 2003 and land use has increased by 50% between 1961 and 2014 (He et al., 2019). Crop NUE has constantly decreased from 70% in 1960 to 35% in 2012 (Zhang et al., 2015).

The Chinese government has dedicated separate action plans to solve China's GHG emissions, $PM_{2.5}$ air pollution, agricultural water pollution and dietary health issues, yet potential of production-side and demand-side strategies to simultaneously address these issues from the agricultural sector have been neglected. Existing policies include the "Three-year Action Plan Fighting for a Blue Sky" (DRC, 2018) which for the first time identifies NH_3 mitigation from agriculture in efforts to address $PM_{2.5}$ air pollution, "Zero N fertilizer use increase by 2020" (MOA (Chinese Ministry of Agriculture), 2015) which is a first step towards a future reduction in N use, “Action Plan of converting animal manure to nutrients (2017-2020)”, a goal of “75% processing (processing indicated manure disposal without harm to the environment) rate of livestock and poultry waste” (MOA (Chinese Ministry of Agriculture), 2017), a drafted regulatory standard that tightens allowed NH_4^+-N and NO_3^--N contents in animal waste discharged to rivers (Ministry of Ecology and Environment of the P.R. China,), a goal of having 400 counties demonstrate ‘High Efficiency and High Yield Agriculture’, and a “Citizen Nutrition Plan for 2017-2030” which calls for enhancing public education and providing dietary guidelines for population with different needs (children, pregnant women, the elderly, people with diabetes, etc).

Agricultural pollution has not been regulated in many countries. Because it is a non-point source (Grosjean et al., 2016), policymakers have difficulty in quantifying pollution and monitoring effectiveness of changes in practices (USDA Agricultural Air Quality Task Force, 2014). In addition, Chinese policymakers highly prioritize food security. Currently the European Union regulates agricultural pollution through the EU Nitrate Directive, e.g. National Emission Ceilings (NEC) for NH_3 (EC, 2010). The U.S. Environmental Protection Agency requires that livestock farmers obtain licenses before discharging manure to rivers but does not set a quantitative allowance for the amount of manure discharged.

The environmental and health challenges posed by Chinese agriculture has cultivated a need for scientific research on the interconnections between agricultural practices and various environmental and health consequences, as well as co-benefits and trade-offs associated with strategies on the production and consumption side in addressing health and environmental issues.

Previous research explores the potential of various strategies in mitigating agricultural environmental damage, however, the design of mitigation scenarios remains simple, the tools utilized are not sophisticated, and the impacts examined are not inclusive. Bodirsky et al analyzes, in order to meet food demand in 2050, how strategies including increased NUE, reduced food waste and reduced consumption of animal products can reduce total reactive nitrogen burden for major regions in the world, using a socioeconomic model and N budgeting tools (Bodirsky et al., 2014). Pelletier et al analyzes effectiveness of dietary transitions including replacing beef consumption with poultry and replacing meat protein with soy in reducing GHGs, biomass appropriation and reactive nitrogen mobilization (Pelletier and Tyedmers, 2010). Bouwman et al analyzes the potential of various strategies including shifts of mixed/industrial animal production towards pastoral production, increased animal feed efficiency, improved manure storage, N fertilizers replaced with animal manure and partial replacement of beef with poultry in reducing nitrogen and phosphorous mobilization (Bouwman, 2013). Liu et al (2019) analyzes the potential of NH_3 emission mitigation for reducing $PM_{2.5}$ air pollution in China and implications for acid rain and nitrogen deposition. However, their work neglects opportunities of improved N management, through which NH_3 emission reduction must be realized, for reducing other reactive water pollution and increasing crop yield. There're a few more recent studies examining impacts of dietary transitions on the environment (Westhoek et al., 2014; Heller and Keoleian, 2015; Song et al., 2015; Behrens et al., 2017; Lei and Shimokawa, 2017), in some cases, they have explored how dietary changes can simultaneously obtain co-benefits of reducing GHGs and improving dietary health (Friel et al., 2009; Lock et al., 2010; Song et al., 2017; Springmann et al., 2018).

My dissertation provides the first attempt to comprehensively evaluate the full-range of dietary health and environmental impacts of various solutions from the perspective of both agricultural production and food consumption. The concepts of co-benefits are always emphasized since the reason why many environmental issues remain unsolved is uneven distribution of costs and benefits among different stakeholders. My analyses pay attention to localized environmental benefits such as air quality improvements, dietary health and yield increases. Localized benefits can incentivize changes of behavior, compared to environmental benefits dispersed at global and regional scale such as GHGs mitigation, water resource savings, etc.

Chapter 2 and 3 focus on opportunities from agricultural production-side nitrogen management in China. **Chapter 2** analyzes technological, social and policy options for increasing nitrogen use efficiency (NUE) in Chinese agriculture based on a literature review and data collection. I evaluate the effectiveness of various N fertilizer application technologies (reduced N fertilizer application, split application, efficient fertilizers, and N deep placement) in reducing emissions of reactive nitrogen (Nr) at an agroecological regional level for major grain crops (maize, wheat and rice). I monetize all the benefits of improved practices and compare them to various costs. I also collect data of China's farm sizes and adoption rate of various technologies. I then explore the socioeconomic barriers that prevent farmers from adopting environmentally-friendly practices. Lastly I provide policy remedies for improving practices of small and large farmers.

Chapter 3 analyzes the potential of agricultural nitrogen management opportunities in simultaneously mitigating China's severe $PM_{2.5}$ air pollution through reducing NH_3 emissions and securing food supply, reducing water pollution and emissions of GHGs. I designed five nitrogen management scenarios including: *Reduced N Application*, *Efficient Fertilizer, Machine Application*, *Manure Management*, and *Combined*. For each opportunity I estimate the $PM_{2.5}$ air quality improvements in 2012 using an NH_3 emission model and an air quality model, as well as yield effects, greenhouse gas (GHG) emission reductions, nitrogen use efficiency (NUE) improvements, water pollution reduction, and impacts on acid rain and nitrogen deposition using additional data. Across these scenarios I find that these approaches, applied to Chinese agriculture, can annually achieve 6.4%-34% NH_3 emission reductions nationally, 0-8 $\mu g/m^3$ reduction of secondary inorganic aerosols locally and ~5,000-30,000 avoided $PM_{2.5}$-related premature mortalities. I find multiple co-benefits from these approaches when applied to grain crops and livestock of: 36-52 Mega tonnes (7%-9%) increase in national crop yields, 0.01- 0.05 Tg N (9%-50%) decrease of N_2O emissions, 0.23-0.62 Tg N(14%-38%) decrease of NO_3-leaching, 0.06-0.14 Tg N (23%-56%) decrease of NO_3-runoff, 4.7 - 38 Mega tonnes CO_2-eq decrease of GHGs, and up to 2.5 Tg N (21%) decrease of N loss to water from animal farms. Total benefits from all N management strategies combined are US$30 (25; 50) billion/annum, compared to costs of US$18 (12; 27) billion/annum.

Chapter 4 examines historical dietary transitions in China and implications for health and the environment, and the potential of future dietary changes in generating environmental and dietary health benefits. We identify four potential Chinese dietary choices: *U.S. diet (US), Soy Replaces Red Meat (SRRM,* red meat protein replaced by soy bean protein*), Chinese Nutritional Guideline Diet (CNG)* and *Lancet-EAT Nutritional Recommendations (EAT,* non-political dietary recommendations*).* For each dietary choice compared to Chinese baseline diet in 2011, we evaluate environmental implications including NH_3 emissions, $PM_{2.5}$ air quality, food-consumption based lifecycle GHGs, total water footprint (TWF) and land appropriation (LA) and land-use carbon (LUCC) emissions, as well as health implications through exposure to $PM_{2.5}$ and nutrient intake. All future diets examined deliver dietary health benefits yet distinct environmental impacts. Two balanced diets, *Chinese Nutritional Guideline Diet* (*CNG*) and *Lancet-EAT Nutritional Recommendations* (*EAT*), each avoid 1.3 and 1.1 million diet-related premature mortalities, and two extreme diets, *U.S. Diet (US)* and *Soy Replace Red Meat Diet (SRRM),* respectively avoid 0.02 and 0.3 million premature deaths. However, *US* and *CNG* diets significantly increase ammonia emissions, $PM_{2.5}$ concentrations, land and water use, and greenhouse gas emissions (GHGs). *SRRM* decreases these impacts and *EAT* increases some (water and land use) but decreases others (ammonia emissions and $PM_{2.5}$). With *SRRM* providing a win-win for health and the environment, a combination of *SRRM* and *EAT* provides a potentially realistic future change in direction for Chinese diets to optimize health plus environmental objectives. Significant dietary health benefits achieved in the two balanced diets call for developments of production-side pollution-mitigating technologies.

Chapter 5 is the concluding chapter. In chapter 5, I summarize findings in Chapters 2-4 and explore future research needed for this area.

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Chapter 2: Technological, Social and Policy Options for Increasing Nitrogen Use Efficiency in China

1. Introduction

Chinese agriculture is confronted with a nitrogen use challenge. Intensive nitrogen fertilizer use has been essential to food security achieved in China. Chinese agriculture has successfully fed 22% of the world's population with 7% of world's arable land (FAO, 2018), yet consumed as much as 28.4% of world's N fertilizer in 2015 (FAO, 2015). Average N application rates in China are 227 kg N/ha, compared to 78 kg N/ha in the United States and 192 kg N/ha in the Netherlands (FAO, 2015). Nitrogen use efficiency (NUE, defined as the fraction of nitrogen harvested as yield divided by total nitrogen input including biological fixation, N deposition, N in irrigation, N fertilizer application and manure N applied to cropland) in China is quite low, i.e. 25-28%, compared to a world average of 42%-47% and high NUE achieved in Europe and North America of 52% and 68% (Lassaletta *et al.*, 2014b; Zhang *et al.*, 2015b). The marginal return of the last dollar spent on N fertilizer purchase is only \$0.7, indicating increased profits realized through reduced N use (Huang *et al.*, 2008).

Lack of manure nitrogen management remains another major problem. China became the largest meat and egg producer and the third largest milk producer globally in 2005. From 2005 to 2010, production of meat, eggs and milk has further increased by 14%, 28% and 38%, respectively (Yang, 2013). However, large amounts of manure excreted by animals have been poorly managed. In 2010, only 33% of manure N was recycled compared to

81% and 74% in the European Union and the United States (U.S.) (Bai *et al.*, 2016). The feed conversion efficiency (fraction of nitrogen in animal feed that ends up in animal products) in China was only 16% in 2010, compared to 22% achieved in the U.S. back in 1999 (Ma *et al.*, 2013).

Poor agricultural N management in China has resulted in various adverse environmental consequences, including soil acidification, NH_3 emissions and $PM_{2.5}$ air pollution, eutrophication and groundwater contamination. Cropland soil in China has been observed to be more acid than conventional soils (Guo *et al.*, 2010), as denitrification of ammonium-based fertilizers release H^+. Ammonia (NH_3) is emitted during N fertilizer application and manure disposal. The agricultural sector accounts for 93% of China's national NH_3 emissions which contributes ~10% to fine particle matter ($PM_{2.5}$) concentrations in the North China Plain, Yangtze River Delta and the Pearl River Delta [*S Wang et al.*, 2011] and even more (~18%) during haze events [*L Wang et al.*, 2013]. Livestock manure contributes to 60-78% and 20-74% of nutrient loads in Bohai Gulf and South China Sea (Strokal *et al.*, 2016), since 30-70% of manure N was directly discharged into water bodies across major Chinese sub-basins. Toxic algal blooms happened for 68 times in the year 2017, affecting a total area of 3679 km^2 (State Oceanic Administration, 2018). N leakage to groundwater causes frequent violations of national standard for nitrate in drinking water (Gao *et al.*, 2012).

Projections show that in the absence of improvements in agricultural nitrogen use, future environmental damage from the agricultural sector in China will worsen. In the business-as-usual case, China's N fertilizer consumption is projected to double in 2050 compared to 2012 (Gu *et al.*, 2015a).Riverine export of dissolved N will increase by up to 90% (Strokal *et al.*, 2017), GHGs emissions from manure will increase to 800 million tons CO_2-equivalent (Ranganathan *et al.*, 2016). In a case where China prioritizes food security and self-sufficiency over environmental protection, reactive nitrogen (Nr) losses from agriculture to air and water systems will increase by 45% and 55% between 2010 and 2050 (Wang *et al.*, 2017).

The world is also facing an agricultural nitrogen use challenge. A nitrogen planetary boundary, proposed to protect sensitive water systems, has been exceeded by almost twice (Steffen *et al.*, 2015). The European N assessment estimates a 70-320 billion euro/yr economic loss due to the damaging impacts of reactive nitrogen on air, water quality and climate (Sutton et al., 2011). Lessons learned from developed countries are that even if mitigation practices took place now, it would take decades for Nr pollution in water systems to be mitigated due to their very long residual time (Bouwman *et al.*, 2017).

The Chinese government has dedicated a number of policy targets and action plans to address the issue of nitrogen use. These include the Guideline for Live Pig Production (2016-2020) which requires small pig farmers located near the southern river network to either improve their manure handling or exit the market, a drafted livestock waste discharge[1] regulatory standard under public review which tightens the regulation of NH_4-

[1]http://www.mep.gov.cn/gkml/hbb/bgth/201103/W020110328492079265579.pdf

N emission from concentrated animal farms from the current 80mg/L to 40mg/L and for the first time regulates total N (<70mg/L), and several policy goals for the year 2020 including 'Zero Fertilizer Use Increase', '60% cropland area fertilized with machinery', '75% manure processing (manure disposed with no harm to the environment) rate', '50% manure nutrient recycled back to cropland', '40% fertilizer use efficiency for Jing-Jin-Ji region and Yantze River Delta region', and '95% small to medium scale breeding farms equipped with animal waste treatment facilities[2]'. In 2018, Chinese government for the first time encouraged reduction of agricultural NH_3 for reducing $PM_{2.5}$ air pollution in the 'Three-year Action Plan: Fighting to Protect the Blue Sky'[3].

Despite great policy relevance, research that analyzes the benefits and costs of unexploited N agricultural management technologies in China is lacking, neither are there sufficient insights provided for removing socioeconomic barriers against effective technology transfer.

Here, for the first time, we provide technical, social and policy solutions to the agricultural nitrogen use challenge faced by Chinese agriculture. We integrate agronomic field research, economic studies, behavioral science and agricultural policy analysis. We utilize both qualitative and quantitative research methods including unpublished data analysis, cost-benefit calculations, and scenario design (See Section 2 Methods). The rest of the chapter is organized as follows. Section 3 summarizes China's current status of

[2]http://english.gov.cn/policies/latest_releases/2017/06/12/content_281475684141592.htm
[3]ttp://www.gov.cn/zhengce/content/2018-07/03/content_5303158.htm?gs_ws=weixin_636662351573937202&from=timeline&isappinstalled=0

agricultural nitrogen management, providing a benchmark to evaluate effectiveness of potential improvements. Section 4 summarizes China's farm size distribution, which affects the magnitude of costs and benefits associated with farmers' improved practices. Section 5 investigates the costs and benefits (yield gains, reduced environmental damage and labor savings) of individual management technology (N fertilizer application amount reduction, machine application, new efficient fertilizer products, manure management technologies, etc). Section 6 identifies socioeconomic barriers against technology adoption. Section 7 provides policy recommendations in order to improve N management of smallholders and large farmers.

2. Methods

This paper combines quantitative and qualitative analysis tools as follows:

2.1 Literature review: extensive literature review has been conducted to summarize current state of knowledge about reactive nitrogen pollution (air, water, soil and GHGs) caused by Chinese agriculture, features of nitrogen fertilizer application for crops and vegetables/fruits, NUE in different agri-ecological zones, crop farm sizes and manure handling technologies.

2.2 Farm size and fertilizer application method dataset: We analyze an unpublished dataset, provided by Weifeng Zhang, providing crop- and region- specific farm sizes and percentage coverage of machine application (original data provided in Table S1). The dataset consists of 1000 farm surveys nationwide with survey methods provided in Zhang *et al.* (2017) where partial surveyed results were published. Machine application is estimated based on consumption of machinery oil.

2.3 Calculating benefits and costs of technologies

2.3.1 Reduced N Application Amount

We evaluate the economic and environment benefits associated when farmers reduce N application levels from current levels to profit-optimizing levels (profit = grain sales minus fertilizer and seed costs). These profit-optimizing levels are region- and major crop- (wheat, maize, and rice) specific, generated from a statistical analysis of 4660 nationwide field experiments published in Chen Xinping (2016), Wu (2014a) and Wu (2014b).

For one crop in one agri-ecological zone, the total benefit is calculated as follows:

$$Benefit = P_{crop} \cdot \Delta Yield - P_{fert} \cdot \Delta Q_N - DCost_{GHGs} \cdot LCE_{GHGs} \cdot \Delta Q_N - DCost_{NH3} \cdot \Delta E_{NH3} - DCost_{N2O} \cdot \Delta E_{N2O}$$

(1)

where $P_{crop} \cdot \Delta Yield$ is crop sales change, $-P_{fert} \cdot \Delta Q_N$ is nitrogen fertilizer purchase savings, $DCost_{GHGs} \cdot LCE_{GHGs} \cdot \Delta Q_N$ is reduced damage costs from reduced GHGs emitted during fertilizer manufacture, transportation and application, and $-DCost_{NH3} \cdot \Delta E_{NH3}$ $and - DCost_{N2O} \cdot \Delta E_{N2O}$ are reduced damage costs from reduced NH_3 emissions and NO_3^--leaching and runoff. We exclude damage cost of N_2O because it is already represented in GHGs calculation.

In detail, ΔQ_N is the recommended N fertilizer amount reduction, $\Delta Yield$ is the difference between crop yield under recommended N use levels and current N level. Both are provided in Chen Xinping (2016), Wu (2014a) and Wu (2014b). P_{crop} is wholesale grain price for the year 2015 from *Chinese NDRC (national development and reform council) (2016)*, i.e. 2.76/kg for rice, 2.33/kg for wheat and 1.88/kg for maize. P_{fert} is fertilizer prices from Xia *et al.* (2016), i.e. urea price of 4 ¥ kg^{-1} N. A currency conversion rate of 1USD=0.14rmb is used when needed which is representative of conversion rates from 2012 until now.

$DCost_{CO2}$ is the social cost of carbon – we use the price of CO_2 of 20.4 dollars per ton for European market in 2008 (Schiermeier, 2009). LCE_{CO2} is life-cycle emission of GHGs emitted during production, transportation and application of one ton of N fertilizer which is 13.5 tons CO_2-eq GHGs per ton of N (Zhang *et al.*, 2013). ΔE_{NH3}, and ΔE_{N2O} are the difference of Nr emissions under recommended N levels and current levels. They are calculated using crop- and region- specific relationships between N input levels and NH_3 emissions (linear, Cui et al, unpublished, see Table S2), and between N input levels and NO_3-leaching and runoff rates (exponential, (Cui *et al.*, 2013a) (Cui *et al.*, 2014), see Table S3). These relationships are obtained from a statistical analysis of thousands of filed experiments.

$DCost_{NH3}$ *and* $DCost_{NO3-leaching}$ are damage costs (including global warming, health, eutrophication, and acidification) of reactive nitrogen species, i.e. \$4.65/kgN for NH_3 and

\$1.32/kg N for NO_3^-. Monetizing the damage impacts of Nr can be of large uncertainty. Studies generate costs that range several orders of magnitude, e.g. from less than \$0.001/kgN to greater than \$10/kgN in one U.S. study for Nr (Keeler *et al.*, 2016), from 2 to 20 euro/kg N for health impacts of NH_3 and secondary NH_4^+ in one European study (Sutton *et al.*, 2011). Here we adopt China-specific values from Ying *et al.* (2017) (listed in Table S4) which summarizes previous Nr damage costs studies for China. We make additional adjustments to the public health damage of NH_3 which is originally estimated from European damaging cost by assuming mortalities from one unit of NH_3 emission is the same for China and Europe but only considering the differences in willingness to pay for mortality reduction and population density in China and Europe. This assumption is hardly true given $PM_{2.5}$ production from NH_3 emissions and dose-response relationships of $PM_{2.5}$ exposure is highly nonlinear. We utilize findings of a recent atmospheric chemistry study that reducing agricultural NH_3 emissions by 6.3×10^{10} molecules/(s·m^2) can reduce $PM_{2.5}$-related mortalities by 52000 people/yr in Europe and instead reducing agricultural NH_3 emissions by 1.68×10^{11} molecules/(s·m^2) can reduce $PM_{2.5}$-related mortalities by 106000 people/yr in East Asia (Pozzer *et al.*, 2017). We estimate that one unit of NH_3 emissions in Europe is associated with 1.3 times of mortality in Europe than that in China, thus we divide the original NH_3 damage cost by 1.3 to obtain our value.

2.3.2 Splitting N Fertilizer Application

To estimate the labor cost of a second application, we use time efficiencies of 3.75-15 h/ha for hand broadcasting and 1.5-7.5 h/ha for machine application (Table S5), labor costs of ¥75/d (Table S6) and fertilizer application service charge of ¥400/time (Table S12). We estimate that labor cost for a second application ranging ¥21- 400/ha.

Benefit of a second application includes yield increase and emission reduction of Nr pollutants:

$$Benefit = P_{crop} \cdot Yield_{current} \cdot R_{yield} - DCost_{NH3} \cdot E_{NH3_{current}} \cdot R_{NH3} - DCost_{N2O} \cdot \cdot E_{N2O_{current}} \cdot R_{N2O} - DCost_{NO3} \cdot E_{NO3_{current}} \cdot R_{NO3}$$

(2)

where R_{yield}, R_{NH3}, R_{N2O}, and R_{NO3} are grain (maize, wheat and rice) averaged yield increase factor (5.9%), NH_3 emission reduction factor (-32%), N_2O emission reduction factor (-5.4%) and NO_3-leaching and runoff reduction factor (-25%) from Xia *et al.* (2017). E_{NH3} and E_{NO3} are current emission levels of Nr under a certain N application level and $DCost_{NH3}$ and $DCost_{NO3}$ are damage costs of NH_3 and NO_3 emissions, calculated as in section 2.5. E_{N2O} are current emission level of N_2O calculated using statistical emission model by Cui *et al.* (2013a) and Cui *et al.* (2014) in Table S3. $DCost_{N2O}$ is damage cost of N_2O from Ying *et al.* (2017) (Table S4). Current yield and N application levels are as in Section 2.5.

2.3.3 Machine Application

$$Benefit = Labor_{sav} + P_{crop} \cdot Yield_{current} \cdot R_{yield} - DCost_{NH3} \cdot E_{NH3_{current}} \cdot R_{NH3} - DCost_{N2O} \cdot\cdot E_{N2O_{current}} \cdot R_{N2O} - DCost_{NO3} \cdot E_{NO3_{current}} \cdot R_{NO3}$$

(3)

It is difficult to estimate machine use cost by upfront purchasing cost and depreciation cost. Instead, we reference machine rental charges, i.e. ¥150/d in Chongqing and estimate machine rental cost is ¥150/ha.

2.3.4 Efficient Fertilizers

We design four scenarios where new efficient fertilizers, in their most suitable regions, replace conventional fertilizers with N application rate unchanged. For each scenario, the total benefit is as follows:

$$Benefit = P_{crop} \cdot Yield_{current} \cdot R_{yield} - DCost_{NH3} \cdot E_{NH3_{current}} \cdot R_{NH3} - DCost_{N2O} \cdot \cdot E_{N2O_{current}} \cdot R_{N2O} - DCost_{NO3} \cdot E_{NO3_{current}} \cdot R_{NO3} - (P_{fert_{efficient}} - P_{fert_{conv}}) \cdot Q_{N\_conv} \quad (4)$$

where R_{yield}, R_{NH3}, R_{N2O}, and R_{NO3} are crop- and new efficient fertilizer type- specific yield and Nr emission change rate factors from Xia et al (2017). E_{NH3} and E_{NO3} are current emission levels of Nr under a certain N application level, calculated through statistical emission models as in section 2.5. E_{N2O} are current emission level of N_2O calculated using statistical emission model as in Table S3. $-(P_{fert_{efficient}} - P_{fert_{conv}}) \cdot Q_{N\_conv}$ is the increasing costs of fertilizer purchase calculated with fertilizer prices (Table S7) and current N application rates as in section 2.5.

3. Agricultural nitrogen management in China at present

3.1 N Fertilizer Application Amount

N fertilizer use in China is intensive, resulting in low nitrogen use efficiency. 28.9 million tons of nitrogen fertilizer was added to Chinese cropland in 2010. In addition, 4.6 million tons of nitrogen were fixed biologically and air deposited another 2.7 million tons (Gu *et al.*, 2015a). 30-35% of the summed N inputs ended up in crops (Lassaletta *et al.*, 2014a); another study found 25% (Zhang *et al.*, 2015b). Table 1 summarizes N application amount, yield productivity and N excess for crops and vegetables.

Table 1. Nitrogen fertilizer consumption and yield productivity for main grain crops, vegetables and fruits in China.

	Wheat	Rice	Maize	Vegetables	Fruits
Share of Nitrogen Fertilizer Consumption* (Wu, 2014a)	14%	16%	19%	20%	15%
Nitrogen Fertilizer Application Rates (kg N/ha) (Zhang *et al.*, 2013)	197 ±134	209 ± 140	231 ± 142	383 ± 263	550 ± 381
Nitrogen Balance** (kg N/ha) (Zhang *et al.*, 2013)	89	102	87	357	464
Partial Productivity of Nitrogen Fertilizer*** (kg/kg) (Wu, 2014a)	27	37	34	94	45

*Total N fertilizer consumption was 28.9 million tons N in 2010

** Nitrogen Balance = Synthetic fertilizer N + manure N – aboveground uptake

*** Partial Productivity of Nitrogen Fertilizer = mass of yield/mass of nitrogen in fertilizers

The over use of N is much more severe for vegetables and fruits than grain crops. Rice, wheat and maize, three major crops in China, respectively consumed 16%, 14% and 19% of total nitrogen fertilizers during the time period of 2007-2009 (Wu, 2014b), with cultivating areas of 30.2 million ha, 24.1 million ha and 36.7million ha in 2016 ((EOCSSB), 2016). Nitrogen application rates are 209 ± 140 kg N/ha for rice, 197 ±134 kg N/ha for wheat and 231 ± 142 kg N/ha for maize in 2009 according to a national survey of 20000 farms (Zhang *et al.*, 2013). Crop excess N (manure N + fertilizer N –

yield N) is estimated to be 80-100 kg N/ha (Zhang *et al.*, 2013). Figure S1 provides basic information of cropping structure and fertilizer application. Instead, vegetables and fruits, respectively consumed 20% and 15% of total nitrogen fertilizers, have cultivating areas of 15 million ha and 8.9 million ha. N application rates are 550 $\pm$ 381 kg N/ha for fruits and 383 $\pm$ 263 kgN/ha for vegetables in 2009 (Zhang *et al.*, 2013). Fruits and vegetables have higher excess nitrogen, i.e. 464 kg N/ha and 357 kg N/ha (Zhang *et al.*, 2013).

The effectiveness of N fertilizer inputs in boosting yields has been much lower in China than that in more developed countries, e.g. the U.S.. A widely-used metric, i.e. partial productivity of N (PFP_N, the fraction of fertilizer N that ends up in crop yields), indicates the effect of N fertilizer in increasing yield. PFP_N for maize is 67 kg/kg in the United States in 2005 and 34 kg/kg in China during 2007-2009. PFP_N for fruits and vegetables are respectively 220 kg/kg and 195 kg/kg for the United States in 2009-2010 and only 45 kg/kg and 94 kg/kg for China during 2007-2009 (Wu, 2014a).

3.2 N Fertilizer Application Frequency

Management regarding frequencies of N application is less of a concern in China compared to excess N application, yet there's room for improving the timing of application and the partitioning between N applied as starter fertilizer and dressing. Chinese farmers have recognized the importance of applying dressings, the majority apply dressings, although sometimes farmers' dressing dates are several days earlier or later than the most suitable time (Zhang *et al.*, 2016). One-time application issue prevails in some regions including arid northwestern China, double-cropped rice region in the south, wheat production in the lower basin of the Yantze River and maize in the northeast. In addition, farmers tend to use more (less) N as starter fertilizer (dressing) than needed. They will need to decrease the ratio of starter fertilizer N to dressed N in order to increase crop yields (Xia, 2011; Liu *et al.*, 2012).

3.3 N Fertilizer Application Method

Machine application is widely used for certain crops and regions in China, but its coverage needs to be increased further to achieve Chinese government's policy goal of having 40% cropland soils fertilized with machine application. More than half of rice production regions, most maize production regions and some wheat production regions in China are still dominated by hand application, according to our analysis of an unpublished dataset provided by Weifeng Zhang (see Methods). Machine application of fertilizer is above 50% in wheat production regions except for Sichuan province, North China Plain, Jiangsu and Zhejiang provinces. All maize production regions have low machine adoption rate except for Northeast China, North China Plain, Neimenggu, Shanxi, and Shaanxi provinces (Figure 2).

Machine application is extremely time-efficient compared to hand application, although geography can be a limitation. Most farmers spend more than 1h/mu applying fertilizer by hand compared to less than 0.3h/mu applying fertilizer by machine. Farmers in Qinghai, Jiangxi and Guizhou provinces spend extremely long hours hand applying fertilizers, >10h/mu, due to hilly geography conditions.

Starter fertilizers, compared to dressing fertilizers, are more often applied by machines; maize and wheat, compared to rice, are more easily fertilized with machine application. In North China Plain, farmers use machines to apply starter fertilizer together with seeds or simultaneously conduct tillage. In rare cases they apply dressing fertilizer also with machine. Similar to wheat, when machine is used for starter fertilizer application, seeding and tillage are finished too and dressing is rarely applied with machines. Less than 10% farmers use machine application of N for rice, except for Northeastern China where plot is vast and machinery levels are advanced (Figure 3). The more southern, the more hours farmers spend applying fertilizer in field by hand, both due to geography and the many times that dressing is applied.

Recent innovations in agricultural mechanics provide opportunities for farmers' to more easily use dressing machines. High-ground-clearance spreader for paddy rice (Chen *et al.*, 2012) , high-ground-clearance spreader for maize's dressing, and liquid fertilizer machine, have made dressing application using machines more feasible.

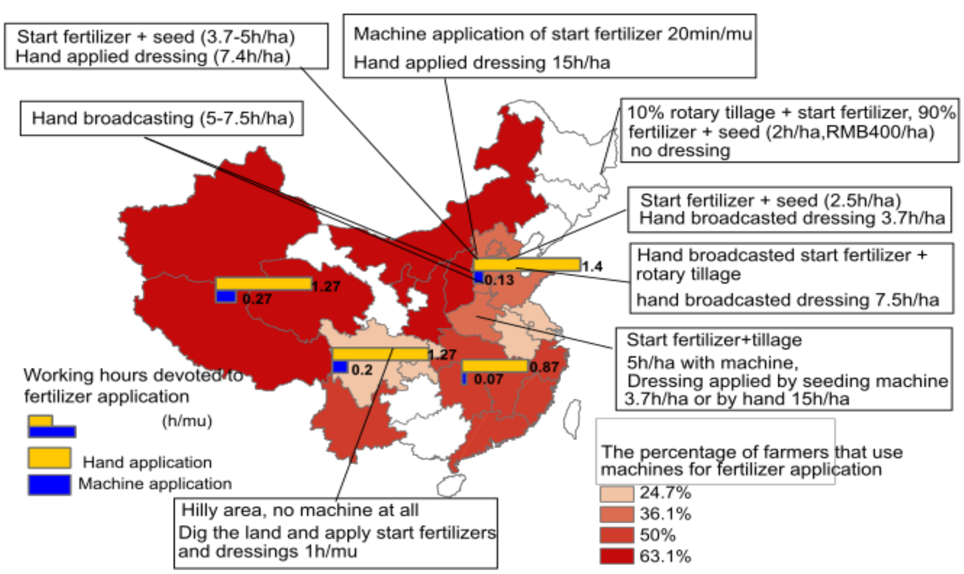

Figure 1. Nitrogen fertilizer application methods for wheat production regions in China. Percentage of farmers that use machines to apply fertilizers in each production region (shown in red shades) and their annual working hours devoted to fertilizer application (Unit: hour/mu, mu is an unit for land area widely used in China, 1ha=15mu) by hand (bars in yellow) and by machine (bars in blue). Text in boxes are methods of starter fertilizer and topdressing application methods and time needed at locations indicated by the lines.

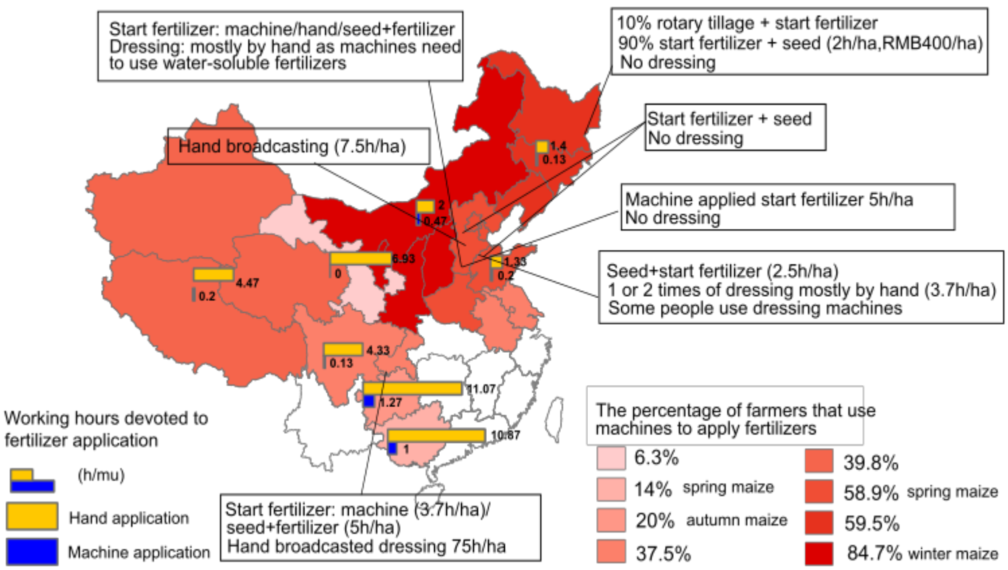

Figure 2. Nitrogen fertilizer application methods for maize production regions in China. Percentage of farmers that use machines to apply fertilizers in each production region (shown in red shades) and their annual working hours devoted to fertilizer application (Unit: hour/mu, mu is an unit for land area widely used in China, 1ha=15mu) by hand (bars in yellow) and by machine (bars in blue). Text in boxes are methods of starter fertilizer and topdressing application methods and time needed at locations indicated by the lines.

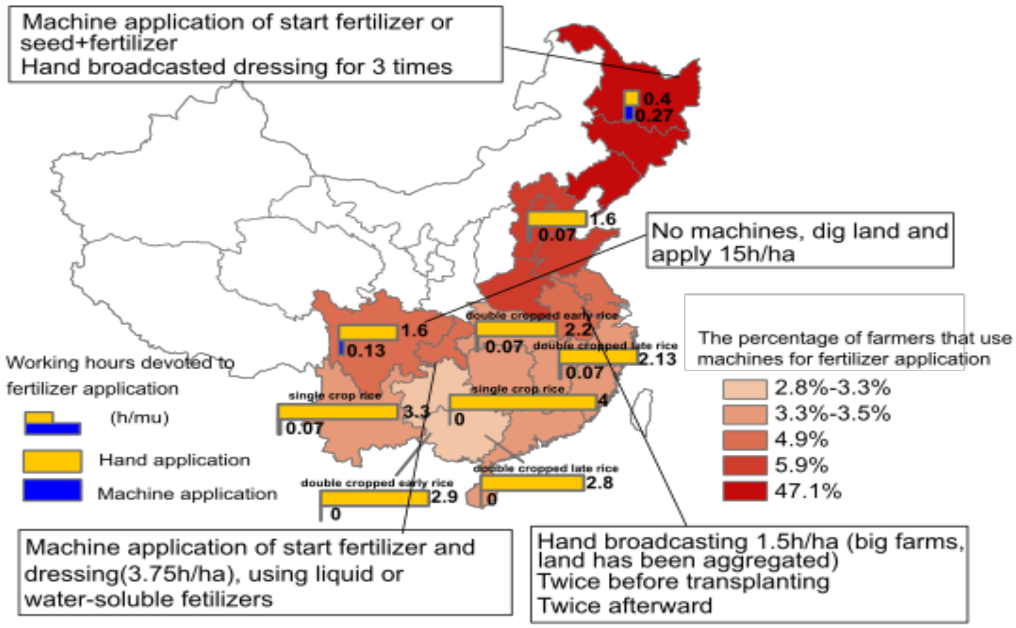

Figure 3. Nitrogen fertilizer application methods for rice production regions in China. Percentage of farmers that use machines to apply fertilizers in each production region (shown in red shades) and their annual working hours devoted to fertilizer application (Unit: hour/mu, mu is an unit for land area widely used in China, 1ha=15mu) by hand (bars in yellow) and by machine (bars in blue). Text in boxes are methods of starter fertilizer and topdressing application methods and time needed at locations indicated by the lines.

3.5 Manure Production and Manure Management in China

Geographically a few provinces are the most concentrated with animals, e.g. Shandong and Chongqing, followed by other provinces such as Hunan, Guangdong, Fujian, Henan, Anhui, Zhejiang, Hubei, Liaoning and Jilin. These provinces are hotspots for manure production and pollution, according to nationals statistics in 2016 (2-4 LU/h) (Table S8)((EOCSSB), 2016) (Ma *et al.*, 2012).

Animal manure can receive treatments in animal houses, during storage, treatment or disposal processes, yet its management in animal houses and during storage are the most important to avoid substantial Nr emissions to air, soil and water. Nr emissions during animal housing and direct discharge of manure are the two most important N loss processes (Zhao *et al.*, 2017). 39% of manure N is volatilized as NH_3 during housing and storage, amounting for 6.7Tg NH_3-N emissions in 2010 (Bai *et al.*, 2016).

Cattle, among other animals, contributes the most to manure N excretion, yet cattle manure is the least managed. Beef cattle contributes to ½ of manure N production in China, pigs 1/6, sheep and goat another 1/6, poultry and dairy cattle in 2010 (Bai *et al.*, 2016; Bai *et al.*, 2017). Manure N recycling rates is the lowest for dairy cows, only 3%, compared to 10% for chicken manure and 43% for pig manure (Ju *et al.*, 2005). N loss per kg of animal product is the largest for diary, followed by pig and poultry (Zhao *et al.*, 2017).

A systematic summary of typical manure management practices by animal system and region, and the nitrogen use efficiencies therein, is not possible due to limited research. A few case studies indicate large variations in management. For example, a survey of 301 representative animal farms in North China Plain found distinct floor types of animal housing (concrete v.s. slatted), various manure collection (separate or mix the solid and liquid manure), storage (covered v.s. uncovered; underground v.s. aboveground) and treatment methods (anaerobic fermentation, composting, oxidation pond, gasification and combustion for ash based fertilizers, pyrolysis for biochar, digestion[4]) (Zhao *et al.*, 2017). Another survey of 23 milk cattle farms show that the peri-urban animal farms (usually with > 100 cows) may systematically have a higher nitrogen use efficiency of 0.273 (defined as the fraction of N in feed excreted as milk) than smallholder subsistence farms (usually with < 10 cows, NUE=0.229) and cooperative farms (usually >100 cows, managed by companies and owned by multiple persons, NUE=0.224) (Wang *et al.*, 2014).

4. Crop Farm Sizes in China

Smallholder farmers in China account for 85% of total farmers and cultivate half of the cropland. Figure 4 summarizes China's farm size distribution. The majority of Chinese farmers have farm sizes smaller than 1ha (Chinese NDRC (national development and reform council), 2016) and all together cultivate less than 50% of total cropland (Figure 4)[5]. As of 2013, 85% of farmers (224 million) own cropland smaller than 10 mu (1ha=15mu); around 1.2% of farmers (3.1 million) own land larger than 50 mu (Figure 5) (Chinese NDRC (national development and reform council), 2016).

[4] Personal communications with Yong Hou

[5] Using data of farmers' population for each farm size interval, total number of farmers and total cropland area (Figure 5), we can estimate that farms with sizes smaller than <10mu altogether cannot take up more than 46% of cropland.

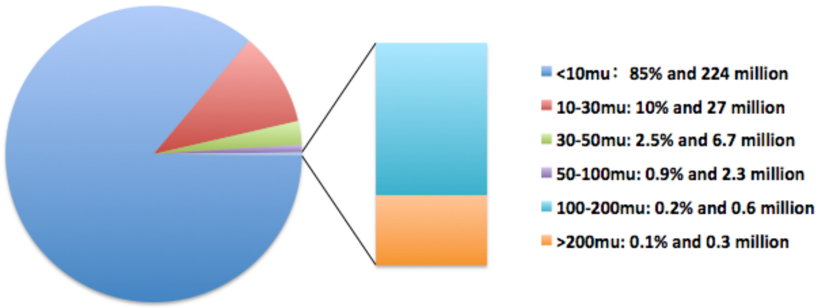

Figure 4. The shares and absolute numbers of Chinese farmers who cultivate farms of various sizes in 2013 (Chinese NDRC (national development and reform council), 2016).

Geographically farm sizes vary greatly. Crop- and region- specific farm sizes are presented in Figure 5, 6 and 7, integrating analysis of an unpublished farm size dataset (See Methods). In general, farms in the west and Northeast are larger than those in central and southern China. For example, wheat farms in the Northwest are nearly 3ha yet elsewhere are mostly below 1ha. Maize farms in Xizang province is 3.6 ha yet elsewhere mostly below 1ha. Rice farms on average are the smallest of all crops but have the largest South-North contrast. Rice farms in south China have an average size of only 0.06ha and are mostly <1mu; rice farms in Northeast China are several hundreds mu.

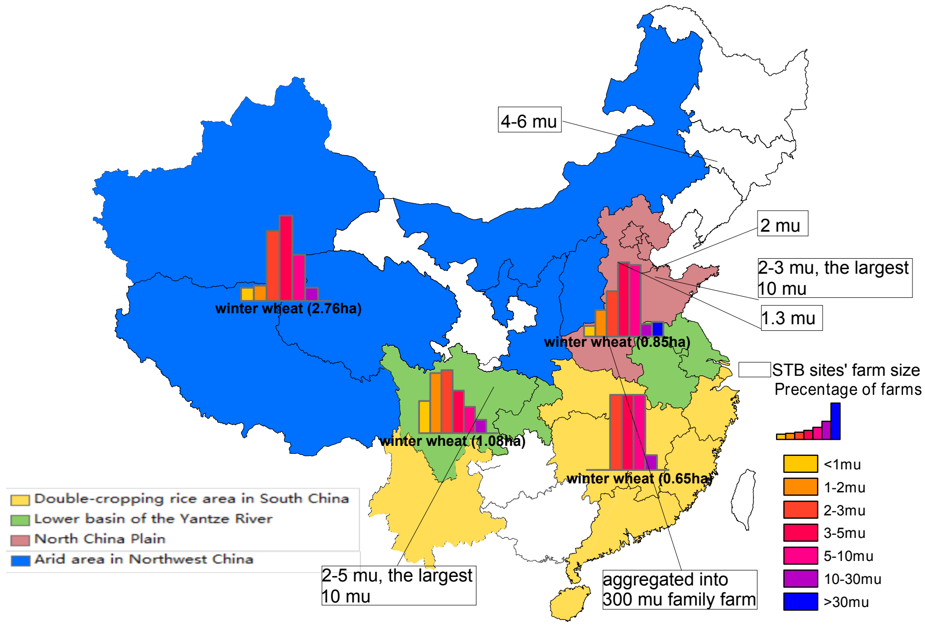

Figure 5. Farm size distribution (in colored bars) and average farm sizes (black numbers in bracelets below the bars; in ha) for wheat production systems in China (definition of ag-ecological zones adopted from Zhang (Zhang *et al.*, 2017)).

China's cropping agriculture in the past thousands of years has been precision agriculture with intensive labor inputs. However, this can no longer sustain in the modern society since increasing rural population has left the countryside for city jobs. Agriculture needs to transform itself into a modern one which utilizes mechanization and capital investments to achieve economies of scale. Rural population can decrease but their management skills have to improve. Experiences in developed countries shows that agricultural population is destined to decrease as agricultural labor shifts to the service sector (Lipton, 1977; Varshney, 2014).

Most recently, ~15-30% of cropland in China has been transferred, creating large farms which can potentially benefit from economies of scale. The Chinese government recently separated cropland cultivating right from contracting right. It also encouraged farmers to transfer the cultivating right to others if they are not cultivating, and has set economic incentives and institutional arrangements. Farmers can get preferrable loans and subsidies to pay for land rents, and discounted prices for buying yield insurances and agricultural inputs (irrigation water, drainage and storage infrastructure and machinery) (Huang and Ding, 2016). Land transfer centers have been established at village and township levels to help match demand for land with supply and to solve possible conflicts. Table 2, adopted from (Zhao *et al.*, 2017a), summarizes land transfer activities in 2011 and 2013. In 2013, 53 million rural households rent out the cultivating rights of their land, transfering 26.1% cropland in eastern China and 30.5% cropland in middle China. Major recipients are individuals (71% of transferred land), farmers' professional cooperatives (FPCs) and agribusinesses (Huang and Ding, 2016). In Northeast and North China, non-individual stakeholders managed 25% of cropland, with average farm sizes above 100 ha.

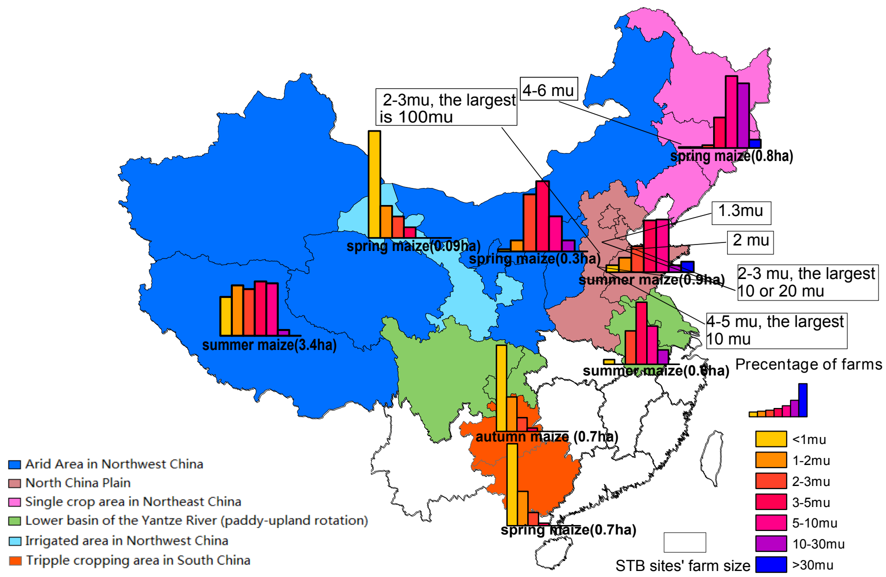

Figure 6. Farm size distribution (in colored bars) and average farm sizes (black numbers in bracelets below the bars; in ha) for maize production systems in China (definition of ag-ecological zones adopted from Zhang (Zhang *et al.*, 2017)).

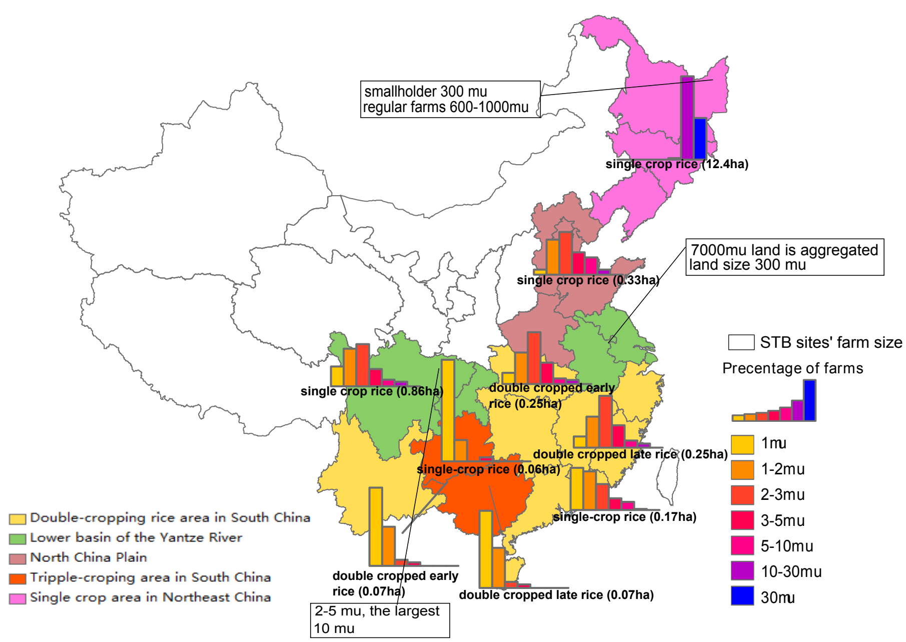

Figure 7. Farm size distribution (in colored bars) and average farm sizes (black numbers in bracelets below the bars; in ha) for rice production systems in China (definition of ag-ecological zones adopted from Zhang (Zhang *et al.*, 2017)).

Table 2: **Land transfer in eastern, middle and western China in 2011 and 2013. This table is adopted from (Cheng *et al.*, 2010)(Zhao *et al.*, 2017a)** [6]**.**

	2011			2013		
	Eastern China	Middle China	Western China	Eastern China	Middle China	Western China
Total land area contracted to farmers (0.1billion mu)	3.55	5.15	4.08	3.59	5.25	4.43
Transferred land area (0.1billion mu)	0.66	1.04	0.58	0.94	1.61	0.86
Land transfer percentage	18.56	20.20	14.25	26.06	30.64	19.53

Unfortunately, farmers' poor management knowledge at present have prevented the newly-emerged large farms from achieving increasing economies of scale. National statistics reports that large-scale farmers have lower net revenue per hectare than small

[6] http://www.aisixiang.com/data/97233-2.html

farmers, i.e. ¥714.17/mu versus ¥2614.42/mu (Chinese NDRC (national development and reform council), 2016). Data collected for Northeast and North China shows that productivity and profit per farm first increases as farm size increases and then decreases (Huang and Ding, 2016), with the causal mechanisms unclear. Many large farmers lack local-specific planting techniques because their previous career has been in real estate and business sectors.

Separate policy incentives are needed to improve N management practices in small (>200 million small farmers, <1ha) and large farms (3 million large farms). Firstly, small and large farmers are equally important since they each roughly account for half of cropland. Secondly, although small and large farms are confronted with the same learning cost of new technologies and the same environmental and yield benefit per hectare land($/ha), large farms benefit substantially more from technological improvements due to economies of scale and greater resilience against risks. Thirdly, considering limited fiscal budget, educating small number of large farmers are a lot cheaper than educating large numbers of small farmers. We provide more detailed policy remedies in Section 6.

5. Technological Solutions to the NUE Challenge and Their Costs and Benefits

5.1 N Fertilizer Strategies

5.1.1 Reduce Over-Application

There's great potential to reduce N use to secure food supply while decreasing environmental impacts. China's national nitrogen fertilizer input could be reduced by 28% with current management practices unchanged, and yields won't decrease, according to a meta-analysis of 232 experiments nationwide for major crops [*L Xia et al.*, 2017]. In regions where historical heavy nitrogen inputs have led to substantial nitrate accumulation in soil, even more ambitious reduction is possible, e.g. 70% in maize systems in Shaanxi province (Zhang *et al.*, 2015a).

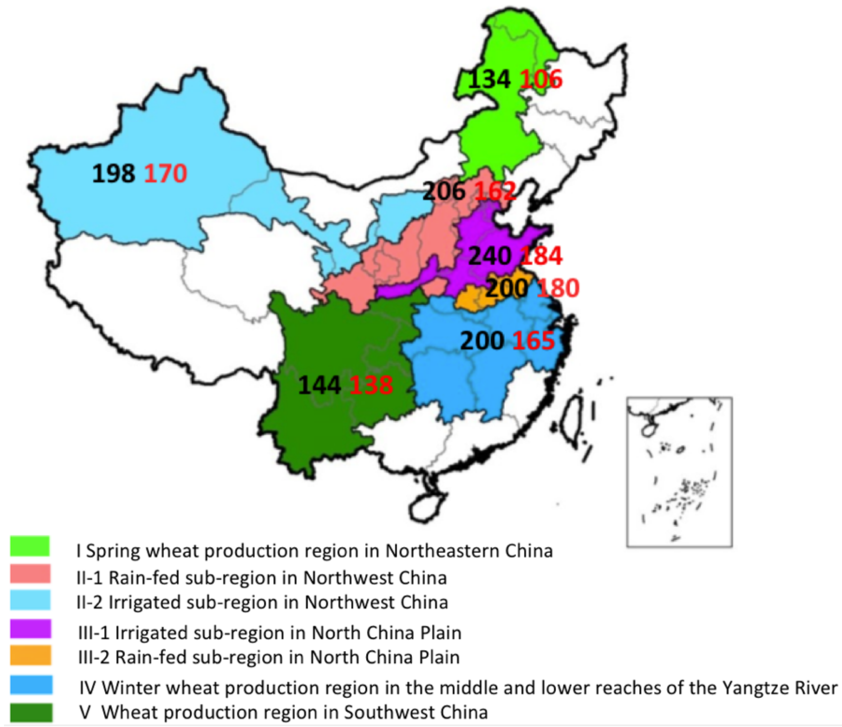

Figure 8. Current Chinese farmers' nitrogen fertilizer application amount (numbers in black, data source (Chen Xinping, 2016)) and recommended N input amount that optimizes farmers' profits (red, derived from 1757 wheat experiments, data source (Wu, 2014b; Wu, 2014a)) in major wheat production agri-ecological zones in China. Map is appropriated from [*WU*, 2014].

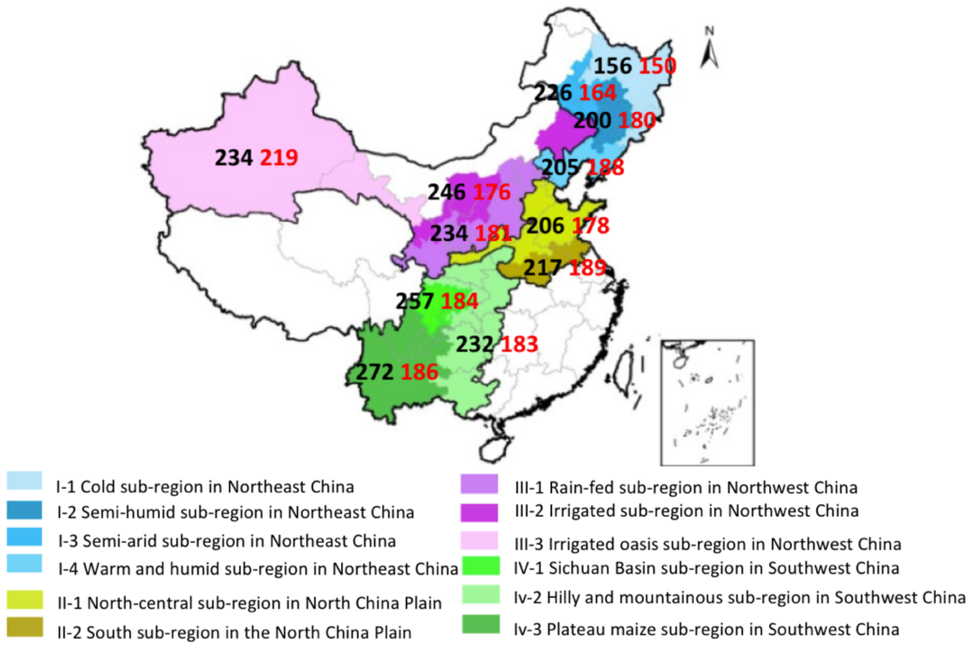

Figure 9 Current Chinese farmers' nitrogen fertilizer application amount (numbers in black, data source (Chen Xinping, 2016)) and recommended N input amount that optimizes farmers' profits (red, derived from 1726 national maize experiments, data source (Wu, 2014b; Wu, 2014a)) in major maize production agri-ecological zones in China. Map is appropriated from [*WU*, 2014].

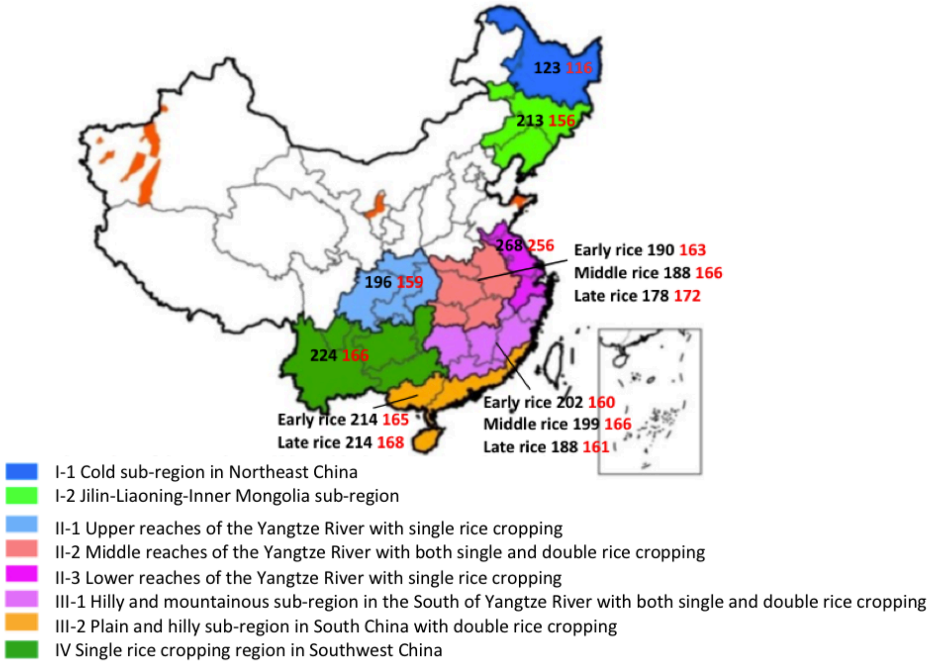

Figure 10 Current Chinese farmers' nitrogen fertilizer application amount (numbers in black, data source (Chen Xinping, 2016)) and recommended N input amount that optimizes farmers' profits (red, derived from 1177 national rice experiments, data source (Wu, 2014b; Wu, 2014a)) in major rice production agri-ecological zones in China. Map is appropriated from [*WU*, 2014].

Here we focus on crop- and region- specific N use recommendations that can optimize farmers' net returns (i.e. profit, grain sales minus fertilizer and seed costs) and calculate the associated environmental and economic benefits (see Methods). Profit-optimizing N input levels strike a balance between environmental integrity and farmers' profitability. These levels are provided in Chen Xinping (2016), Wu (2014a) and Wu (2014b) and selected through statistical analysis of 4660 nationwide field experiments (Figure 8, 9, and 10). Figure 11 and 12 present associated monetarized benefits, i.e. environmental benefits from reduced NH_3 emission, NO_3-leaching and runoff and GHGs emission, as well as economic benefits of N fertilizer purchase savings, and crop yield increases.

In order to achieve profit maximization, fairly large N reductions are needed for a number of regions: maize in Yunnan Province (a 86 kgN/ha reduction, which is 31.6% of the current application level), rice in upper Yantze River Basin (a 58 kgN/ha reduction, which is 25.9% of the current application level), early rice in Lower Yantze River Basin (a 42 kgN/ha reduction, 20.8% of the current application level), and wheat in North China Plain (a 56 kgN/ha reduction, which is 27% of the current application level).

Yield increase benefit will be the primary driver for farmers to reduce N use. It is one order of magnitude larger than all other types of benefits for all crop production regions (Figure 11). For example, the largest yield benefit achieved is $1177/ha for wheat production in III-1, $766 for maize production in I-3, and $1228 for rice production in II-3 (Figure 11 and Table S10). In comparison, the largest total environmental benefit is $44/ha for wheat production in III-1, $94.2/ha for maize production in IV-3 and $26/ha for rice production in IV (Figure 12, Table S9 and S11).

There's a geographical mismatch between where the largest economic benefits occur and where the largest environmental benefits occur (Figure 12). For rice production, I-1 has the largest economic benefits of $1228/ha while the smallest environmental benefits of $3/ha. Given environmental benefits are external to farmers, policy interventions such as environmental-friendly payments and demonstration projects are needed to incentive farmers in agri-ecological regions that have high (low) environmental (yield) benefits to improve practices. These prioritized regions include wheat production in III-1 and II-1, maize production in IV-3 and IV-1, and rice production in IV (Figure 12 and Table S11).

Policymakers may also prioritize N use reduction for one crop over others if specific environmental goals (e.g. air pollution, water pollution or climate mitigation) are prioritized (Figure 12). N use reduction for wheat generate NH_3 emission and GHGs mitigation benefits that are comparable or much larger than NO_3-leaching and runoff mitigation benefits. N use reduction for maize generate NO_3-leaching and runoff mitigation benefits that are much larger than NH_3 and GHGs mitigation benefits. N use reduction for rice generates significant GHGs mitigation benefits but little NH_3 mitigation benefits. If the eutrophication issue in southern China’s river network is prioritized, N use reduction for maize production needs to be prioritized over other crops. If the issue of high NH_3 emission and severe $PM_{2.5}$ air pollution over North China Plain is prioritized, N use amount reduction for wheat needs to be prioritized. Sum of all types of environmental benefits achieved through N reduction is the largest for maize, followed by wheat and rice. N fertilizer purchase savings are larger for rice and maize than wheat.

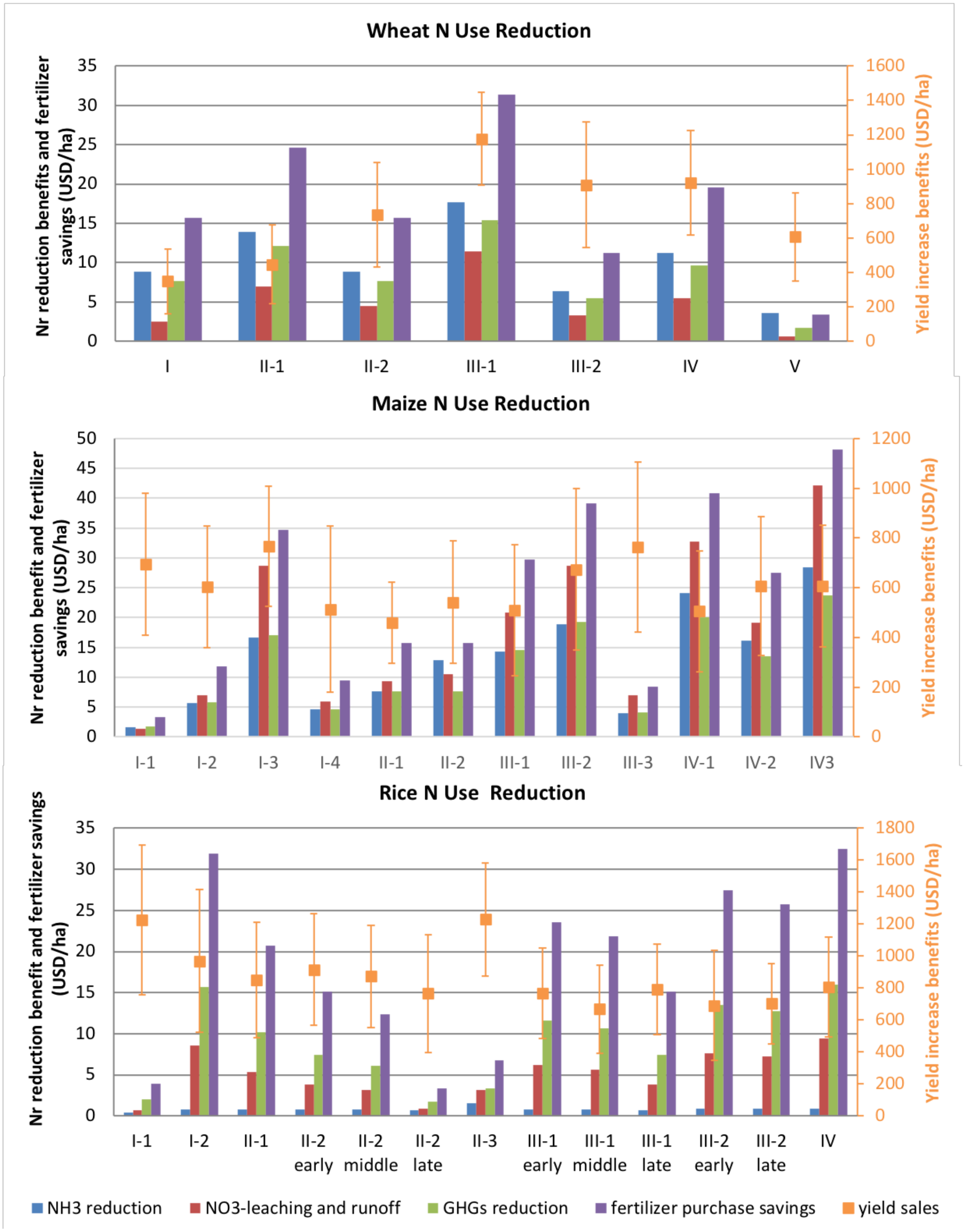

Figure 11 Environmental benefits, i.e. NH_3 reduction (bars, in blue), NO_3-leaching and runoff reduction (bars, in red) and GHGs reduction (bars, in green), and fertilizer purchase savings (bars, in purple) when N application amount is reduced to profit maximization

levels for different wheat, maize and rice agri-ecological zones. Increase of yield sales are also provided (markers, in orange, secondary y axis).

Despite large yield benefits from our analysis, yields achieved enjoyed by farmerse will highly depend on climate conditions in one specific year and how well farmers can master the technology. Yield effects from researchers' field experiments cannot always been fully replicated by farmers. Farmers' capabilities of reproducing yields achieved in researchers' plots vary. Furthermore, even though these yield benefits are larger than other types of benefits, they are small compared to farmers' off-farm earning, which represent an opportunity cost of farmers' time (detailed discussion in Section 6) and can be a major impediment for technology adoption.

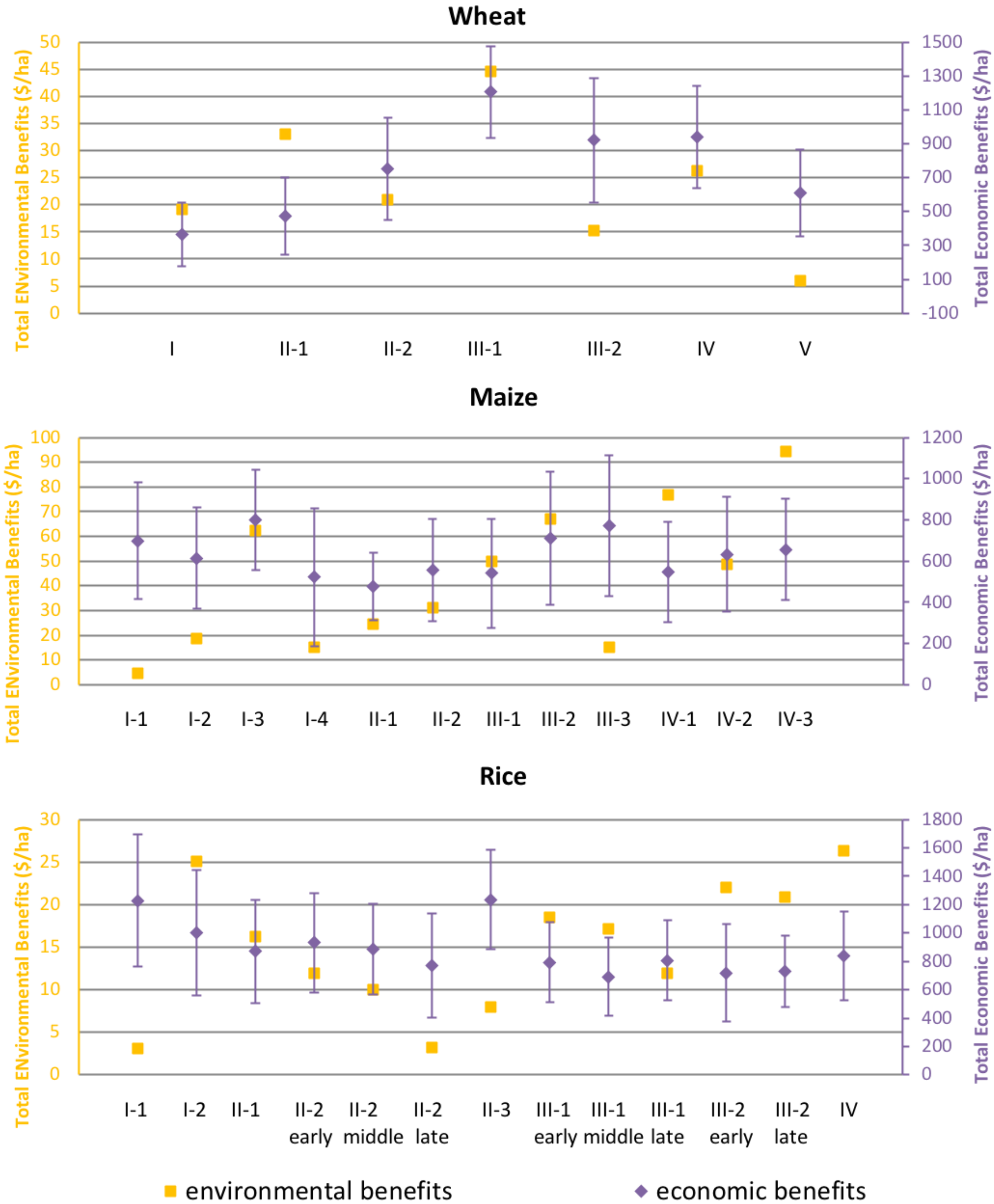

Figure 12 Total environmental (yellow) and economic (purple, secondary y axis) benefits when reducing N use amount to profit maximization levels for different wheat, maize and rice agri-ecological zones.

5.1.2 Optimized N Application Frequency

Theoretically two applications instead of one increase grain crop yields by 5.9% and decrease emissions of nitrous oxide by 5.4%, ammonia by 32% and nitrate leaching and runoff by 25%, according to a meta-analysis of 241 filed experiments for major crops in China (Xia *et al.*, 2017).

We find that yield benefit is the largest type of benefit provided by splitting application. In Table 3 we estimate labor costs, yield and environmental benefits associated with a second application (see Methods) for rice and wheat production regions. Yield benefit is \$100-150/ha and environmental benefit \$30-80/ha with reduced damage from NH_3 emission being the major component. Total benefits justify labor costs of \$3-56/ha.

Table 3 Costs and benefits of conducting a second fertilizer dressing application.

Technology recipient	Labor cost (\$/ha)	Benefits (\$/ha)			
		Yield gains	NH_3 reduction	NO_3^- reduction	N_2O reduction
IV rice	3-56	159.6	64.0	14.2	0.9
III-1 wheat	3-56	111.6	29.1	6.4	0.7
II-2 rice	3-56	129.9	55.1	12.2	0.7
IV wheat	3-56	105.9	24.5	5.4	9.7

Cost-benefit calculation of splitting N application can vary by region because of large uncertainties in labor cost and yield effects. Labor costs in farming season v.s. non-farming season (e.g. ¥150/d v.s. ¥100/d) vary, and labor costs in different provinces vary (e.g. ¥200-400/d in Heilongjiang province v.s. ¥60 in Hebei province). Yield increase achieved always depend on other factors such as plot quality, planting techniques, management and climate conditions.

Besides N application frequency, farmers can optimize the partitioning between N applied as starter fertilizer and as dressing. Current N applied as starter fertilizers are unnecessarily high; more N can be applied as dressing instead. Reducing starter fertilizer N by 10% or more increased yield by 4.1% and reduced various Nr emissions and especially NH_3 by 61.5% (Xia *et al.*, 2017). For rice production in Ningxia and Hebei provinces, changing the ratio of based/top-dressed N ratio from current 0.3 to 0.5 increases yield by 19.6% and decreases N_2O by 3.4%, NH_3 by 10% and N-leaching by 12%(Xia, 2011; Liu *et al.*, 2012).

5.1.3 Machine Application

Using machinery to place N fertilizers deeply near crop roots can substantially reduce N loss and reduce labor input, compared to conventional hand application. Nationally on average for three grain crops (maize, wheat and rice), machine application increases yield by 10% and reduces N_2O and NO_3 emissions by 15% for all crops. NH_3 losses to the air is reduced by 35% for wheat and rice and by 70% for maize (Xia et al. 2016).

Machine application provides two types of labor savings: 1) machine has higher time efficiency of application, i.e. 1.5-7.5h/ha compared to 3.75-14h/ha through hand broadcasting (Table S5), 2) machines can simultaneously conduct seeding, tillage or irrigation when applying fertilizers, generate additional labor savings. In this analysis, only the first type of labor savings is quantified due to lack of data.

Table 4 summarizes the cost and benefit of machine application for several rice, wheat and maize production regions. Similar to the previous two technologies examined, yield benefit also dominates the total benefits. Our analysis also indicates that machine application is affordable if farmers have access to machine rental services. Although machine prices on the Chinese market range from ¥4k-30k and present a high-upfront cost compared to a rural household's annual income, e.g. ¥1.2k in 2016 ((EOCSSB), 2017), farmers can rent machines from rental centers or from farmers' professional cooperatives at low or no cost.

Table 4 Costs and benefits of using machine to deeply place nitrogen fertilizers.

Technology recipient	Costs ($/ha)	Benefits ($/ha)				
	Machine rental	Yield gains	Labor savings	NH_3 reduction	NO_3^- reduction	N_2O reduction
III-1 wheat	21	189.2	3-16	31.8	4.4	2.1
IV wheat	21	179.4	3-16	26.8	3.1	27.1
IV-3 maize	21	194.8	3-16	34.6	11.4	3.5
II-1 maize	21	189.5	3-16	23.6	6.1	2.6

IV rice	21	270.5	3-16	70.0	4.2	2.5
II-2 rice	21	220.2	3-16	60.3	3.2	2.0

5.1.4 New-efficient fertilizers

New-efficient fertilizers, i.e. coated time-release fertilizers and inhibitor fertilizers can generate large benefits of reduced Nr pollution. Coated fertilizers allow nutrients to be released gradually with time, thus avoid a second application. When combined with conventional fast-released N, crop nitrogen demand can be better accommodated thus higher NUE can be achieved. Dry-farmed land is not suitable for coated fertilizers as low moisture content in soil prevents N from being released as designed (Trenkel, 2010). Coated fertilizers increase NUE in rice systems by 5-10%[7] and increase yield by 20%[8], and decrease total emissions of nitrogen pollutants by more than 50% in China (Xia *et al.*, 2016). Rice production in southern China, mostly located in ecological sensitive zones and near river network, can benefit significantly from reduced N leaching. Published experiments have found little change in NUE and yield using coated fertilizers in maize and wheat systems, yet these results are puzzling as emissions have decreased by 60% for ammonia, 49% for nitrate and 38% for nitrous oxide (Xia *et al.*, 2016). Various coated fertilizer products available on the Chinese market remain more pricy than conventional fertilizers (Table S13) and has a very low adoption rate.

[7] Personal communications with Tingyu Li
[8] Personal communications with Tingyu Li

Nitrification inhibitors (NI) reduces the conversion rate of ammonium to nitrite and the conversion rate of nitrite to nitrate. Experiments in China have found that NIs reduce nitrous oxide by 40% and nitrate leaching by 82%, increase yield by 10%, yet increase ammonia emissions by 28% averaged for three major crops nationally (Xia *et al.*, 2016). This is because North China Plain has alkaline soils where NH_3 will increase when using NI. Such side-effect can be mitigated if urease inhibitor (UI) is used together with NI in alkaline soil conditions. UIs reduce the hydrolysis conversion rate of urea to ammonium, thus efficiently reduce ammonia emissions, by up to 50% according to experiments in China. Nitrous oxide emissions decrease by 28% as well (Xia *et al.*, 2016).

We design four scenarios where new efficient fertilizers, in their most suitable regions, replace conventional ones. The 100%CRF_rice scenario targets rice production in a biodiversity-rich area in southern China, i.e. single rice cropping in southwest China (IV in Figure 11). A relatively large N use reduction is expected, from current 224 kg N/ha to 166 kg N/ha to optimize farmers' profit (Figure 11). Instead of using 224kg N/ha as urea at a price of 4 /kg urea N, sulfur-polymer coated fertilizer (the cheapest CRF on Chinese market) at a price of 5.6 /kg N is applied. This will result in 60.8% reduction of NH_3 emission, 49% reduction of NO_3-leaching, 50% reduction of N_2O emission, 5-10% increase of yield and avoid labor cost for a second N application. The 50%CRF_50%urea_wheat scenario targets wheat production in irrigated sub-region in North China Plain (III-1 in figure 9). In this scenario, instead of using 240 kg N/ha as urea, 120 kg N/ha is supplied with CRF and 120kg N/ha as urea to better meet crop N demand. In UI (NBPT)_rice scenario, urease inhibitor fertilizer is used for early rice production in middle Yantze River Basin (II-2 in Figure 11). In NI (DCD)_wheat scenario, nitrification inhibitor fertilizer is used for wheat production in the middle and lower Yantze River Basin (IV in Figure 9) where eutrophication frequently happens in Lake Tai and Lake Boyang.

Benefits and costs of implementing these four new efficient fertilizers are summarized in Table 5. In all scenarios, yield gains alone would justify the increased fertilizer purchase costs. Reduced Nr benefits are a lot higher than that achieved in the nitrogen fertilizer use amount reduction strategy (Section 5.1.1), due to the very large percentage Nr reduction enabled with various types of efficient N fertilizers.

Table 5 Costs and benefits of implementing four scenarios where new efficient fertilizers replace conventional fertilizers.

	Technology recipient	Costs ($/ha)	Benefits ($/ha)				
Strategy		Fertilizer costs increase	Yield gains	Labor savings	NH_3 reduction	NO_3^- reduction	N_2O reduction
100%CRF_rice	IV rice	50.2	202.9	3-56	121.7	13.7	8.3
50%CRF_50%urea_wheat	III-1 wheat	26.9	0	3-56	55.2	14.4	5.3
UI (NBPT*)_rice	II-2 rice	15.8	156.4	0	86.2	0.0	1.6
NI (DCD**)_wheat	IV wheat	2.7	215.3	0	-21.1	17.0	90.2

*NBPT is used at 0.05% of N use rate (Rawluk *et al.*, 2001)

** DCD is used at a rate of 1.25kg DCD/ha (Yang *et al.*, 2016)

5.1.5 Other techniques and highly-sophisticated farming

In addition to the strategies discussed in previous sections targeting the amount, method and type of N fertilizer applied, there're other strategies such as reducing flood irrigation, returning crop residues back to cropland, frequent crop growth monitoring and flexible nitrogen application prescription, and using water-soluble fertilizers that integrate irrigation and nutrient management. These strategies can benefit both yield and pollution mitigation. For example, furrow irrigation for maize in comparison to conventional irrigation reduces NH_3 emissions by 9-27% and increases yield by 10-16%, according to field experiments in Shaanxi Province (Han *et al.*, 2014). Compared to burning wheat straw, returning that straw to the soil reduces ammonia emissions by 84%, according to experiments in Hebei Province (Zhang *et al.*, 2011).

Nitrogen application amount could be further reduced (compared to the reduction proposed in 5.1.1 with all other management unchanged) if nitrogen management (application frequency, timing, depth, etc) and planting techniques (crop variety, planting density, etc) can simultaneously be improved so crop nitrogen supply can be better synchronized with crop demand. Integrated soil-crop system management (ISSM) modeling and in-season root-zone N fertilization strategies (Chen *et al.*, 2011; Cui *et al.*, 2013b; Chen *et al.*, 2014) use models with inputs of localized climate and crop information to provide recommendations regarding both planting and N management. This system was experimented with maize and wheat production and prescribed improvements such as deep tillage, a certain fraction of N to be applied during shooting stage (60% or wheat but 50% for maize in North China Plain), exact timing of dressing to be applied (e.g. during 6~8 leaf stage for continuous maize in Northeast China), etc. When management follows these recommendations, maize yields doubled at 66 planting sites, with no fertilizer use increase. This system has been promoted nationally through a ten-year campaign involving local technicians, fertilizer producers, farmers and researchers. It generates large savings of chemical inputs, i.e. 1.2 Mt reduction of nitrogen fertilizer use per Mg grain produced, and large reductions of emissions, i.e. 94 kg, 129 kg and 115 kg CO_2-eq reduction of greenhouse gas emissions respectively per

Mg of maize, rice and wheat produced and ~1.5kg reduction of N per Mg of grain produced (Cui *et al.*, 2018).

5.1.6 Summary of N Fertilizer Strategies

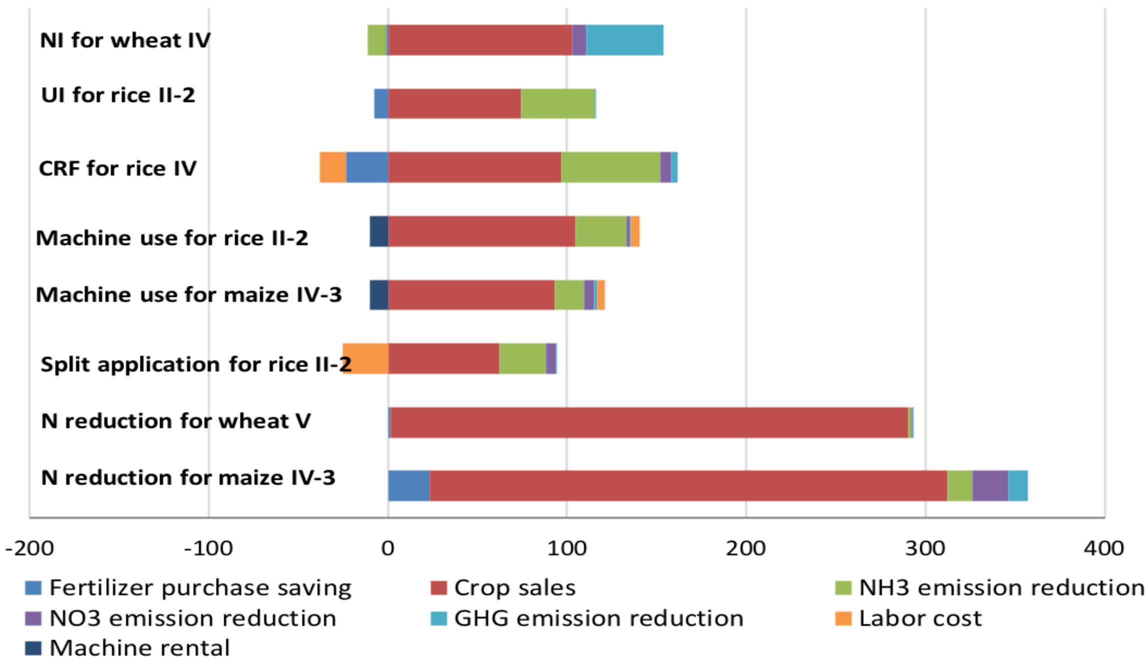

Figure 13 Costs and benefits (in rmb/mu, including fertilizer purchase savings, crop sales, NH_3 emission reduction, NO_3^- emission reduction, GHG emission reduction, labor cost and machine rental) associated with various representative fertilizer N strategies implemented on a specific grain crop production region. NI is short for nitrification inhibitor, UI for urease inhibitor, and CRF for controlled-release fertilizer.

Figure 13 summarizes the costs and benefits associated with the implementation of various N fertilizer application technologies. Yield increase benefit dominates total benefits for all strategies. Yield increase benefit alone, in all cases, justifies total costs. New efficient fertilizers have extremely large potential for pollution reduction, e.g. CRF is the technology that generates the largest unit NH_3 reduction per land area and NI generates the largest unit GHG reduction. Our dataset for N use reduction predicts larger yield benefits than all other individual technologies, indicating these experiments may not perfectly control other N management, it is likely multiple N management have occurred in these experiments.

5.2 Manure Management

Advanced technologies exist for reducing emissions during animal feeding and housing, manure collection, composting, storage, treatment and spreading.

5.2.1 Low-crude protein animal feed

Conventional animal feed includes 15-20% more N and 20-30% more P than animals actually need[9]. Low-crude protein animal feed reduces these nutrients without harming animal growth and reduces nutrient content in animal manure excreted (N by 10% and P by 10-20%). It reduces the production cost of animal feed and Nr leakage from animal manure, providing a win-win solution for both the environment and animal raisers.

5.2.2 Animal housing technologies

Conventional practices leave animal manure inside animal houses for an extended period time before removing them by flushing animal floor with large amount of water. This results in large fraction of manure N emitted as NH_3, as high as 15-20%. A conveyor belt system can automatically remove manure in a timely manner thus reducing NH_3 in animal

[9] Personal communications with Lin Ma

houses by 50-80%[10]. Such system works extremely well for small to middle-sized chicken and pig farms which account for 70% of China's chicken and pig production. It generates electricity savings, e.g. 2kwh/day for a typical 10k chicken farm, because ventilation is saved since manure has been removed timely so order is avoided. For a chicken farm with 20k chicken, the belt system costs ¥80k and consumes 2-3kwh/d electricity. Another option for pig manure is separating dry and wet manure instead of flushing with lots of water, which reduces NH_3 emission and water withdrawal by up to 50%[11]. This can be realized at a cost of ¥30-50 per pig.

5.2.3 Manure composting

Traditional open-space composting emits 43% of manure N as NH_3. Aerobic composting within reactors can reduce NH_3 emission by 50-60%, reduce N-leaching to zero and increase N content in residues by 10-20%[12]. The reactor allows shorter composting period, one week compared to months in conventional composting. Although upfront cost is high, e.g. a reactor with 40t capacity costs ¥1500k for a 10000-head pig farm, this cost can be recovered in two or three years through inorganic fertilizer sales, a by-product of composting, e.g. 400-600t/yr. There are some additional costs, including electricity consumption of ¥15k/year, labor cost of ¥30k/year, and reactor depreciation cost of ¥10k/year. The reactor is extremely suitable for landless concentrated animal systems.

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[10] Personal communications with Baoming Li
[11] Personal communications with Baoming Li
[12] Personal communications with Lin Ma

Anaerobic digestion producing biogas is another valuable option. One ton of manure can generate 300-500m^3 of biogas[13]. Manure needs to be in the biogas storage facility for one month to release 75% of biogas production potential. In Germany, farmers have manure be within the facility longer, for 2-4 months before another 4 months for composting, to release 90% of biogas production potential. During biogas production, most manure N can be converted to ammonium and potentially incur NH_3 emissions. Most biogas stations are built underground to avoid large fluxes of NH_3 to the air. A 10-15m^3 capacity station costs ¥2k; larger-scale stations cost ¥2.5k per m^3 biogas capacity.

5.2.4 Manure storage

Conventional open-space manure storage emits 15-25% of N as NH_3. Adding H_3PO_4 to manure, especially suitable for landless concentrated systems[14], can reduce NH_3 emissions by 60-70% In China, H_3PO_4 spray facilities are still under development in China since H_3PO_4 cannot be sprinkled in a way that causes harm to animals' skin. A H_3PO_4 spray facility in Denmark costs ¥7 million.

5.2.5 Manure spreading

Injecting manure into land using machines reduces NH_3 emissions by 30-50% compared to surface manure application[15]. Manure N can also substitute N supplied with synthetic N fertilizers. Assuming manure N replacing 25% of nitrogen fertilizers, manure produced

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[13] Personal communications with Renjie Dong
[14] Personal communications with Lin Ma
[15] Personal communications with Zhaohai Bai

from a 5000 head pig farm can serve 730 ha cropland in a typical wheat-maize rotation system.

5.2.6 Waste treatment system

China's livestock waste discharge standard has in the past regulated only NO_3-N content in waste water. As a result, many animal farms use technologies that convert N to NH_4-N to avoid violation of standard, however, this results in substantial loss of N as NH_3-N. To address this issue, a new standard has been drafted to regulate both NO_3-N and NH_4-N and is now under public review. This will increase demands for waste treatment systems that can reduce all types of N emission, e.g. total N content by 57% and total P by 96%[16]. Upfront cost of a system is ¥5.6million and operation cost is ¥107k/year.

6. Socioeconomic Barriers Against Agricultural Technology Transfer

Although our analysis of the various technologies above indicates that yield benefits are larger than visible costs, farmers' adoption rates of these technologies have been surprisingly low. We identified major barriers, including high learning cost due to lack of support, and high opportunity cost of not engaging in off-farm activities and low total benefits due to small plot sizes.

Existing work indicated that Chinese farmers have very poor nutrient knowledge. Many scholars have argued that the primary reason for China's N overuse is that farmers simply don't know they are overusing – they stick to their way of using responsive inorganic N

[16] Personal communications with Lin Ma

fertilizers in 1960-70s and think yield will continue to increase when additional N is applied as in the past (Huang *et al.*, 2008). Many farmers prefer buying fertilizers that are well packaged, with beautiful colored granules, and advertised as being produced from advanced facilities. Farmers even frequently buy false products or products that have passed the usage date. Researchers working with farmers within Hebei Province observed that the majority of agricultural labor are poorly-educated elder people and women. Less than one third of farmers understand the meaning of N-P-K formulation and the way N, P and K nutrients work.

Lack of systematic support from the public and private sectors have contributed to smallholders' poor nutrient management knowledge. Although farmers can obtain some information from public extensions and fertilizer industry, support from these stakeholders have been weak, resulting in high learning costs faced by farmers.

China's public agricultural extension agents spend limited time helping famers and have conflicts of interests. In 1970s, the central government substantially reduced funding for agricultural public extension. As a result, extension staff turned to commercial activities for funding. This created conflicts of interests. In order to obtain rebates, some agents would recommend more fertilizers, pesticides and seeds than farmers actually needed. Also, the more time agents spend raising money, the less time they spend helping farmers in the field. In 2000s, extension agents spent only 81 days delivering technical help to farmers, yet 56 days on commercial activities, 135 days in office and 92 days on other activities. For stations that obtain no funding from central government and remain selffunded, agents' time spent on commercial activities are even more, i.e. 190 days (Hu *et al.*, 2012b). In addition, agents don't have enough financial resources to offer farmers' support. In 2002, the average budget per agent (including salaries and extension expenditures) was only ¥14304 per year.

Fertilizer producers can help with knowledge transfer. Fertilizer producers have increasingly engaged in fertilizer application demonstration projects in order to advertise their products and provided fertilizer application guidance to farmers as part of their customer service. However, since producers' ultimate goal is to expand customer base, their service strives to cover as many villages as possible instead of equipping farmers with nutrient management techniques. Furthermore, China's fertilizer producers in the past have produced products based on company's production capability, available technologies and equipment, rather than based on crops' localized needs. As a result, uniform formula fertilizers have dominated the Chinese market until recently when local- and crop- specific formulas appeared.

For the way forward, fertilizer producers are likely to be profit-maximizing through value-added innovative fertilizers. Fertilizer producers in the future will likely focus on adding micro-nutrients such as zinc and microorganisms to fertilizers. These can strengthen crop roots and improve fruit/vegetable quality. The more appealing the products are to farmers, the more profitable they are.[17] Prescribed formulas that can benefit Nr pollution mitigation are not appealing to producers, because designing local

[17] Personal communications with Yanfeng Wang, Vice president at Xin Yangfeng Fertilizer

specific formulas requires costly soil test yet prescriptions can easily be copied by other companies, given China's limited intellectual property protection.

In addition, due to the very small plot size, yield benefits achieved through improved management for farmers are still too small compared to earnings from off-farm jobs. Figure 12 indicates that yield benefits from implementing technologies are ~¥100/mu, in rare cases yield benefits are as high as ¥300/mu when technologies are combined. However, this only translates into earnings of ¥500-1500 for a typical 5-mu farm in China. In comparison, the earnings from off-farm jobs are very high, e.g. per capita rural household income from off-farm jobs was ¥7745 in 2014, accounting for 72% of total per capita net income[18]. 274 million farmers engaged in off-farm jobs in 2014.

Yield benefits still remain too small for farmers to care. Considering the small plot size, management practices that cause a 5% yield increase are not appealing to farmers.[19] Farming is not a prestigious nor profitable business in China. The profit margin for major crops is only ¥0.23/kg, calculated using a grain yield level of 470kg/mu, monetized production cost of ¥1069/mu and grain price ¥2.5/kg ((EOCSSB), 2015). Agricultural tax has not been removed until 2000.

7. Opportunities and Policy Recommendations

7.1 Improving N management is cost-effective for large farmers

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[18] http://www.stats.gov.cn/tjsj/zxfb/201704/t20170428_1489334.html

Although for small farmers economic benefits are too small to care, for large farmers total benefits increase with farm sizes and can justify technology learning cost and opportunity costs. Large farms take up more than 50% of China cropland area, meaning their improvements of practices are critical to solve the N challenge China faces. To educate a limited number of large farmers are a lot easier and less costly than to educate millions of small farmers. There can be a win-win situation for large farms and the society.

7.2 Farmer organizations as a learning community for small farmers

Farmers' professional cooperatives (FPCs) are important institutional arrangements that can enable knowledge sharing and mutual learning among smallholders and have been succesfull in countries such as Japan. In Japan, similar organizations at village level are effective in delivering management knowledge. They gain villagers' respect and trust through helping resolve conflicts and taking care of the disadvantaged. FPCs can serve as middlemen between the government and farmerse. They can also increase farmers' bargaining power in front of food retailers and processing companies.

FPCs in China has a long way to go before being capable of fostering technology sharing and transfer. The Chinese government recognized the legal status of FPC in 2006. It further encouraged formation of new FPCs by providing preferable taxation policies and subsidies for FPC-scale machinery purchase. Statistics show by 2016 1.8 million FPCs are registered and are present at 21% Chinese villages. However, less than 1% of these cooperatives actually serve the expected functionality. They are mostly formed to obtain governmental subsidies. For the way forward, we expect FPCs to help facilitate mutual learning within smallholders.

7.3 Researchers help with education programs and formula fertilizer prescription

Researchers have played an increasingly effective role in educating both small and large farmers. Compared to extension agents who spend a limited number of days in field, there're researchers working in the countryside and providing on-site help for farmers. For example, China Agricultural University has built 71 rural working stations in 21 provinces, where master students and professors live in the countryside for years and interact with farmers on a daily basis. Once long-term trust relationships between researchers and farmers are formed, farmers become increasingly willing to take researchers' suggestions regarding planting and nutrient management[20]. One station in Quzhou, Hebei has realized a 20% increase in crop yields after five years' intervention to farmers' conventional practices. Education efforts require long-term investments in human capital and funding and in this case, $200,000 has been provided by the local Quzhou government. Scaling up this education effort to nationwide is estimated to cost $600 million per year (Zhang *et al.*, 2016).

Researchers have also been actively harnessed the industry and public extension in order to increase yields and improve N management (Cui *et al.*, 2018). Researchers designed formulas with optimized N:P:K rationing for local crops for fertilizer company. For example, CAU researchers have prescribed formulas for 3 major crops and 17 cash crops,

[20] Personal communications with Weifeng Zhang

help built 18 demonstration sites and reached 31,448 fertilizer salesmen and farmers during 2011-2015.[21]

7.4 Private fertilizer application service contracting

The newly-emerged fertilizer application service contracting in China can avoid irrational high use of agricultural chemical inputs since service suppliers will prioritize profit-maximization. Private agribusiness companies in China offers contracting services including seeding, pesticide application, fertilizer application, crop harvesting and selling.

Service contractors can be quite innovative with designing solutions that harness efforts from relevant stakeholders to save N use and boost yields. Contractors are responsible for purchasing agricultural inputs and the entire crop production processes. Crop sales will be split between contractors and farmers. A pricing chart of various contracting services is provided in Table S12. Risky averse small farmers especially are attracted to service contracting, because they are guaranteed with economic returns at no risk. Contracting companies also benefit from economies of scale; they operate like a large farm except for compared to conventional farmers companies are equipped with planting techniques, management knowledge, etc and are more risk-resilient.

[21] from materials provided by Zhenling Cui

Knowledge of management and local farming techniques and agricultural chemical input market is essential for contracting service companies to be successful. Large farmers who aggregate land to cultivate but are not familiar with farming could lose money. The success of the contracting service relies on knowledge about local agricultural input markets and local cropping techniques.

Service contracting companies is still pretty new in China (Table S12). However, there're informal contracting activities among farmers. For example, farmers who own machines rent their machines out or help others apply fertilizers and seeds, charging ¥450/ha or ¥150/100kgN applied (doesn't include fertilizer purchase fee).

Production contracting is not new globally (Eaton and Shepherd, 2001). It generates risks and opportunities for both farmers and agribusinesses and have positive implication for farmers' earnings and equity(Zhang, 2012). For example, in Kenya a sugar company contracts with 1800 farmers and assign one field officer to help with transplanting, spacing, fertilizer application, cultivation and harvesting for every 65 sugarcane growers. In India agribusinesses contract with tomato growers resulting increased yield and income levels.

7.5 Environmental-friendly payment and performance-based indicators

Policies that encourage environmental-friendly practices can be complemented by performance-based indicators. Both the European Union (EU) and U.S. have agri-environmental payments delivered to farmers to reduce negative externalities of agricultural production (N and P pollution, biodiversity loss, etc) and to foster rural development (reward agriculture's positive externality such as attractive landscape, etc) (Baylis *et al.*, 2008).

Development of crop nitrogen censors can help policymakers monitor and assess outcomes of farmers' nitrogen management. Besides time-consuming lab tests of crop tissue, a number of in-the-field options (spectroradiometers, reflectometers, satellite imagery) can provide fast determination of crop N supply based on leaf optical and electrical properties (Muñoz-Huerta *et al.*, 2013). Furthermore, N use in large mechanized farms can be monitored by adding N amount meters to fertilizer application machines.

7.6 Public extension system reform

Regarding the inefficiencies of public extensions, in 2003 Chinese government launched a nationwide pilot program covering 12 counties to separate the public and commercial functions of extensions, and experimented with farmers' monitoring and evaluation mechanisms in 30 villages (Hu *et al.*, 2012a). Although until now there's no sign that the reform will scale to nationwide.

7.7 Remove N fertilizer subsidies

High subsidies for N fertilizers in China has been criticized as responsible for farmers' overuse of N, fortunately these subsidies have been removed in 2015. N fertilizer production in used to get preferable prices for natural gas purchases and electricity consumption, railway transportation and value-added tax (VAT) was also waived. Research estimates that no VAT alone subsidizes ¥56 for one ton of urea and total subsidies add up to ¥20 billion for 40 million ton urea production(Cheng *et al.*, 2010).

7.8 Agricultural policy reform

Chinese agriculture has not yet transformed into a professional and skillful one due to historical agricultural policies. There are research pointing out that systematic discrimination against the rural poor and peasantry existed as a result of a city-prioritized state-capitalism development model (Whyte, 2010) (Naughton, 2004) (Huang, 2008) (Remington, 2015). In the 1960s, grain prices were kept low to guarantee the well-being and satisfaction of urban dwellers in 1960s (Oi, 1993) (Bates, 1981). It was until 2004, agricultural taxation that has lasted for a century was terminated (Qu *et al.*, 2011). The Hukou system (person/household needs to be registered as city or countryside) has operated from 1960s until now, binding people to where they are born and constraining easy relocation. When migration constraints to cities was later relaxed, rural migrants engaged in the dirty, difficult and dangerous city jobs (construction, hauling, domestic service and street-corner commerce). They were less likely to be employed in state-owned enterprises and government agencies and could hardly enjoy the same social benefits as people with ‘city Hukou’.

Chinese farmers’ management skills need to be improved with knowledge and financial support of the government. Farmers who are bound to their cropland do not view themselves as proud and professional suppliers of food for people in China. Instead, life is not very promising since growing crops receives less revenue than engaging in other decent jobs such as the financial sector, real estate sector, small businesses, etc. Under this socioeconomic condition, farmers lack the capital, land and incentives to achieve better nutrient management.

8. Uncertainties and Limitations

Despite that this Chapter has made great efforts pulling together available knowledge across multiple disciplines, some conclusions have been drawn upon limited data support. Here below I point out where validity of conclusions can be improved:

This work can be more comprehensive if more private sector experts can be included for their insights on the fertilizer industry and service contracting services. In comparison, collaborators of this chapter, although mostly from China Agricultural University, are prestigious researchers in crop science and have broad collaborations with all major agricultural research institutions and universities within China.

For the overview of current N fertilizer application in China, finds regarding adoption of new efficient fertilizers are less solid compared to N application amount and method. We developed our understanding of adoption rate of new efficient fertilizer products based on collaborators' knowledge with a heavy representation of practices in the North China Plain. Instead, current N application amount has been based on thousands of surveys and based on several governmental projects researching farmers' N application in order to prescribe N fertilizer application. Results regarding machine application of N is also quite solid, i.e. estimated from machinery oil consumption summarized from thousands of surveys conducted with national representation.

Our analysis of the Nr mitigation effects associated with implementation of various N management heavily relies on one meta-analysis paper which statistically summarizes available field experiment results conducted for major grain crops in China (Xia et al., 2017). There's a lack of research for effects of various management combined.

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Chapter 3: Improving Air Quality and Nitrogen Use Efficiency in China via Cost-effective Agricultural Nitrogen Management

1. Introduction

China's population-weighted annual $PM_{2.5}$ (fine particulate matter with aerodynamic diameter less than 2.5 μm) reached 54.3 μg/m^3 in 2013, among the highest in the world (Brauer *et al* 2016), and has large associated adverse impacts on public health (Burnett *et al* 2018). To date, China's control strategies have focused primarily on pollutants other than ammonia (NH_3), reducing emissions of primary $PM_{2.5}$ by 35%, nitrogen oxides (NO_x ($NO_x = NO + NO_2$)) by 17% and sulfur dioxide (SO_2) by 62% from 2010 to 2017, resulting in reductions in $PM_{2.5}$ concentrations by 33% over this period (Zheng *et al* 2018). In contrast, emissions of NH_3 have remained high (Zhang *et al* 2018) and stable between 2010-2017 at~12 Tg (Zheng *et al* 2018).

However, NH_3 controls can in some cases be the most effective mechanism for reducing $PM_{2.5}$ in winter in the eastern U.S. (Pinder *et al* 2007), Europe (Banzhaf *et al* 2013, Megaritis *et al* 2013), and China' s Pearl River Delta, and in summer in Europe (Wang *et al* 2011). NH_3 emission reductions reduce $PM_{2.5}$ by reducing the inorganic fraction of $PM_{2.5}$, i.e. secondary inorganic aerosols (SIA, the sum of ammonium, nitrate and sulfate aerosols). Although at high NH_3 concentrations, reduction of NO_x and SO_2 are generally more effective at reducing $PM_{2.5}$ formation, reductions in NH_3 can be more effective at moderate and lower NH_3 levels (Seinfeld and Pandis 2016, Meng *et al* 1997). In general, as marginal costs of further SO_2 and NO_x emission reductions increase, NH_3 emission controls become increasingly attractive. Europe, which already has stringent controls on SO_2 and NO_x, for example, has recently set national ceilings for NH_3 (EC 2010). China's 2018 "Three-year Action Plan Fighting for a Blue Sky" for the first time encourages NH_3 emission reductions from agricultural management changes, although it has not set a quantitative target.

Research on the benefits of NH_3 emission reductions needed to inform policy has faced a number of limitations including a lack of accurate, geographically varied NH_3 emission inventories and incomplete aerosol formation chemistry in atmospheric chemistry models. Earlier bottom-up NH_3 emission inventories have low spatial resolution and no seasonality, (e.g. the NH_3 emission inventory used by the GAINS model [Greenhouse Gas and Air Pollution Interactions and Synergies]), or use NH_3 emission factors primarily developed for other regions (Zhang *et al* 2018, Paulot *et al* 2014). Previous inventories of NH_3 emissions in China vary by a factor of two and have discrepancies in spatial distribution and seasonality (Zhang *et al* 2018). In this paper, we address the limitation of past NH_3 emission inventories by using a new improved NH_3 emission inventory, which adds dependence of NH_3 emission rates on climate and soil conditions (Paulot and Jacob 2014) and local agricultural practices (Zhang *et al* 2018, Huang *et al* 2012).

A critical question for public policy is typically the extent to which benefits exceed costs and whether a full range of policy impacts are included in a cost-benefit analysis. A recent paper found that with a 50% reduction in NH_3 emissions in China, annual benefits due to reduced $PM_{2.5}$ related mortalities and reduced nitrogen deposition valued at US$12 billion barely exceeded the annual costs of control technology and yield reductions resulting from worsened acid rain together valued at US$9.4 billion (Liu *et al* 2019a). That paper, however, employed little of the emerging work on management opportunities in China's agricultural sector through which NH_3 emission mitigation can be realized and did not consider the large co-benefits of improved agricultural practices beyond air quality and nitrogen deposition. Our work utilizes their findings but in addition conducts a comprehensive analysis of various management opportunities. We evaluate a full range of associated impacts including air quality, nitrogen use efficiency (NUE), crop yields, greenhouse gas emissions (GHGs) and reactive nitrogen water pollution (Figure 1) and we estimate technological adoption costs using data collected for China.

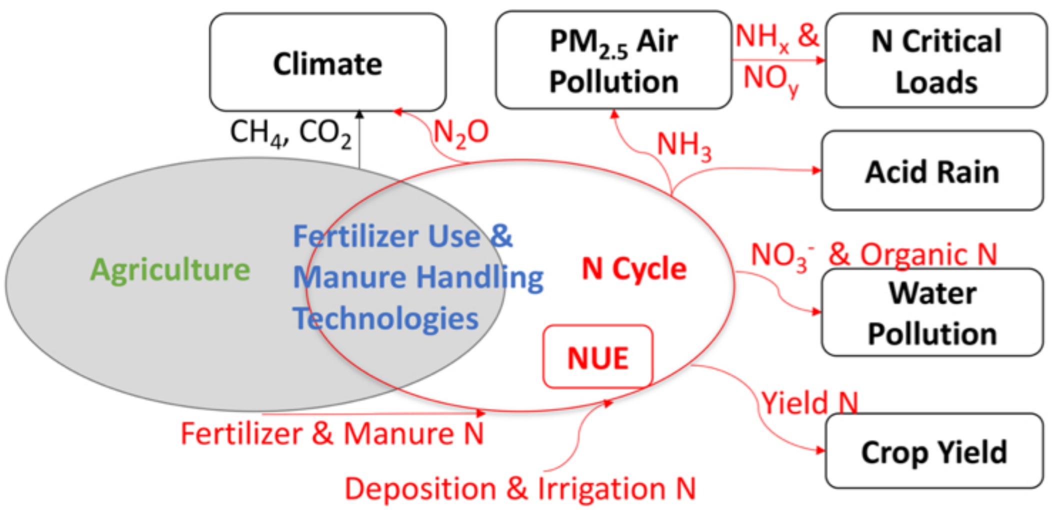

Figure 1. Effects of changes in agricultural practices. NH_x for reduced N and NO_y for oxidized N.

The opportunities of improving fertilizer application and manure handling we identified help improve China's extremely low NUE and reduce air pollution. China's total agricultural NUE (including grain crops, cash crops, fruits and vegetables) is only 25-28%, compared to high NUE achieved in Europe and North America of 52% and 68% (Zhang *et al* 2015, Lassaletta *et al* 2014). Manure disposal in China results in leakage of 2/3 of manure N to the environment (Bai *et* *al* 2016a, 2017), resulting in increased air pollution, algal blooms, and groundwater contamination (Sun *et al* 2012).

In this study, key agricultural nitrogen management technologies for China are identified. The potential of each technology to reduce NH_3 emissions and $PM_{2.5}$ formation is examined using a recent process-based NH_3 emission model for China at $0.25° \times 0.25°$ latitude $\times$ longitude resolution (Zhang *et al* 2018) and the WRF-Chem (Weather Research and Forecasting – Chemistry) air quality model at 27km $\times$ 27km for January and July of 2012. In addition, implications for NUE (defined as $N_{crop}/(N_{dep} + N_{bfix} + N_{irri} + N_{fert} + N_{manu})$, where N_{crop}, N_{dep}, N_{bfix}, N_{irri}, N_{fert}, and N_{manu} represent, respectively, N content in harvested crop, atmospheric deposition, biological fixation, irrigation water, fertilizer application, and manure used as fertilizer), crop yields, GHG mitigation, and water quality improvements are quantified. Furthermore, we compare benefits achieved (including value of avoided premature deaths, yield gains, labor savings, GHG mitigation, reduced Nr pollution, and reduced N deposition from Liu et al (2019)) with total costs (including worsened acid rain adopted from Liu et al (2019) and our estimated technological adoption costs (20-26)).

2. Results

2.1 NH_3 mitigation technologies for China

We focus exclusively on measures that have substantial potential to be economically cost-effective and utilize the following scenarios (see Table S1 and SI for details):

Reduced N application describes realistic N fertilizer use reduction for China's crops, vegetables and fruits. China has lower crop production while using more fertilizer nitrogen inputs than most other parts of the world. Excess N fertilizer application increases NH_3 emissions and decreases crop yields (Wu 2014, Wu *et al* 2015a, 2014, Chen Xinping 2016). Excess N supply decreases yields by allowing rapid growth of crops, especially growth of stems near crop roots, resulting in ineffective tiller and low earing rate (for maize and rice), high chance of lodging (for rice), pest diseases and crop senescence (for all grain crops) (Ghorbani *et al* 2010), as well as prohibiting root elongation (for maize) (Mi *et al* 2010). In the *Reduced N Application* scenario, N application rates for maize, wheat and rice are reduced to levels that optimize farmers' net return (profit minus seed and fertilizer input cost) (Wu 2014, Wu *et al* 2015a, 2014, Chen Xinping 2016). The quantity reduced is generated by a statistical analyses of ~5500 controlled field experiments during 2005-2010 (Table S2 and SI)(Wu 2014, Wu *et al* 2015a, 2014, Chen Xinping 2016). 1/3 of N is applied during sowing and 2/3 of N at 5th-leaf stage. This can increase grain yield by ~5-10% across agroecological regions and reduce emissions of Nr (Wu *et al* 2015a, 2014). Nitrogen application rates for vegetables and fruits are reduced following recommendations derived from field experiments (Feng 2014, Liyou Tang and Xuejun Zhou 2014, He *et al* 2016, Dong *et al* n.d., Wang *et al* 2013, Zhao *et al* 2017a, Shen 2014).

Efficient Fertilizer describes the use of controlled-release and urease inhibitor fertilizers, which substantially reduce NH_3 emission rates by 40-70% depending on crop type and N application rates (Trenkel 2010).

Machine Application describes replacing hand application of fertilizers by machine deep placement of N in croplands other than hilly regions. Only 30% of Chinese cropland is fertilized using machine application of N (MOA (Chinese Ministry of Agriculture) 2015). A Chinese government policy document calls for increasing the percentage of machine fertilized area to 40% of cropland by 2020 (MOA (Chinese Ministry of Agriculture) 2015). Deep placement of N near crop roots can substantially reduce NH_3 losses to the air (i.e., by 35% for wheat and rice systems and by 70% for maize) (Xia *et al* 2016). We also discuss the implications of two variations of this scenario: deep placement of N by hand and machine broadcasting.

Manure management is based on a recent review paper showing the effectiveness of acidification and aerobic composting as well as cropland injection of manure for reducing NH_3 emissions during storage and manure spreading, and the effectiveness of reducing unnecessarily high protein content in animal feed to reduce total N content in manure (Cao *et al* 2018).

The *Combined scenario* combines all improvements in the four scenarios above.

2.2 NH_3 emissions

NH_3 emissions in the *Baseline* case (Figure S1) and N management scenarios are summarized in Table 1. Total and spatial distributions of reductions of NH_3 emissions achieved through implementation of each N management scenario are presented in Table 1 and Figure 2, respectively. The most ambitious scenario, *Combined*, reduces NH_3 emissions by up to 34% annually (Table 1). Three scenarios, i.e., *Efficient Fertilizer*, *Machine Application*, and *Manure Management*, each result in the same moderate annual reduction of ~11% and up to 30% reduction locally in January and 50% in July (Figure S2).

Table 1 China's national total NH_3 emissions (annual, January and July, in unit of Tg) in the base case (year 2012) and agricultural N management scenarios. NH_3 emissions reductions achieved through implementation of each N management scenario are shown as a percent of *Baseline* NH_3 emissions.

Scenario	Annual NH_3 Emissions (Tg)	January (Tg)	July (Tg)	Annual reduction (%)	January reduction (%)	July reduction (%)
Baseline	14.0	0.66	1.66	-	-	-
Reduced N Application	13.1	0.64	1.54	6.4	3	7
Efficient 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

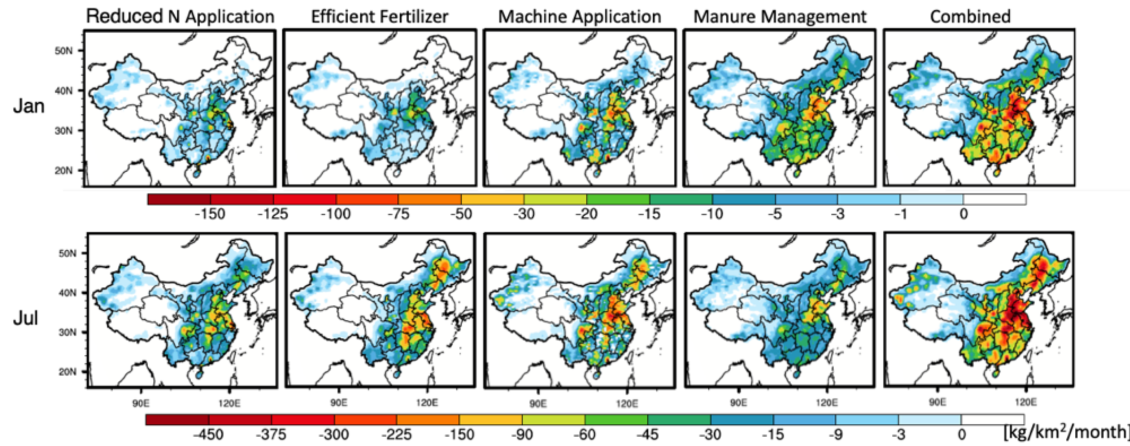

Figure 2. Changes of NH_3 emissions (in kg/km^2/month, negative means reduction) in N management scenarios compared to *Baseline* NH_3 emissions in January and July of the year 2012.

Wintertime reduction of NH_3 emissions relies mainly on improved manure management, which results in a 10-20% nationwide reduction (Table 1 and Figure S2). Summertime reduction of NH_3 emissions can be achieved in multiple scenarios, including *Machine Application*, *Efficient Fertilizer* and *Manure Management*, which results in a 10%-14% nationwide reduction.

2.3 $PM_{2.5}$ air quality benefits

We evaluate modeled NH_3(g) against satellite observations (see Supplementary Information). Modeled NH_3(g) concentrations at 918hPa are in good agreement with AIRS (Atmospheric Infrared Sounder) observations over China, especially in July of 2012 (Warner *et al* 2016) (Figure S3 and S4 and Table S3). Modeled daily and monthly NH_3 column density are in better agreement with IASI (Infrared Atmospheric Sounding Interferometer) observations for January and July of 2012 (Damme *et al* 2017) (Figure S5, S6, S7 and S8) than other NH_3 emission inventories such as NH_3 in MIX (Li *et al* 2017a, Kang *et al* 2016).

We evaluate modeled $PM_{2.5}$, ammonium (NH_4^+), nitrate (NO_3^-) and sulfate (SO_4^{2-}) concentrations for the *baseline* scenario during January and July of 2012 against $PM_{2.5}$ observations at U.S. embassy locations (Anon n.d.) and as many published speciated $PM_{2.5}$ observations during the same time period as available, including observations collected in Beijing (US Embassy at Beijing 2012)(Sun *et al* 2015), Shanghai (Wang *et al* 2016), Xiamen, Quanzhou, Putian and Fuzhou (Wu *et al* 2015b), Guangzhou (Lai *et al* 2016) and Jinsha (Zhang *et al* 2014). The WRF-Chem model in general reproduces observations well especially in July (Figures S9 and S10). In January, WRF-Chem's underestimation of sulfate is prominent at four coastal sites but barely noticeable in Beijing and Shanghai. Modeled daily and hourly $PM_{2.5}$ and speciated $PM_{2.5}$ correlate well with observations (Figure S11, Table S4 and S5).

As NH_3 emissions primarily affect the inorganic fraction of $PM_{2.5}$, mainly secondary inorganic aerosols (SIA, [SIA] = [NH_4^+] +[NO_3^-] +[SO_4^{2-}]), we present SIA concentrations at the surface (the lowest model level with a thickness of 18m), as a result of NH_3 emission reductions, achieved through implementation of each N management scenario in Figure 3. In the *Combined* scenario, up to 8 μg/m^3 (28%) reduction of SIA can be achieved in both January and July (Figures 3 and S12). *Manure Management* and *Machine Application* scenarios are the most effective scenarios to reduce SIA in January, with reductions up to 4 μg/m^3. In July the four individual management scenarios each generate SIA reductions across China of between 0.5-4

$\mu g/m^3$. Regions that have the largest SIA reduction are central and southern China in January, and the North China Plain in July.

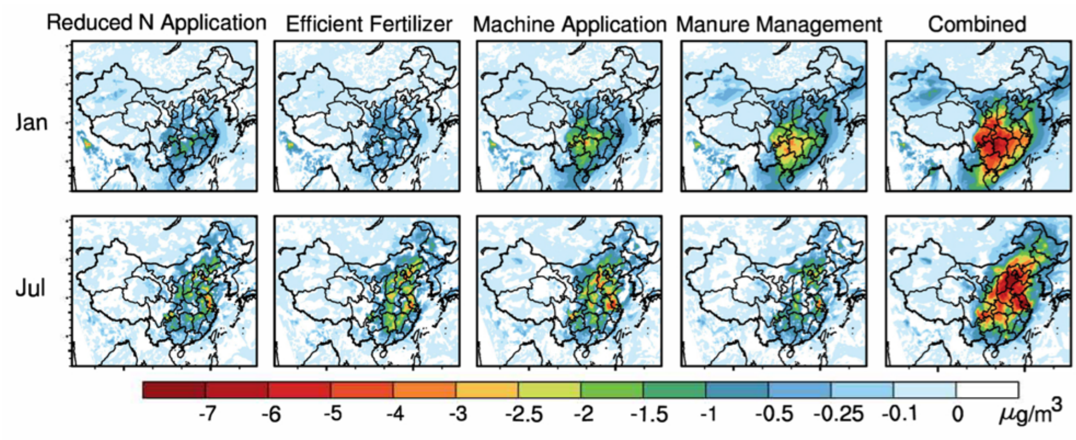

Figure 3. Changes in ground-level (surface layer in WRF-Chem is 18m thick) concentrations (in unit of $\mu g/m^3$, negative values mean reductions) of secondary inorganic aerosol (SIA, the sum of ammonium, nitrate and sulfate aerosols) in management scenarios compared to the *Baseline* simulation, in January and July of 2012.

Reductions of SIA are driven mainly by reductions of nitrate and ammonium aerosols with sulfate concentrations remaining unchanged in both January and July (Figures S13 and S14). This is because changes in NH_3 emissions do not directly impact sulfate aerosol, but changes in other aerosols (ammonium and nitrate) can slightly modulate the simulated oxidants of SO_2 (e.g., O_3, H_2O_2) and further formation of sulfate. Increased (decreased) formation of ammonium and nitrate will decrease (increase) the available oxidants for SO_2.

Meanwhile, as found in Liu et al. (2019), reduced NH_3 emissions will increase the acidity of precipitation in China. This mainly occurs by favoring production of NH_4HSO_4 which is more acidic than $(NH_4)_2SO_4$ and shifting $NH_3(g)$-$NH_3(aq)$-$NH_4^+(aq)$ equilibrium towards $NH_3(g)$

which releases more H^+(aq) (Liu *et al* 2019b). Reduced NH_3 emissions will also decrease N deposition and are thus beneficial for sensitive ecosystems. These two effects are not simulated in this study but are incorporated into the cost-benefit analysis section by adopting results from Liu et al (2019).

2.4 Mortality effects of reduced $PM_{2.5}$: We estimate reductions of premature mortality due to reduced exposure to $PM_{2.5}$ for our scenarios (Table S6). The *Combined* scenario saves 30,500 (95% confidence interval (CI): 18,400; 39,900) lives, which is nearly 3% of the 1.3 million premature deaths resulting from exposure to $PM_{2.5}$ in China in 2012. Other scenarios reduce premature mortality by 5300-8800 persons. Considering the uncertainties of existing NH_3 emission inventories for China (ranging from 10Tg to 17Tg (Huang *et al* 2012, Gu *et al* 2012, Kang *et al* 2016), compared to 14Tg in this study), we estimate that the range of reduced mortality for the *Combined* scenario is 27400 to 30700 persons, which is -10% to +0.7% of our estimate of 30500 persons (see Supplementary Information, Figure S15 and Table S7).

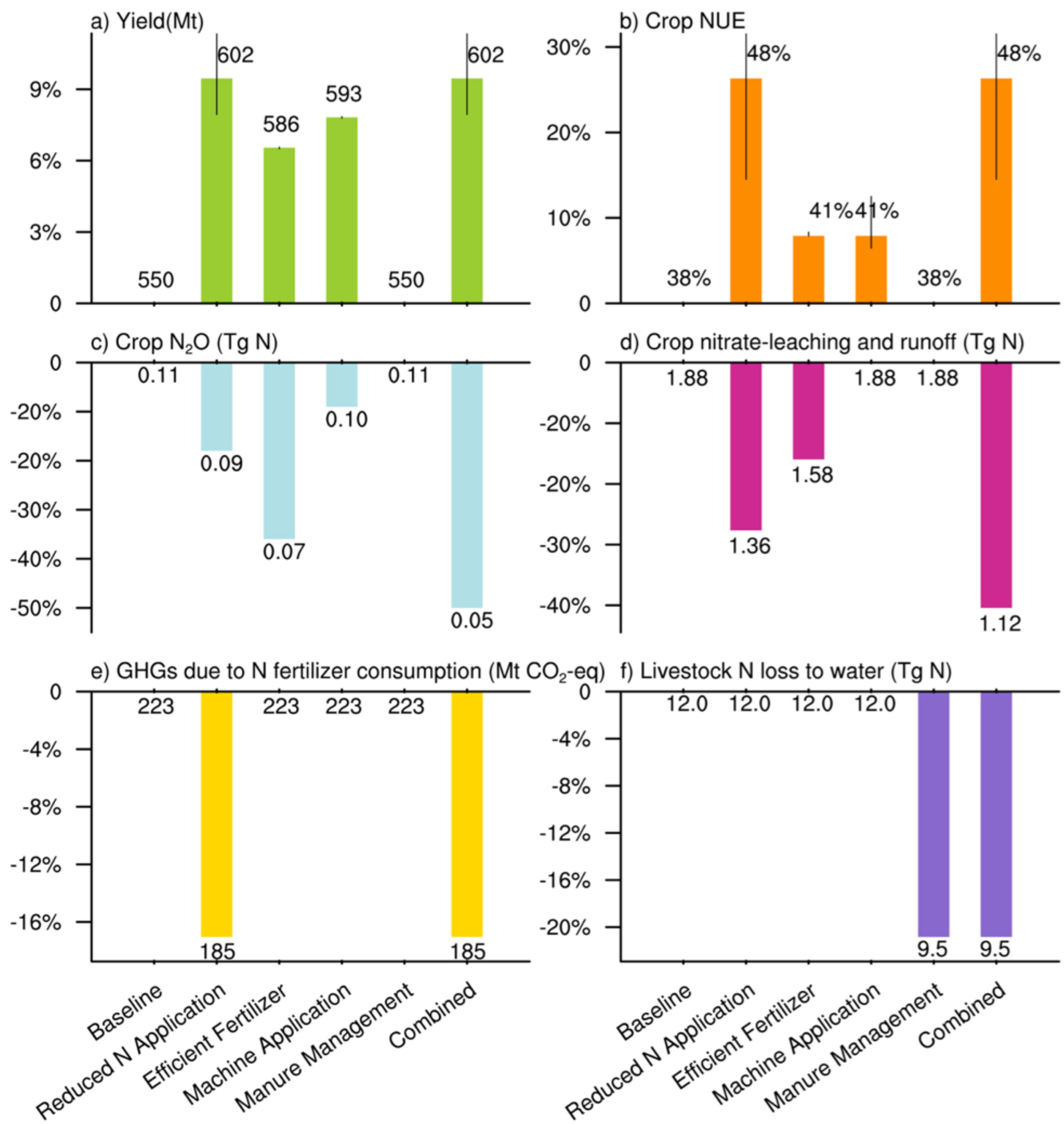

Figure 4. Co-benefits achieved through implementing NH_3-mitigating nitrogen management scenarios. Y-axes show % changes of metrics in each scenario, (scenario – baseline) / baseline. Numbers associated with each bar show the absolute value of the metric. a) China's national total grain yields (wheat, maize and rice, in Mega tonnes), b) average grain crop system NUE, c) grain crop N_2O emissions (in Tg N), d) grain crop NO_3-leaching and runoff (in Tg N), e) life-cycle GHG emitted during N fertilizer manufacture, transport and application for grain crops (in

Mega tonnes CO_2-eq), and f) livestock N loss to water (in Tg N). Error bars in plot a) indicate the range of estimated metrics that originate from the standard deviation of surveyed crop yields and surveyed N application rates in each grain crop production region (Wu 2014). Error bars in plot b) originate from the range of NUE achieved in individual grain crop production regions.

2.5 Yield effects:

We estimate that the *Reduced N Application, Machine Application,* and *Efficient Fertilizer* scenarios respectively increase national crop yields of the major grains (maize, wheat and rice) by 52, 43 and 36 Mega tonnes (9%, 8% and 7% of the *Baseline* crop yields) (Figure 4 and Table S8). We conservatively estimate that yield gain achieved in the *Combined* scenario is the same as that in *Reduced N Application* scenario.

2.6 Change in NUE

Average cropping system NUE for grain crops increases from the current value of 38% to 48% in *Reduced N Application* scenario, 41% in *Machine Application* and *Efficient Fertilizer* scenarios and as high as 48% in *Combined* scenario (Figure 4, SI Table S9 and S10).

2.7 Greenhouse Gases

Reduced N application rates in *Reduced N Application* and *Combined* scenarios reduces life-cycle GHG emissions from N fertilizer manufacture, transport, application and post-application by 38 Mega tonnes CO_2-eq. *Machine Application* and *Efficient Fertilizer* scenarios reduce N_2O emissions equivalent to 4.7 Mega tonnes CO_2-eq and 18.8 Mega tonnes CO_2-eq, respectively (Figure 4 and Table S9 and S10).

2.8 Water pollution: NO_3^--leaching and runoff from croplands are lowered by 28%, 16% and 40% in *Reduced N Application*, *Efficient Fertilizer* and *Combined* scenarios, respectively. The *Manure Management* and *Combined* scenario 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).

2.9 Cost-benefit analysis

Table 2 shows economic benefits achieved and costs incurred through implementation of each agricultural N management scenario. The last three lines of Table 2 provide total benefits, total costs, and net benefits for each scenario. For metrics that have large uncertainty, Table 2 presents results with a range of monetary values in the form of medium (low to high) estimate(s) considering possible ranges of economic parameters selected[22]. In Tables S11, S12, and S13, we provide details of the origin of these estimates. Table S14 provides nine benefit/cost ratios for each scenario when benefits and costs are each at low, medium and high levels.

Table 2: Economic benefits & costs calculated for agricultural nitrogen management scenarios in US$ billions/annum, as well as values of economic parameters used. We provide a range of estimates, in the form of a mid-range (low; high estimate), for cost and benefit metrics that are sensitive to the selected values of economic parameters.

Metrics	*Reduced N Application*	*Efficient Fertilizer*	*Machine Application*	*Manure Management*	*Combined*	Economic parameters used (with ranges)

[22] A currency conversion rate of US$1=RMB6.95 is used. Currency exchange rates in 2012 and 2018 are both approximately this value. Prices for grain and labor costs are for the year 2012.

Value of lives saved from $PM_{2.5}$ air pollution related deaths	0.70 (0.51; 1.41)	1.08 (0.78; 2.18)	0.65 (0.47; 1.32)	1.08 (0.79; 2.19)	3.77 (2.74; 7.61)	Value of statistical life US$0.124 million /person (Xu *et al* n.d.) (US$0.09; 0.25 million /person) (Xie 2011, Nielsen and Ho n.d.)
Reduced conventional N fertilizer purchase expenditure[1]	1.62	0	0	0	1.62	Urea price: US$0.6 kg^{-1} N (Table S15)
Increased crop sales	15.7	12.3	14.1	0	15.7	2012 crop wholesale prices in China: Rice US$0.40 kg^{-1} Wheat US$0.34 kg^{-1} Maize US$0.27 kg^{-1} (Chinese NDRC (national development and reform council) 2016)
Reduced damage from NO_3- leaching and runoff	0.61	0.41	0.06	0.33	0.65	Damage cost of NO_3-N is $1.32/kg N, with $0.2/kg N from drinking water health impacts and $1.12/kg N from eutrophication (Gu *et al* 2012, Ying *et al* 2017) (Table S16)
Reduced social cost of carbon (CO_2-eq) emissions	0.76 (0.38; 1.89)	0.28 (0.14; 0.7)	0.16 (0.08; 0.4)	0	0.76 (0.38; 1.89)	Social costs of carbon $20.4/tonne CO_2-eq (Schiermeier 2009) (10; $50/tonne CO_2-eq)

Labor savings	0	0	0.45 (0.2;0.7)	0	0.45 (0.2;0.7)	Decreases in time needed for machine application compared to hand application 4h/ha (2.5; 7.5h/ha). Costs of labor varies across provinces (US$7.34; 43.2 d^{-1}), provided in Table S17
Reduced N deposition (by scaling benefits estimated by Liu et al)(Liu *et al* 2019a)	0.24	0.41	0.41	0.41	1.29	
Organic fertilizer sales	0	0	0	5.14 (2.74-20.5)	5.14 (2.74-20.5)	Organic fertilizer sale price US$43.2 $tonne^{-1}$ (US$28.8; 143.9 $tonne^{-1}$)
Yield reductions from worsened acid rain (scaled from Liu et al)(Liu *et al* 2019a)	0.64	1.07	1.07	1.07	3.39	
Technological costs of improving N management resulting in NH_3 emission	0	4.09 (2.75; 5.42)	3.92 (1.31; 7.84)	6.35 (4.2;10.8)	14.36 (8.26;24.06)	Fertilizer application machine rental price US$5.8 ha^{-1} (US$2.9; 8.6 ha^{-1})

reduction (this study)						Composting reactor lifetime 30 yr(20yr; 50yr) Discount rate 4% (2%; 15%) Controlled-release fertilizer price: US$0.9 kg^{-1} N (0.8; 1.0 kg^{-1} N) NBPT urease inhibitor price: US$14.4 kg^{-1} applied at 0.001kg kg^{-1} N (Xia *et al* 2017) (Table S15)
Technological costs of achieving NH_3 emission reduction (obtained by scaling costs from Liu et al)(Liu *et al* 2019a)	0.84	1.41	1.41	1.41	4.47	
Total benefits[2]	19.6(19.1;22.2)	14.5(14.0; 16.0)	15.8(15.3; 16.3)	7.0 (4.3; 23.4)	29.4 (25.3;50.0)	
Total costs[3]	0.64	5.16 (3.82; 6.49)	4.99 (2.38; 8.91)	7.42 (5.27; 11.87)	17.75 (11.65; 27.45)	
Net benefits[4]	19.0 (18.5;21.5)	9.3 (7.6; 12.2)	10.8 (6.4;13.9)	-0.46 (-7.6; 18.2)	11.7 (-2.12; 38.3)	

[1]Subsidies for fertilizer production in China including subsidies for electricity, natural gas, and rail transportation for fertilizer producers was removed in 2015. For urea, subsidies accounted for ~3% of

price. Reduced conventional N fertilizer expenditure here is for farmers, savings for society is 103% of this value.

[2]Total benefits = reduced $PM_{2.5}$ mortality + reduced fertilizer expenditure + increased crop sales + reduced NO_3^- leaching and runoff + labor savings + reduced social costs of carbon + reduced N deposition + organic fertilizer sales

[3]Total costs = acid rain costs + technological costs estimated by this study

[4] Net benefits (medium estimate) =Total benefits (medium estimate) – Total costs (medium estimate estimate)

Net benefits (low estimate) =Total benefits (low estimate) – Total costs(high estimate)

Net benefits (high estimate) =Total benefits (high estimate) – Total costs(low estimate)

Our individual N management options remain cost-effective and generate net benefits ranging from US$9.3 to19 billion/annum, except for the *Manure Management* scenario alone which we only accounts for some of its benefits (see Discussions and Conclusions). *Reduced N Application* has benefit/cost ratios of 30-35, *Machine Application* 2-7, *Efficient Fertilizer* 2-4 and *Manure Management* 0.4-4. Our *Combined* scenario remains cost-effective except under high cost and low benefit estimates and delivers benefits of US$29 (25; 50) billion per year against costs of US$18 (12; 27) billion/annum (benefit/cost ratio of 0.9-4.3).

Our comprehensive cost-benefit analysis incorporates our calculations of yield, GHG mitigation and Nr pollution mitigation benefits of NH_3-mitigating N management, and includes acid rain and nitrogen deposition impacts calculated by Liu et al (Liu *et al* 2019a). By including these additional components we find US$9.4-19 billion/annum net monetary benefits across NH_3-

mitigating measures (*Reduced N Application, Machine Application, Efficient Fertilizers and Combined)* which reduce NH_3 emissions by 6-34%. In contrast, Liu et al (2019) estimated that for a 50% reduction in NH_3 emissions net benefits totaled only US$0.4 billion.

Unlike Liu et al. (2019), we include yield benefits from improved agricultural management valued at US$12.3 to 15.7 billion/annum (range for medium estimates across the scenarios) and various NUE and GHG related benefits each valued at ~ US$1 billion/annum. All scenarios reduce $PM_{2.5}$ related premature deaths by US$0.65 to 3.77 billion/annum. All scenarios reduce damage from NO_3^--leaching and runoff by US$0.06 to 0.65 billion/annum and reduce N deposition damage by US$ 0.24 to 1.29 billion/annum (adopted from Liu et al (2019)). *Reduced N Application* and *Combined* scenarios, through reduction of nitrogen use for grain crops, reduce N fertilizer expenditure by US$1.62 billion/annum and reduce life-cycle GHGs related to N fertilize manufacture/production/transport by US$0.76 billion/a. *Machine Application*, through deeply placing N near plant roots in a time efficient way, saves US$0.2 to 0.7 billion/annum of labor compared to conventional surface broadcasting by hand. *Machine Application* and *Efficient Fertilizer,* through deep placement of N and slow-release of N, respectively, reduced N_2O emissions valued at US$0.16 and US$0.28 billion/annum, respectively. Total benefits sum to US$7 to 30 (4;50) billions/annum across scenarios. Of these benefits, Liu et al (2019) only included reduction in N-deposition and air pollution impacts totaling only $US12 billion for an NH_3 emission reduction (50%) much larger than any of our scenarios (6-34%).

Because of the large total benefits we obtain, we find benefits exceed total costs (technological adoption costs + acid rain costs) in most cases despite the fact that our technological adoption costs, estimated using data collected for China, are 250% to 450% of those obtained by scaling Liu's costs from the GAINS model. We find that although *Machine Application, Efficient Fertilizer* and *Manure Management* scenarios achieve similar levels of national NH_3 emission reductions, their technological adoption costs are dramatically different. *Machine Application* has a cost of US$3.92 (1.31; 7.84) billion/annnum, which is less than 60% of *Manure Management* costs of 6.96 (4.27; 23.43). Scaling Liu et al (2019) technological adoption costs based on levels of NH_3 emission reduction achieved would generate a uniform cost of US$1.41 billion/annum for these three scenarios. Acid rain costs valued at US$0.64-3.39 billion across scenarios (scaled from Liu et al (2019)) are included in our estimate of total costs.

Manure management measures are less clearly beneficial in summer compared to other scenarios, due to large fixed costs and price uncertainties. However, in winter the benefits of manure management for reducing $PM_{2.5}$ are substantial. Using lower protein feeds reduces animal feed purchase costs and has no adverse impacts on the animals and hence is always beneficial. Costs of purchasing and operating manure injection machines are more than fully compensated over the machine lifetime by savings from reduced N fertilizer purchases (SI) however logistical complications can be an impediment. Using aerobic composting reactors for manure storage introduces cost uncertainties, because of large variations in sale prices for organic fertilizer, discount rate and machine lifetime (e.g. organic fertilizers of low and high quality sold at US$29 tonne^{-1} and US$144 tonne^{-1} respectively, machine lifetime of 20-50yr and discount rate ranging from 2 to 14%). As a result of these uncertainties, the *Manure Management* scenario is beneficial only under high benefit and low/medium cost estimates.

3. Discussion and Conclusions

China's rapid economic development and success in addressing food insecurity have created multiple environmental challenges including air pollution, water pollution, unsustainable agricultural N use, increased greenhouse gas emissions, etc. To address these concerns, the Chinese government has signaled a change in direction for agricultural nitrogen use by setting a number of short-term policy goals for 2020. These include the "Three-year Action Plan Fighting for a Blue Sky" (DRC, 2018) which for the first time identifies NH_3 in efforts to address $PM_{2.5}$ air pollution, "Zero N fertilizer use increase by 2020" (MOA (Chinese Ministry of Agriculture) 2015) which is a first step towards a future reduction in N use, "Action Plan of converting animal manure to nutrients (2017-2020)", a goal of "75% processing rate of livestock and poultry waste" which identifies the importance of managing manure disposal to avoid damage to the environment (MOA (Chinese Ministry of Agriculture) 2017), a tightened regulatory standard for discharge of animal waste to rivers drafted for public review (Ministry of Ecology and Environment of the P.R. China n.d.), and a goal of having 400 counties demonstrate 'High Efficiency and High Yield Agriculture'.

Contradictory to previous work that views Chinese air pollution and NUE challenges as completely separate issues (Liu *et al* 2019a), in this study we identify four agricultural N management technologies that address multiple environmental problems including air quality, and three out of four technologies prove to be independently cost-effective.

Our agricultural nitrogen management scenarios reduce NH_3 emissions by up to 34% nationally and reduce $PM_{2.5}$ air pollution by up to 8 μg/m^3 both in January over central China (i.e. Hunan, Hubei, Guizhou and Chongqing provinces, etc) and in July over the North China Plain (i.e. Hebei, Beijing, Tianjin and Shandong provinces, etc).

The evaluation of yield, NUE, reactive N water pollution and GHG implications of improved N management are largely neglected in other studies. We include them and find they represent ~80% of total benefits (49–51)(Liu *et al* 2019a). We find that co-benefits from N management applied to grain crops alone and livestock husbandry include 36 -52 million tonnes (7%-9%) increase in national crop yields, 0.01- 0.05 Tg N (9%-50%) decrease of N_2O emissions, 0.23-0.62 Tg (14%-38%) decrease of NO_3-leaching, 0.06-0.14 Tg (23%-56%) decrease of NO_3-runoff, increase of average grain crop NUE from the current value of 38% to up to 48%, 4.7 -38 Mega tonnes CO_2-eq decrease of life-cycle GHGs, and 0-2.5 Tg N (0-21%) decrease of N loss to water from animal farms. . Incorporating acid rain and N deposition impacts from Liu et al (2019), we find that individual N management options deliver net benefits of US$9.4-19 billion/annum and remain cost-effective, except for the *Manure Management* scenario under low and medium benefit estimates. Our *Combined* scenario delivers benefits of US$30 (25; 50) billion/annum against costs of US$18 (12; 27) billion/annum.

Our estimated benefits remain conservative. For example, recent findings indicate that urease inhibitors can reduce NH_3 emission rates from fertilized fields by 84% in China (Li *et al* 2017b), while we use 42-58%. Our analysis of yield, GHGs and water pollution co-benefits leaves out potential gains of improved fertilizer management for Chinese vegetables and fruits, as well as crops other than the three major grain crops (i.e. maize, wheat, and rice). Our yield gain estimation for the *Combined* scenario is conservative and assumes no cumulative effects of multiple nitrogen management strategies due to lack of field experiments. Our *Manure Management* scenario does not include technologies that deal with Nr emissions within animal houses (e.g. conveyer belts to collect chicken manure) and that have more economic potential, e.g. anaerobic digestion that will generate biogas), nor does it tackle direct manure discharge to water bodies (which is estimated to be 30-70% of total manure in the year 2000 (Strokal *et al* 2016)) and is a major source of water pollution (Chadwick *et al* 2015)) and damage effect of manure odor and pathogen content. Including these benefits could make manure management improvements appear more attractive. Our *Machine Application* scenario assumes deep placement of N by machinery. However, deep placement of N by hand is more cost-effective (Table S20), mainly because the NH_3, N_2O, NO_3-leaching and runoff benefits remain with deep placement, the loss of labor savings is small, and the cost of large machine rental is removed. In contrast, machines that broadcast fertilizers (frequently dressing machines) are not cost-effective since the only benefit they achieve is saving labor costs while machine rental costs are estimated to be 10 times larger. Our valuation of avoided premature deaths from reduced exposure to $PM_{2.5}$ is also conservative, since the values of statistical life is from 2010 and will increase as the Chinese economy continues to develop.

The yield changes estimated in this study exclude other factors influencing yields such as depletion of aquifers and ground-level ozone pollution which decrease crop yields, as well as climate change for which the impact is uncertain. Climate change (with CO_2 fertilization effect included) is projected to increase Chinese grain crop yield (except maize) slightly by 3-15% in the 2020s (Piao *et al* 2010). However, climate change (with CO_2 fertilization effect excluded) is projected to decrease grain crop yields by 37% (Erda Lin *et al* 2005), making it even more crucial to implement other yield-increasing technologies to secure food supply. Research finds that improved N management, together with improved planting techniques and seed varieties can increase crop yields in China by ~15-30% (Chen *et al* 2014, Cui *et al* 2018).

Compared to the alternative $PM_{2.5}$ control strategies of reducing NO_x and SO_2 emissions, reducing NH_3 emissions incur technological adoption costs (estimated here to range US$0 to ~4000/tonne NH_3 across technologies and estimated to be US$1500/tonne NH_3 by Liu et al (2019)) are lower or comparable to technological adoption costs of reducing SO_2 emissions in China (ranging from US$2500-5000/tonne SO_2 across major mega cities (Kanada *et al* 2013)). However, since the same mass reduction of SO_2, NO_x and NH_3 result in different levels of $PM_{2.5}$ reduction, and different externalities are associated with NO_x and SO_2 emission reductions (reduced O_3 pollution and acid rain) compared to NH_3 emission reductions, future research is needed to provide comprehensive cost-benefit analyses for unexploited SO_2 and NO_x mitigating technologies.

Despite our finding of net economic benefits, financing and socioeconomic barriers and uneven distribution of costs and benefits prevent Chinese farmers and livestock ranchers from easily improving their N management practices. The largest opportunities exist with the 15% of farmers in China that cultivate approximately half of the cropland (Anon n.d.) and with industrialized animal farms, because improvements in their practices realize economies of scale and incur lower transaction costs than smallholders. Educating the 224 million smallholders (cultivating land <2/3ha) who cultivate the other half of available cropland (Anon n.d.) has high transaction costs. One rural research station in the North China Plain, funded by the central and local government, has helped local farmers increase yields from 68% to 97% of attainable level by forming trust relationships between researchers and farmers allowing for improved management and planting techniques. Scaling such education programs to nationwide smallholders would cost US$ 60 billion (Zhang *et al* 2016). In addition, except for improvements in yields and reduced labor costs, benefits of changed farming practices accrue to society at large rather than to the farmer. Agro-environmental payments and 'polluters pay' rules are needed to address the unequal distribution of costs and benefits (Matthews 2013, Baylis *et al* 2008, Knowler and Bradshaw 2007).

Overall, our findings support an increased emphasis on improved agricultural nitrogen management to address multiple challenges China faces including air pollution, food security, low agricultural nitrogen use efficiency and climate change.

4. Materials and Methods

4.1 Agricultural nitrogen management scenarios.

We design five agricultural N management scenarios based on a literature review (SI Table S1 and S19).

4.2 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 (Figure S1) and NH_3 emissions in the five N management scenarios. The NH_3 emission model we use was published in Zhang et al (Zhang *et al* 2018) and is the most updated and best available bottom-up high-resolution NH_3

emission estimation tool for China. The model represents N fertilizer application for 21 crops (including spring and summer maize, winter and spring wheat, early and late rice, potato, sweet potato, rapeseed, soybean, groundnut, tobacco, cotton, citrus, banana, grape, apple, pear, other fruits, vegetables). Crop NH_3 emission factors are parametrized including fertilizer application timing, rate, type, and method, as well as a number of climate variables (temperature, wind, etc.) and local soil (pH) conditions. The model represents major animal production (cattle, goat, sheep, pig and poultry) in grazing, intensive and free-range systems. Total ammonium nitrogen (TAN) content excreted by outdoor animals are subject to NH_3 volatilization and are without further management. TAN excreted by indoor animals goes through several stages of management, i.e. animal housing, manure storage and manure spreading, with each stage suspect to NH_3 volatilization. NH_3 emissions are gridded at $0.25° \times 0.25°$ resolution.

4.3 Air quality simulation. We use the Weather Research and Forecasting – Chemistry (WRF-Chem) model v3.6.1 to simulate $PM_{2.5}$ formation in the base case and in agricultural N management scenarios. WRF-Chem is an online-coupled meteorology-chemistry model widely used for air quality research (Gao *et al* 2016, Qin *et al* 2017). The chemical and physical schemes, emission inventories and initial/boundary conditions used are provided in the SI.

We conduct 6 sets of simulations: one *Baseline* and five agricultural management scenarios. The only difference between the *Baseline* and management scenario simulations are modified NH_3 emissions due to various agricultural N management options. Each simulation set includes one month of simulation for January and one month of simulation for July (in all cases after six days of spin-up) for the year 2012. The model resolution is 27 km by 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 levels extending from the surface to 50hPa with an 18m deep surface layer.

4.4 Estimating yield, GHGs, NUE and water pollution implications. We evaluate co-benefits of yield, NUE, NO_3^--leaching and runoff to water, GHGs, labor savings and fertilizer purchase expenditure through implementation of improved N management scenarios exclusively for grain crops (wheat, maize and rice) and livestock husbandry. All calculations are conducted at agroecological regional levels and aggregated to national estimates using crop planting areas listed in the 2009 China Statistical Yearbook. Definition of wheat, maize and rice regions are the same as that defined in (22). For the *Baseline* scenario, we estimate emissions of N_2O, NO_3-leaching and NO_3-runoff based on the amount of N fertilizer used using statistical relationships built from field measurements (Cui *et al* 2013, 2014) (listed in Table S18). *Baseline* nitrogen fertilizer application amounts and yields are from over 5000 large-scale household farm surveys conducted under National 948 Project (2003-Z53) and Agricultural Research Project (200803030) (Table S2) (Wu 2014, Wu *et al* 2015a, 2014, Chen Xinping 2016). For *Machine Application* and *Efficient Fertilizer* scenarios, we estimate Nr emissions by scaling the *Baseline* Nr emissions and yields by management-specific factors which represent the effect of technology on yield and Nr emissions. These factors are obtained from a meta-analysis of field experiments (listed in Table S19) (Xia *et al* 2017). We estimate Nr emissions in the *Combined* scenario by scaling emissions in the *Reduced N Application* scenario with management technology impact factors (listed in Table S19) (Xia *et al* 2017). We estimate yield increases in the *Combined* scenario using yield achieved in *Reduced N* Application scenario. We estimate reduced N loss to water from animal farms achieved in the *Manure Management* scenario using estimates of total

N lost from animals and contribution by various manure handling stages (Bai *et al* 2016b, 2018). We calculate NUE for each scenario, using additional data of atmospheric N deposition from the atmospheric model (Zhao *et al* 2017b), manure N production (Zhang *et al* 2018, p 3), N content in irrigation water (Gu *et al* 2015)and crop N content (Bouwman 2013). We calculate GHGs mitigation for scenarios that have reduced N application rates using a reported value of 13.5 tons CO_2-eq GHGs (including N_2O, CH_4 and CO_2 with GWP at 100-yr time scale) emitted per ton of N fertilizer consumed in China during nitrogenous fertilizer manufacture, transport, application and postapplication (Zhang *et al* 2013). We calculate GHGs mitigation for scenarios that have conventional N application rates by converting reduced N_2O emission due to management to CO_2-eq.

4.5 Economic analysis. We monetize all the benefits using value of statistical life, social costs of CO_2-eq emissions (Schiermeier 2009), damage costs (including global warming, health impacts, eutrophication, and acidification) of N_2O and NO_3-leaching and runoff (Table S15), wholesale grain prices (Chinese NDRC (national development and reform council) 2016), fertilizer sale prices (Xia *et al* 2017), and labor costs (SI). We conduct our own cost analysis for implementing each N management scenario. We calculate costs and benefits when economic parameters used are at low, medium and high levels.

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Chapter 4: Environmental and Dietary Health Co-benefits and Tradeoffs in Potential Future Chinese Dietary Choices

1. Introduction

China needs to address increasing environmental damage from agriculture. China's very high nitrogen (N) fertilizer use and poor manure handling has resulted in substantial N loss to the environment. For example, ammonia emissions (NH_3) that are ~5 times of that in the U.S.(Hoesly et al., 2018), contributing ~10%-18% to health-damaging particulate air pollution ($PM_{2.5}$)(Wang et al., 2011a). Direct emissions of greenhouse gas (GHG), including nitrous oxide from fertilizer application and methane from livestock husbandry, account for 8% of the country's total GHG emissions. With indirect GHG emissions (land use change, food processing and transportation) included, agriculture is responsible for 18% of national GHGs(Chen and Zhang, 2010; Vermeulen et al., 2012). Agriculture withdraws 64% of surface and ground water and occupied 72% of land(FAOSTAT, 2015).

Demand-side dietary strategies can help mitigate agricultural environmental damage, yet recent dietary transitions in China and in most places of the world have not followed a pro-environmental path. Diets have shifted towards food products such as meat, fruits and vegetables which are more resource- and emission- intensive than stable crops(Eshel et al., 2014). As a result, environmental burdens posed by agriculture have increased (Figure 1).

Despite increased environmental burden, recent dietary transitions also overall have worsened dietary health. Although China's undernourishment rate has dropped, overweight, obesity and premature deaths attributable to dietary risks have surged in the past decade. As of 2017, more than ¼ of global premature deaths attributable to dietary risks, i.e. 3.4 million deaths, happened in China. Diet-related deaths in China in 2017 is 2 times of that back in 1990. These dietary premature deaths (mainly heart disease, stroke, Type II diabetes and site-specific cancers) are attributable to risk factors including low intake of whole grains, nuts, seeds, fruits and vegetables and high intake of sodium and red meat(World Cancer Research Fund/American Institute for Cancer Research, 2018).

Previous studies have examined impacts of dietary changes in European, U.S. and China on the environment (Westhoek et al., 2014; Heller and Keoleian, 2015; Song et al., 2015; Behrens et al., 2017; Lei and Shimokawa, 2017), in some cases, exploring dietary change strategies that can bring simultaneous benefits for dietary health and GHG mitigation(Friel et al., 2009; Lock et al., 2010; Song et al., 2017; Springmann et al., 2018). To our knowledge, there has not yet been study evaluating effectiveness of dietary changes for reducing $PM_{2.5}$ air pollution through reducing emissions of ammonia (NH_3), although NH_3 emissions are mainly emitted from agriculture and have been found to limit formation of inorganic $PM_{2.5}$ in winter in the eastern U.S.(Pinder et al., 2007), Europe (Banzhaf et al., 2013; Megaritis et al., 2013), and China' s Pearl River Delta, and even in summer in Europe(Wang et al., 2011b). In addition, several Chinese governmental action plans (e.g. the "2018 Action Plan for Fighting for Agricultural Pollution"(Ministry of Agriculture and Ministry of Ecology and Environnment of P.R. China, 2018), "Three-year Action Plan Fighting for a Blue Sky" (DRC, 2018),"Citizen Nutrition Plan for 2017-2030"(State councile of P.R. China, 2017), etc) exist but tackle agricultural environmental pollution and nutrition issues separately. For the way forward, it is crucial to identify possible dietary choices that can reverse current tendency of diets being increasingly harmful for the environment and health. This can better inform China's air pollution, climate change, agricultural nitrogen use and nutrition policies, and help realize multiple Sustainable Developmental Goals (including Zero Hunger, Good Health and Well-Being, Responsible Consumption and Production and Climate Action).

In this study, we draw lessons from historical evolvement of Chinese diets as well as associated dietary and environmental consequences and analyze four potential future Chinese diets for their impacts on health and the environment. We construct future dietary scenarios that represent worst/best cases for dietary health and environment. When estimating baselined Chinese diets, we utilize China Health and Nutrition Survey(Popkin et al., 2009; Carolina Population Center at University of North Carolina at Chapel Hill and Chinese Center for Disease Control and Prevention, 2011; He et al., 2018), which is more representative of people's actual food intake, compared to national statistics of food consumption, which suffers from over-reporting especially regarding livestock products(Zhong, 1997; Fuller et al., 2000; Ma et al., 2004) and other errors during the surveying process conducted by local bureaucracies(Peng et al., 2019). We evaluate environmental impacts including NH_3 emissions, $PM_{2.5}$ air pollution, consumption-based GHG emissions, total water footprint (TWF), land appropriation (LA), and land use change carbon emissions (LUCC) resulting from growing the consumed food. We evaluate health impacts of changes in $PM_{2.5}$ air quality triggered by changes in food demand thus production, as well as dietary health impacts.

2. Results

2.1 A historical perspective: Chinese diets, health & environment

Fig. 1 provides an overview of evolving Chinese diets, dietary health and major food-related environmental burdens during the last decade, in the context of the global and U.S. trends (as we include a future scenario of U.S. diet capturing potential westernization of Chinese diets). Structural changes of diets in China have occurred at rates more rapid than that globally and in the U.S..

In the past decade, China has experienced dietary shifts towards more animal products, associated with decreased undernourishment but increased obesity and dietary related premature deaths (Fig. 1). During 1998-2016, per capita meat protein supply in China have increased by 46%, at a rate much more rapid than global rates. The share of starchy food in total calories dropped from 60% in 1998 to 50% in 2011, approaching the global level. Undernourishment in China has dropped from 16% in 1999 to 9% in 2016. In contrast, obesity rates have surged from 2% in 2002 to 7% in 2012(Bygbjerg, 2012; National Health and Family Planning Commission, 2015; World Bank, 2017). China's all-cause mortality rates from dietary risks have increased from 0.14% in 1990 to 0.22% in 2017, in comparison to a global stabilization around 0.14%.

All indicators of agricultural environmental damage have worsened during this time period, except for NH_3 emissions. Food consumption-based per capita life-cycle GHG emissions increased by 13% during 1997-2011, which have been less than ½ of that in the U.S.. National total food-related GHGs have increased from 700Mt in 1997 to 842 Mt in 2011, accounting for

15% of total national GHGs in 1997 and 7% in 2011(Climate Action Tracker for China (available at https://climateactiontracker.org/countries/china/),). Food consumption-based per capita total water footprint (TWP) also has increased by 25% during 1997-2011, much higher than that in the U.S.. National total food-related water withdrawal accounted for only 22% of the country's renewable internal freshwater back in 1997 but as much as 30% in 2011(The World Bank DataBank(available at https://data.worldbank.org/indicator/ER.H2O.INTR.K3?locations=CN),). Food consumption-based per capita land use has increased by 50%. Nitrogen use efficiency (NUE) for all crops in China has decreased from 70% in 1960 to 35% in 2012, in comparison to crop NUE globally that also declined but stabilized at around 50%. Low NUE indicates large fraction of agricultural N lost to the environment as nitrate (NO_3^- leaching and runoff) causing eutrophication, N_2O emission causing climate change, and $PM_{2.5}$-contributing NH_3 emissions which have been 12Tg/yr.

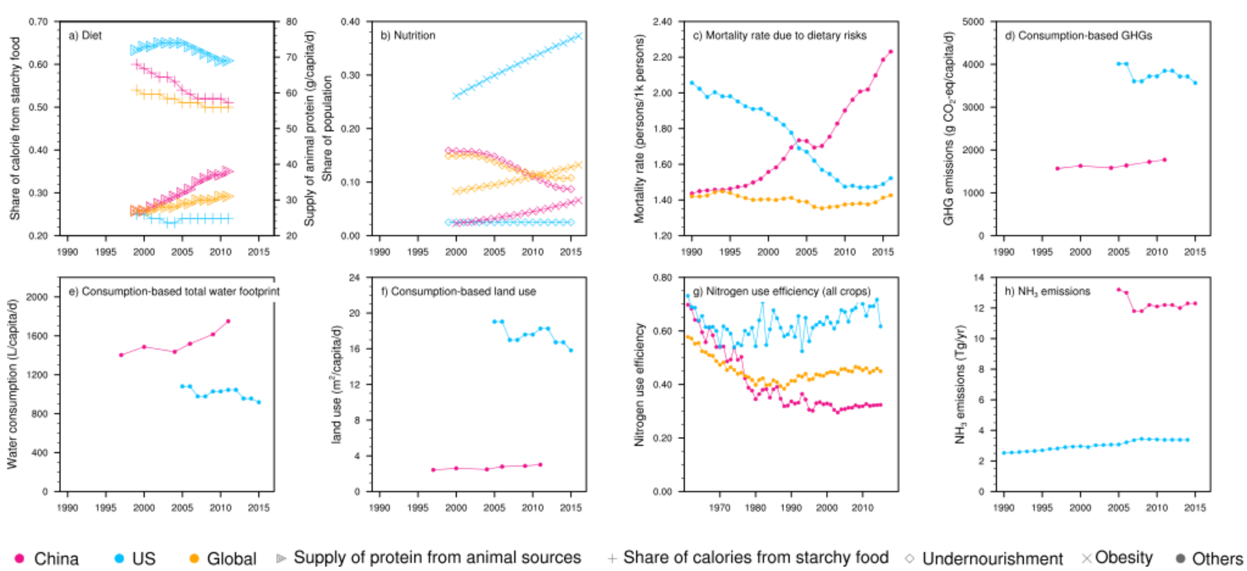

Fig. 1 Historical trends of diet and associated health and environmental implications in China, the United States (US) and the globe. a) Diet: Share of calories from starchy food (including cereals, roots and tubers, left) and supply of protein from animal sources (right) (g/capita/day), adopted from FAO Food Security Indicators [17–19]; b) Nutrition: rates of undernourishment and obesity, adopted from FAO Food Security Indicators (FAOSTAT, 2015); c) Mortality rate due to dietary risks (persons/1000 persons, adopted from Global Burden of Diseases 2017 Study (Institute for Health Metrics and Evaluation (IHME), 2018); d) Food consumption-based greenhouse gas emissions (g CO_2-eq/capita/day) adopted from ; e) Food consumption-based total water footprint (TWF) (L/capita/day) adopted from (IMPLAN's regional economic research data for the United States (available at http://www.implan.com/data/),; He et al., 2019); f) Food consumption-based land use (m^2/capita/day) adopted from (IMPLAN's regional economic research data for the United States (available at http://www.implan.com/data/),; He et al., 2019); g) Nitrogen use efficiency: average for all crops and h) NH_3 emissions (Tg/yr) adopted from (Zhang et al., 2018, p. 3; Hoesly et al., 2018). Colors indicate country/globe; Shapes indicate characteristics (see legend above).

2.2 *Baseline* diet and four future dietary scenarios:

Changing Chinese dietary choices have adversely impacted health and the environment in China, for the way forward, it is critical to identify possible consumer-side dietary change strategies that can benefit public health and environmental protection. We model individuals' (see Methods) choices of food among ~4000 Chinese food products and estimate national food consumption under *Baseline* condition and four dietary scenarios, in order to analyze their implications for health and the environment:

Baseline diet describes Chinese diets in 2011, constructed by mapping the latest China Health and Nutrition Survey (CHNS) for 2011(Carolina Population Center at University of North Carolina at Chapel Hill and Chinese Center for Disease Control and Prevention, 2011) to national population based on eating habits and socioeconomic status. Supplemental Appendix Table S1 and S2 provides *Baseline* per capita (national total consumption divided by national population) daily food intake and consumption (consumption = intake + food loss (food loss ratios provided in Table S3)) without and with standardization of food items. The standardization process (See Supplemental Appendix materials and Table S4) defines a reference food item within a specific major food group with certain content of nutrients and then converts consumption of other sub-group food items to consumption of this referenced 'standard' food item.

US Diet describes that Chinese intake of nutrients will match those included in a typical US diet, through choosing among food products available on the Chinese market. Nutrient intake of the U.S. people is from the U.S. Centers for Disease Control and Prevention's National Health and Nutrition Examination Survey (NHNES)(Centers for DIsease Control and Prevention,) during 2005-2016.

Soy Replaces Red Meat (SRRM) describes all red meat (goat, sheep, beef and pork) consumption in *Baseline* diet is removed with loss of protein compensated by soybean products. This scenario potentially can benefit both environmental and health by reducing red meat production/consumption(Melina et al., 2016).

Chinese Dietary Guideline Diet (CDG) describes that Chinese follow a nutritionally healthy diet, i.e. Balanced Dietary Patterns (recommended by the Chinese government's '2016 Chinese Dietary Guideline') based on their daily activity levels. The recommendations of food intake are by major food groups (e.g. fruits, leafy vegetables, whole grains) with intake of each sub-group food items not specified (e.g. apple v.s. banana, etc). We assume that people's preferences among various sub-group food items are the same as that in *Baseline.*

Lancet-EAT Dietary Recommendations (EAT) describes that Chinese adults (>20yrs old) follow a nutritionally healthy and environmentally sustainable diet recommended by the EAT-Lancet Commission(Willett et al., 2019). Diets of those below <20 yrs old are the same as *Baseline.*

Supplemental Appendix Table S5 and Figure S1 and Fig. 2 provides national food intake and consumption in *Baseline* and dietary scenarios. Food loss during production and consumption processes in dietary scenarios are estimated using *Baseline* loss ratios.

Supplemental Appendix Table S6 provides national food production in dietary scenarios as a ratio of *Baseline*. We estimate production by assuming production of a food product will scale up/down following the same ratio that consumption has changed, except for crops that are used for animal feed (soybean, vegetables, maize, wheat and rice). Changes in production of these crops reflect both changes in human consumption and changes in production for animal feed. We also assume that net import of food (e.g. soybean) as a share of total domestic supply (Supplemental Appendix Table S7) in dietary scenarios is the same as that in *Baseline*.

2.3 Environmental Implications:

2.3.1 NH_3 emissions. Two future scenarios, *US* and *CDG* diets significantly increase NH_3 emissions in China. Baseline NH_3 emissions are 13.9 Tg NH_3/yr, with 5.3 Tg NH_3 from nitrogen fertilizer application and 6.8 Tg NH_3 from manure management (Table 1). *US* diet results in NH_3 emissions of 40.2 Tg (~290% of that in *Baseline*) because of its high consumption of fruits, poultry, dairy products and nuts which are respectively 4, 3, 16 and 8 times of consumption of *Baseline* (Fig. 2). *CDG* diet results in NH_3 emissions of 29.8 Tg (~220% of that in *Baseline*) because of its high consumption of fruits, vegetables, eggs, dairy products, aquatic products, soy products, root vegetables and nuts which are respectively 10, 2, 2, 16, 5, 2, 4 and 32 times of *Baseline* (Fig. 2) despite its low consumption of red meat, poultry and grain which is respectively ½, 4/5 and 3/5 of that in *Baseline*.

Two future scenarios, *SRRM* and *EAT* diets significantly decrease NH_3 emissions in China. *SRRM* diet results in NH_3 emissions of 8.7 Tg (~63% of that in *Baseline)* because it removes N-intensive production of pigs, beef cattle and goat. Associated animal feed production also decreased, e.g. maize (46% decrease), wheat (28% decrease) and rice (6% decrease). Locally NH_3 emission reduction achieved through shifting to *SRRM* can be as high as 20% in eastern China and 60% in northeastern, middle and western China where animal density is high (Fig. 2).

EAT diet results in NH_3 emissions of 11.4 Tg (72% of that in *Baseline*). Although *EAT* requires significant (moderate) increase consumption of fruits, soy products and nuts (vegetables and root vegetables) compared to *Baseline*, it also significantly cuts consumption of red meat (by 77%), poultry (by 33%) and eggs (62%) (Fig. 2 and Table S5). Locally NH_3 emission reduction can be as high as 60%, with spots in western China, lower basin of Yantze River and eastern China experience increased NH_3 emissions in January and July (Fig. 3).

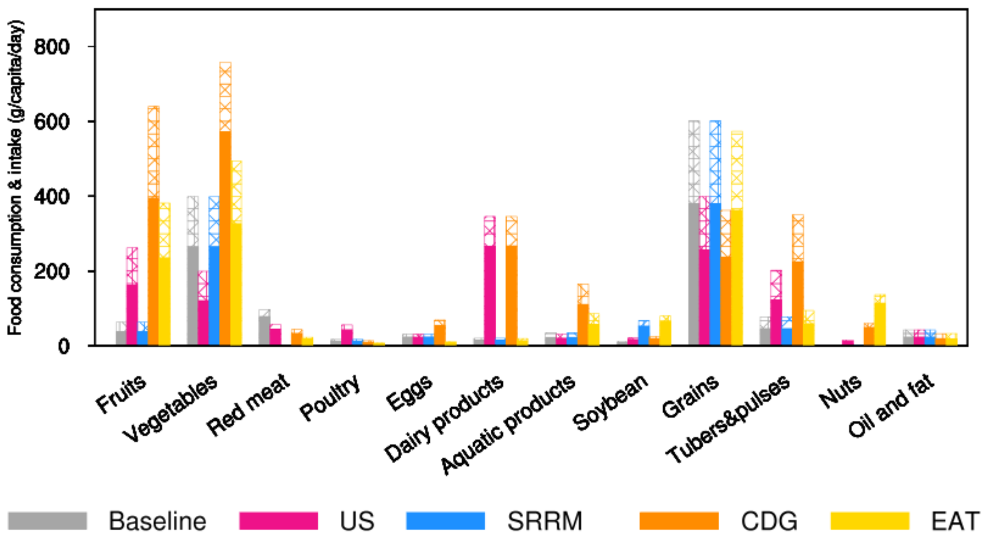

Fig. 2. Food intake (solid bars; fresh food weight with non-edible portion included), food loss (bars with filled patterns; including food loss during production, processing and consumption) and food consumption (solid bars plus bars with filled patterns) by food type for *Baseline* Chinese diet (estimated from the latest China Health and Nutrition Survey (CHNS) for 2011(Carolina Population Center at University of North Carolina at Chapel Hill and Chinese Center for Disease Control and Prevention, 2011)) and four future dietary change scenarios (*US*: typical 2011 US diet; *Soy Replaces Red Meat (SRRM)*: All red meat replaced with soy products; *Chinese Dietary Guideline (CDG)*: Recommendation of Chinese dietary guidelines; *Lancet-EAT Dietary Recommendations (EAT)*: healthy and sustainable diet recommended by Lancet EAT project). The unit is g/capita/day. Definitions of vegetables and fruits are based on Chinese

habits, e.g. cucumber, tomato, loofah and zucchini are categorized as vegetables; watermelon and muskmelon as fruits. All scenarios have daily per capita calorie intake of around 1800-1900 kcal.

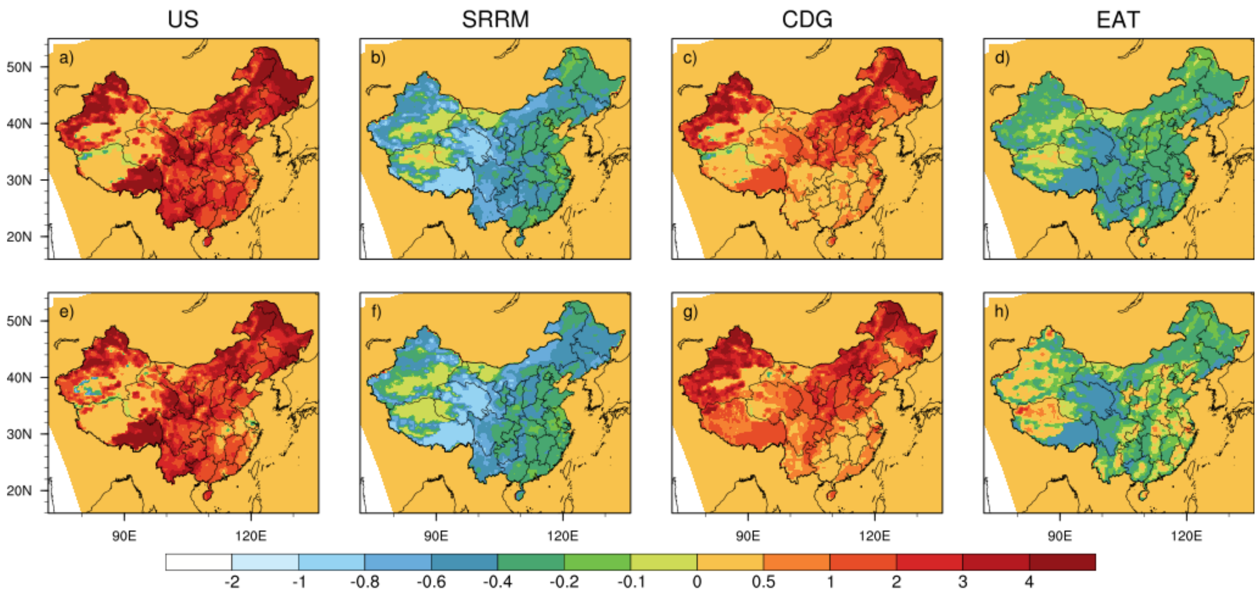

Fig. 3. Changes of NH_3 emissions in dietary change scenarios compared to *Baseline* NH_3 emissions divided by *Baseline* NH_3 emissions (scenario-*Baseline*/*Baseline*) in (a-d) January and (e-h)) July of the year 2012. The dietary change scenarios including US (a, e), SRRM (b, f), CDG (c, g), and EAT (d, h) are described in the text.

2.3.2 $PM_{2.5}$ air quality. Evaluation of WRF-Chem's simulation of NH_3 and speciated $PM_{2.5}$ concentrations and meteorology against satellite and ground-based observations can be found in Supplemental Appendix Figures S2-S8 and Table S8-S10 and Guo et al (2019).

Heavily increased (decreased) NH_3 emissions in the *US* and *CDG* (*SRRM* and *EAT*) scenarios increases (decreases) $PM_{2.5}$ air pollution in China. The *US* and *CDG* scenarios increase secondary inorganic aerosols (SIA, secondary inorganic fraction of $PM_{2.5}$ including sulfate, nitrate, and ammonium) concentrations nationwide and by up to 10 μg/m^3 locally, e.g. in wintertime eastern China and summertime North China Plain.

EAT and *SRRM* diets achieve comparable SIA reduction in winter (up to 12 μg/m^3 reduction in middle China). In summer, *EAT* generates slightly smaller SIA reduction than SRRM, e.g. 6-10 μg/m^3 in *EAT* vs. 10-12 μg/m^3 in *SRRM* over the North China Plain. However, *EAT* reduces annual NH_3 emissions by only 18% (compared to 37% achieved by *SRRM)*. This indicates that SIA production in summertime eastern China is not limited by NH_3 (but NO_x and SO_2), thus the differences between changes in NH_3 emissions achieved in *EAT* and *SRRM* does not heavily impact SIA reduction achieved in these two scenarios.

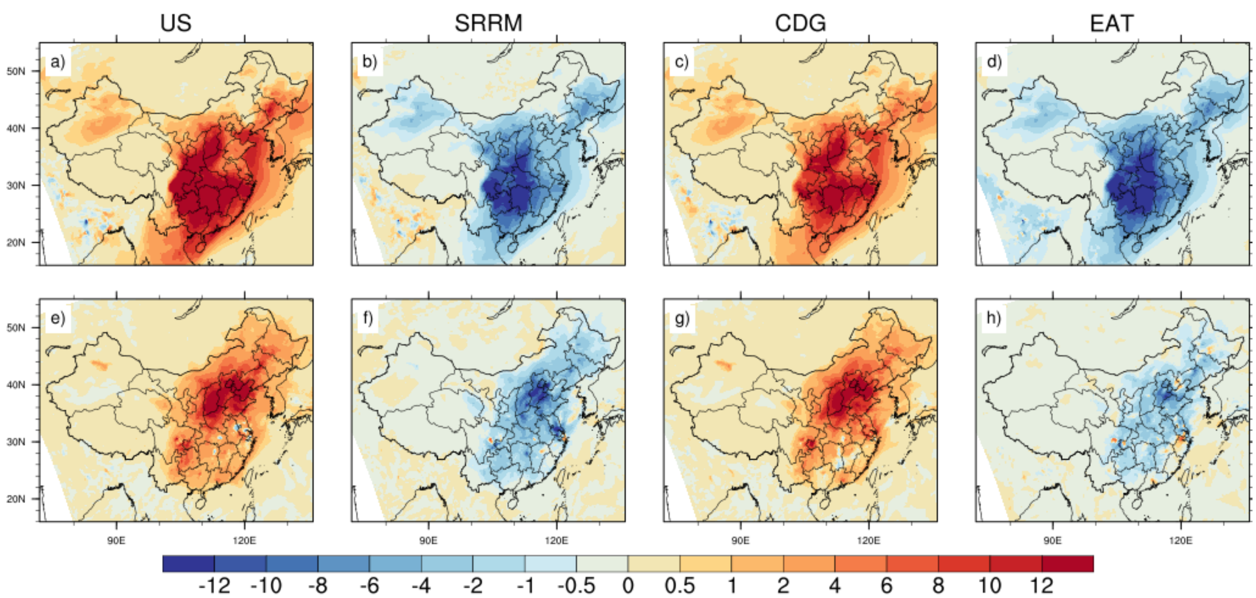

Fig. 4. Changes in ground-level concentrations of secondary inorganic aerosol (SIA) (in unit of μg/m^3, negative values mean reductions) in dietary change scenarios compared to the *Baseline* simulation, in (a-d) January and (e-h) July of 2012.

2.3.3 Other Environmental Impacts. Implications of dietary changes for consumption-based lifecycle GHGs, total water footprint, land appropriation and land-use change carbon emissions are presented in Fig. 5 (Supplemental Appendix Table S11-S14). Shifting from *Baseline* to three scenarios, *US, CDG* and *EAT,* worsens multiple environmental impacts, i.e. respectively increases GHG emissions by 20%, 40% and 0, increases TWF by 33%, 67% and 78%, increases LA by 56%, 100% and 38%, and increases LUCC emissions by 138%, 88% and 0 (Table 1). The only scenario that reduces these environmental impacts are *SRRM* where GHGs, TWF, LA and LUCC are respectively reduced by 30%, 0, 6% and 29% compared to *Baseline*.

Changes in consumption of meat, eggs/dairy products, and vegetables are major drivers of changes in environmental burden (Fig. 5). The *US* scenario consumes less pork (more poultry and beef) than baseline and results in increased GHGs and LUCC but decreased LA. Other scenarios all consume less red meat than baseline thus mitigate all environmental impacts associated with red meat consumption. High consumption of eggs/dairy, poultry and aquatic products in *US* and *CDG* scenarios resulting significant increases of environmental burdens. Increased soybean consumption in *SRRM* and *EAT* scenarios result in large TWF and LA requirements, yet negligible GHGs and LUCC impacts. High vegetable consumption in *US*, *CDG* and *EAT* scenario worsens TWF, LA and LUCC.

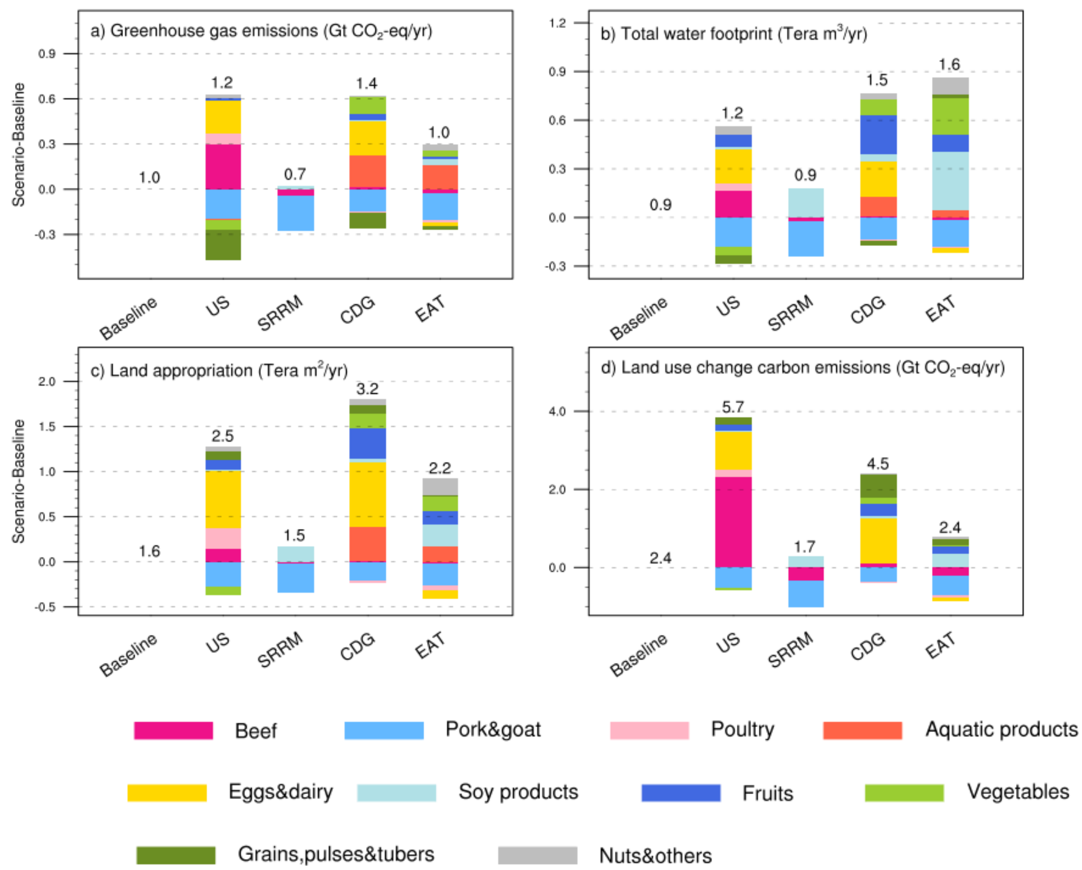

Fig. 5 Environmental impacts of food consumption in China in baseline and four dietary change scenarios. The bars represent changes of metrics when shifting from baseline to each dietary scenario. Numbers above each bar show the absolute value of the metric. a) Life-cycle food consumption-based GHG emissions emitted during food production and transportation (Giga tonne CO_2-eq/yr); b) Food consumption-based total water footprint (TWF) (Tera m^3/yr); c) Food consumption-based land appropriation (Tera m^2/yr); d) Land-use change carbon emissions indicating the opportunity cost of land (Giga tonne CO_2-eq/yr). For any food produced outside China and later imported for Chinese consumption, their environmental impacts are accounted by assuming emission factors during production/transportation are the same as those produced within China for indicator a), b) and c). Colors denote impacts from changes in consumption of different food types.

2.4 Health Implications. *US* and *CDG* respectively increase $PM_{2.5}$ related premature mortalities by 0.08 million and 0.06 million. Instead, *SRRM* and *EAT* respectively reduce premature mortality due to exposure to $PM_{2.5}$ by 0.057 and 0.055 million.

All scenarios generate significant dietary health benefits ranging from 0.02-1.4 million avoided premature mortalities. *CDG* generates the largest dietary benefit of 1.4 million avoided deaths, followed by *EAT* of 1.1 million. In both cases increased fruit intake acts as the major driver followed by increased vegetable intake. *SRRM* generates dietary benefits of 0.36 million avoided deaths with 0.056 million from reduced red meat consumption and 0.25 million from increased legume intake. *US* diet generates the smallest dietary health benefits of 0.02 million avoided premature deaths due to mixed effects of increased fruit intake (saves 0.38 million), reduced red meat intake (saves 0.02 million) and reduced vegetable intake (increases 0.64 million).

Combining $PM_{2.5}$ and dietary risk factors, *CDG* and *EAT* achieve public health benefits of reducing respectively 1.6 and 1.3 million premature deaths, followed by *SRRM* (0.29 million). The *US* case increases premature deaths by 0.08 million.

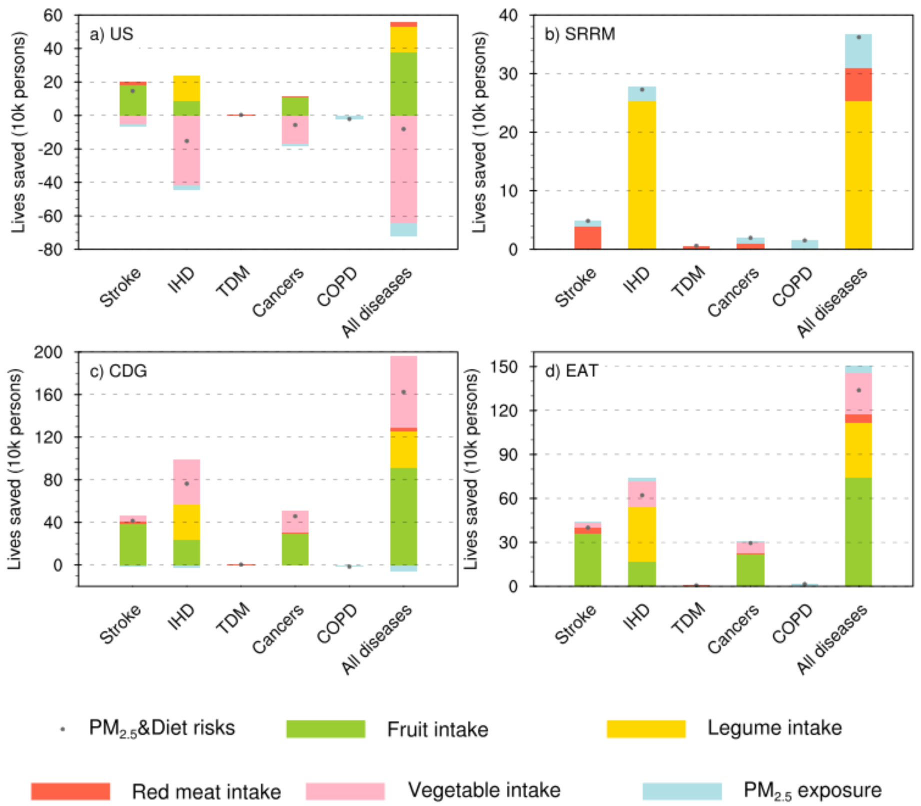

Fig. 6 Lives saved (10000 persons) from five diseases in four dietary change scenarios compared to baseline scenario, due to changes in food consumption and changes in exposure to $PM_{2.5}$ air pollution. Colors of bars indicate risk factors and grey dots denote all individual risks combined. End-point diseases considered include stroke, ischemic heart disease (IHD), Type II Diabetes (TDM), cancers (including colon and rectum cancers, lung cancer and other cancers, chronic obstructive pulmonary disease (COPD), and all these diseases.

Table 1. Environmental and health implications of Chinese dietary shifts from 2011 *Baseline* diet towards four potential future diets. For each metric, both *Baseline* values and changes in dietary change scenario, scenario-*Baseline*, are provided. Negative values mean mitigation of environmental impacts and lives saved.

Metric			Baseline value	Scenario-baseline			
				US	SRRM	CDG	EAT
Environment	NH_3 emissions (Tg/yr)	Total	13.9	26.3	-5.1	15.8	-2.5
		Fertilizer	5.3	0.8	-1.1	4.8	1.4
		Manure	6.8	25.5	-4.1	11.1	-3.8
	GHG emissions (Gt CO_2-eq/yr)		1	0.2	-0.3	0.4	0.0
	Total Water Footprint (Tera m^3/yr)		0.9	0.3	0.0	0.6	0.7
	Land appropriation (Tera m^2/yr)		1.6	0.9	-0.1	1.6	0.6
	Land-use change carbon emissions (Gt CO_2-eq)		2.4	3.3	-0.7	2.1	0.0
Premature mortalities (10k persons)	Exposure to $PM_{2.5}$		N/A	7.9	-5.7	6	-5.5
	Four dietary risks		N/A	-2.0	-29.3	-136.4	-110.9
	Fruit intake		N/A	-37.8	0	-91.3	-74.2
	Legume intake		N/A	-15.4	-25.3	-33.6	-37.6
	Red meat intake		N/A	-2.5	-5.6	-3.4	-5.3
	Vegetable intake		N/A	64.1	0	-68.5	-28.4
	$PM_{2.5}$ and dietary risks		N/A	8.11	-36.2	-162.6	-133.9

3. Conclusion

Food security in China achieved through increasing N fertilizer application and intensive resource use, as well as recent dietary shifts towards more animal products have resulted in rising environmental and dietary health concerns. These include increasing GHGs, land and water use, as well as increasing obesity, overweight and premature deaths attributable to dietary risks. It is critical that policies can shape future dietary pathways towards that are both beneficial to public health and the environment. Experiences from more developed regions such as the U.S.(Wang et al., 2014; Rehm et al., 2016) and Europe(Novaković et al., 2014) indicate that as society further develops, high-income groups will shift towards balanced diets.

Here we analyze the health and environmental co-benefits and trade-offs associated with dietary shifts from 2011 *Baseline* Chinese diets towards four potential future dietary choices: *U.S. diet (US), Soy Replaces Red Meat (SRRM,* red meat protein replaced by soy bean protein*), Chinese Dietary Guideline Diet (CDG)* and *Lancet-EAT Dietary Recommendations (EAT,* non-political dietary recommendations*)*. For the first time we evaluate impacts of dietary changes on NH_3 emissions and $PM_{2.5}$ air quality, as well as a full range of environmental (GHGs, water, and land) and health implications (through exposure to $PM_{2.5}$ and through intake of nutrients).

Our analysis for China indicates that healthy-eating does not necessarily benefit the environment and multiple environmental impacts may not change in the same direction. All four dietary scenarios reduce diet-related premature deaths by 0.02-1.4 million, yet two scenarios, *US* and *CDG* degrade all environmental impacts examined and one scenario *EAT* delivers mixed impacts. *EAT* increases water and land use, yet mitigates NH_3 emissions and $PM_{2.5}$ air pollution and keeps GHG emissions unchanged.

We find opportunities for substantially improving dietary health, i.e. shifting from *Baseline* diet towards all dietary scenarios, especially towards the two balanced diet scenarios. The two balanced diets, *CDG* and *EAT,* each avoid 1.4 and 1.1 million premature deaths attributable to the four dietary risks considered. Instead, *US* and *SRRM* respectively only avoid 0.02 and 0.3 million premature deaths. Dietary health benefits can be gained only if consumption of fruits, vegetables, nuts and whole grains increase and consumption of red meat decreases.

We also find opportunities for mitigating $PM_{2.5}$ air pollution, i.e. shifting from *Baseline* diet towards *SRRM* and *EAT* diets. *SRRM* and *EAT* respectively reduce NH_3 emissions by 37% and 18% thus decrease $PM_{2.5}$ concentrations by up to 8 ug/m^3, avoiding ~0.0057 and 0.0055 million premature deaths due to exposure to $PM_{2.5}$. To note, the $PM_{2.5}$-related health benefits are rather small compared to dietary health benefits.

We also find opportunities for addressing climate change and natural resource conservation, i.e., shifting towards *SRRM* diet. *SRRM* reduces GHGs by 30%, LA by 6% and LUCC by 29%. In other dietary scenarios, land appropriation, GHG and LUCC emissions increase mainly driven by increased dairy products and red meat consumption. Instead, increased total water footprint is mainly driven by increased vegetables, soybean, and dairy consumption.

Policymaking in the real world need to optimize dietary health and all environmental objectives. Although *SRRM* provides a win-win strategy for health and the environment, it remains a choice too radical for consumers to immediately accept. *EAT* is a no-regret choice, it delivers dietary health benefits that are ~3.7 times of *SRRM*, mitigates China's air pollution by up to 8 ug/m^3, keeps GHG emissions unchanged, although slightly increases land use and water use. *CDG* is not as good as *EAT* since it worsens China's $PM_{2.5}$, largely due to the very high dairy consumption requirement. However, *CDG* presents a better option than the *US* diet. In the *US* diet, dietary health benefits (from increased fruit and legume consumption and decreased red meat consumption) are largely offset by increased health risks from decreased vegetable consumption and worsened air quality. The high beef consumption and high dairy consumption in *US* diet is detrimental for all environment impacts examined in this study. Rapid westernization of Chinese diets in the past two decades needs to be taken with caution.

National dietary guidelines provide important references for the general public to improve their diets, our analysis indicates great opportunities for Chinese dietary guideline to improve. These include reducing recommended share of red meat within total meat, as well as adjust recommended milk intake levels. In the past, national dietary guidelines have solely considered the nutrition aspects of diets, however, for the way forward taking environmental footprints into consideration can be increasingly important. Given multiple food choices can fulfill one specific dietary purpose, there is definitely room to minimize environmental impacts while keeping micro-nutrients supply unchanged.

Nutrition, compared to air pollution, matters more to overall public health, although individuals can choose their diets but not clean air based on their wish. The significant dietary health benefits achieved through balanced diets are significantly larger than health benefits of improved air quality achieved through reduced consumption and production of nitrogen-intensive products in *EAT* and *SRRM*. This demonstrates the need to enhance public education of nutrition, as well as the need of adopting production-side mitigation technologies to mitigate adverse environmental impacts associated with balanced diets. Guo et al (2019) shows that cost-effective improved N management can reduce NH_3 emission by up to 34% nationwide. This allows some space for increasing production of fruits and vegetables to gain some dietary benefits while keeping NH_3 emission and $PM_{2.5}$ air pollution unchanged.

There's also a need for better understanding the gaps between baseline diets indicated by dietary surveys and macro-level statistics to better understand people's actual diets. In this study, we utilized Chinese nutritional surveys to construct baseline Chinese diets because of macro-level food consumption data for China obtained from Chinese national statistics suffer from 'human errors' (Supplemental Appendix) and Chinese nutritional survey data realistically captures dietary preferences of people with different socioeconomic status and eating habits. We have noted that for many countries in the world the macro-level food consumption data has a higher estimation of livestock product intake and a lower estimation of grain intake than dietary surveys(Del Gobbo et al., 2015). For China, macro-level data indicates substantially higher intake of vegetables and fruits than the data we used and the Global Dietary Database (Supplemental Appendix Table S15). Although dietary surveys can be vulnerable to under-reporting, which has been shown especially serious among population with severe obesity issues, macro food consumption data suffers from aggregation of errors during surveying food production, export/import, other utilizations and stock, etc. More research is needed to improve the quality of baseline diet data, since baseline diet is critical for understanding how far people are from a healthy and sustainable diet.

Overall our analysis indicates opportunities with changing diets to improve dietary health and environmental objectives including air pollution and climate change mitigation, and water and land use reduction.

4. Methods

4.1 *Baseline* diet and dietary scenarios. *Baseline* diet is from China Health and Nutrition Survey (CHNS) for 2011(Carolina Population Center at University of North Carolina at Chapel Hill and Chinese Center for Disease Control and Prevention, 2011) which sampled 10000 random people in twelve provinces with distinct socioeconomic and demographic backgrounds and tracked their food intake (types and weights) in three consecutive days. We then match diets of individuals outside the sample areas to those within based on socioeconomic conditions (indicated by income) and eating habits (indicated by province of residence), using demographic information of CHNS samples and of all Chinese provided by China Family Panel Studies (CFPS).

U.S. Diet is estimated using intake of major nutrients reported by U.S. Centers for Disease Control and Prevention's National Health and Nutrition Examination Survey (NHNES)(Centers for DIsease Control and Prevention,) during 2005-2016 period. *Soy Replaces Red Meat (SRRM)* describes all red meat (goat, sheep, beef and pork) consumption is removed and loss of animal protein is fulfilled by increased intake of soybean products. This scenario demonstrates a situation where potentially environmental and health co-benefits can be achieved, since reduced livestock production and corresponding animal feed production will lower environmental damage and reduced red meat intake will reduce health risks(Melina et al., 2016).

Chinese Nutritional Guideline Diet (CNG) is based on China's Balanced Dietary Patterns from 2016 Chinese Dietary Guideline, which includes intake amount of 14 food groups for people at 11 energy requirement levels. We determine the exact combination of foods in sub- food groups (the Chinese Food Content Tables, 2002 & 2004 version) by randomizing individual's choices within each food group for 10000 times through Monte-Carlo simulations while keeping people's dietary preferences within each food group the same as their baseline diets indicated by 2011 CHNS survey.

Lancet-EAT Nutritional Recommendations (EAT) is based on diets provided by EAT-Lancet Commission(Willett et al., 2019), which applies universally to all adults regardless of age and country of origin.

4.2 Estimates of agricultural production. *Baseline* agricultural production for the year 2012 is obtained from Chinese Statistical Yearbook. Production of one food type in dietary scenarios are estimated by scaling *Baseline* with factors equal to the ratio of food consumption in dietary scenarios to *Baseline* consumption. This rule applies to all agricultural products except for animal feed crops (maize, wheat, rice and soybean). Changes of animal feed crop production should reflect both changes in human food demand and changes in animal feed demand as a result of changed animal production (see Supplemental Appendix).

4.3 NH_3 emissions. We utilize an NH_3 emission model to obtain baseline NH_3 emissions and NH_3 emissions in dietary change scenarios. The NH_3 emission model we use was published in Zhang et al (Zhang et al., 2018) and is the most updated and best available bottom-up high-resolution NH_3 emission estimation tool for China.

4.4 Air quality simulation. We use the Weather Research and Forecasting – Chemistry (WRF-Chem) model v3.6.1, an online-coupled meteorology-chemistry model, to simulate $PM_{2.5}$ formation in baseline and scenarios at 27km by 27 km resolution for January and July of 2012 (after six days of spin-up). Details of model set-up are provided in Supplementary Appendix.

4.5 Estimate consumption-based lifecycle GHG emissions, water, land use and land-use carbon emissions. For baseline and each dietary scenario, GHG emissions are estimated using 300 lifecycle assessment (LCA) covering the emissions from cradle to farm gate worldwide following the methodology in He et al (2019)(He et al., 2019). Total water footprints are estimated using the Water Footprint Network database(Water Footprint Network (available at https://waterfootprint.org/en/),), which includes average water consumption for 352 plant-based and 106 animal-based products during the period of 1996-2005. Land appropriation are estimated using Food and Agriculture Organization Statistics (FAOSTAT) field data for plant-based food (averaged for the time period of 1996-2005) in China provided and plant to animal meat conversion factors. Land use change related carbon emissions are estimated using food-specific factors reported in Searchinger et al 2018(Searchinger et al., 2018). This metric goes beyond life-cycle consumption-based GHGs and land use metrics since it considers the opportunity cost of one piece of land if it is used for producing one specific type of food compared to for producing another type of food/biofuel and compared to simply remaining as forests for global carbon storage purposes. It measures the carbon efficiency of the land analyzed compared to global-average carbon efficiency.

We introduce the Monte Carlo simulation to estimate the uncertainty of the impacts of diets on the environment due to uncertainties in climate, technologies, errors from various evaluations, etc. We run a simulation repeated for 10000 trials. In each trial, environmental impact factors of each food group are generated from the assumed distribution with a specific mean and standard deviation retrieved from the dataset of environmental impact factors. We assume log normal distributions for GHG emissions of each food group based on the distribution of factors of our collection of LCA studies, and retrieve the mean and standard deviation for each food group. For water consumption, we assume a normal distribution for each of the 352 plant-based and 106 animal-based products from the Water Footprint Network database, and a 15% of the means as the standard deviations for each product following a previous study [35]. For land appropriation, we assume normal distributions and 5% of the means from the FAOSTAT data as the standard deviations for each food group due to the observations of the flat change in productivity over time in FAOSTAT. The simulation is repeated for 100 trials. We then link these generated factors to the CHNS dataset to evaluate the individual dietary environmental impacts.

4.6 Health impacts of exposure to $PM_{2.5}$ and diets. We calculate premature mortality of four diseases (chronic obstructive pulmonary disease (COPD), lung cancer, ischemic heart disease (IHD) and ischemic stroke) due to exposure to $PM_{2.5}$ for adults ($\geq$ 25 y old).

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

$$Mort_{i,P} = POP_P \times Mortbase_{i,P} \times (1 - \frac{1}{RR_{i,P}})$$

where $Mort_{i,P}$ is the number of premature mortality in province P from disease *i*; $POP_{j,P}$is the number of exposed targeted population in province P considering adults (≥ 25 y old) in 2012 from 2013 China Statistical Yearbook(All China Marketing Research Co. Ltd, 2014); $Mortbase_{i,P}$ is the baseline mortality rate in province P for disease *i* in 2012 from Global Burden of Disease study (Burnett et al., 2014); $RR_{i,P}$ is the relative risk factor for one disease *i* adopted from (Burnett et al., 2018). Relative risk factors for IHD and stroke are by age groups. There are 12 age groups considered, i.e. 25-29, 30-34, 35-39, 40-44, 45-49,50-54,55-59,60-64,65-69,70-74,75-79 and over 80 y old. Relative risk factors for lung cancer and COPD are the same for all people ≥ 25 y old.

We consider four dietary risk factors (intakes of red meat, vegetables, fruits and legumes) and six end-point diseases (coronary heart disease (CHD), stroke, type II diabetes (T2DM), colon and rectum cancers, lung cancer and other cancers). We estimate the mortality attributable to dietary risk factors by calculating “population attributable fractions (PAFs)” using relative risk factors reported in Aune et al(Aune et al., 2017), Kim et al(Kim Kyuwoong et al.,) and those used in Springmann et al(Springmann et al., 2018) (Supplemental Appendix Table S16).

In cases of one disease attributable to multiple risk factors, we assume PAFs combine multiplicatively, i.e. $PAF_{TOT} = 1 - \prod_i(1 - PAF_i)$.

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Chapter 5: Conclusions

Food is essential to human survival. Food production has created a wide range of severe environmental problems and food consumption has important dietary health consequences. Chinese agriculture feeds 21% of the global population mainly through intensive use of inputs such as synthetic nitrogenous fertilizers, water, land, etc, however, farmers' management skills and machinery coverage remain low. This results in substantial reactive nitrogen leakage to the environment causing air and water pollution, as well as greenhouse gas emissions and land appropriation. Such environmental damage has been further worsened by recent dietary shifts towards more animal products. However, despite decreased undernourishment rates over the past decade, mortalities attributable to dietary risks and obesity rates in China have increased due to underconsumption of fruits, vegetables, whole grains and nuts and overconsumption of red meat, sugar, etc.

We are at a critical juncture confronted with grand challenges of food security, climate change, air pollution, water pollution, resource exploitation and rural development; policy interventions are needed to transform agricultural production and consumer diets into ones that are healthy for the economy, the environment and public health. My dissertation explores the effectiveness of strategies from both the agricultural production side and consumer side in China in addressing one or multiple challenges posed by agriculture. Special attention has been paid to strategies that can generate more than one type of benefit or benefits that are internal to stakeholders (e.g. yield benefit, dietary health gains, etc). This is because co-benefits and benefits internal to stakeholders are more likely to incentivize changes in action compared to benefits that are largely dispersed at regional and global levels (e.g. climate change mitigation).

Chapter 2 focuses on costs and benefits associated with production-side technologies that can increase Chinese agricultural nitrogen use efficiency (NUE) and explores how to enhance technology and knowledge transfer by removing existing barriers. I find that adopting NUE-increasing management, i.e. reduced N fertilizer application, split application, use of new efficient fertilizers (controlled-released fertilizers and inhibitors) and machine deep placement, brings yield benefit of ¥70-300/mu (1ha=15mu). The yield benefit is much larger than any other type of benefit, including labor savings (<¥10/mu), fertilizer purchase savings (<¥30/mu), reactive nitrogen emission reduction (<¥60/mu), and greenhouse gas emission reduction (<¥30/mu)), but still remains too small for farmers to care, compared to relatively high off-farm earnings. Reasons that prevent farmers from improving practices also include high learning costs due to weak knowledge support from governmental extension schools and fertilizer sellers. I identify separate mechanisms for smallholders (who cultivate cropland <10mu) and large farmers (who cultivate cropland > 10mu and altogether cultivate 50% of China's cropland) through which technology transfer can be enhanced. Smallholders' improvement of practices relies on the development of farmers' professional cooperatives, private service contractors, as well as long-term education programs. Large farmers can benefit from economies of scale when they improve management and can be educated at significantly lower transaction costs.

Chapter 3 focuses on the potential of production-side technologies in reducing emissions of NH_3 and $PM_{2.5}$ air pollution in China and generating co-benefits of yield increase, water pollution mitigation, GHGs mitigation, etc. We identify cost-effective agricultural nitrogen (N) management opportunities in China: *Reduced N Application*, *Efficient Fertilizer, Machine Application*, *Manure Management*, and *Combined*. We find that when these approaches are applied to Chinese agriculture, they can annually achieve 6.4%-34% NH_3 emission reductions nationally in 2012, 0-8 μg/m^3 reduction of secondary inorganic aerosols locally and ~5,000-30,000 avoided $PM_{2.5}$-related premature mortalities. We find multiple co-benefits from these approaches when applied to grain crops and livestock of: 36-52 Mega tonnes (7%-9%) increase in national crop yields, 0.01- 0.05 Tg N (9%-50%) decrease of N_2O emissions, 0.23-0.62 Tg N(14%-38%) decrease of NO_3-leaching, 0.06-0.14 Tg N (23%-56%) decrease of NO_3-runoff, 4.7 - 38 Mega tonnes CO_2-eq decrease of GHGs, and up to 2.5 Tg N (21%) decrease of N loss to water from animal farms. Total benefits from all N management strategies combined are US$30 (25; 50) billion/annum, compared to costs of US$18 (12; 27) billion/annum.

Chapter 4 focuses on the potential of consumer-side dietary shifts in reducing emissions of NH_3 and $PM_{2.5}$ air pollution in China, water and land use and greenhouse gas emissions, as well as reducing premature mortalities attributable to dietary risks. All future diets examined deliver dietary health benefits yet with distinct environmental impacts. Two balanced diets, *Chinese Nutritional Guideline Diet* (*CNG*) and *Lancet-EAT Nutritional Recommendations* (*EAT*), each avoid 1.4 and 1.1 million diet-related premature mortalities, and two extreme diets, *U.S. Diet (US)* and *Soy Replace Red Meat Diet (SRRM),* respectively avoid 0.02 and 0.3 million. However, *US* and *CNG* diets significantly increase ammonia emissions, $PM_{2.5}$ concentrations, land and water use, and greenhouse gas emissions (GHGs). *SRRM* decreases these impacts and *EAT* increases some (water and land use) but decreases others (ammonia emissions and $PM_{2.5}$). With

SRRM providing a win-win for health and the environment, a combination of *SRRM* and *EAT* provides realistic future change in direction for Chinese diets to optimize health plus environmental objectives. Significant dietary health benefits achieved in the two balanced diets call for developments of production-side pollution-mitigating technologies.

Based on the findings from my dissertation, I suggest the following future research needed to better identify strategies to mitigate environmental and health damage from the agricultural sector and beyond:

1. **Comprehensive cost-benefit analyses for controlling NO_x, SO_2 and NH_3 emissions**

   NO_x and SO_2 emissions, in addition to NH_3 emissions, are important precursors to $PM_{2.5}$ air pollution. Unlike that NH_3 emissions are mostly from the agricultural sector, NO_x and SO_2 emissions are from energy, residential and transportation sectors. Effectiveness of NH_3 mitigation for reducing $PM_{2.5}$ will decrease as emissions of NO_x and SO_2 decrease. Instead, effectiveness of NO_x and SO_2 emission mitigation for reducing $PM_{2.5}$ will increase as emissions of NO_x and SO_2 decrease.

   Recent command-and-control regulations in China have reduced NO_x and SO_2 emissions by 17% and 62%, respectively, during 2010 to 2017. The $PM_{2.5}$ air quality benefits of mitigating NH_3 emissions claimed in Chapter 3 were based on air quality modeling of emissions for the year 2012. Mitigation of NH_3 emissions at present will lead to smaller reduction in $PM_{2.5}$, considering emissions of NO_x and SO_2 have decreased. Future research is needed to fully understand how the effectiveness of reducing NO_x, SO_2 and NH_3 emissions for reducing $PM_{2.5}$ concentrations evolves over time.

One obstacle for a comprehensive cost-benefit analysis of controlling NO_x and SO_2 emissions is to include evaluation of co-benefits provided beyond $PM_{2.5}$ air quality improvements. Reducing NO_x and SO_2 emissions will mitigate acid rain. Reducing NO_x emissions will also mitigate ground-level O_3 air pollution.

Another obstacle is lack of knowledge about current adoption of emission control technologies, unexploited technologies and their costs. Technologies within various sectors (energy, transportation, residential, etc) need to be considered. Knowledge about current status of technology adoption and unexploited technologies, at provincial level, is needed. This type of comprehensive review work has not yet existed for China, either.

GAINS-China model provides a good tool analyzing the mitigation potential of precursor emissions and associated costs at provincial levels. Coupling this tool with atmospheric chemistry modeling could inform China's future air pollution policymaking.

2. **Future global burden of reactive nitrogen under different socioeconomic pathways**

Future agricultural and industrial N mobilization, dependent upon population growth, dietary changes and production technologies, will have implications for future global burdens of reactive nitrogen, as well as associated water and air pollution and climate change. Since recent work finds that the global planetary boundary has been exceeded by almost twice

(Steffen et al., 2015), this work will help identify how severe future challenge of reactive nitrogen will be for the globe.

Previous work has been limited by idealized future projections of agricultural activities and simple modeling tools not capable of fully capturing interactions of global agriculture with the environment. Previous research has either utilized projections for future livestock and crop production that lack geographical variations (e.g. livestock production under different socioeconomic pathways provided by IIASA[23] and fertilizer consumption provided by CMIP6) or modeling tools that are not process-based (e.g. N flow/budgeting models (Bodirsky et al., 2014; Billen et al., 2014)).

Utilizing future projections of agricultural activities that are of high geographical resolution (work of Dr. Benjamin Bodirsky) and a flow of agricultural nitrogen model that is coupled to the community earth system model (published in Riddick et al. (2015) and Vira et al., (2019) can improve our understanding of the future global burden of N.

Also with future development of land model within earth system model (ESM), ESM will provide a great tool for studying interaction of agriculture with the environment, including N cycling, C cycling and climate-agriculture interactions. ESM will be complementary to existing models at farm- and regional levels such as EPIC, GEPIC and IMAGE.

Modeling agriculture and its impacts are difficult because agricultural management that varies greatly geographically currently is poorly understood and represented in models. Also, interaction of agriculture with the environment is dependent on local climate and atmospheric

[23] https://tntcat.iiasa.ac.at/SspDb/dsd?Action=htmlpage&page=10

chemistry conditions. A few efforts that aim to improve these representations of management and climate dependence have been at country-level or using parameterization (Paulot et al. 2014; Zhang et al., 2018). There has been a few recent work which couples nutrient models with earth system models to study impacts of nitrogen management on water pollution and air pollution (Lee et al., 2014; Riddick et al., 2015; Nevison et al., 2016).

3. **Optimizing Chinese dietary strategies for multiple environmental objectives**

Chapter 4 of this dissertation identifies a dietary change strategy, i.e. soy replaces red meat (SRRM), that provides a win-win for $PM_{2.5}$ air quality, greenhouse gas emissions (GHGs), total water footprint, land use and dietary health. However, this SRRM dietary change strategy is not optimal since the associated dietary health benefit is only less than ¼ of that achieved through balanced diet examined (i.e. the Chinese Dietary Guideline diet and the Lancet-EAT diet recommendations). There're opportunities of slightly increasing fruit and vegetable consumption to an extent that other environmental objectives remain no worse than the status quo but additional dietary health benefits can be gained.

An optimization framework is needed to address the dilemma that dietary changes cannot always simultaneously benefit dietary health and various environmental objectives including air quality, GHGs, water and land use. Dietary change such as increased fruits and vegetable consumption substantially benefit dietary health but requires substantial nitrogen, water and land input to agriculture. An optimal diet for Chinese can be determined only when different weights are assigned to objectives including dietary health, air quality, water use, land use and greenhouse gas emissions.

4. **Improving data quality and better understanding current diets**

Large gaps exist between diets indicated by dietary surveys and national statistics, which affects the validity of conclusions in Chapter 4. Although expected improvements are out of scope of this dissertation, future work is needed to improve understanding of current Chinese diets.

Chinese baseline diets indicated by nutritional surveys and national statistics are distinct especially regarding fruit, vegetable, legume and red meat intake. Dietary surveys would show that the Americans eats more fruits and legume, but less vegetables and red meat than Chinese (He et al 2018; Global Dietary Database). Statistics data agrees that Americans eat less vegetables than Chinese but indicates that Chinese eats more legume and less red meat than China (FAO Food Balance Sheet).

Both data have caveats. National statistics for food consumption in China have been found to suffer from 'human errors' (Zhong et al., 1997; Ma et al., 2004). Local governmental officials would over-report agricultural production especially for livestock product since agriculture is part of their political performance to be examined. For many countries in the world statistics of food consumption reports higher livestock product intake and lower grain intake than dietary surveys. Furthermore, national statistics data in China suffers from aggregation of errors because it requires surveys conducted by local governmental agents regarding food production, export/import, other utilizations and stock, etc. Food consumption is estimated by calculating the balance of different food usages. In contract, dietary surveys can be vulnerable to under-reporting, which has been shown especially serious among population with severe obesity issues in the US (Freedman et al., 2014). More research is needed to improve the quality of baseline diet data, since baseline diet is critical for understanding how far people are from a healthy and sustainable diet.

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Appendix for Chapter 2

Technological, Social and Policy Options for Increasing Nitrogen Use Efficiency in China

Figure S1 major cropping systems in China, as well as their N fertilizer application features. Data source: Zhenling Cui.

Table S1: Farmers' machine use during fertilizer applications for three major crops in their primary production regions in China, data from a survey of 1000 farms across China provided by Weifeng Zhang. Crop codes: 101 = spring maize; 102 = summer maize; 103 = autumn maize; 201 = winter wheat; 301 = single-crop rice; 302 = double-cropped early rice; 303 = double-cropped late rice. Area codes: I = triple-cropping area in South China, II = double-cropping rice area in South China, III = lower basin of the Yangtze River (paddy–upland rotation area), IV = North China Plain, V = irrigated area in Northwest

China; VI = arid area in Northwest China, VII = single crop area in Northeast China. Data compiled and delivered to me by Dan Zhang.

Crop Region		No. of farms	Farm size (ha)	Machine application coverage		Machine rental cost (¥/ha)	Time spent on fertilizer application (h/ha)		N use (kg N/ha)	Annual fertilizer purchase (¥/ha)
				% of land area	% of farmers		Hand	Machine		
Maize	I101	50	0.07	14	14	350	163	15	253	2766
	I103	40	0.07	20	20	350	166	19	232	2547
	III102	32	0.60	38	31	1050	65	2	208	1764
	IV102	703	0.92	59	35	376	20	3	214	2282
	V101	16	0.09	6	6	566	104	N/A	328	2439
	VI101	128	0.30	40	8	566	67	3	287	2534
	VI102	85	3.44	85	2	434	30	7	246	1991
	VII101	195	0.78	59	22	350	21	2	231	2790
Wheat	II201	16	0.65	50	50	300	13	1	180	1772
	III201	162	1.08	25	19	838	19	3	209	1817
	IV201	800	0.85	36	23	360	21	2	254	2513
	VI201	111	2.76	63	7	410	19	4	210	2297
Rice	I301	30	0.06	3	3	1200	60	N/A	178	2109

I302	104	0.07	3	3	1200	44	N/A	185	2463
I303	105	0.07	3	3	1200	42	N/A	188	2475
II301	142	0.17	4	4	1200	49	1	154	1912
II302	88	0.25	3	2	1200	33	1	154	1661
II303	91	0.25	3	2	1200	32	1	156	1695
III301	206	0.86	5	4	1500	24	2	251	1991
IV301	51	0.33	6	4	1800	24	1	304	2407
VII301	85	12.40	47	5	602	6	4	114	1424

Table S2: NH_3 volatilization (kg N/ha) under a certain level of fertilizer nitrogen input (kg N/ha) for maize, rice and wheat in North, Central and South China. Unpublished work from Cui et al.

Crop type	Region	NH_3 volatilization (Y; kg N/ha) with respect to fertilizer N input (x; kg N/ha)
Maize	North China	Y=2.53+0.058x
	Central China	Y=7.98+0.099x
	South China	Y=1.93+0.071x
Rice	North China	Y=3.83+0.1x
	Central China	Y=-0.54+0.2x
	South China	Y=4.95+0.17x
Wheat	North China	Y=3.21+0.068x
	Central China	Y=2.69+0.069x

	South China	Y=-0.61+0.13x

Table S3: modeled N_2O emission and NO_3^--leaching (kg N/ha) under a certain level of fertilizer nitrogen input (kg N/ha) for maize ((Cui *et al.*, 2013)), rice and wheat (both adopted from (Cui *et al.*, 2014)).

Crop	N_2O emission (Y; kg N/ha) with respect to fertilizer N input (x; kg N/ha)	NO_3^- leaching (Y; kg N/ha) with respect to fertilizer N input (x; kg N/ha)	NO_3^- runoff (Y; kg N/ha) with respect to fertilizer N input (x; kg N/ha)
Spring maize	Y=0.576exp(0.0049x)	Y=4.46exp(0.0094x)	N/A
Summer maize	Y=0.593exp(0.0045x)		
Wheat	Y=0.33exp(0.0054x)	Y=2.7exp(0.0088x)	
Rice	Y=0.37exp(0.0061x)	Y=2.2exp(0.0071x)	Y=2.12exp(0.0071x)

Table S4 Damage costs of NO_3-N, N_2O and NH_3-N in Ying *et al.* (2017) and references therein and NH_3-N with adjustments by author.

	Health $/kg N	Eutrophication $/kg N	Acidification $/kg N	Global warming $/kg N	Total $/kg N
NO_3-N	0.2	1.12	_	_	1.32
N_2O-N	0.3	_	_	11.2	11.5
NH_3-N	3.3	0.24	1.87	_	5.41
NH_3-N (adjusted by author)	2.54	0.24	1.87	_	4.65

Table S5: Available machinery options for starter fertilizer and dressing on the Chinese market by crop, together with their prices and time efficiencies in comparison to hand application (based on information available online).

Purpose	Machine type	Price (¥)	Time efficiency (h/ha)	Hand application time efficiency (h/ha)	Notes
Applying starter fertilizer for maize	Seed and fertilizer application machine	3-4k plus a tractor (10k)	5-7.5		

	Deep tillage and fertilizer application machine				
Applying dressing for maize	High ground-clearance dressing machine			5	
	Hand-push liquid/water-soluble fertilizer application machine				
Applying starter fertilizer for wheat	Seed and fertilizer application machine	3k plus a tractor (10k)	2.5-5	7.5-15	Save 37.5kg/ha seeds
	Rotary tillage and fertilizer deep placement machine		2-5		
Applying dressing for wheat	Wheat seeding machine		3.75	3.75-15 depending on geography	
	Hand-push fertilizer machine				

				and farmers' experience	
Applying starter fertilizer for rice	No-tillage machine or rotary tillage machine (Northeast China)	30k plus a tractor (10k)	1.5	15	
	Water-soluble fertilizer application machine (South China)		3.75		
Applying dressing for rice	Water-soluble fertilizer application machine (South China)		3.75	15	

Table S6: labor cost in China in 2015 by province, unit ¥/day; source: *Information Summary on the Production Costs and Revenues of National Agricultural Products*. Labor cost is the amount of payment a person with mid-level working capacity should receive for working for one day (8 h); it also represents the opportunity cost of time. Labor cost per day in one year = rural household family earnings per person in last year*the village's population/the village's population with employment/working days in a year (=250 days)

Province	Labor cost (¥/day)	Province	Labor cost (¥/day)
Beijing	95	Hubei	75
Tianjin	84	Sichuan	73
Hebei	68	Fujian	98
Shanxi	48	Jiangxi	74
Neimenggu	70	Hunan	71
Heilongjiang	79	Guangdong	85
Jiangsu	56	Guangxi	54
Liaoning	65	Hainan	77
Jilin	81	Chongqing	65
Shanghai	110	Guizhou	43
Zhejiang	75	Yunan	54
Anhui	61	Shaanxi	46
Shandong	68	Gansu	71
Henan	63	Ningxia	53
Qinghai	60	Xinjiang	72

Table S7: Prices (¥ kg^{-1} N) of regular fertilizers, controlled-released fertilizers, and fertilizers with nitrification inhibitor (NI) and urease inhibitor (UI), from (Xia *et al.*, 2017).

Fertilizer type		Category[a]	Price or cost
Regular fertilizer (Xia *et al.*, 2016)		Urea	4 ¥ kg^{-1} N
		P fertilizer	3 ¥ kg^{-1} P_2O_5
		K fertilizer	4 ¥ kg^{-1} K_2O
Controlled-release fertilizer		CRF (sulfur-coated)	5.6 ¥ kg-1 N
		CRF (polymer-coated)	7.2 ¥ kg-1 N
		CRF (polymer and sulfur-coated)	6.4 ¥ kg-1 N
Slow-released fertilizer	Nitrification inhibitor	DCD	15 ¥ kg^{-1}
		DMPP($C_5H_8N_2 \cdot H_3O_4P$)	150 ¥ kg^{-1}
		CP	100¥ kg^{-1}
		HQ	70 ¥ kg^{-1}
	Urease inhibitor	NBPT	100 ¥ kg^{-1}
		NAM	25 ¥ kg^{-1}
		THA	15 ¥ kg^{-1}
		ASQ	15 ¥ kg^{-1}

a. The price of urea, P and K fertilizer, staple grains referred to (Xia *et al.*, 2016). The price of controlled-release fertilizers referred to Chinese Fertilizer Website (www.fert.cn). The price of various nitrification and urease inhibitor products referred to Chinese GuideChem Website (china.guidechem.com).

Table S8 the top four provinces of cattle, lamb and pig production in China in 2016((EOCSSB), 2016).

	Cattle (million head at the end of the year)	Lamb (million head at the end of the year)	Pig (million head)
No.1	Sichuan (9.8)	Neimenggu (5777)	Sichuan (72)
No.2	Henan (9.3)	Xinjiang (3995)	Henan (61)
No.3	Yunan (7.6)	Shandong (2235)	Hunan (60)
No.4	Neimenggu (6.7)	Gansu (1939)	Shandong (48)

Table S9 Current N fertilizer application amount, recommended N use levels (Chen Xinping, 2016), (Wu, 2014a) and (Wu, 2014b) and the environmental benefits of N use reduction for major crop (wheat, maize, rice) production regions.

Region	Current N (kg N/ha)	Recom-mended N (kg N/ha)	NH_3 reduction (kg N/ha)	$ of NH3 reduction ($/ha)	NO_3 reduction (kg N/ha)	$ of NO_3 reduction ($/ha)	GHGs reduction (kg CO_2-eq/ha)	$ of GHGs reduction ($/ha)
Wheat								
I	134	106	1.90	8.85	1.92	2.53	378.0	7.7
II-1	206	162	2.99	13.91	5.31	7.01	594.0	12.1
II-2	198	170	1.90	8.85	3.37	4.45	378.0	7.7
III-1	240	184	3.81	17.70	8.68	11.46	756.0	15.4
III-2	200	180	1.38	6.42	2.53	3.34	270.0	5.5
IV	200	165	2.42	11.23	4.16	5.49	472.5	9.6
V	144	138	0.78	3.63	0.49	0.65	81.0	1.7
Maize								
I-1	156	150	0.35	1.6	1.1	1.4	81.0	1.7
I-2	200	180	1.22	5.7	5.3	7.0	283.5	5.8
I-3	226	164	3.60	16.7	21.7	28.7	837.0	17.1
I-4	205	188	0.99	4.6	4.5	6.0	229.5	4.7
II-1	206	178	1.62	7.6	7.2	9.4	378.0	7.7
II-2	217	189	2.77	12.9	7.9	10.5	378.0	7.7
III-1	234	181	3.07	14.3	15.8	20.8	715.5	14.6
III-2	246	176	4.06	18.9	21.7	28.7	945.0	19.3
III-3	234	219	0.87	4.0	5.3	7.0	202.5	4.1

IV-1	257	184	5.18	24.1	24.8	32.7	985.5	20.1
IV-2	232	183	3.48	16.2	14.6	19.2	661.5	13.5
IV-3	272	186	6.11	28.4	31.9	42.1	1161.0	23.7
				Rice				
I-1	123	116	3.26	0.40	0.25	0.66	94.5	2.0
I-2	213	156	26.51	0.83	3.20	8.61	769.5	15.7
II-1	196	159	34.41	0.76	1.97	5.30	499.5	10.2
II-2 Early	190	163	25.11	0.74	1.43	3.83	364.5	7.4
II-2 Middle	188	166	20.46	0.74	1.16	3.13	297.0	6.1
II-2 Late	178	172	5.58	0.71	0.31	0.84	81.0	1.7
II-3	268	256	11.16	1.50	1.16	3.12	162.0	3.3
III-1 Early	202	160	33.20	0.79	2.29	6.17	567.0	11.6
III-1 Middle	199	166	30.83	0.78	2.11	5.67	526.5	10.7
III-1 Late	188	161	21.34	0.73	1.41	3.78	364.5	7.4
III-2 Early	214	165	38.73	0.87	2.85	7.66	661.5	13.5
III-2 Late	214	168	36.36	0.88	2.70	7.26	621.0	12.7
IV	224	166	45.85	0.92	3.51	9.44	783.0	16.0

Table S10 Current N fertilizer application amount, recommended N use levels (Chen Xinping, 2016), (Wu, 2014a) and (Wu, 2014b) and the economic benefits of N use reduction for major crop (wheat, maize, rice) production regions.

Region	Current N (kg N/ha)	Recom-mended N (kg N/ha)	Fertilizer purchase savings (rmb/ha)	Crop yield change (t/ha)	Crop sales change (rmb/ha)
				Wheat	
I	134	106	112	1.07±0.58	2493±1351
II-1	206	162	176	1.37±0.7	3192±1631
II-2	198	170	112	2.26±0.93	5265±2167
III-1	240	184	224	1.76±0.83	8410±1934
III-2	200	180	80	2.79±1.12	6500±2610
IV	200	165	140	2.82±0.93	6571±2167
V	144	138	24	1.86±0.79	4334±1841
				Maize	
I-1	156	150	24	2.64±1.08	4963±2030
I-2	200	180	84	2.29±0.93	4305±1748
I-3	226	164	248	2.91±0.92	5471±1730
I-4	205	188	68	1.95±1.27	3666±2388
II-1	206	178	112	1.75±0.62	3290±1166
II-2	217	189	112	2.06±0.94	3873±1767
III-1	234	181	212	1.94±1	3647±1880

III-2	246	176	280	2.56±1.23	4812±2312
III-3	234	219	60	2.9±1.3	5452±2444
IV-1	257	184	292	1.92±0.92	3610±1730
IV-2	232	183	196	2.3±1.1	4324±1993
IV-3	272	186	344	2.3±0.93	4343±1748
				Rice	
I-1	123	116	28	3.17±1.21	8749±3340
I-2	213	156	228	2.51±1.15	6928±3174
II-1	196	159	148	2.2±0.94	6072±2594
II-2 Early	190	163	108	2.37±0.9	6541±2484
II-2 Middle	188	166	88	2.26±0.83	6238±2291
II-2 Late	178	172	24	1.98±0.95	5465±2622
II-3	268	256	48	3.18±0.91	8777±2512
III-1 Early	202	160	168	1.99±0.73	5492±2015
III-1 Middle	199	166	156	1.73±0.71	4775±1960
III-1 Late	188	161	108	2.05±0.73	5658±2015
III-2 Early	214	165	196	1.79±0.89	4940±2456
III-2	214	168	184	1.82±0.65	5023±1794

Late					
IV	224	166	232	2.09±0.81	5768±2236

Table S11 Environmental and economic benefits of reducing N use amount for different agri-ecological regions for major crops.

Region	Environmental benefits ($/ha)	Economic benefits ($/ha)
Wheat		
I	19.08	364.7
II-1	33.02	471.5
II-2	21	752.7
III-1	44.56	1208.7
III-2	15.26	921.2
IV	26.32	939.5
V	5.98	610.1
	Maize	
I-1	4.7	698.1
I-2	18.5	614.4
I-3	62.5	800.6
I-4	15.3	522.7
II-1	24.7	476.2
II-2	31.1	557.9
III-1	49.7	540.2
III-2	66.9	712.8
III-3	15.1	771.6
IV-1	76.9	546.2

IV-2	48.9	632.8
IV-3	94.2	656.1
	Rice	
I-1	3.06	1228.7
I-2	25.14	1001.8
II-1	16.26	870.8
II-2 Early	11.97	930.8
II-2 Middle	9.97	885.6
II-2 Late	3.25	768.4
II-3	7.92	1235.5
III-1 Early	18.56	792.4
III-1 Middle	17.15	690.3
III-1 Late	11.91	807.2
III-2 Early	22.03	719.0
III-2 Late	20.84	728.9
IV	26.36	840

Table S12: Various fertilizer application services existing in rural China and their prices.

Service	Price
Fertilizer top dressing	¥ 400/time
Fertilizer deep placement	¥ 700/time
Soil N test	¥500/ha
Machine rental	Chongqing: ¥150/d Shandong: ¥30/d Hebei: free, ¥180/d
Fertilizer + seed application machine rental or Drivers help broadcast fertilizers + seed with machine	¥450/ha
Machine application of fertilizer (fertilizer purchase cost excluded)	¥150/100kgN
Hand broadcasting fertilizer	¥450/ha or ¥100-150/d (different crops vary)

Table S13: New-efficient fertilizer products on the Chinese market, i.e. controlled-release fertilizers and inhibitor fertilizers, their formulation, producers, controlled-release N or inhibitor content and prices (based on information online).

Product Name	Formulation (N-P-K)	Description	Producer	Controlled-release N content	Ex-work price (no Vat ¥/t)[24]	Retail price (¥/t)[25]

[24] This is primary dealer's price for domestic product and country dealer's price for imported product

[25] Usually is 2-3 times of ex-work price for imported product and the same as ex-work price for domestic product

Wofute For crops	28-6-6	Polymer-coated + Sulphur-coated urea, 90 days release period, bulk blending (BB) Fertilizer	KINGSTA	8%	2200	3900
蓝膜缓控释 For crops	27-5-8	Polymer-coated + Sulphur-coated urea, 90 days release period, BB Fertilizer	Nongda	8%	2200	4300
Kongdejiu 控得久 For crops	26-10-12	Polymer-coated urea, 60days release period (50%) + regular urea (50%), BB Fertilizer	Maoshi	9%	2700	4500
Agroleaf 爱果利丰 For flowers	16-10-16	POLY-S Emax Polymer-Coated BB fertilizer	ICL	10%	6200	9000
Agromaster 易迈施 For flowers	16-5-22	Emax Polymer-Coated NPK fertilizer	ICL	16%	6300	11000

OSMOCOTE 奥绿肥 For flowers	11-11-18	OSMOCOTE Polymer-Coated NPK fertilizer	ICL	11%	12000	240000
Basaka 巴萨克 For flowers	16-8-12	Polymer-Coated NPK compound NPK fertilizer	Compu	16%	12000	240000
nitrification inhibitor + urease inhibitor	N/A	Imported: US Bach, Nitrapin/entrench from Dow, DMPP from Germany BASF, Agrho N from Norway Solvi Domestic: Zhejiang aofutuo		100%	Imported: 300000-350000 Domestic: 130000-160000	N/A
Compu 康普	15-15-15	DMPP Nitrification inhibitor NPK fertilizer	Compu	NI 0.1%-0.08%	4700	6600
Nitrophoska 狮马	15-15-15	DMPP Nitrification inhibitor NPK fertilizer	Eurochem	NI 0.1%-0.08%	4200	5800-6600
Agrotain	20-8-11	NBPT Urease inhibitor NPK fertilizer	Koch Agronomic Services Agrium	UI 0.1%-0.08%	4200	6600

Table S14: Various agricultural services and their pricings in Shandong province, 2014, with wheat and maize as an example.

Service item	Content	Market price (¥/mu)	Cooperative price with discounts (¥/mu)	Farmers' savings (¥/mu)
Tillage	In spring and fall	70	55	15
Seed supply	Wheat 15kg/mu	80	65	15
	Maize 1.75kg/mu	50	35	15
Fertilizer supply	Wheat 50kg/mu	145	145	Formula fertilizers require less N input, save 20
	Maize 50kg/mu	145	145	
Apply fertilizer together with seed	Wheat	75	65	10
	Maize	35	30	5
Pest control	5 applications	105	90	15
Irrigation	3 applications	105	75	30
Harvest	Wheat	65	55	10
	Maize	65	55	10
Straw return	Maize	50	45	5
Drying	Dry the maize for farmers	55	55	Increase quality save 45
Total		1045	915	195
Buying grain	Price is 4 cents/kg higher than market price.			

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Appendix for Chapter 3

Improving Air Quality and Nitrogen Use Efficiency in China via Cost-effective Agricultural Nitrogen Management

1. Agricultural NH_3 mitigation technologies and scenarios

1.1 Reduced N Application

This scenario describes appropriate and realistic N fertilizer use amount reduction for China's crops, vegetables and fruits. FAOSTAT shows that in China's N application rates in 2013 was 223 kgN/ha, higher than 79 kgN/ha in the U.S. and 195 kgN/ha in Netherlands. However, China's yields of cereal (5.9 t/ha), vegetables (23.9 t/ha), and fruits (excluding melons) (11.1 t/ha) are all significantly lower than U.S.'s and Netherland's yields of cereal (7.3 t/ha and 8.6t/ha), vegetable (32.6 t/ha and 57.1 t/ha), fruits (23.8 t/ha and 33.7 t/ha). In response to fertilizer N input, crops' yield will first increase and then flattens out after reaching a plateau (Zhang et al., 2015).

In case of enough excess N supply, yield even decreases (Wu, 2014). In-the-field experience shows that:

1) Over-application of N fertilizer will cause over-growing, ineffective/unproductive tiller and low earing rate for maize and rice, thus adversely affect grain absorption of nutrient in later growing stages. 2) Over application of N also causes over-growing of stems near root areas, making crops more susceptible to lodge thus adversely affects yield. 3) Over application of N results in rapid growth of crop and high cropland crown density, increasing the incidence of pest diseases and crop senescence (Ghorbani et al., 2010). 4) For maize, over application of N can

inhibit root elongation thus hurt yield. Most farmers improperly apply all N fertilizers in one application, resulting in high nitrate levels near crop root areas adversely impacting growth of lateral roots. More detailed description of mechanisms can be found in Mi et al (2010) and references therein (Mi et al., 2010).

Controlled experiments find lowered yield at higher N application rates: 1) For rice, increase of N application rates from optimized levels to 150% of optimized levels cause a decrease of yield in all production regions (reference the columns called 'yield at MN' and 'yield at 150% MN' in Table 2 of Wu et al 2015 (Wu et al., 2015a)). 2) For maize, increase of N application rates from RNR to 150%RNR decreases yields (Table 1 in Gao et al (2012)(Gao et al., 2012))

Instead, optimized N fertilizer application (reducing N application rates accompanied with split application) has resulted in increased yield. For maize, recommended N application rates (Table 2 a column called N rate under regional N management approach N) are lower than farmers' average practices (Table 1 a column called 'N rate') in most regions yet grain yields under recommended practices are higher than conventional practice (Wu et al., 2014). In detail, the recommended practices include reduced N rate and split application, i.e. 'approximately one-third of the granular urea was applied by broadcasting at sowing, while the remainder was applied as a side-dressing at the six-leaf stage'.

In reality, farmers' N application rates vary substantially across individuals (e.g. 238$\pm$107 kgN/ha across 523 maize farmers in North China Plain Region II, compared to recommended rate of 177kg N/ha in (Wu et al., 2014)). It is important that both farmers who apply excess N and not sufficient N can adjust their application rates, which will lead to higher total yields.

Lastly, N fertilizer input is not linearly correlated with grain N intake for many reasons: Research finds that N absorption by above ground crop during maturity stage (N_m hereafter) can increase even if N application rates decrease, e.g. for rice in Northeastern China, when N application amount is optimized to 85-145kg N/ha, compared to the current farmers' practice of 77-185 kgN/ha, yield increases to 7.9-9.6 t/ha compared to current yield of 6-9.16t/ha. N absorption during maturity stage by above ground fraction increases to 95-115 kgN/ha, compared to current value of 80.2-112.5kgN/ha (Peng et al., 2006, 2007).

Crop need for nutrient is different across agroecological regions. Rice grown south of the Yantze River Basin absorbed more N during maturity stage, i.e. 135-196 kgN/ha yet the yield is lower, i.e. 7.2-7.6t/ha. This is because rice in northeastern China grows less leaves, have lower plant height, and experiences lower accumulated temperature than others (e.g. rice south of Yantze River), thus absorb less N to grow its above ground fraction during maturity stage (9).

Under similar N application levels, maize grown in Northeastern China can have higher yield, because of high organic matter content in soil, summer precipitation in synchrony with maize growing season, deep seeding with machinery. One research finds that due to these preferable conditions, even without N fertilizer application, maize, single cropped in Jilin province, has yield as high as 7.6t/ha. Optimized N application amount will increase yield to 9.8t/ha (Gao et al., 2012). In comparison, the North China Plains have wheat-maize rotation and has much higher NH_3 emission volatilization rate at similar levels of N due to alkaline soils. In southwestern hilly maize fields, abundant precipitation concentrates in summer, resulting in a lot of N lost through leaching and runoff.

Yield N content does not increase linearly with yield. As maize yield increases, the amount of N needed per unit of maize decreases. Considering 6 ranges of grain yield (<7.5 Mg/ha, 7.5–9 Mg/ha, 9–10.5 Mg/ha, 10.5–12 Mg/ha, 12–13.5 Mg/ha and >13.5 Mg/ha), N uptake requirements per Mg grain yield were 19.8, 18.1, 17.4, 17.1, 17.0 and 16.9 kg respectively (Hou et al., 2012).

Reduced N Application scenario reduces N application for China's grain crops (wheat, maize and rice) to levels that optimize farmers' profit (Table S2) and for China's vegetables and fruits based on field experiment recommendations. We utilize data on current N application rates and crop yields from over 5000 large-scale household farm surveys conducted under National 948 Project (2003-Z53) and Agricultural Research Project (200803030) (Wu, 2014; Wu et al., 2014, 2015a; Chen Xinping, 2016) and data on recommended nitrogen application levels and yields derived statistical analysis of ~5500 controlled field experiments during 2005-2010 (Wu, 2014; Wu et al., 2014, 2015a; Chen Xinping, 2016). We reduce N use levels for vegetables and fruits are reduced following recommendations derived from field experiments (Dong et al.,; Hong et al., 2006; Wang et al., 2013; Feng, 2014; Shen, 2014; He et al., 2016; Zhao et al., 2017a).

1.2 Machine Application

This scenario describes increased machinery use during fertilizer application. China's machinery coverage rate is still relatively low. More than half of rice production regions, most maize production and some wheat production are dominated by hand application[26]. The Chinese government set a policy target that by 2020 machinery use for fertilizer application will increase to 60%. Machine Application can place N fertilizers deeply near crop root areas and thus substantially reduce NH_3 losses to the air, i.e., by 35% for wheat and rice systems and by 70% for maize (Xia et al., 2016). Persuading farmers to use less N is difficult as risk-averse farmers will fear yield loss and changing behavior of millions of small farmers incur high transaction costs. In comparison, promoting machinery use appears more attractive to farmers because it is equipment upgrading and one machine rental center can serve all farmers in one village. Literature review shows that starter fertilizer and dressing machinery options exist in the Chinese market for major crops and vegetables, i.e., seed and starter fertilizer application machine, hand-push liquid fertilizer application machine, high ground-clearance dressing machine, etc (Chen et al., 2012).

In *Machine Application* scenario, we switch all hand broadcasting to machine application, except for rice production in hilly areas where machines are difficult to operate and fruit production where current N is already applied through deep placement.

1.3 Efficient Fertilizer

This scenario describes the use of controlled-release fertilizer and urease inhibitor fertilizer, both substantially reducing NH_3 emission rates by more than 50%. Controlled-release fertilizers are regular fertilizers that are coated, allowing nutrients to release gradually with time (Trenkel, 2010). Urease inhibitor fertilizers reduce hydrolysis rate of urea, prevent rapid accumulation of NH_4^+, and thus reduce NH_3 emission (Trenkel, 2010). N use amount reduction and machine application both require change of farmer behavior, yet efficient fertilizers can easily be promoted by governmental subsidies and fertilizer seller marketing. In this scenario, reduced NH_3 emission rates are by crop type and by N application rates and presented in Table S19 based on reference (Xia et al., 2017).

1.4 Manure Management

This scenario depicts improved manure storage, manure spreading and animal feed. Each step of manure handling, i.e., feed N intake, animal housing, manure collection, composting, storage, treatment and spreading, is susceptible to N loss as NH_3. Animal housing and final disposal are two most important processes for N loss. For example, 39% of manure N is volatilized as NH_3 during housing and storage, amounting to 6.7 Tg NH_3-N emissions in 2010(Bai et al., 2016). Due to great variations in management across animal types and farms, and absence of systematic recommendation for manure handling. In manure management scenario, through acidifying manure to store and utilizing aerobic composting reactors, NH_3 emission rates during storage are reduced by 60% for all indoor animals. Other analyses have found that through injecting manure into land with specially-designed machines, NH_3 emission rate during spreading is reduced by ~80% (Cao et al., 2018).To conservatively estimate this potential, we assume that only 25% of farms adopt this spreading of manure. Through using low-protein animal feed, N content of manure for chicken, cattle and pig decreases by 10%.

1.5 Combined

This scenario combines management improvements in the four scenarios described above. This means N fertilizer is applied at reduced rates, in the forms of new-efficient fertilizers and with machine. Animal housing, manure spreading and animal feed is also improved. This scenario provides an NH_3 emission reduction ceiling, demonstrating potentials of the several most realistic and easy-to-implement agricultural management improvements. NH_3 emission reduction through technologies combined is modeled by NH_3 emission model(Zhang et al., 2018). Because there is little experimentation with these combined results, we estimate that the lower rend of benefits would be the same as that achieved in *Reduced N Application* scenario and the higher end of benefits would be accumulative benefits from individual technologies.

The four individual NH_3-mitigating N management opportunities we proposed are easier-to-implement than other production-side technologies and demand-side dietary transitions. For example, we didn't include in our scenario we didn't include in our scenario such as applying water-soluble or liquid fertilizer together with irrigation water, and precision agriculture which provide guidance on how much N to apply during which crop growth period through detailed crop modeling (Chen et al., 2011), etc. In reality, many farmers in China spend considerable amount of time in off-farm jobs yet limited time farming. Reducing meat consumption, thus lower animal production, can reduce NH_3 emissions. It can be very difficult for Chinese consumers for now. Because it is only within the past decade that meat consumption has increased due to increased household income. It is more difficult to change people's mind than to introduce people with new technologies, e.g. machinery, new-type of fertilizers, etc.

Even among the four individual management opportunities, some is easier than others. For example, improved manure management is easier to implement than all fertilizer application management, mainly because animal production in China is more industrialized than crop production. There're a few big monopolies of animal production companies who raise animals themselves but also contract with small animal farmers to collect their produce. It would incur much smaller transaction costs for the government to impose regulations on these larger producers and they can work with their contract farmers, compared to the government works with millions of small crop farmers.

2. NH_3 emission factors in NH_3 emission model by Zhang et al (2018)

Emission factors of NH_3 from fertilizer application is estimated as a function of soil pH, cation exchange capacity, fertilizer type, and application mode adapted from Bouwman et al. (2002) (Bouwman et al., 2002) (cf. Figure 4 of Zhang et al (2018)). The estimated NH_3 emission factors are in good agreement with an ensemble of measured emission factors in China from the literature with values ranging from 2.2% to 50.9% (cf. Figure 6 of Zhang et al. (2018)). The function of Bouwman et al. (2002), although is developed mainly based on NH_3 emission factor measurements in Europe and the US, generally captures the emission factor measurements in China, except for those for ammonium bicarbonate, which requires 70% increase to match the Chinese measurements.

Baseline NH_3 emissions in China in the year 2012 were 14 Tg NH_3, with 0.66 Tg emitted in January and 1.66 Tg emitted in July (Table 1). High NH_3 emissions in January occur in the North China Plain (612 kg/km^2/month [Figure S1]) due to concentrated agricultural activities and alkaline soils susceptible to NH_3 volatilization. High NH_3 emission regions in July include central China, northeastern China and the North China Plain (rates as high as 1530 kg/km^2/month) due to high summer temperatures and heavy fertilizer application.

3. Air quality simulation

We use the Weather Research and Forecasting – Chemistry (WRF-Chem) model v3.6.1 to simulate $PM_{2.5}$ formation in the base case and in agricultural N management scenarios. WRF-Chem is an online-coupled meteorology-chemistry model widely used for air quality research (Gao et al., 2016; Qin et al., 2017). The chemical and physical schemes used are Carbon-Bond Mechanism Version Z (CBMZ) for gas-phase chemistry, 4-bin Model for Simulating Aerosol Interactions and Chemistry (MOSAIC) for aerosol chemistry, the Rapid Radiative Transfer Model for General Circulation Models (RRTMG) scheme for shortwave and longwave radiation, the Morrison scheme for cloud microphysics (Morrison et al., 2005), the Yonsei University scheme for boundary layer parameterization (Hong et al., 2006), and the Noah land surface model for surface processes (Chen and Dudhia, 2001). Meteorological boundary conditions come every 6 hours from the 2012 National Centers for Environmental Prediction (NCEP) Final Analyses data. Chemical initial and boundary conditions are from a 2014 simulation of the global chemical transport model, Model for Ozone and Related Tracers Version 4 (MOZART-4).

Anthropogenic emissions of air pollutants (except for NH_3) are from the Multi-resolution emission inventory for China (MEIC) (http://www.meicmodel.org)(Li et al., 2017) for the year 2012 and from HTAP (Hemispheric Transport of Air Pollutants) v2.2 outside China (Janssens-Maenhout et al., 2015) for the year 2010. Biogenic emissions are calculated online using the Model of Emissions of Gases and Aerosols from Nature (MEGAN) scheme (Guenther et al., 2006) and open biomass burning emissions are from Global Fire Emission Database version 4 (GFED v4, www.globalfiredata.org) for the year 2012.

We conduct 6 sets of simulations: one *Baseline* and five agricultural management scenarios. The only difference between the *Baseline* and management scenario simulations are modified NH_3 emissions due to various agricultural N management options. Each simulation set includes one month of simulation for January and one month of simulation for July (in all cases after six days of spin-up) for the year 2012. The model resolution is 27 km by 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 levels extending from the surface to 50hPa with an 18m deep surface layer. We turn off aerosol-radiation interactions (ARIs) to minimize the impact of aerosol concentration changes due to perturbations in boundary-layer meteorology. We keep aerosol-cloud interactions (ACIs) on. Our tests indicate that turning it off generates a large and uniform impact on $PM_{2.5}$ concentrations in eastern China and because developments of ACIs in the model are more recent and less certain than ARIs we keep the indirect feedback on.

4. Evaluation of 2012 baseline NH_3(g) concentration

We conduct evaluations of modeled NH_3 concentration against satellite observations, i.e. modeled NH_3(g) concentration at 918hPa against AIRS (Atmospheric Infrared Sounder) satellite observation, and modeled column density of NH_3 (g) against IASI (Infrared Atmospheric Sounding Interferometer) satellite observation. Our NH_3 emission inventory provides reasonably good comparison over China in January and July of 2012. Our inventory outperforms MIX NH_3 inventory (the NH_3 emission inventory published in Kang et al (2016)(Kang et al., 2016) and Huang et al (2017) (Huang et al., 2012) and used in Liu et al (2019) and MIX inventory (Li et al., 2017)).

We run both WRF-chem and GEOS-chem (another well-known atmospheric chemistry model frequently used by atmospheric scientists) with NH_3 emissions from Zhang et al (Zhang et al., 2018, p. 3) (used in this study) and MIX NH_3 inventory (the NH_3 emission inventory published in Kang et al (2016)(Kang et al., 2016) and Huang et al (2017) (Huang et al., 2012)and used in Liu et al (2019)). We extracted modeled NH_3 concentration at 918hPa averaged over regions that AIRS satellite provides reliable observations (AIRS mask region, Figure S3). The AIRS observations are NH_3(g) concentration before smoothing through two-dimensional penalized least squares provided by Warner J.K.. Warner's paper provide AIRS observations that are after smoothing (Warner et al., 2016).

AIRS observation shows a strong summer-winter contrast of NH_3 concentration, which is better captured by NH_3 emission inventory used in this study. Zhang et al (2018) NH_3 outperforms MIX NH_3 especially in July, where both WRF-chem and GEOS-chem simulation with MIX NH_3

is -25%~-75% lower than AIRS observation yet both WRF-chem and GEOS-chem simulation with Zhang et al (2018) NH_3 is within 25% of AIRS observation (Figure S4 and Table S3).

We evaluate NH_3 column density simulated by WRF-chem using both Zhang et al (2018) NH_3 (this study) and MIX NH_3 emissions (published in Kang et al (2016) (Kang et al., 2016) and Huang et al (2017) (Huang et al., 2012)) against IASI observations for January and July of 2012 over China. We processed IASI column NH_3 density according to v2.2 instruction (Damme et al., 2017)[27]. Only observations with relative error <100% are used and unweighted averages are used. IASI pixels are then mapped to closest WRF-chem grids (27km ×27km resolution, 9.0-57.6°N and 59.6-156.3°E domain) following two principals: 1) multiple pixels could be mapped to the same grid. 2) In this case, average values of IASI are used for comparison. WRF-chem grids and periods without valid IASI observations are removed.

We first compare daily values of mapped IASI data to WRF-chem simulation using two NH_3 emission inventories using orthogonal distance regression (Figure S5 and S6). We then compare monthly means of all valid daily mapped IASI and WRF-chem simulations (Figure S7 and S8). Overall Zhang et al (2018) NH_3 is much better than MIX NH_3. The correlation of IASI and WRF-chem simulation (regardless of which NH_3 emission inventory is used) is much worse in January than in July, probably due to large IASI uncertainty in winter.

5. Evaluation of 2012 baseline $PM_{2.5}$ simulation

[27] http://cds-espri.ipsl.fr/etherTypo/index.php?id=1730&L=1

We evaluate modeled $PM_{2.5}$, ammonium (NH_4^+,) nitrate (NO_3^-) and sulfate (SO_4^{2-}) concentrations during January and July of 2012 against as many published speciated PM observations during the same time period as possible, i.e., Beijing from (US Embassy at Beijing, 2012)(Sun et al., 2015), Shanghai (Wang et al., 2016), Xiamen, Quanzhou, Putian and Fuzhou (Wu et al., 2015b), Guangzhou (Lai et al., 2016), and Jinsha (Zhang et al., 2014). Observational sites mainly represent the North China Plain, Yantze-River Delta and coastal regions. Unfortunately, there is a lack of representation for inland China during wintertime. The WRF-Chem model well captures observations, especially in July (Figures S9 and S10). In July, modeled mean ammonium, nitrate, sulfate and $PM_{2.5}$ concentrations averaged across eight sites are 3.4, 3.9, 7.1 and 33.4 $\mu g/m^3$, decently close to observations of 3.1, 2.7, 6.9 and 36.2 $\mu g/m^3$ (Figure S10). In January, modeled mean ammonium, nitrate, sulfate and $PM_{2.5}$ concentrations averaged across six sites are 4.3, 8.1, 6.0 and 66.9 $\mu g/m^3$, compared to observations of 8.5, 10.1, 14.0 and 83.7 $\mu g/m^3$ (Figure S9). The model underestimation of sulfate in winter is prominent at four coastal sites in Fujian province, but barely noticeable in Beijing and Shanghai. In January at Beijing, modeled ammonium, nitrate, sulfate and $PM_{2.5}$ concentrations are 6.9, 9.4, 10.5 and 174.9 $\mu g/m^3$, compared to observations of 6.9, 8.3, 10.2 and 175.3 $\mu g/m^3$. In January at Shanghai, modeled ammonium, nitrate, sulfate and $PM_{2.5}$ concentrations are 7.5, 13.5, 9.6 and 93.4 $\mu g/m^3$, compared to observations of 7.5, 10.2, 11.6 and 53.1 $\mu g/m^3$. The fact that our model does not include the latest sulfate production mechanism – heterogeneous SO_2 oxidation at the surface of liquid aerosols, results in underestimation of sulfate. Model's underestimation at coastal sites originate from other factors such as models' flaws in simulating complex terrain geography and sea breeze as well as flaws in emission inventory in that region.

6. **Health impacts of exposure to $PM_{2.5}$**

We calculate premature mortality of four diseases due to exposure to $PM_{2.5}$ for adults (≥ 25 y old) in five agricultural nitrogen management scenarios. The four diseases considered are chronic obstructive pulmonary disease (COPD), lung cancer, ischemic heart disease (IHD) and ischemic stroke.

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

$$Mort_{i,P} = POP_P \times Mortbase_{i,P} \times (1 - \frac{1}{RR_{i,P}})$$

where $Mort_{i,P}$ is the number of premature mortality in province P from disease *i*; $POP_{j,P}$ is the number of exposed targeted population in province P considering adults (≥ 25 y old) in 2012 from 2013 China Statistical Yearbook and (All China Marketing Research Co. Ltd, 2014); $Mortbase_{i,P}$ is the baseline mortality rate in province P for disease *i* in 2012 from Global Burden of Disease study (Burnett et al., 2014); $RR_{i,P}$ is the relative risk factor for one disease *i* adopted from (Burnett et al., 2018). Relative risk factors for IHD and stroke are by age groups. There are 12 age groups considered, i.e. 25-29, 30-34, 35-39, 40-44, 45-49,50-54,55-59,60-64,65-69,70-74,75-79 and over 80 y old. Relative risk factors for lung cancer and COPD are the same for all people ≥ 25 y old.

7. Estimating co-benefits achieved in scenarios. We exclusively evaluate co-benefits generated through improved N management in grain crop (wheat, maize and rice) production. Benefits due to improved management in vegetable and fruit production are excluded, due to lack of data for reliable estimation.

We divide China's crop production into seven wheat production regions, thirteen maize regions and thirteen rice regions, following the definitions in (2).For each region, we obtain current N fertilizer use amount and yield for the base case from more than 5000 large-scale household farm surveys in 17 provinces conducted in 2008 and 2009 under the umbrella of national projects including 948 Project (2003-Z53) and Agricultural Research Project (200803030). Survey questionnaires were designed by China Agricultural University, researching farmers' fertilizer application timing, amount and fertilizer types. Surveys are conducted by 27 researchers in research institutions and local soil-fertilizer station agents. Two to three counties were picked from each province, three to five towns picked from each county, two to three villages picked in each town and eight to ten farmers picked from each village.

For each region, we obtain recommended reduced N use amount and corresponding yield for the *Reduced N Application* scenario using statistical analysis of nationwide field experiments (~2000 for wheat, ~2000 for maize, ~1500 for rice) conducted under 3141 Experiment Project by the Soil Testing and Fertilizer Prescription Program by Chinese Ministry of Agriculture during 2005-2010 (Wu, 2014).

These field experiments aim to test the effect of nitrogen fertilizer application levels on crop yields under a as much as possible controlled setting. Each experiment includes four sets of treatment: no N fertilizer application, an agent-recommended N application level, 50% of recommended N application level and 150% of recommended N application level. Agent-recommended N application levels were come up with by local agricultural technicians depending on local soil tests and experiences, and recommended levels varied across plots. Management aims to stay consistent in all treatments of experiments: 1) urea is the selected N fertilizer to apply. 2) 1/3 N is applied as base fertilizer and 2/3 used as dressing. 3) Application of P fertilizers are all the same and there is no organic fertilizer added. 4) Seed varieties are determined by agronomists and chosen among seeds used by local farmers. During the experiments, farmers are in charge of daily management such as pest control and agricultural technicians are in charge of fertilizer application and testing yields.

Gathering experiment data of yields under various N application levels for each crop agroecological region, a curve estimating the relationship of yield with respect to N input is produced using statistical analysis. A N application level that optimizes farmers' net returns (yield sales minus N fertilizer purchase costs) is generated for each crop agroecological zone

For each region in the *Baseline* and *Reduced N Application* scenario, we then estimate emissions of NH_3, N_2O, NO_3-leaching and NO_3-runoff based on N use amounts using statistical relationships built from field measurements (Cui et al., 2013, 2014)

For each region, we estimate Nr emissions and crop yields in the *Machine Application* and *Efficient Fertilizer* scenarios through scaling the *Baseline* Nr emissions and yields by management-specific factors which represent the effect of technology on yield and Nr emissions. These factors are obtained from a meta-analysis of field experiments (Xia et al., 2017). For each region, we estimate Nr emissions in the *Combined* scenario by scaling emissions in the *Reduced N Application* scenario with management technology impact factors (Xia et al., 2017).

We estimate life-cycle greenhouse gas emissions (CH_4, N_2O, and CO_2) during nitrogenous fertilizer manufacture, transport, application and postapplication using the reported value of 13.5 tons CO_2-eq GHGs (including N_2O, CH_4 and CO_2 with GWP at 100-yr time scale) emitted per ton of N fertilizer consumed in China (Zhang et al., 2013). Zhang et al (2013) used 'a life cycle assessment approach to estimate GHG emissions due to the main components of the N fertilizer chain in China, primarily using Chinese-specific parameters rather than IPCC tier 1 default values. GHGs during fossil fuel mining are estimated using two published literatures. GHGs during ammonia synthesis are estimated using surveys of 230 fertilizer companies conducted by Chinese Nitrogen Fertilizer Industry Association. GHGs during fertilizer manufacture are estimated using specific energy consumption rate of each fertilizer product from surveys and a literature. GHGs during transporting energy and N fertilizer products are estimated using average transportation distances by train and truck for coal, crude oil and N fertilizer from China's national statistics 2) estimate multiplied and GHG emission factors of these processes from IPCC.' Regarding N_2O emission from post-application, Zhang et al (2013) 'discriminated between upland fields and paddy fields, analyzed published field measurements in China (a total of 853) through meta-analysis method and estimated direct and indirect N_2O emission factors for these two field types.'

For crop system NUE calculations, we utilize N content in crop yield (Bouwman et al., 2005, pp. 1970–2030), N biological fixation and deposition (Zhao et al., 2017b), N manure production

(Zhang et al., 2018) and N in irrigation water (estimated from irrigation water use and observed N content at surface waters at observational sites) (Gu et al., 2015).

We then aggregate all crop regional estimates to obtain national total values by using crop planting areas listed in the 2009 China Statistical Yearbook.

The *Manure Management* scenario includes two technologies that reduce manure N loss to water. One technology is improved animal feed applied to cattle, pig and chicken husbandry. This technology reduces manure N content from these animals by 10%. We assume that N loss to water from these animals will be proportionally reduced by 10%. *Baseline* manure N loss to water from cattle, pig and chicken husbandry is estimated to be 10 Tg, using N loss to water from all animal manure in China in 2010 (i.e. 12 Tg) (Bai et al., 2018) and a ratio of 5/6 which is the relative contribution of cattle, pig and chicken manure to livestock total N loss to the environment (Bai et al., 2016). The other technology is composting reactors applied to pig farms. This technology reduces N loss to water during manure storage by 100%. We estimate that *Baseline* N loss to water during storage of pig manure is 1.5 Tg, using N loss to water from all animal manure in China in 2010 (i.e. 12 Tg), a ratio of ~50% which is the contribution of manure storage among all manure handling stages to total N loss to the environment (Bai et al., 2016), and a ratio of ¼ which is the contribution of pigs among all animal types to total N loss to water (Bai et al., 2016).

8. Estimating benefits in monetary terms

We translate increased yield, mitigated greenhouse gas and mitigated reactive nitrogen pollution (including NH_3, N_2O and NO_3-leaching and runoff) into economic monetary values.

For one crop in one agri-ecological zone, we conduct the following benefit calculation:

$$Benefit = P_{crop} \cdot \Delta Yield - P_{fert} \cdot \Delta Q_N - DCost_{GHGs} \cdot LCE_{GHGs} \cdot \Delta Q_N - DCost_{N2O} \cdot \Delta E_{N2O} - DCost_{NO3} \cdot \Delta E_{NO3} \quad (1)$$

where ΔQ_N is the amount of N fertilizer application reduction compared to the baseline, $\Delta Yield$ is the difference between crop yield in scenario compared to the baseline, P_{crop} is grain wholesale price for the year 2012 (Chinese NDRC (national development and reform council), 2016), i.e. 2.76/kg for rice, 2.33/kg for wheat and 1.88/kg for maize. P_{fert} is fertilizer prices (Xia et al., 2017), i.e. urea price of 4 ¥ kg^{-1} N. $DCost_{CO2}$ is the social cost of carbon – we use the price of CO_2-eq of 20.4 dollars per tonne for European market in 2008 (Schiermeier, 2009) as well as prices of 10, 30, 40 and 50 dollars per tonne to represent to highly uncertain nature of the social cost of carbon. LCE_{CO2} is life-cycle emission of GHGs during the production, transportation and application, i.e. 13.5 tons CO_2-eq GHGs per ton of N fertilizer consumed in China (Zhang et al., 2013). ΔE_{N2O} and ΔE_{NO3} are the change of Nr emissions in scenario compared to baseline. $DCost_{NO3-leaching}$ is damage costs (including global warming, health, eutrophication, and acidification) of reactive nitrogen species (Gu et al., 2012; Ying et al., 2017). $DCost_{N2O}$ is damage cost of N_2O on public health through depleting stratospheric ozone, as the warming effect already taken into consideration in the GHG damage cost calculation. A currency conversion rate of 1USD=0.14¥ is used when needed.

We use the value of statistical life (VSL) to calculate economic effect of lives saved from reduced exposure to $PM_{2.5}$. The VSL estimates vary among countries and income groups and introduces controversies of judging value of lives. We use the value of 0.86 million Chinese Yuan, equivalent to 0.124 million USD (Xu et al.,). We also provide a range of economic values using two more estimates of Chinese VSL providing lower (0.09 million USD)(Nielsen and Ho,) and higher (0.25 million USD) (Xie, 2011) range.

9. Estimating costs of improving agricultural nitrogen management

Reduced N Application scenario

There's no additional cost associated with reduction of N application. Farmers' phycological cost of changing practices are difficult and excluded in this analysis.

Machine Application scenario

We calculate machine rental prices and labor savings if 60% of China's cropland (currently not fertilized with machines) can be fertilized with machine application instead of hand application. We estimate that machine application, compared to hand application, increases time efficiency of N fertilizer application by 2.25-7.5h/ha. The range represents variations across crop types, plot sizes and different levels of farmers' experience with hand application. Using provincial labor cost, we calculate labor savings for each agroecological region. We estimate that machine rental cost ranges 180 -560 ¥/ha. We estimate that 60% of cropland has the potential to be fertilized with machine, excluding around 30% which has already been dominated by machine and 10% hilly area not suitable for machine to operate. We exclude calculating upfront cost of purchasing new machines assuming current machine rental centers at township and county levels as well as machine leasing among villagers are sufficient.

Efficient Fertilizer scenario

We calculate fertilizer purchase expenditure change when farmers shift from purchasing conventional fertilizers (e.g. urea) to controlled-release and urease inhibitor fertilizers(Xia et al., 2017). We use coated CRF with prices varying ¥5.6-7.2/kg N and NBPT urease inhibitor applied at 1200mg/kg urea ("Abstract: Interaction of NBPT with Two Nitrification Inhibitors on Multiple Soils. (ASA, CSSA and SSSA International Annual Meetings (2016)),") at a price of 100 ¥ kg^{-1}.

Manure Management scenario

A composting reactor which is suitable for a pig farm of 10000 heads incurs an upfront cost of ¥1500k. Varying the lifetime between 20-50yr and discount rate at 2%, 4% and 15%, we estimate the annual equipment cost varying between ¥47.7 k to 240 k/annum. Annual operational costs include electricity bill and labor cost. Electricity consumption costs ¥15k/annum. Labor which helps add manure and collect organic fertilizer costs ¥30 k/annum. Annual organic fertilizer production is 400-600t, assumed sold at a medium price of ¥300/tonne with a possible lower price of ¥200/tonne and higher price of ¥1000/tonne.

We scale estimate above to all pig farms, given China raised 476 million heads of pigs in 2012 (National Bureau of Statistics of the People's Republic of China, 2012). Since pig manure accounts for only 1/6 of all animal manures (Wang et al., 2018) and grazing animal manure v.s.

indoor animal manure is around 1:5(Wang et al., 2018), we estimate total cost through scaling up the costs of composters applied to pig to all indoor animals.

The upfront cost of a manure deep-placement machine is ¥10k. For a pig farm of 5000 heads, we assume manure urine N replaces 25% of nitrogenous fertilizers. These deep-placed N help serve N fertilizer needed by 11000ha of wheat/maize rotation systems, worthy of ¥8.71 million N fertilizer purchase expenditures. Cost information is provided by Lin Ma who commercialized the reactor and published in (Cao et al., 2018).So, manure injection machines in the long term generate net economic benefits which outweigh the upfront costs.

Low-protein animal feed should sell at a lower price than conventional feed, ideally generating animal feed purchase savings.

Combined scenario

We calculate fertilizer purchase expenditure in this scenario using reduced nitrogen application rates and prices of new-efficient fertilizers. We add to that costs for machine use and manure management.

10. Uncertainties and constraints

10.1 Sensitivity of air quality and public health benefits to uncertainties in NH_3 emissions in China.

Among more recent NH_3 emission inventories constructed for China, the lowest estimate is 10Tg/a by Huang et al (2017) and the highest estimate 17Tg/a by Gu et al (2015) (Gu et al., 2015), compared to 14Tg/a by Zhang et al (2018) which is used in this study. Perturbing (increase and decrease) our current NH_3 emission by 20% roughly exhausts the lower and higher bounds of existing NH_3 emission inventory estimations.

We first constructed two new 'perturbed' baselines – one with NH_3 emissions that are 20% higher than standard baseline (referred as 1.2NH_3base hereafter) and one with NH_3 emissions 20% lower than standard baseline (referred as 0.8NH_3base hereafter). On top of each new baseline, we constructed NH_3 emissions when all technologies are combined, by assuming NH_3 emission reduction achieved in percentage for each grid is the same as that achieved in *Combined* scenario compared to standard baseline. We conducted WRF-chem simulations for January and July of 2012.

Figure S15 shows the SIA reduction achieved in current All Combined scenario (first column), that achieved in All_combined_1.2NH_3base scenario if baseline NH_3 should be 1.2 times of Zhang et al (2018)'s estimate and that achieved in All_combined_0.8NH_3base scenario if baseline NH_3 should be 0.8 times of Zhang et al (2018)'s estimate. Figure S15 shows that SIA reduction (in ug/m^3) through implementing all N management technologies combined are similar even if baseline NH_3 emission estimate is 20% higher or lower than that estimated by Zhang et al (2018).

These results, surprising and interesting, make sense. To note, even though the same percentage NH_3 reduction for each grid are imposed on standard NH_3 emissions, 1.2NH_3base and 0.8NH_3base, the absolute NH_3 emission reduction achieved in All_combined_0.8NH_3base (All_combined_1.2NH_3base) is 20% lower (higher) than that in All_combined. However, since the SIA reduction achieved are similar across All_combined_0.8NH_3base, All_combined, and All_combined_1.2NH_3base, the SIA reduction per unit NH_3 emission reduced probably increased by 40 – 50% when we changed from All_combined_0.8NH_3base to All_combined_1.2NH_3base. This indicates that the simulation shifted from a NH_3-rich regime to a NH_3-limited regime when NH_3 emissions changed from 120% of the baseline to 80% of the baseline.

We also calculated the avoided mortalities due to reduced $PM_{2.5}$ exposure if all nitrogen management technologies are combinedly implemented. Avoided mortalities are 30437 persons with current baseline, 27424 persons if NH_3 emissions are actually 20% higher than the current, and 30659 persons if NH_3 emissions are actually 20% lower than the current. Across the full range of uncertainties for NH_3 emission inventory, the range of public health benefits are around -10%~+0.7% of current results.

10.2 Air Quality Modeling Uncertainties

Uncertainties exist in the WRF-Chem simulations for estimating $PM_{2.5}$ concentration changes. Here we turn off aerosol-radiation interactions (ARIs) and keep the aerosol-cloud interactions (ACIs) on. Our model's dust emission is calculated online and is sensitive to surface wind speed which would be affected by background aerosol concentrations if we turn on ARIs in the simulations. Although the best case is to fully turn off both ARIs and ACIs, given the scientific community's knowledge about ACIs is a lot less than ARIs, and our test of turning off ACIs gives a significant and geographically wide-spread effect on SIA concentrations in eastern China, we decided to keep ACIs on in the model as it is. In reality, when NH_3 emission reduction happens, changed aerosol concentrations will affect meteorology which may in return affect distribution and production of air pollutants.

Although our model captures observed speciated $PM_{2.5}$ concentrations in regions, current model chemistry does not include the most updated sulfate aerosol pathways, which may result in underestimation of sulfate and overestimation of nitrate in wintertime under meteorological conditions unfavorable for dispersion of pollutants. The absence of the most recent aerosol formation mechanism is not uncommon in most atmospheric chemistry models. Models include classic O_3/H_2O_2/OH oxidation pathways for producing sulfate aerosol, but not heterogeneous reactions on surfaces of particles including oxidation by NO_2 in aerosol water, through transition metal ions as catalysis, and by criegee intermediates (sCl) (21-22).

10.3 Co-benefits Analysis Uncertainties

Our agricultural analysis, although helpful in illustrating the magnitude of co-benefits achieved through implementation of NH_3 emission reduction technologies, is based on data with uncertainties. For example, in field experiments with reduced N application rates, scientists do provide farmers instructions on crop variety selection and pest control timing and frequencies. Although those instructions are consistent across experiments within agroecological region and across all nitrogen application rates, this advice could alter farmer practices for both baseline and reduced application rates compared to nonexperimental conditions. In addition, statistical relationships we use that relate Nr emissions to N application rates, although they differ by crop type and crop region, cannot by itself factor in some dependence of emissions on localized practices at atmospheric chemistry model grid level.

When estimating yield and N flows to demonstrate co-benefit achieved, we utilized a number of data sources (i.e., yields in field experiments, technological impact factors, statistical Nr emission models (see Methods)), conducted estimations at crop region levels and aggregated them to national values. Uncertainties of this bottom-up approach includes regional representation of yields and N use, accuracy of planting area statistics, lack of representation of localized-climate and soil conditions in Nr emission model, etc. However, when calculating the differences between scenarios and baseline, these terms should cancel out. We believe this approach is sufficient to demonstrate the magnitude of various co-benefits achieved.

Our baseline calculations are comparable to other published work. For example, we estimate crop N_2O emission is 0.11Tg N, together with animal N_2O emission of 0.3 Tg N reported in (Bai et al., 2018), close to the value of 0.4Tg N of total N_2O emission (Cui et al., 2013). Our annual yield estimation (550 million t) are close to, although slightly higher than, yield (496 million t) reported in China Statistical Yearbook for the year 2010. The discrepancy occurs probably because our yield data represent the average of a three-year time period (2008-2010).

Climate change and water supply are important yield-affecting factors which are excluded from this analysis. A few climate indicators have observed to have changed over time (Piao et al., 2010): 1) **Temperature.** 'China's overall mean annual temperature has significantly increased over the past five decades. The largest warming is found in northeastern China with a trend of 0.36 ℃ per decade, and Inner Mongolia, with 0.4 ℃ per decade. The smallest warming trend is found over southwest China with a trend of 0.15 ℃ per decade.' 2) **Precipitation.** Agricultural crop production in middle and northeastern China is observed to experience an increase in drought from 1960 to 2005, yet grassland in northwestern China is observed to experience a decrease in drought. Overall within China annual number of rain days have decreased. 3) **River runoff**. Among the two major rivers, the Yangtze River shows a small and statistically insignificant increase in annual runoff since 1960 and the Yellow River shows a persistent decline in runoff. A decrease in glacier mass has been observed, resulting in increased local runoff. 4) **Pest diseases.** Partially induced by climate change, cropland area exposed to crop diseases and insect pests have constantly increased by 250% from 1971 to 2007.

Climate change can increase crop yield through enhanced CO_2 fertilization and decreases crop yield through increased heat waves and droughts and decreased precipitation. One research using crop model estimates that climate change overall will increase crop yield of rainfed and irrigated rice by 3% and wheat by 15% and rainfed maize by 10%, yet decrease crop yield of irrigated maize by 1% in the 2020s in China (Piao et al., 2010). Excluding the CO_2 fertilization effect, yields of grain crops (rice, wheat and maize) can decrease by 5-20% in the 2020s (the range is projections made under different future emission change scenarios) (Piao et al., 2010). Another crop and climate model research estimates a 37% decrease of crop yield from climate change without CO_2 fertilization effect (Erda Lin et al., 2005).

Despite these estimations, the magnitude of CO_2 fertilization effect on crop yields is still debated. One research finds that CO_2 level above 450ppm would have an adverse effect on grain quality. There is also contrasting research conclusion on the impact of temperature increase on crop yield – one research estimates 9.8% and 3.2% increase of crop yield for wheat and maize in China by 2090s considering purely impacts of climate warming (with CO_2 fertilization effect excluded). This is because high temperature positively affects crop's water use efficiency and thus yield (Erda Lin et al., 2005).
Uncertain climate model projection of rainfall because of imperfect simulation of clouds in climate model further complicates projecting future crop yield. Also, current crop model excludes the impacts of environmental forcers such as ground-level ozone exposure (which hurts crop yields), decrease of glacial water supply (which hurts crop yields), etc in its crop yield estimation.

Since overall climate change and water supply change is likely to decrease crop yield, this makes the development of agro-technology such as nitrogen management, planting techniques (better seed variety, appropriate planting density), etc even more necessary to secure global food supply.

Table S1 Detailed descriptions of NH_3 mitigation scenarios.

NH_3 mitigation scenarios	Description
Reduced N Application	N application rates for maize, wheat and rice are reduced to levels that optimize farmers' net return (profit minus seed and fertilizer input cost) (Table S2). N application rates for vegetables and fruits are reduced following recommendations reported in the literature, i.e., potato (20%) (Feng, 2014), rapeseed (28%), tobacco (8%), citrus (36%)(Liyou Tang and Xuejun Zhou, 2014), banana (30%)(He et al., 2016), grape (30%) (Dong et al.,), apple (50%) (Wang et al., 2013)(Zhao et al., 2017a), pear (24%), sweet potato (20%)(Feng, 2014) and vegetables (20%)(Shen, 2014). Reduction of NH_3 emissions are calculated by our NH_3 emission model (Zhang et al., 2018, p. 3).
Machine Application	Machine application replaces hand application in all grain croplands except for hilly regions in southern China. Reduction of NH_3 emissions are calculated by our NH_3 emission model (Zhang et al., 2018, p. 3), which uses impact of application mode on emission factors from Bouwman et al (Bouwman et al., 2002).
Efficient Fertilizer	Controlled-release fertilizer is used for rice and maize thus reducing NH_3 emissions by 70%, 43% and 55% when N application rates are below 200kgN/ha, between 200-300kgN/ha and over 300kgN/ha. Urease inhibitor fertilizer is used for wheat, reducing NH_3 emissions by 58% and 42% when N application rates are below and above 200kgN/ha.

Manure Management	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%. 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 to surface application. Low-protein animal feed replaces conventional animal feed which includes more protein-N than chicken, cattle and pigs need. This results in a 10% decrease in N content of manure from these animals.
Combined	N application rates for grain crops, fruits and vegetables are reduced as in *Reduced N Application* scenario. These N are in applied in forms of new-efficient fertilizers as in *Efficient Fertilizer* scenario and placed deeply in cropland as in *Machine Application* scenario. Manure management is also improved as in *Manure Management* scenario.

Table S2 Current and recommended N fertilizer use amount, corresponding yield levels and N application reduction ratio for different production regions of wheat, maize and rice in China. SR, LR and ER are short for single rice cropping, late rice and early rice, respectively.

Crop	production region	Current N use amount (kg N/ha/a)	Current yield (Mg/ha)	Farmers' Profit Maximization N use amount (kg N/ha/a)	Yield under Farmers' Profit Maximization N use amount (Mg/ha)	Recommended N use amount reduction

wheat	NW2	198	4.83	171	6.40	13.6%
	NE	134	4.58	99	4.80	26.1%
	NW1	206	4.60	161	5.59	21.8%
	NCP1	240	6.75	179	6.78	25.4%
	NCP2	200	6.08	193	6.73	3.5%
	YR	200	5.15	184	5.84	8.0%
	SW	144	3.88	131	4.45	9.0%
maize	NW1	234	6.93	181	8.13	22.6%
	NE2	201	9.03	150	9.18	25.4%
	NE1	156	8.59	150	8.85	3.8%
	NE3	226	8.80	164	9.01	27.4%
	NE4	205	8.68	188	8.76	8.3%
	NW2	246	8.22	176	10.38	28.5%
	SW2	234	7.15	219	9.83	6.4%
	NCP1	206	7.68	178	8.13	13.6%
	NCP2	217	7.14	177	8.37	18.4%
	SW1	257	5.41	174	7.46	32.3%
	SW2	232	5.36	183	7.71	21.1%
	SW3	272	5.59	186	8.10	31.6%
	NW3	234	7.15	221	10.33	
rice	NE1_SR	131	7.60	114	7.78	13.0%
	NE2_SR	163	8.07	155	8.90	4.9%
	UYR_SR	199	7.07	163	8.40	18.1%
	MYR_ER	197	6.66	167	6.75	15.2%
	MYR_LR	193	6.94	170	7.10	11.9%

	MYR_SR	206	7.05	172	7.88	16.5%
	LYR_SR	316	7.61	224	8.97	29.1%
	SC1_ER	206	7.00	162	7.03	21.4%
	SC1_LR	212	6.85	162	6.95	23.6%
	SC1_SR	223	7.43	171	8.14	23.3%
	SC2_ER	217	6.54	163	6.69	24.9%
	SC2_LR	244	6.47	164	6.70	32.8%
	SW_SR	203	7.10	162	7.13	20.2%

Table S3: Monthly mean NH_3 concentrations (ppbv) at 918hPa observed by AIRS and simulated by WRF-Chem and GEOS-Chem with NH_3 emissions by Zhang et al (2018) and with MIX NH_3 emissions (Kang et al (2016)(Kang et al., 2016); Huang et al (2017) (Huang et al., 2012)).

NH_3 concentration at 918hPa (ppbv) over AIRS mask		January of 2012	July of 2012
Satellite	AIRS observations before smoothed by using two-dimensional penalized least squares (Warner et al. (2017) (Warner et al., 2016))	1.57	5.76
Model	WRF-Chem simulation using Zhang et al (2018) NH_3	0.62	6.35
	WRF-Chem simulation using MIX NH_3	0.96	2.98
	GEOS-Chem simulation using Zhang et al (2018) NH_3	0.53	4.61
	GEOS-Chem simulation using MIX NH_3	0.51	2.48

Table S4: Evaluation of modeled daily SNA concentrations in Beijing against observations provided by (Chen et al, 2015) in January and July of 2012.

Variables	Pairs of data	Obs (ug/m^3)	Model (ug/m^3)	Mean bias (ug/m^3)	R	IOA*
SO_4 (January)	29	10.27	10.94	0.66	0.69	0.78
NO_3 (January)	29	8.43	9.96	1.53	0.59	0.67
NH_4 (January)	29	7.05	7.12	0.08	0.68	0.81
SO_4 (July)	29	14.92	14.50	-0.42	0.25	0.52
NO_3 (July)	29	12.16	13.66	1.49	0.15	0.44
NH_4 (July)	29	8.60	9.41	0.81	0.19	0.51

*Index of agreement (IOA) was proposed by Willmott et al 1981 On the validation of models *Phys. Geogr. 2(2) 184-194*

Table S5: Evaluation of modeled hourly $PM_{2.5}$ concentrations in Beijing, Shanghai, Guangzhou and Chengdu against U.S. embassy observations in January and July of 2012.

$PM_{2.5}$	Pairs of data	Obs (ug/m^3)	Model (ug/m^3)	Mean bias (ug/m^3)	R	IOA[1]
Beijing (January)	670	118.92	112.07	-6.85	0.72	0.81
Shanghai (January)	740	64.44	62.28	-2.16	0.50	0.70
Guangzhou (January)	82	80.16	66.53	-13.62	0.29	0.49
Beijing (July)	688	80.65	70.16	-10.49	0.52	0.69
Shanghai (July)	737	26.73	29.27	2.54	0.81	0.89
Guangzhou (July)	740	29.46	22.97	-6.50	0.46	0.65
Chengdu (July)[2]	729	59.03	55.28	-3.75	0.70	0.84

[1]Index of agreement (IOA) was proposed by Willmott et al 1981 On the validation of models *Phys. Geogr. 2(2) 184-194*

[2] A comparison for Chengdu in January is not available due to too many invalid observations there.

Table S6 Avoided deaths (mean, low and high estimates with 95% confidence interval) from diseases due to reduced exposure to $PM_{2.5}$ as a result of implementing agricultural nitrogen management.

Avoided deaths from diseases due to exposure to $PM_{2.5}$ (persons)	*Reduced N Application*	*Machine Application*	*Efficient Fertilizer*	*Manure Management*	*Combined*
COPD_mean	1,536	1,249	2,378	2,644	8,261
LC_mean	996	907	1,495	1,510	5,200
IHD_mean	2,061	2,101	3,258	3,112	11,414
Stroke_mean	1,037	1,015	1,572	1,496	5,562
Sum_mean	5,630	5,272	8,703	8,761	30,437
COPD_low	895	735	1,401	1,508	4,807
LC_low	722	664	1,098	1,073	3,773
IHD_low	1,386	1,438	2,214	2,095	7,781
Stroke_low	367	381	570	517	2,049
Sum_low	3,370	3,219	5,284	5,193	18,410
COPD_high	1,881	1,514	2,881	3,308	10,133
LC_high	1,140	1,028	1,687	1,762	5,947
IHD_high	2,618	2,642	4,121	3,980	14,471
Stroke_high	1,742	1,678	2,616	2,568	9,307
Sum_high	7,381	6,861	11,306	11,618	39,858

Table S7: Avoided deaths (mean, low and high estimates with 95% confidence interval) from diseases due to reduced exposure to $PM_{2.5}$ as a result of implementing all agricultural nitrogen management combined for current NH_3 emission inventory and two perturbed NH_3 emission inventories.

Type of estimate	Disease Type	All_combined	All_combined_1.2NH$_3$base	All_Combined_0.8NH$_3$base
Medium estimate	COPD	8261	7505	8230
	LC	5200	4680	5231
	IHD	11414	10271	11604
	Stroke	5562	4968	5595
	Sum	**30437**	**27424**	**30659**
Lower estimate	COPD	4807	4370	4770
	LC	3773	3394	3780
	IHD	7781	7029	7927
	Stroke	2049	1849	2070
	Sum	18410	16642	18546
Higher estimate	COPD	10133	9201	10136
	LC	5947	5354	6004
	IHD	14471	13017	14721
	Stroke	9307	8320	9388
	Sum	39858	35892	40248

Table S8 Grain yield (wheat, maize, rice and sum) in baseline and five nitrogen agricultural management scenarios.

Yield (million tonne)		*Baseline*	*Reduced N Application*	*Machine Application*	*Efficient Fertilizer*	*Manure Management*	*Combined*
Crop Type	Wheat	132551	147170	143657	140750	132551	147170
	Maize	214450	240399	232155	226635	214450	240399
	Rice	203213	214921	217228	218845	203213	214921
	Sum	550214	602490	593041	586230	550214	602490

Table S9 Crop yield, crop Nr emissions (N_2O, NO_3-leaching and runoff), crop NUE, crop GHG emission, as well as N loss to water, GHG and N_2O emission from animal production in baseline and management scenarios. All values are China's national total values. Statistics for crop are calculated summing up wheat, maize and rice, excluding vegetables and fruits due to lack of data.

Environmental and economic indicators		*Baseline*	*Reduced N Application*	*Machine Application*	*Efficient Fertilizer*	*Manure Management*	*Combined*
Crop system	Yield (million tonne)	550	602	593	586	550	602
	N_2O emission (Tg N)	0.11	0.09	0.10	0.07	0.11	0.06
	NO_3-leaching (Tg N)	1.62	1.16	1.62	1.39	1.62	1.00
	NO_3-runoff (Tg N)	0.26	0.20	0.26	0.19	0.26	0.12
	NUE	38%	48%	41%	41%	38%	48%
	Lifecycle greenhouse gas emissions (GHGs) due to	223	185	223	223	223	185

	fertilizer manufacture, transport and use (Mega tonne CO_2-eq)						
Animal system	N_2O emission (Tg N)	0.3	0.3	0.3	0.3	0.3	0.3
	N loss to water (Tg N)	12	12	12	12	9.5	9.5

Table S10 Co-benefits (in absolute amount and relative percentage compared to baseline levels) of yield, Nr loss, NUE, and greenhouse gas emissions due to fertilizer consumption achieved in management scenarios.

Types of co-benefit		*Reduced N Application*	*Machine Application*	*Efficient Fertilizer*	*Manure Management*	*Combined*
Crop system	Yield increase (million tonne)	52 (9%)	43 (8%)	36 (7%)	0	52 (9%)
	N_2O emission reduction (Tg N) and GHG reduction (Mega tonne CO_2-eq)	0.02 (18%) and 9.4	0.01 (9%) and 4.7	0.04 (36%) and 18.8	0	0.05 (50%) and 23.5
	NO_3-leaching reduction (Tg N)	0.46 (28%)	0	0.23 (14%)	0	0.62 (38%)

	NO_3-runoff reduction (Tg N)	0.06 (23%)	0	0.07 (27%)	0	0.14 (56%)
	NUE	0.10 (26%)	0.03 (8%)	0.03 (8%)	0	0.10 (26%)
	GHGs reduction due to reduced N fertilizer consumption (Mega tonne CO_2-eq)	38 (17%)	0	0	0	38 (17%)
Animal system	N loss to water reduction (Tg N)	0	0	0	2.5 (21%)	2.5 (21%)

Table S11 Low estimate of economic benefits & costs calculated for different agricultural control scenarios

(US$ billions/year) and value of economic parameters used.

Metrics	*Reduced N Application*	*Machine application*	*Efficient fertilizer*	*Manure management*	*Combined*	Economic parameters used
Value of lives saved from $PM_{2.5}$ air	0.51	0.47	0.78	0.79	2.74	Value of statistical life

pollution related deaths						0.09 million USD/reduced person mortality
Reduced social cost of carbon (CO_2-eq) emissions	0.38	0.08	0.14	0	0.38	Social cost of carbon 10$/tonne CO_2-eq
Labor savings	0	0.2	0	0	0.2	Differences in time efficiencies of *Machine Application* and hand application 2.5h/ha
Organic fertilizer sales	0	0	0	2.74	2.74	
Estimated costs for improving N management (this study)	0	1.31	2.75	-16.2	-12.4	Fertilizer application machine rental price 20¥ ha^{-1} Composting reactor lifetime 50yr Discount rate 2% Organic fertilizer sale price 1000¥ t^{-1}

Table S12 Medium estimate of economic benefits & costs calculated for different agricultural control scenarios

($USD billions/year) and value of economic parameters used.

Metrics	*Reduced N Application*	*Machine application*	*Efficient fertilizer*	*Manure management*	*Combined*	Economic parameters used
Value of lives saved from PM2.5 air pollution related deaths	0.70	0.65	1.08	1.08	3.77	Value of statistical life 0.0124 million USD/reduced person mortality
Reduced social cost of carbon (CO_2-eq) emissions	0.76	0.16	0.28	0	0.76	Social costs of carbon 20.4 $/tonne CO_2-eq (Schiermeier, 2009)
Labor savings	0	0.45	0	0	0.45	Differences in time efficiencies of Machine Application and hand application 4h/ha
Organic fertilizer sales	0	0	0	5.14	5.14	

Estimated costs for improving N management (this study)	0	3.92	4.09	0.01	8.02	Fertilizer application machine rental price 40¥ ha^{-1} Composting reactor lifetime 20yr Discount rate 4% Organic fertilizer sale price 300¥ t^{-1}

Table S13 High estimate of economic benefits & costs calculated for different agricultural control scenarios

($USD billions/year) and value of economic parameters used.

Metrics	*Reduced N Application*	*Machine application*	*Efficient fertilizer*	*Manure management*	*Combined*	Economic parameters used
Value of lives saved from PM2.5 air pollution related deaths	1.41	1.32	2.18	2.19	7.61	Value of statistical life 0.25 million US$/reduced person mortality
Reduced social cost of carbon	1.89	0.4	0.7	0	1.89	Social costs of carbon

(CO_2-eq) emissions						50 $/tonne CO_2-eq
Labor savings	0	0.7	0	0	0.7	Differences in time efficiencies of Machine Application and hand application 7.5h/ha
Organic fertilizer sales	0	0	0	20.5	20.5	
Estimated costs for improving N management (this study)	0	7.84	5.42	8.89	22.5	Fertilizer application machine rental price 60¥ ha^{-1} Composting reactor lifetime 20yr Discount rate 15% Organic fertilizer sale price 1000¥ t^{-1}

Table S14 Benefit/cost ratio for implementing agricultural nitrogen management scenarios.

Metric ($/$)	*Reduced N Application*	*Efficient Fertilizer*	*Machine Application*	*Manure Management*	*Combined*

high benefit/high cost	34.6	2.5	1.8	2.0	1.8
high benefit/medium cost	34.6	3.1	3.3	3.2	2.8
high benefit/low cost	34.6	4.2	6.8	4.4	4.3
medium benefit/high cost	30.7	2.2	1.8	0.6	1.1
medium benefit/medium cost	30.7	2.8	3.2	0.9	1.7
medium benefit/low cost	30.7	3.8	6.7	1.3	2.5
low benefit/high cost	29.8	2.2	1.7	0.4	0.9
low benefit/medium cost	29.8	2.7	3.1	0.6	1.4

low benefit/low cost	29.8	3.7	6.4	0.8	2.2

Table S15: Prices (¥ kg^{-1} N) of regular fertilizers, controlled-released fertilizers, and fertilizers with nitrification inhibitor (NI) and urease inhibitor (UI), source (Xia et al., 2017)

Fertilizer type	Category[a]	Price or cost
Regular fertilizer (Xia et al., 2016)	Urea	4 ¥ kg^{-1} N
Controlled-release fertilizer	CRF (sulfur-coated)	5.6 ¥ kg-1 N
	CRF (polymer-coated)	7.2 ¥ kg-1 N
	CRF (polymer and sulfur-coated)	6.4 ¥ kg-1 N
Urease inhibitor	NBPT	100 ¥ kg^{-1}

a. The price of urea, P and K fertilizer, staple grains (Xia et al., 2016). The price of controlled-release fertilizers referred to Chinese Fertilizer Website (www.fert.cn). The price of various nitrification and urease inhibitor products referred to Chinese Guide Chem Website (china.guidechem.com).

Table S16 Damage costs for China of NO_3-N, N_2O and NH_3-N emissions adjusted from European studies (Ying et al., 2017).

	Health $/kg N	Eutrophication $/kg N	Acidification $/kg N	Global warming $/kg N	Total $/kg N
NO_3-N	0.2	1.12	_	_	1.32
N_2O-N	0.3**	_	_	11.2	11.5

NH_3-N	3.3	0.24*	1.87*	_	5.41

*We didn't calculate the eutrophication and acidification damage costs from NH_3 emission given a recent study for China shows that reduced NH_3 emission actually worsens acid rain (55).

** We didn't calculate the health impacts from N_2O emission due to controversies of whether current stratospheric ozone levels have recovered to levels higher than desirable so further N_2O reduction is not needed.

Table S17 Labor cost in agroecological regions, mapped from provincial labor cost reported by China Statistical Yearbook(Chinese NDRC (national development and reform council), 2016). Labor cost is defined as the amount of payment a person with mid-level working capacity should receive for working for one day (8 h).

Crop	Production region	Labor cost (¥ /d =¥ /8h)
Wheat	NW2	63
	NE	70
	NW1	51
	NCP1	140
	NCP2	62
	YR	106
	SW	55
Maize	NW1	60
	NE2	81
	NE1	300
	NE3	60
	NE4	65

	NW2	67
	SW2	68
	NCP1	140
	NCP2	62
	SW1	60
	SW2	68
	SW3	55
Rice	NE1_SR	300
	NE2_SR	73
	UYR_SR	60
	MYR_ER	67
	MYR_LR	67
	MYR_SR	67
	LYR_SR	56
	SC1_ER	81
	SC1_LR	81
	SC1_SR	81
	SC2_ER	85
	SC2_LR	85

Table S18: N_2O emission and NO_3^--leaching levels (kg N/ha) under a specific level of fertilizer nitrogen input (kg N/ha) for maize (Cui et al., 2013), rice and wheat in China (Cui et al., 2014).

Crop	N_2O emission (Y; kg N/ha) with respect to fertilizer N input (x; kg N/ha)	NO_3^-- leaching (Y; kg N/ha) with respect to fertilizer N input (x; kg N/ha)	NO_3^-- runoff (Y; kg N/ha) with respect to fertilizer N input (x; kg N/ha)
Spring maize	Y=0.576exp(0.0049x)	Y=4.46exp(0.0094x)	N/A
Summer maize	Y=0.593exp(0.0045x)		
Wheat	Y=0.33exp(0.0054x)	Y=2.7exp(0.0088x)	
Rice	Y=0.37exp(0.0061x)	Y=2.2exp(0.0071x)	Y=2.12exp(0.0071x)

Table S19 Impact of agricultural nitrogen management technologies on crop yield, NH_3, N_2O and NO_3-leaching and runoff in China (Xia et al., 2017).

Technology	condition	yield increase (%)	NH_3 Emission change (%)	N_2O emission change (%)	NO_3-leaching change (%)	NO_{3}- runoff change (%)
Controlled-release fertilizer	if N rate <200	7.64	-70.20	-31.40	-14.05	-39.74
	if 200 < N rate <300	5.17	-42.57	-27.14	-19.37	-15.01
	if N rate >300	17.75	-66.26	-56.70	-38.50	-28.58
Urease inhibitor	if N rate <200	6.02	-58.11	-24.81	N/A	N/A
	if 200 < N rate <300	6.22	-42.24	-48.38	N/A	N/A
	if N rate >300	14.21	-42.24	-35.05	N/A	N/A
	if N rate <200	5.41		-14.60	N/A	-15.47

Machine Application	if 200 < N rate <300	9.00	-14.60	N/A	-15.47
	if N rate >300	8.55	-14.60	N/A	-15.47

Table S20 Cost-benefit analysis for *Machine Application* scenario and its two variations.

Metrics	*Deep Placement of N by hand*	*Machinery surface broadcasting*	*Machine Application* scenario (fertilizer deep placement by machine)
Value of lives saved from $PM_{2.5}$ air pollution related deaths	0.65 (0.47; 1.32)	0	0.65 (0.47; 1.32)
Increased crop sales	14.1	0	14.1
Reduced damage from NO_3-leaching and runoff	0.06	0	0.06
Reduced social cost of carbon	0.16 (0.08; 0.4)	0	0.16 (0.08; 0.4)

(CO_2-eq) emissions			
Labor savings	0	0.2~0.7	0.2~0.7
Reduced N deposition (by scaling benefits estimated by Liu et al)(Liu et al., 2019)	0.41	0	0.41
Labor cost	0.4~1.4*	0	0
Costs of worsened acid rain (by scaling benefits estimated by Liu et al)(Liu et al., 2019)	1.07	0	1.07
Technological costs for improving N management resulting in NH_3 emission	0	3.92 (1.31; 7.84)	3.92 (1.31; 7.84)

reduction (this study)			
Total benefits*	15.38 (15.12; 16.29)	0.45 (0.2;0.7)	15.81(15.31; 16.31)
Total costs**	1.97 (1.47; 2.88)	3.92 (1.31; 7.84)	4.99 (2.38; 8.91)

*we roughly estimate that the time needed for dig soil to apply fertilizers deeply will be two times of that needed for surface broadcasting by hand.

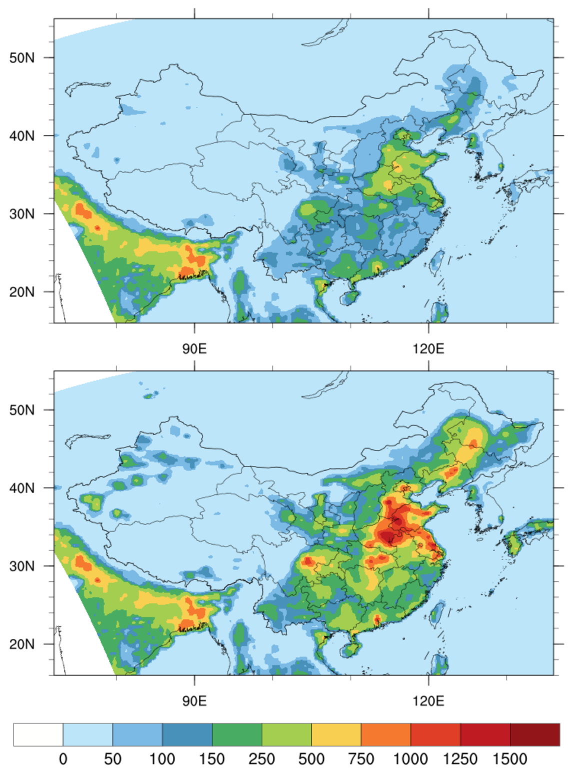

Figure S1 baseline NH_3 emissions (in kg/km^2/month) in January (1st row, 0.66Tg) and July (2nd row, 1.66Tg) of 2012.

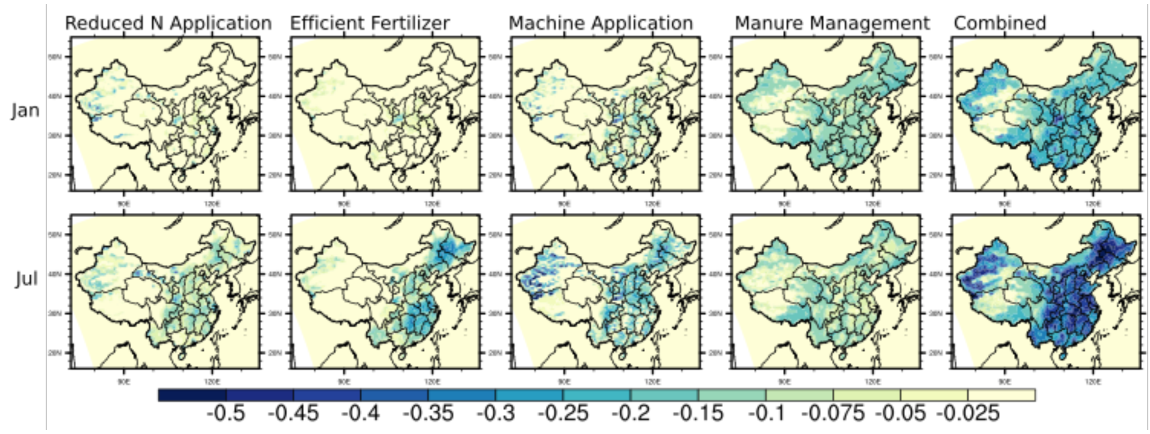

Figure S2 NH_3 emission reduction ratios in management scenarios compared to baseline in January and July of 2012.

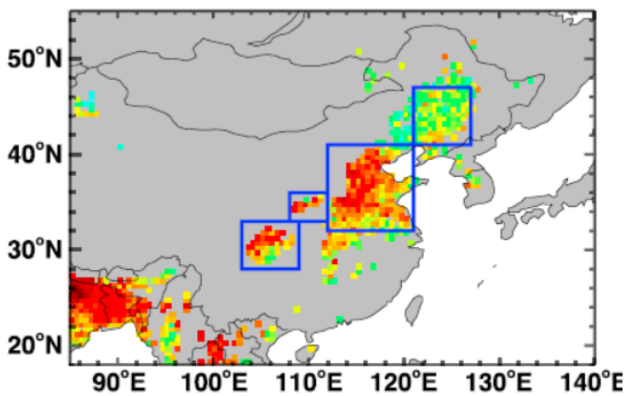

Figure S3: Pixels (the colored grid boxes within blue rectangles) where AIRS observations are extracted for comparison against model NH_3 concentration.

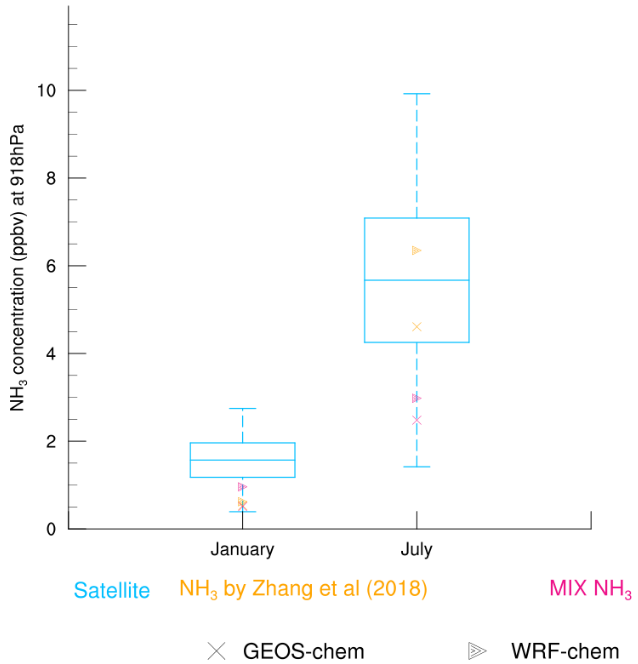

Figure S4: NH_3 concentration (ppbv) at 918hPa observed by AIRS (blue boxes with ±25% and ±75% ranges provided) and NH_3 concentration (ppbv) at 918hPa simulated by WRF-chem (in ×) and GEOS-chem (in rectangle) with NH_3 emissions by Zhang et al (2018) (orange) and with MIX NH_3 emissions (pink) (published in Kang et al (2016)(Kang et al., 2016) and Huang et al (2017) (Huang et al., 2012)) for January and July of 2012.

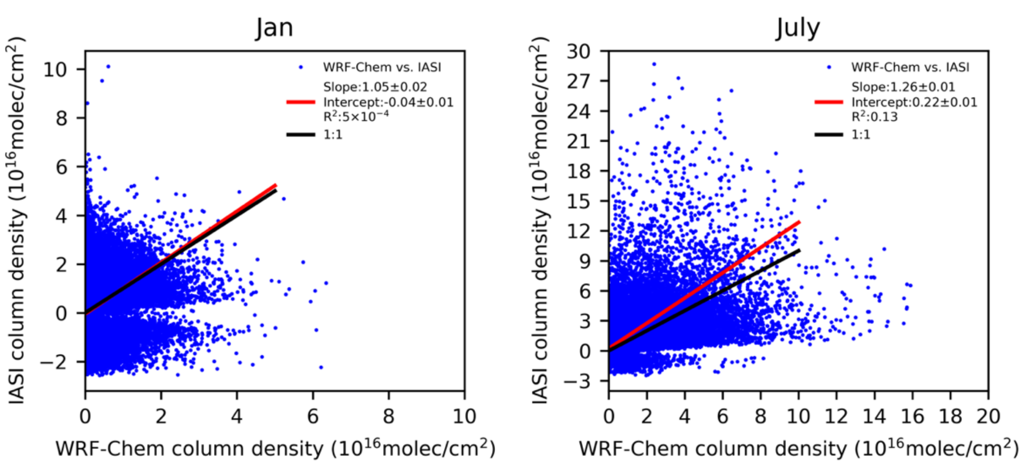

Figure S5: Daily NH_3 column density observed by IASI and simulated by WRF-chem with Zhang et al (2018) NH_3 (this study) for January (number of valid pixels: N=70063) and July (N=96557).

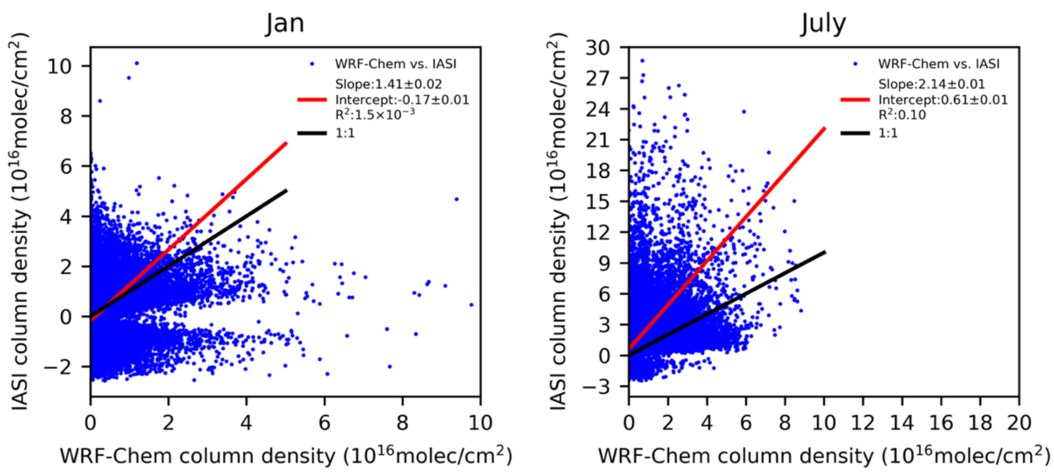

Figure S6: Daily NH_3 column density observed by IASI and simulated by WRF-chem with MIX NH_3 for January (number of valid pixels: N=70063) and July (N=96557).

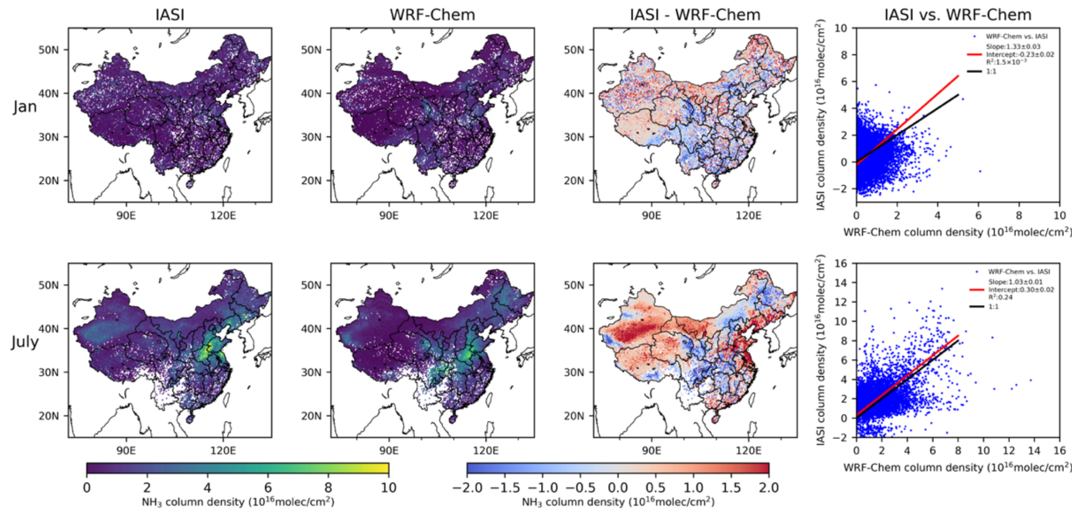

Figure S7 Monthly mean NH_3 column density observed by IASI (1st column) and simulated by WRF-chem with Zhang et al (2018) NH_3 (this study) (2nd column), as well as their difference (IASI-WRF-chem, 3rd column) for January (number of valid pixels: N=23506) and July (N=20767). An orthogonal distance regression (4th column) is also provided.

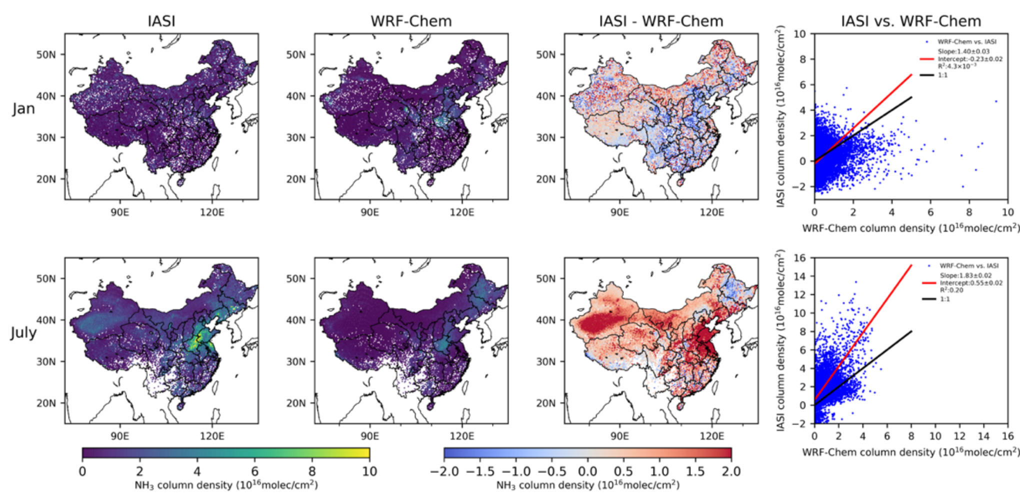

Figure S8 Monthly mean NH_3 column density observed by IASI (1st column) and simulated by WRF-chem with MIX NH_3 emissions (published in Kang et al (2016) (Kang et al., 2016) and Huang et al (2017) (Huang et al., 2012)) (2nd column), as well as their difference (IASI-WRF-chem, 3rd column) for January (number of valid pixels: N=23506) and July (N=20767). An orthogonal distance regression (4th column) is also provided.

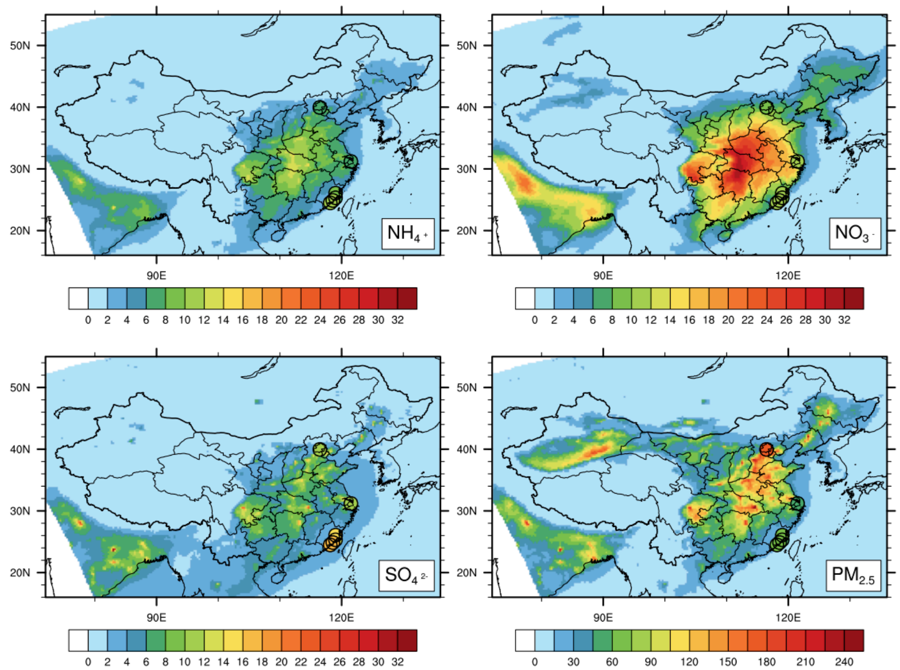

Figure S9 Evaluation of modeled ammonium (NH_4^+,) nitrate (NO_3^-), sulfate (SO_4^{2-}) and $PM_{2.5}$ concentrations (in μg/m^3) against observations during January of 2012. All observational and modeled values are monthly, except for four sites in the southeast where observation was conducted during the first 15 days of January so modeled concentrations are also averaged among that same period of time. Across observational sites, for NH_4^+, observations average at 8.5 ug/m^3 and model estimated 4.3 ug/m^3. For NO_3^-, observations average at 10.1 ug/m^3 and model estimated 8.1 ug/m^3. For SO_4^{2-}, observations average at 14.0 ug/m^3 and model estimated 6.0 ug/m^3. For $PM_{2.5}$, observations average at 83.7 ug/m^3 and model estimated 66.9 ug/m^3.

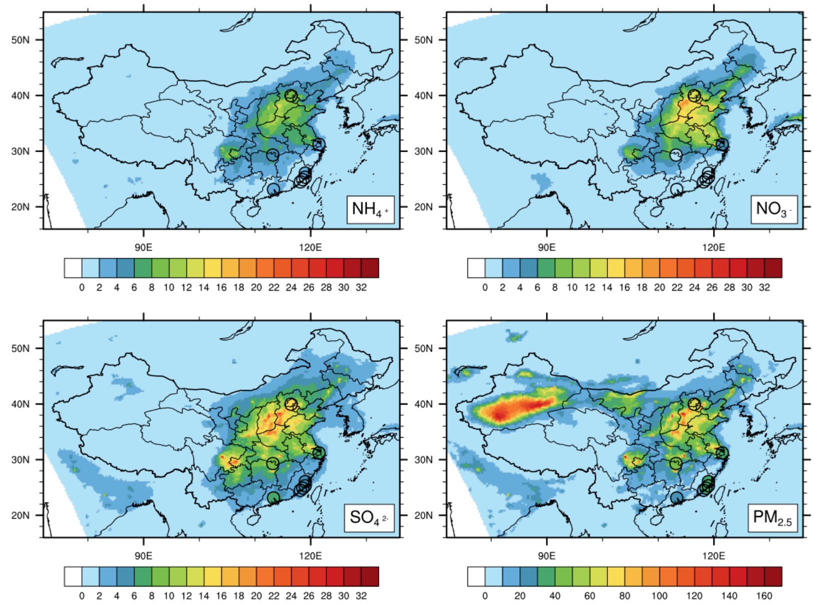

Figure S10 Evaluation of modeled ammonium (NH_4^+,) nitrate (NO_3^-), sulfate (SO_4^{2-}) and $PM_{2.5}$ concentrations (in μg/m^3) against observations during July of 2012. All observational and modeled values are monthly, except for four sites in the southeast where observation was conducted during the first 15 days of July, so modeled concentrations are also averaged among that same period of time. Across observational sites, for NH_4^+, observations average at 3.1 ug/m^3 and model estimated 3.4 ug/m^3. For NO_3^-, observations average at 2.7 ug/m^3 and model estimated 3.9 ug/m^3. For SO_4^{2-}, observations average at 6.9 ug/m^3 and model estimated 7.1 ug/m^3. For $PM_{2.5}$, observations average at 36.2 ug/m^3 and model estimated 33.4 ug/m^3.

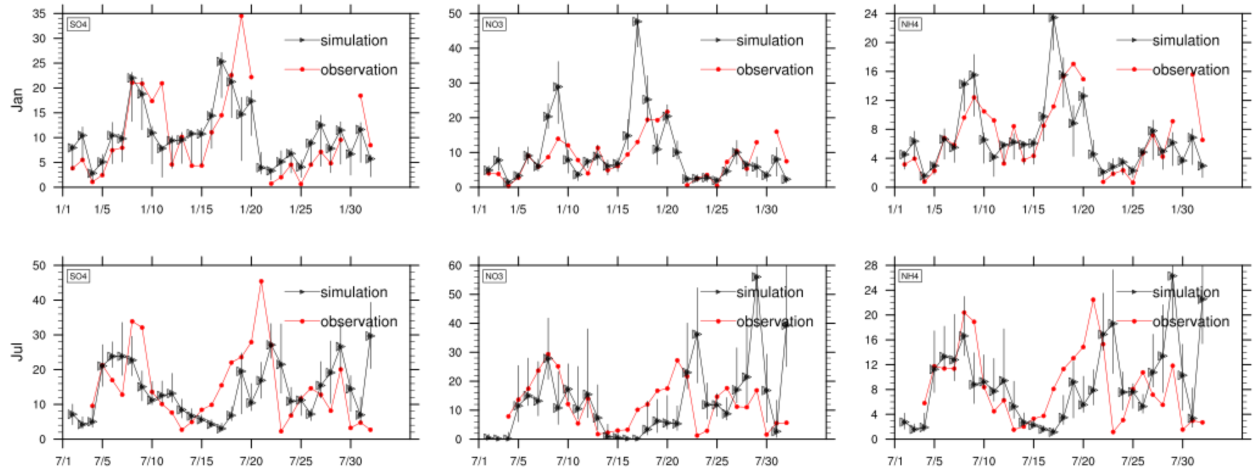

Figure S11: Comparison of modeled daily concentrations of sulfate, nitrate and ammonium aerosols (black triangles) against observations (red dots) in Beijing in January and July of 2012. The error bar of modeled results denotes the range of concentrations in the eight grid cells surrounding the grid cell where Beijing is located. Source of observation data is from Chen et al. (2015).

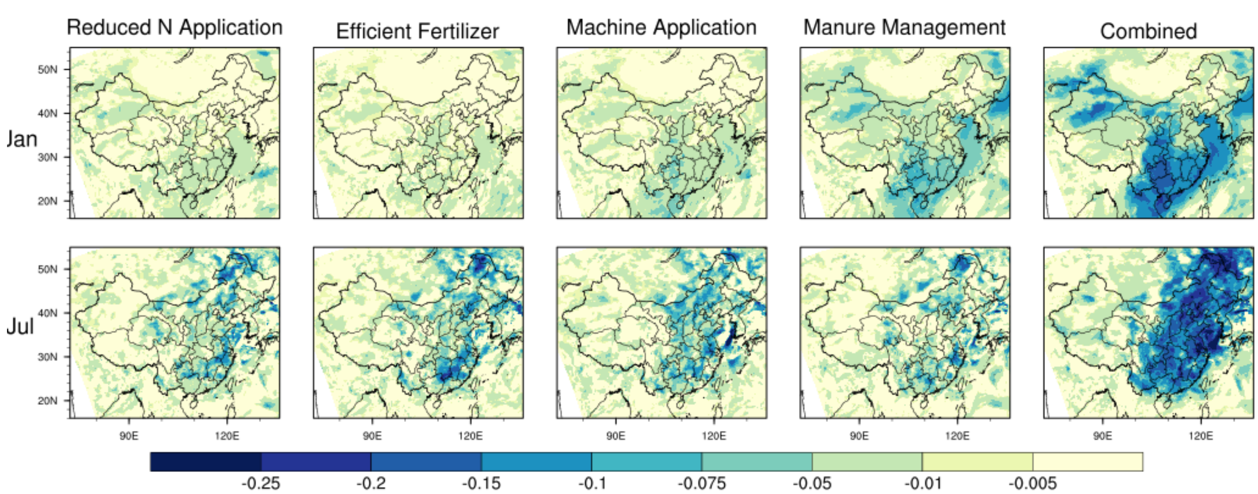

Figure S12 SIA concentration reduction ratios in management scenarios compared to baseline emission in January and July of 2012.

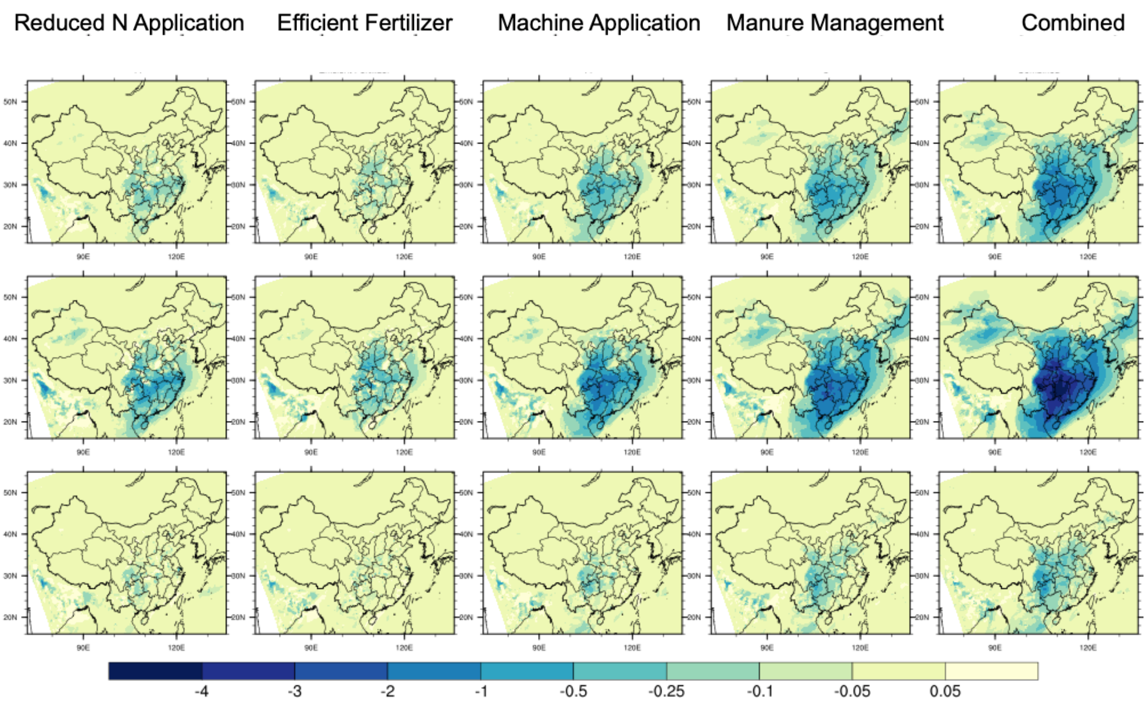

Figure S13 Changes in ammonium (1st row), nitrate (2nd row) and sulfate (3rd row) aerosol concentrations (in μg/m^3) in management scenarios compared to baseline simulation in January 2012.

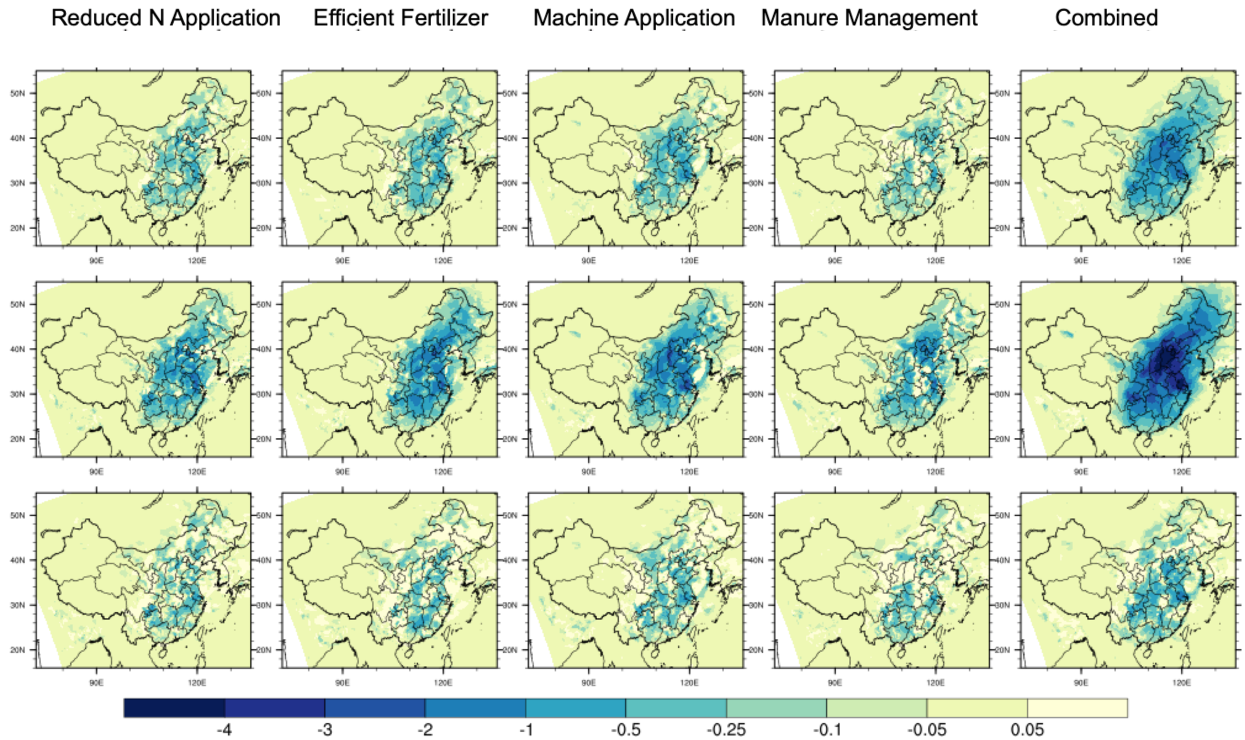

Figure S14 Changes in ammonium (1st row), nitrate (2nd row) and sulfate (3rd row) aerosol concentrations (in μg/m^3) in management scenarios compared to baseline simulation in July 2012.

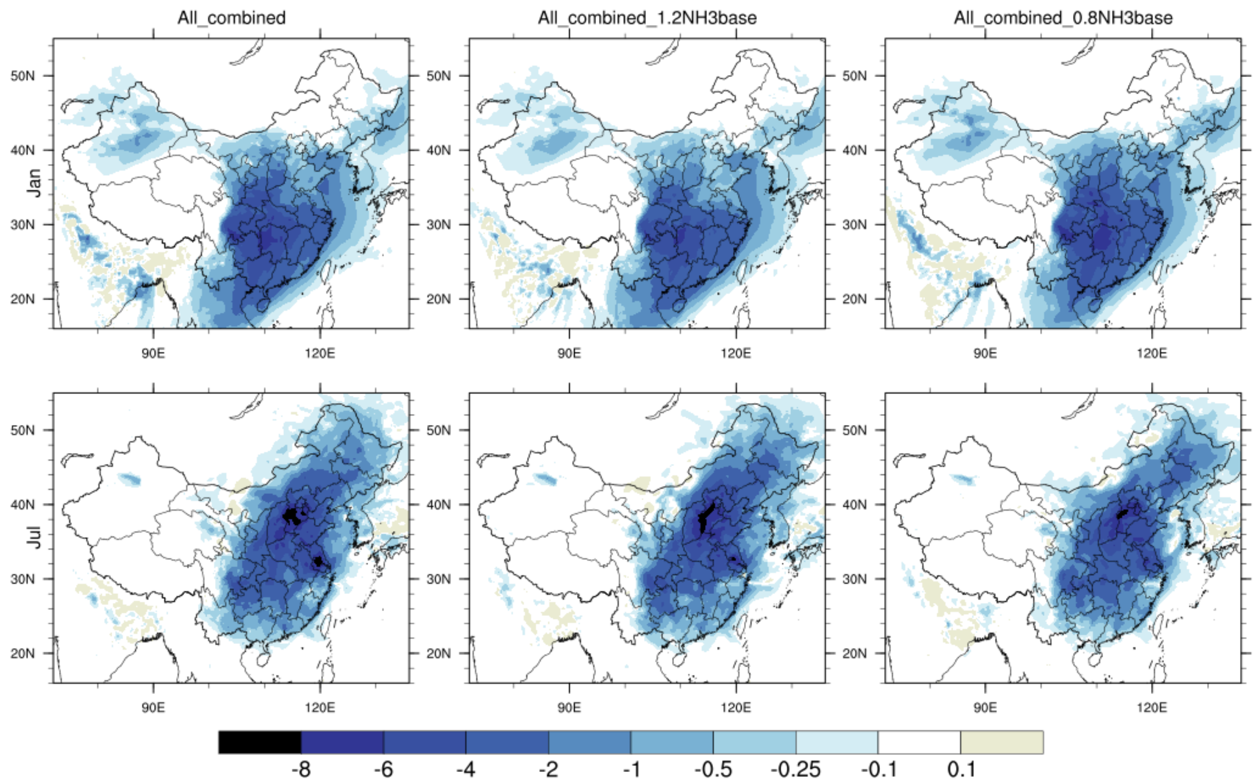

Figure S15: Changes in ground-level (surface layer in WRF-Chem is 18m thick) concentrations (in unit of $\mu g/m^3$, negative values mean reductions) of secondary inorganic aerosol (SIA, the sum of ammonium, nitrate and sulfate aerosols) in management scenarios compared to the *Baseline* simulation, in January and July of 2012 if baseline NH_3 emission is Zhang et al (2018) (first column, all_combined), 20% higher than Zhang et al (2018) (second column, All_combined_1.2NH$_3$base), and 20% lower than Zhang et al (2018) (third column, All_combined_0.8NH$_3$base).

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Appendix for Chapter 4

Environmental and Dietary Health Co-benefits and Tradeoffs in Potential Future Chinese Dietary Choices

Estimating food consumption based on diets

We calculate food consumption by adding food waste to food intake. We account for food waste during production, post-harvesting, food processing, packaging, distribution and consumption, using food waste ratios reported by FAO(Gustavsson *et al* 2011) (Table S1).

We report both food intake and food consumption under baseline, under both non-standardized food types (Table S2) and standardized food types (Table S3). Among the ~5000 types of Chinese food products we model, food products under the same food group can vary significantly in nutrition composition. Cooking methods of food also affect nutrition and energy supply. For example, different types of pork have substantially different fat and protein contents and energy supply. One gram of strawberry has much less calorie supply than one gram of grapes. Cooked rice and rice congee have substantially different energy supply.

It is thus problematic to report raw weight of food given large within-food-group variation; we thus standardize food items to allow better comparison following guidelines of calculating food weight equivalent provided by Chinese Dietary Guidelines. The standardized food types are defined as food with certain amount of nutrients and energy supply (Table S).

Estimating agricultural production based on food consumption

Baseline agricultural production in 2012 in NH_3 emission model is from production data in China Statistical Yearbook. In order to estimate agricultural production in dietary scenarios for the sake of NH_3 emission simulation, we scale up baseline production in NH_3 emission model using a production scale factor.

A production scale factor for a specific agricultural product equals to the ratio of its consumption in a dietary scenario to consumption in baseline diet. This rule applies to all agricultural products except for maize, wheat, rice and soybean.

Maize, wheat, rice and soybean are important animal feed crops. Thus, changes of their production in dietary scenarios should reflect both changes in human food demand and changes in animal feed demand as a result of changed animal production. We follow three steps to estimate their production in dietary scenarios as percentages of baseline production.

First, we obtain the partitioning between crop production (rice, wheat, maize and soybean) for animal feed, for human food and for other purposes (Table 2). Usages of rice, wheat and maize are directly from FAO Food Balance Sheet in 2011. Usages of soybean include soybean for animal feed, for human food, for processing and others. 80% of processed soybean is for animal feed and 20% is for soybean oil[28]. We thus add the processed soybean for animal feed to primary soybean for animal feed as total soybean for animal feed. We assume crop production for other

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[28] http://www.chyxx.com/industry/201805/638760.html

usages stay unchanged in scenarios compared to baseline. The two steps adjust crop production for feed and for food, respectively.

Second, crop production for animal feed in scenarios with respect to that in baseline will be proportionally to total meat (beef, goat, poultry and pork) consumption in scenarios with respect to that in baseline. Our dietary scenario results show that total meat (beef, goat, poultry and pork) consumption in Balanced Diet scenario is 46% of that in baseline, similarly 26% in EAT_Balanced diet, 97% in US Diet scenario, and 32.6% in Zero Red Meat scenario. Thus, crop production for animal feed in Balanced Diet scenario will be 46% of that in baseline, in EAT_Balanced diet 26%, in US Diet scenario 97%, and in Zero Red Meat scenario 32.6%.

Third, we calculate crop production for human food in scenarios by multiplying baseline crop production for human food in baseline and the ratio of human crop consumption in scenarios to that in baseline. Table 2 summarizes production amount in scenarios as percentages of that in baseline by food type.

For one crop, P denotes production. C denotes consumption.

$$P_{base} = P_{base_{animalfeed}} + P_{base_{humanfood}} + P_{base_{others}}$$

$$P_{scenario} = P_{base_{animalfeed}} \times \frac{C_{scenario_{meat}}}{C_{base_{meat}}} + P_{base_{humanfood}} \times \frac{C_{scenario}}{C_{base}} + P_{base_{others}}$$

$$\frac{P_{scenario}}{P_{base}} = \frac{P_{base_{animalfeed}}}{P_{base}} \times \frac{C_{scenario_{meat}}}{C_{base_{meat}}} + \frac{P_{base_{humanfood}}}{P_{base}} \times \frac{C_{scenario}}{C_{base}} + \frac{P_{base_{others}}}{P_{base}}$$

Crop import is quite small quantity compared to domestic production, for wheat (0.4%), maize (2%) and rice (3%), except for soybean (377%) (Table 2). We assume that the ratio of import to domestic production stays unchanged in all dietary scenarios as that in baseline.

Production-based NH_3 emission model

We utilize a production-based NH_3 emission model to obtain baseline NH_3 emissions and NH_3 emissions in dietary change scenarios(Zhang *et al* 2018). The NH_3 emission model is the most updated and best available bottom-up high-resolution NH_3 emission estimation tool for China. The model represents production of eighteen crops (including maize, wheat, rice, potato, sweet potato, rapeseed, soybean, groundnut, tobacco, cotton, citrus, banana, grape, apple, pear, other fruits, vegetables). Crop NH_3 emission factors are parametrized with fertilizer application timing, rate, type, method, as well as a number of climate (temperature, wind, etc.) and soil (pH) conditions. The model represents production of major animals (cattle, goat, sheep, pig and poultry) in grazing, intensive and free-range systems. Total ammonium nitrogen (TAN) content produced by outdoor animals are subject to NH_3 volatilization and are without further management. TAN produced by indoor animals goes through several stages of management, i.e. 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.

Air quality simulation

We use the Weather Research and Forecasting – Chemistry (WRF-Chem) model v3.6.1 to simulate $PM_{2.5}$ formation from baseline emissions and from emissions in agricultural N management scenarios. WRF-Chem is an online-coupled meteorology-chemistry model widely used for air quality research (Gao *et al* 2016, Qin *et al* 2017). The physical and chemical schemes used are Carbon-Bond Mechanism Version Z (CBMZ) for gas-phase chemistry, 4-bin Model for Simulating Aerosol Interactions and Chemistry (MOSAIC) for aerosol chemistry, RRTMG scheme for shortwave and longwave radiation, the Morrison scheme for cloud microphysics (Morrison *et al* 2005), the Yonsei University scheme for boundary layer mixing (Hong *et al* 2006), and the Noah land surface model for land surface (Chen and Dudhia 2001). Meteorological boundary conditions are from the 2014 National Centers for Environmental Prediction (NCEP) Final Analyses data for every 6 hours. Chemical initial and boundary conditions are a 2014 simulation of the global chemical transport model, Model for Ozone and Related Tracers Version 4 (MOZART-4).

Anthropogenic emissions of air pollutants are from the Multi-resolution emission inventory for China (MEIC) (http://www.meicmodel.org) (Li *et al* 2017) and from HTAP (Hemispheric Transport of Air Pollutants) v2.2 outside China (Janssens-Maenhout *et al* 2015). Biogenic emissions are calculated online using the Model of Emissions of Gases and Aerosols from Nature (MEGAN) scheme (Guenther *et al* 2006) and open biomass burning emissions are from Global Fire Emission Database version 4 (GFED v4, www.globalfiredata.org).

We conduct 4 sets of simulations: one baseline and three dietary change scenarios where the only difference from the baseline simulation is modified NH_3 emissions due to dietary changes. Each simulation set includes one month of simulation for January and one month of simulation for July (both after six days of spin-up) for the year 2012. The model resolution is 27 km by 27 km with the domain covering China and parts of other Asian countries (9°N-58°N, 60°E-156°E) and with 37 vertical levels extending from the surface to 50hPa. We turn off direct aerosol-climate feedback to minimize the impact of aerosol concentration change due to meteorology, which would in return feedback to aerosol concentration simulation.

Evaluation of 2012 baseline NH_3(g) concentration

We conduct evaluations of modeled NH_3 concentration against satellite observations, i.e. modeled NH_3(g) concentration at 918hPa against AIRS (Atmospheric Infrared Sounder) satellite observation, and modeled column density of NH_3 (g) against IASI (Infrared Atmospheric Sounding Interferometer) satellite observation. Our NH_3 emission inventory provides reasonably good comparison over China in January and July of 2012. Our inventory outperforms MIX NH_3 inventory (the NH_3 emission inventory published in Kang et al (2016)(Kang *et al* 2016) and Huang et al (2017) (Huang *et al* 2012) and used in Liu et al (2019) and MIX inventory (Li *et al* 2017)).

We run both WRF-chem and GEOS-chem (another well-known atmospheric chemistry model frequently used by atmospheric scientists) with NH_3 emissions from Zhang et al (Zhang *et al* 2018, p 3) (used in this study) and MIX NH_3 inventory (the NH_3 emission inventory published in Kang et al (2016)(Kang *et al* 2016) and Huang et al (2017) (Huang *et al* 2012)and used in Liu et al (2019)). We extracted modeled NH_3 concentration at 918hPa averaged over regions that AIRS satellite provides reliable observations (AIRS mask region, Figure S2). The AIRS observations are NH_3(g) concentration before smoothing through two-dimensional penalized least squares provided by Warner J.K.. Warner's paper provide AIRS observations that are after smoothing (Warner *et al* 2016).

AIRS observation shows a strong summer-winter contrast of NH_3 concentration, which is better captured by NH_3 emission inventory used in this study. Zhang et al (2018) NH_3 outperforms MIX NH_3 especially in July, where both WRF-chem and GEOS-chem simulation with MIX NH_3 is -25%~-75% lower than AIRS observation yet both WRF-chem and GEOS-chem simulation with Zhang et al (2018) NH_3 is within 25% of AIRS observation (Table S8).

We evaluate NH_3 column density simulated by WRF-chem using both Zhang et al (2018) NH_3 (this study) and MIX NH_3 emissions (published in Kang et al (2016) (Kang *et al* 2016) and Huang et al (2017) (Huang *et al* 2012)) against IASI observations for January and July of 2012 over China. We processed IASI column NH_3 density according to v2.2 instruction (Damme *et al* 2017)[29]. Only observations with relative error <100% are used and unweighted averages are used. IASI pixels are then mapped to closest WRF-chem grids (27km ×27km resolution, 9.0-57.6°N and 59.6-156.3°E domain) following two principals: 1) multiple pixels could be mapped to the same grid. 2) In this case, average values of IASI are used for comparison. WRF-chem grids and periods without valid IASI observations are removed.

[29] http://cds-espri.ipsl.fr/etherTypo/index.php?id=1730&L=1

We first compare daily values of mapped IASI data to WRF-chem simulation using two NH_3 emission inventories using orthogonal distance regression (Figure S3 and S4). We then compare monthly means of all valid daily mapped IASI and WRF-chem simulations (Figure S5 and S6). Overall Zhang et al (2018) NH_3 is much better than MIX NH_3. The correlation of IASI and WRF-chem simulation (regardless of which NH_3 emission inventory is used) is much worse in January than in July, probably due to large IASI uncertainty in winter.

Evaluation of 2012 baseline $PM_{2.5}$ simulation

We evaluate modeled $PM_{2.5}$, ammonium (NH_4^+,) nitrate (NO_3^-) and sulfate (SO_4^{2-}) concentrations during January and July of 2012 against as many published speciated PM observations during the same time period as possible, i.e., Beijing from (US Embassy at Beijing 2012)(Sun *et al* 2015), Shanghai (Wang *et al* 2016), Xiamen, Quanzhou, Putian and Fuzhou (Wu *et al* 2015), Guangzhou (Lai *et al* 2016), and Jinsha (Zhang *et al* 2014). Observational sites mainly represent the North China Plain, Yantze-River Delta and coastal regions. Unfortunately, there is a lack of representation for inland China during wintertime. The WRF-Chem model well captures observations, especially in July (Figures S7 and S8). In July, modeled mean ammonium, nitrate, sulfate and $PM_{2.5}$ concentrations averaged across eight sites are 3.4, 3.9, 7.1 and 33.4 $\mu g/m^3$, decently close to observations of 3.1, 2.7, 6.9 and 36.2 $\mu g/m^3$ (Figure S10). In January, modeled mean ammonium, nitrate, sulfate and $PM_{2.5}$ concentrations averaged across six sites are 4.3, 8.1, 6.0 and 66.9 $\mu g/m^3$, compared to observations of 8.5, 10.1, 14.0 and 83.7 $\mu g/m^3$ (Figure S9). The model underestimation of sulfate in winter is prominent at four coastal sites in Fujian province, but barely noticeable in Beijing and Shanghai. In January at Beijing, modeled ammonium, nitrate, sulfate and $PM_{2.5}$ concentrations are 6.9, 9.4, 10.5 and 174.9 $\mu g/m^3$, compared to observations of 6.9, 8.3, 10.2 and 175.3 $\mu g/m^3$. In January at Shanghai, modeled ammonium, nitrate, sulfate and $PM_{2.5}$ concentrations are 7.5, 13.5, 9.6 and 93.4 $\mu g/m^3$, compared to observations of 7.5, 10.2, 11.6 and 53.1 $\mu g/m^3$. The fact that our model does not include the latest sulfate production mechanism – heterogeneous SO_2 oxidation at the surface of liquid aerosols, results in underestimation of sulfate. Model's underestimation at coastal sites originate from other factors such as models' flaws in simulating complex terrain geography and sea breeze as well as flaws in emission inventory in that region.

Consumption-based Life-cycle GHG emissions, water and land use

We utilize GHG emissions per gram of food from over 300 lifecycle assessment (LCA) studies covering the emissions from cradle to farm gate worldwide following the methodology in He et al (2019)(He *et al* 2019). We utilize water consumption per gram of food in China from water footprint database of Water Footprint Network(Anon n.d.). The network includes average water consumption for 352 plant-based and 106 animal-based products during the period of 1996-2005. We utilize land appropriation for plant-based food from the average of 1996-2005 field data for China provided by the Food and Agriculture Organization Statistics (FAOSTAT), while we estimate this indicator for animal-based food using conversion factors. Details of quantification for each of these three footprints are included in SI.

We introduce the Monte Carlo simulation to estimate the uncertainty of the impacts of diets on the environment due to uncertainties in climate, technologies, errors from various evaluations, etc. We run a simulation repeated for 10000 trials. In each trial, environmental impact factors of each food group are generated from the assumed distribution with a specific mean and standard deviation retrieved from the dataset of environmental impact factors. We assume log normal distributions for GHG emissions of each food group based on the distribution of factors of our collection of LCA studies, and retrieve the mean and standard deviation for each food group. For water consumption, we assume a normal distribution for each of the 352 plant-based and 106 animal-based products from the Water Footprint Network database, and a 15% of the means as the standard deviations for each product following a previous study [35]. For land appropriation, we assume normal distributions and 5% of the means from the FAOSTAT data as the standard deviations for each food group due to the observations of the flat change in productivity over time in FAOSTAT. The simulation is repeated for 100 trials. We then link these generated factors to the CHNS dataset to evaluate the individual dietary environmental impacts.

Opportunity cost of land

We quantify the land-use carbon emissions of agricultural production using life-cycle land-change carbon emission factors for crops and livestock products reported in Searchinger et al 2018(Searchinger *et al* 2018). This metric goes beyond life-cycle consumption-based GHGs and land use metrics since it considers the opportunity cost of one piece of land if it is used for producing one specific type of food compared to for producing another type of food/biofuel and compared to simply remaining as forests for global carbon storage purposes. It measures the carbon efficiency of the land analyzed compared to global-average carbon efficiency.

Health impacts of exposure to $PM_{2.5}$

We calculate premature mortality of four diseases due to exposure to $PM_{2.5}$ for adults (≥ 25 y old) in three dietary changes scenarios. The four diseases considered are chronic obstructive pulmonary disease (COPD), lung cancer, ischemic heart disease (IHD) and ischemic stroke. For each province in China, we calculate number of premature deaths of each disease based on

$$Mort_{i,P} = POP_P \times Mortbase_{i,P} \times (1 - \frac{1}{RR_{i,P}})$$

where $Mort_{i,P}$ is the number of premature mortality in province P from disease *i*; $POP_{j,P}$ is the number of exposed targeted population in province P considering adults (≥ 25 y old) in 2012 from 2013 China Statistical Yearbook(All China Marketing Research Co. Ltd 2014); $Mortbase_{i,P}$ is the baseline mortality rate in province P for disease *i* in 2012 from Global Burden of Disease study (Burnett *et al* 2014); $RR_{i,P}$ is the relative risk factor for one disease *i* adopted from (Burnett *et al* 2018). Relative risk factors for IHD and stroke are by age groups. There are 12 age groups considered, i.e. 25-29, 30-34, 35-39, 40-44, 45-49,50-54,55-59,60-64,65-69,70-74,75-79 and over 80 y old. Relative risk factors for lung cancer and COPD are the same for all people ≥ 25 y old.

Health impacts of diets

We quantify the public health implications of various dietary changes considering four dietary risk factors (including intakes of total red meat, vegetables, fruits and legumes) and six end-point diseases (including coronary heart disease (CHD), stroke, total cancers, Type II diabetes (T2DM), Colon and rectum cancers, and lung cancer). In detail, total red meat intake has been found positively associated with mortalities from stroke, T2DM, and colon and rectum cancers.

Vegetable and fruit intakes have been found negatively associated with mortalities from CHD, stroke, total cancer and lung cancer. Legume intake has been found negatively associated with CHD. We estimate the mortality attributable to dietary risk factors by calculating "population attributable fractions (PAFs)" following the formula below,

$$PAF = \frac{\int RR(x)P(x)dx - \int RR(x)P'dx}{\int RR(x)P(x)dx}$$

Where RR(x) is the relative risk of one specific end-point disease for risk factor level x, P(x) is the number of populations exposed to this risk level x in baseline scenario and that P'(x) is the number of people exposed to risk level x in one dietary change scenario. We utilize relative risk factors reported in Aune et al(Aune *et al* 2017), Kim et al(Kim Kyuwoong *et al* n.d.) and those used in Springmann et al(Springmann *et al* 2018)**.** Population data and baseline mortality for diseases of relevance are from Global Burden of Disease (Institute for Health Metrics and Evaluation (IHME) 2018). In cases of mortality of one disease attributable to multiple risk factors, we assume PAFs combine multiplicatively, i.e. $PAF_{TOT} = 1 - \prod_i (1 - PAF_i)$.

Uncertainty analysis

An alternative to nutritional surveys, i.e. macro statistics from FAO Food Balance Sheet (FBS), could also be used to estimate baseline Chinese diets. However, in this study, we decided to rely more on nutritional surveys to construct baseline Chinese diets for two reasons. The first reason is the quality of Chinese national statistics of agricultural production and supply has been criticized as of poor quality. The second reason is that nutritional survey data more realistically capture people's dietary preference variations depending on age, sex and region and more accurately estimate food waste.

FAO Food Balance Sheet's estimation of per capita food supply in China is estimated by subtracting non-human food usages, e.g. food for animal feed, food for export, food for seed, food for processing, etc from total agricultural production reported by Chinese State Statistics Bureau. China's official statistics, in nature, rely on household and enterprise surveys. In particular, a number of research pointed out severe mis-reporting issues(Peng *et al* 2019, Holz 2004). One research finds that increase of meat production reported by statistics during 1990s cannot be explained by stagnation of consumption and decline of livestock product export. Given lack of refrigerated storage facilities particularly in rural China, stock holdings are less likely to be able to explain the discrepancies(Fuller *et al* 2000). That research, through interviews, also finds that 'human errors' probably remains the most important source of data errors since the central government had set targets of agricultural production for regional government to achieve and frequently utilized agricultural production to assess the political performance of bureaucrats at regional and village levels(Fuller *et al* 2000). Another research echoes that China's official livestock production data series has been a level two to three times as high as its consumption data series since the year 1999 and official statistics over years fall short of various statistical tests, indicating poor data quality and consistency(Ma *et al* 2004). Other research finds that fishery output data suffer from similar issues(Li and Haomiao n.d.) and that township and village enterprise output statistics are also overstated(Zheng n.d.).

FAO FBS's data of per capita food supply includes food waste during food processing, cooking, and dine-out, and non-edible portion of food. Obtaining food intake has to involve using models to estimate food waste. A number of studies found that FAO data substantially overestimate individual's total calorie intake, e.g. a Chinese diet of over 3000kcal/day/capita from FAO FBS compared to those reported(Zhou *et al* 2003).

FAO data provides national level per capita food consumption, exclude substantial dietary variations among people in different age groups, of different sexes, at different income levels and with rural or urban backgrounds. Instead, in two of our dietary change scenarios, i.e. Chinese Nutritional Guideline diet and US diet, diets vary depending on people's sex, age, daily calorie intake and activity level. It is thus infeasible to model each person's dietary transitions to these two diets based on FAO data which captures only diets for the nation on average.

Macro statistics data (e.g. FAO Food Balance Sheet) of per capita food supply is the best suitable for cross-country comparison as they are estimated with relatively comparable methodologies using data reported

Similar to previous finds, we find that for China FAO food consumption data, compared to nutritional survey data mapped to nationwide population, overestimates consumption of livestock products but underestimates consumption of grains (Table S9)(Del Gobbo *et al* 2015).

A large body of literature utilize non-macro statistics data to inform diet, nutrition and human health, e.g. survey(Yang *et al* 2017, Miller *et al* 2016, Conrad *et al* 2017)and GENuS model that combines multiple data sources(Smith 2012).

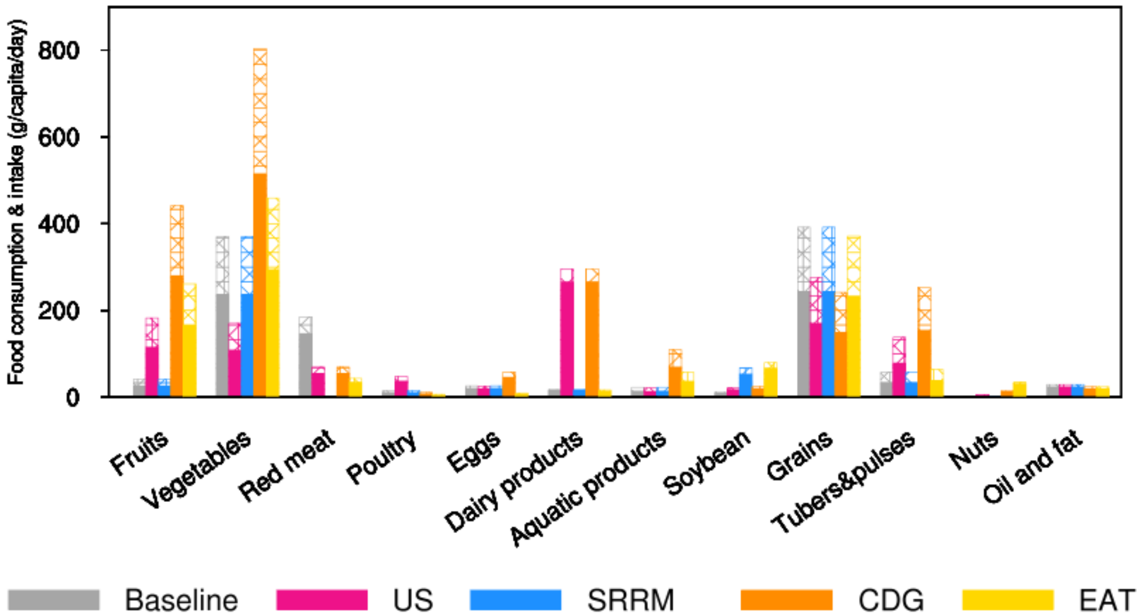

Figure S1. Food intake (solid bars; fresh food weight with non-edible portion included), food loss (bars with filled patterns; including food loss during production, processing and consumption) and food consumption (solid bars plus bars with filled patterns) by 'standardized' food type for *Baseline* Chinese diet (estimated from the latest China Health and Nutrition Survey (CHNS) for 2011(Carolina Population Center at University of North Carolina at Chapel Hill and Chinese Center for Disease Control and Prevention 2011)) and four future dietary change scenarios (*US*: typical 2011 US diet; *Soy Replaces Red Meat (SRRM)*: All red meat replaced with soy products; *Chinese Dietary Guideline (CDG)*: Recommendation of Chinese dietary guidelines; *Lancet-EAT Dietary Recommendations (EAT)*: healthy and sustainable diet recommended by Lancet EAT project). The standardization process (See Supplemental Appendix materials and Table S4) defines a reference food item within a specific food group with certain content of representative nutrients and then converts consumption of all food items within the same food group to consumption of this referenced 'standard' food item. The unit is g/capita/day. Definitions of vegetables and fruits are based on Chinese habits, e.g. cucumber, tomato, loofah and zucchini are categorized as vegetables; watermelon and muskmelon as fruits. All scenarios have daily per capita calorie intake of around 1800-1900 kcal.

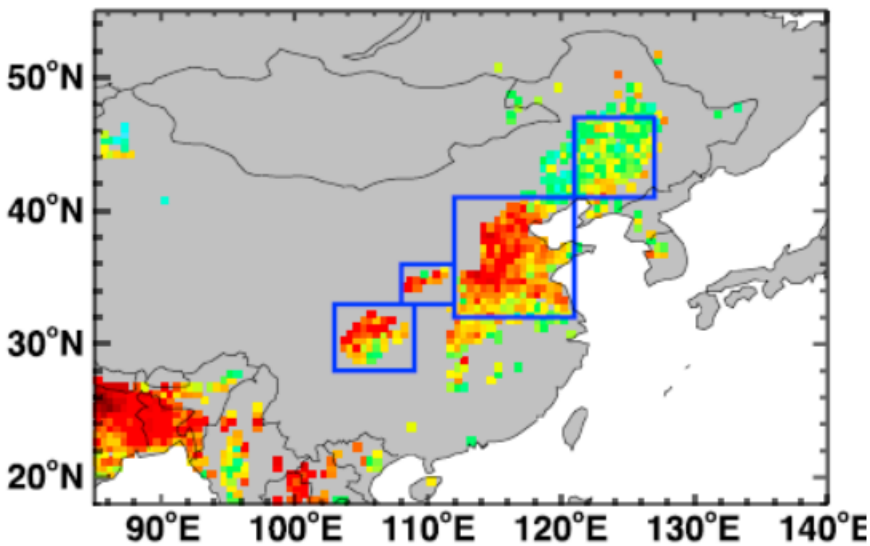

Figure S2 Pixels (the colored grid boxes within blue rectangles) where AIRS observations are extracted for comparison against model results.

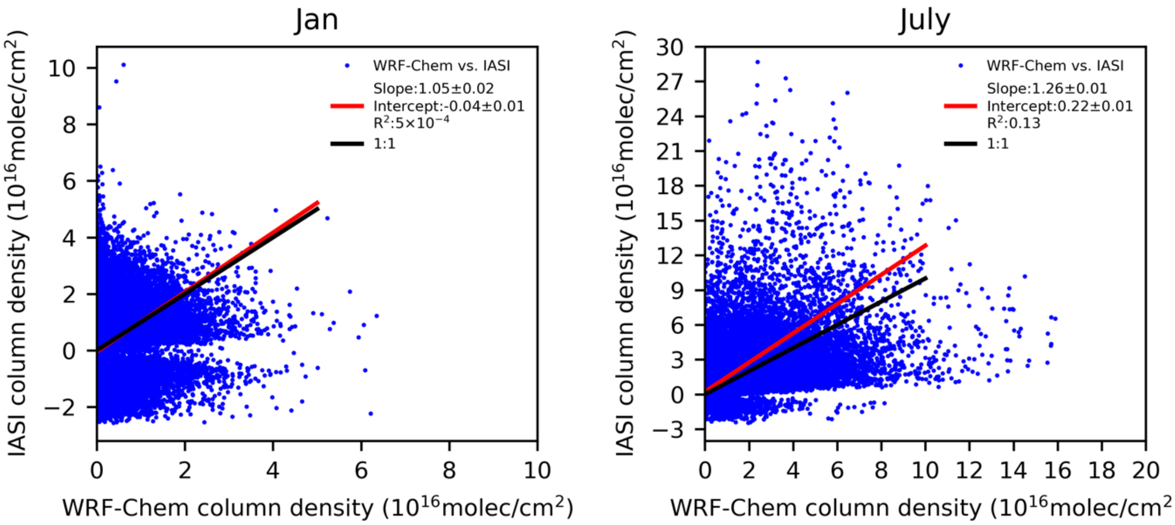

Figure S3: Daily NH_3 column density observed by IASI and simulated by WRF-chem with Zhang et al (2018) NH_3 (this study) for January (number of valid pixels: N=70063) and July (N=96557).

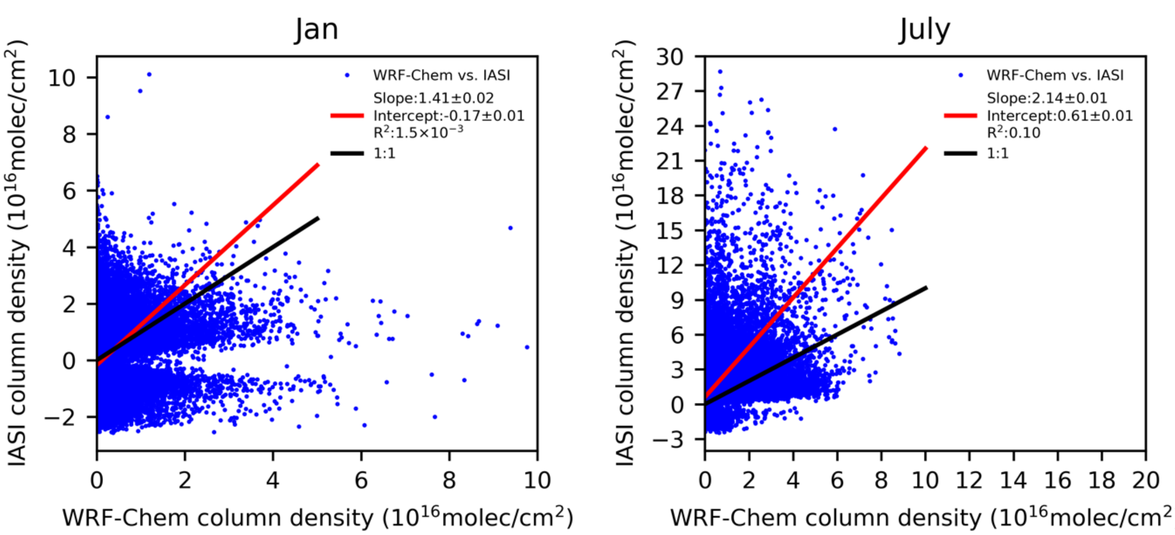

Figure S4: Daily NH_3 column density observed by IASI and simulated by WRF-chem with MIX NH_3 for January (number of valid pixels: N=70063) and July (N=96557).

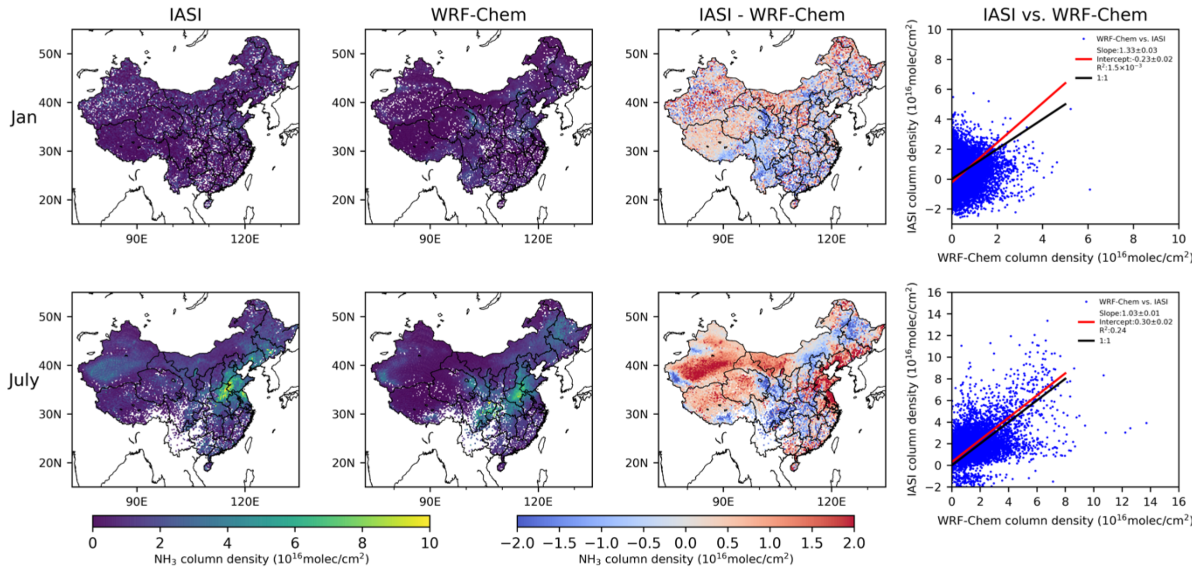

Figure S5: Monthly mean NH_3 column density observed by IASI (1st column) and simulated by WRF-chem with Zhang et al (2018) NH_3 (this study) (2nd column), as well as their difference (IASI-WRF-chem, 3rd column) for January (number of valid pixels: N=23506) and July (N=20767). An orthogonal distance regression (4th column) is also provided.

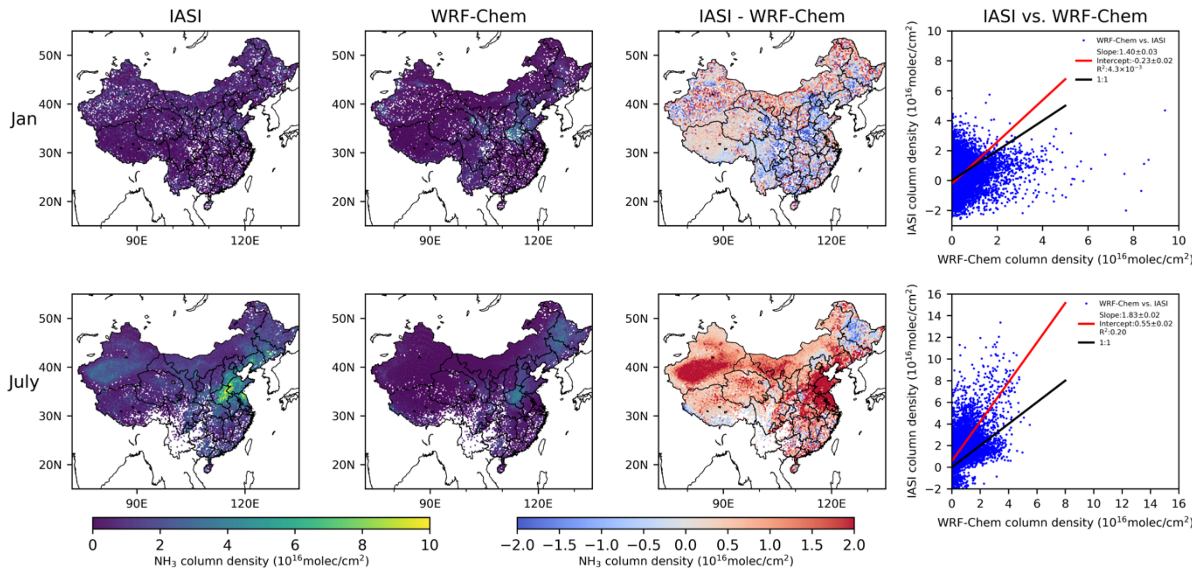

Figure S6: Monthly mean NH_3 column density observed by IASI (1st column) and simulated by WRF-chem with MIX NH_3 emissions (published in Kang et al (2016) (Kang *et al* 2016) and Huang et al (2017) (Huang *et al* 2012)) (2nd column), as well as their difference (IASI-WRF-chem, 3rd column) for January (number of valid pixels: N=23506) and July (N=20767). An orthogonal distance regression (4th column) is also provided.

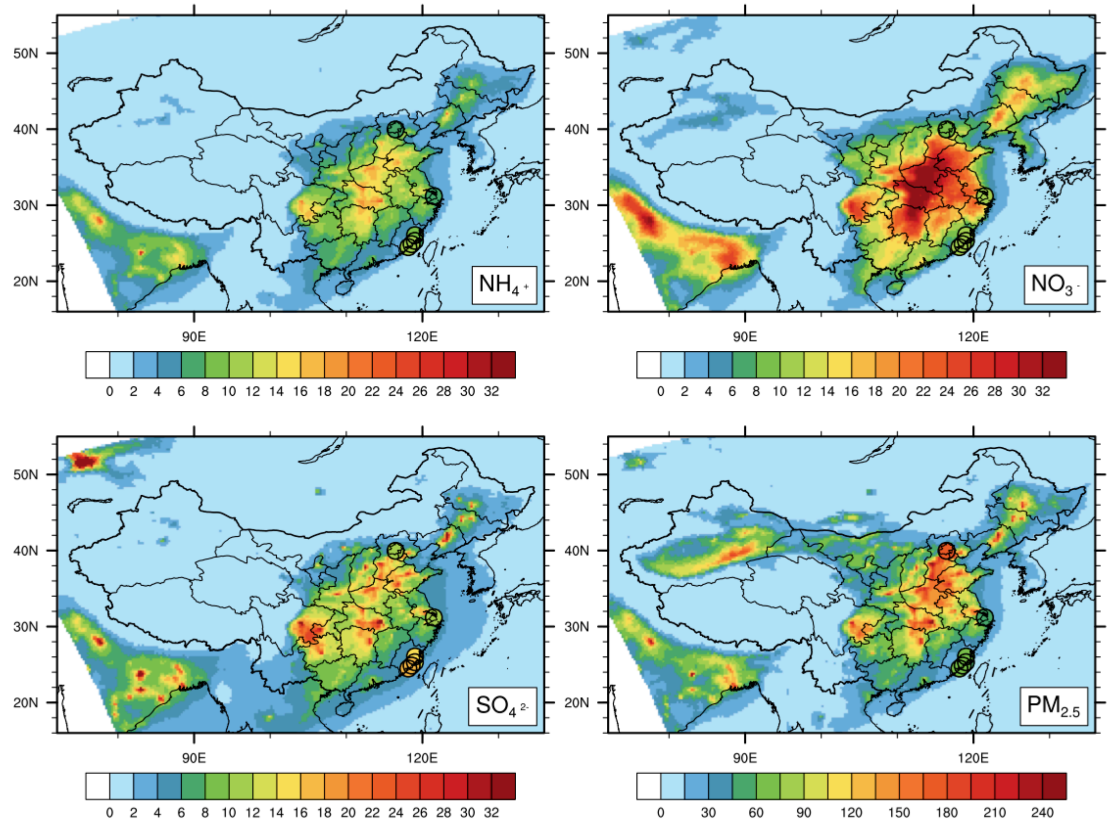

Figure S7 Evaluation of modeled ammonium (NH_4^+,) nitrate (NO_3^-), sulfate (SO_4^{2-}) and $PM_{2.5}$ concentrations (in μg/m^3) against observations during January of 2012. All observational and modeled values are monthly, except for four sites in the southeast where observation was conducted during the first 15 days of January so modeled concentrations are also averaged among that same period of time.

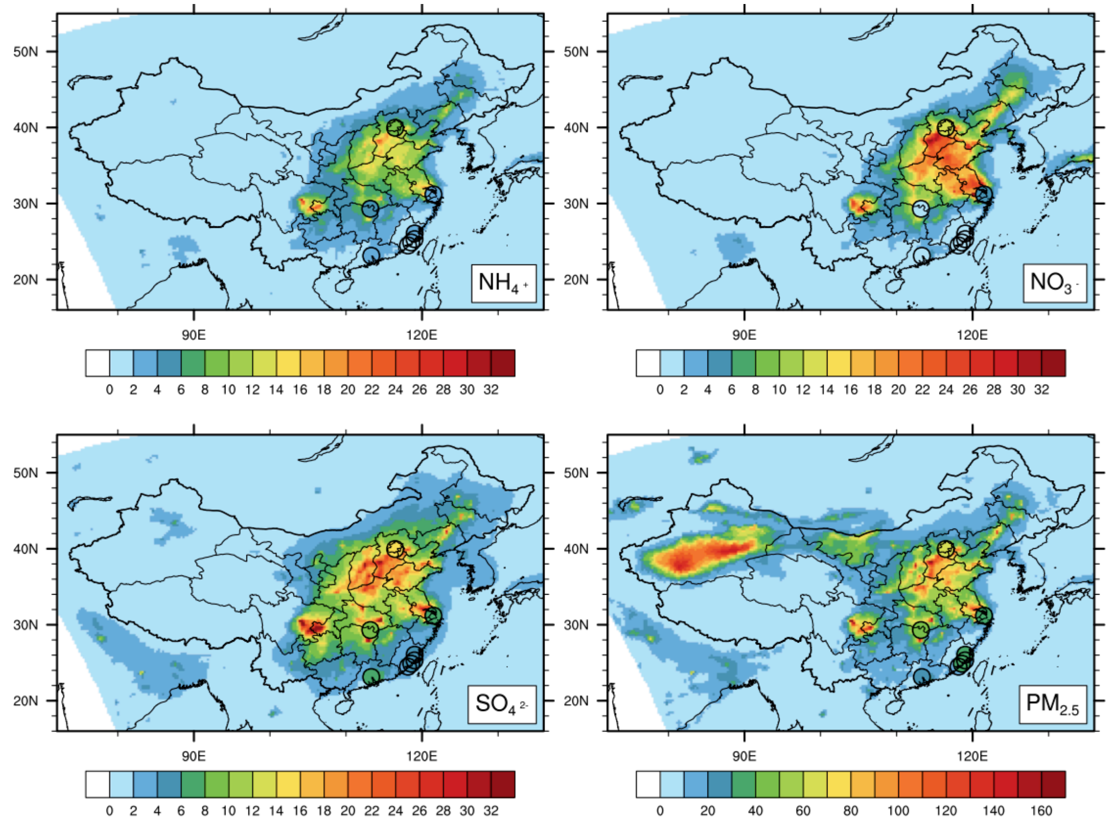

Figure S8 Evaluation of modeled ammonium (NH_4^+,) nitrate (NO_3^-), sulfate (SO_4^{2-}) and $PM_{2.5}$ concentrations (in μg/m^3) against observations during July of 2012. All observational and modeled values are monthly, except for four sites in the southeast where observation was conducted during the first 15 days of July, so modeled concentrations are also averaged among that same period of time.

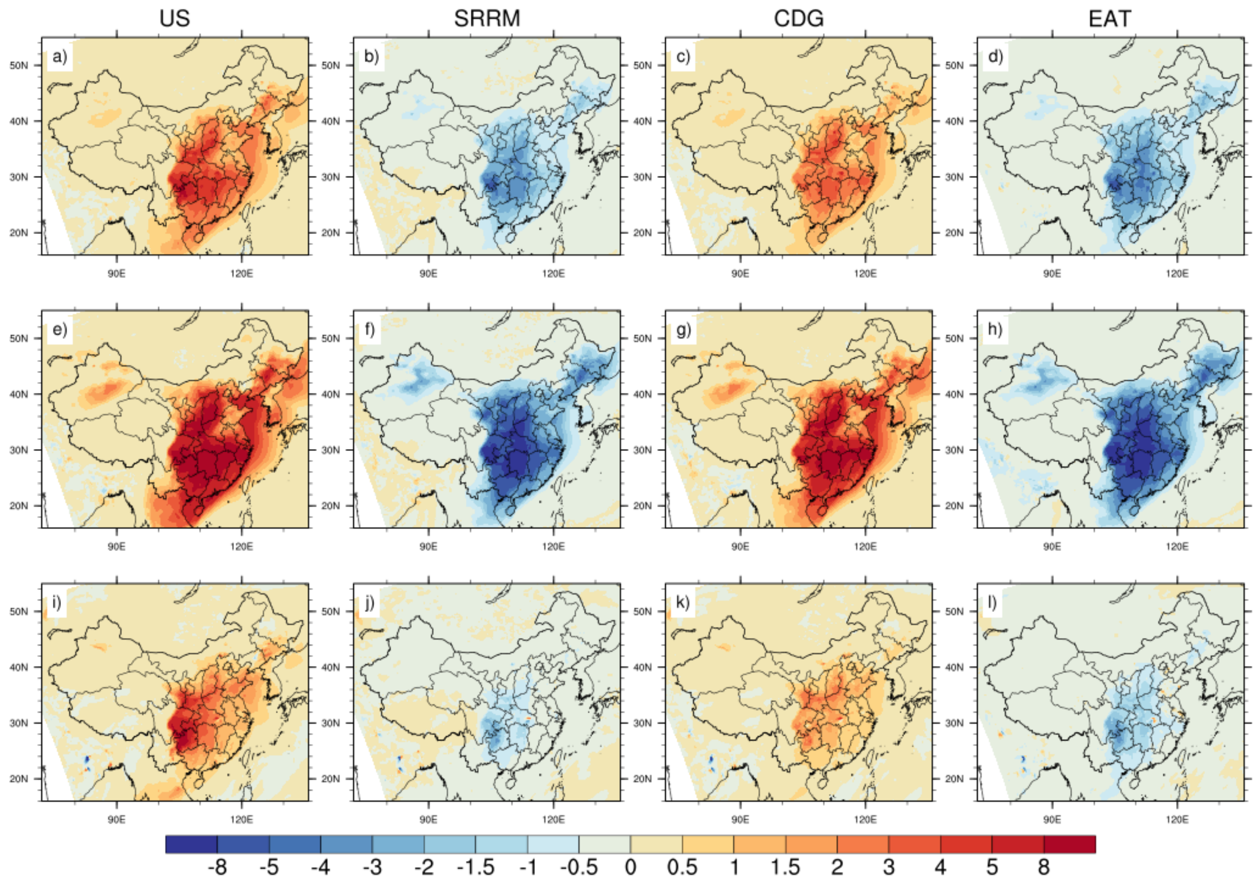

Figure S9 Changes in ammonium (1st row, a)-d)), nitrate (2nd row, e)-h)) and sulfate (3rd row, i)-l)) aerosol concentrations (in μg/m^3) in dietary change scenarios compared to baseline simulation in January 2012.

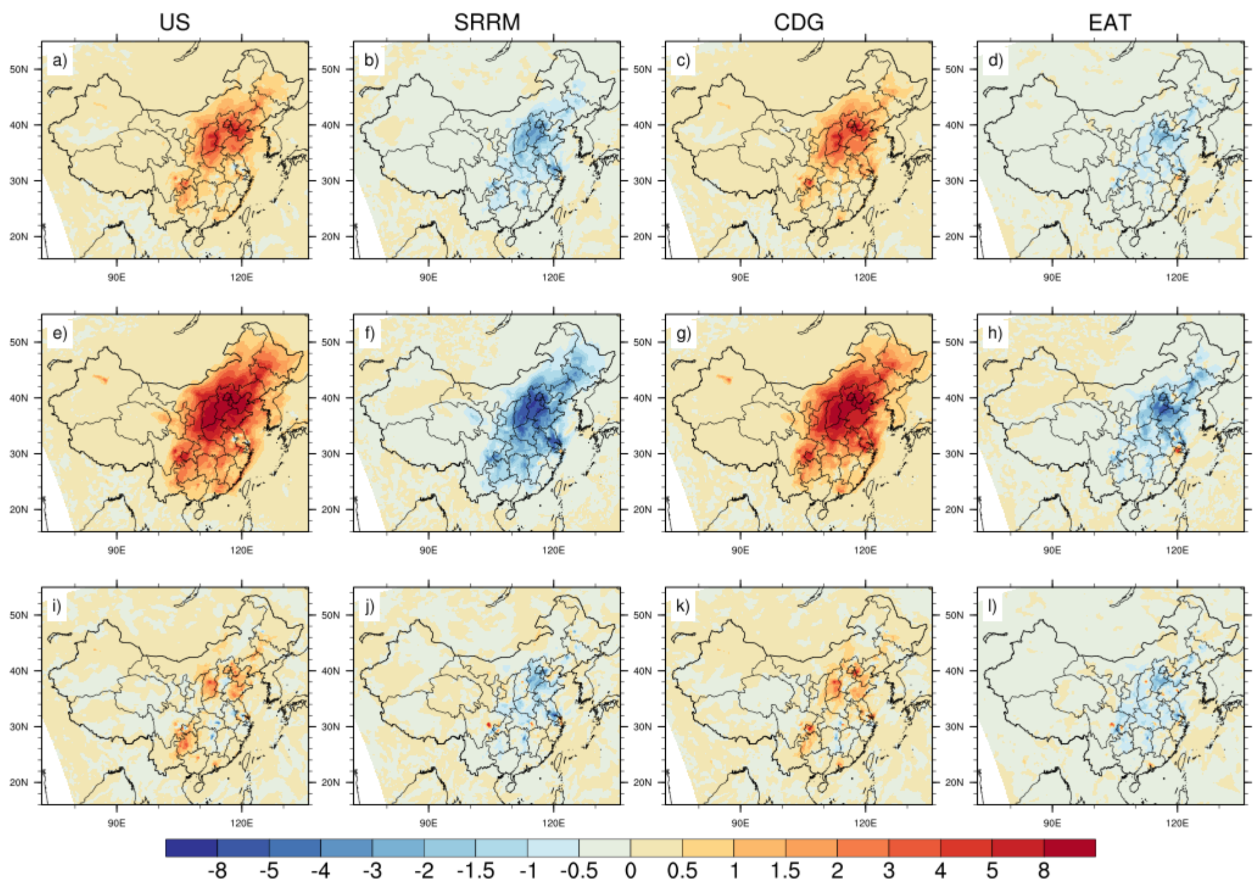

Figure S10 Changes in ammonium (1st row, a)-d)), nitrate (2nd row, e)-h)) and sulfate (3rd row, i)-l)) aerosol concentrations (in μg/m^3) in dietary change scenarios compared to baseline simulation in July 2012.

Table S1: *Baseline* food intake (in g/day/capita; net of food waste and non-edible portion) and consumption (in g/day/capita; including food waste and non-edible portion) in China in 2011 for each type of 'non-standardized' food product.

Food type (non-standardized)	Intake (g/day/capita)	Consumption (g/day/capita)

apple	12.4	20.3
banana	0.7	1.2
citrus	2.8	4.2
pear	1.5	2.4
grape	0.6	1
Other fruits	21.1	34.6
vegetables	264	396.7
Pig meat	75	93.6
Poultry meat	13.6	17.3
beef	3.8	4.7
Goat and sheep	0.4	0.5
Egg products	25.1	31.8
Dairy product	16	20.9
Aquatic products	23.3	34.8
rice	231	365.2
wheat	142.9	221
maize	6.5	9.7
potato	25.9	49.1
Sweet potato	1.5	2.7
Other coarse pulse	14.7	21.6
groundnut	1.1	1.2
Other nuts	0.5	0.6
Rapeseed oil	24.4	28.2

Soy products	21.6	21.6

Table S2: *Baseline* food intake (in g/day/capita; net of food waste and non-edible portion) and consumption (in g/day/capita; including food waste and non-edible portion) in China in 2011 for each type of 'standardized' food product. The standardization process converts food items under the same food group but with diverse nutrition composition to a pre-defined reference 'standard' food item. We follow the process elaborated in He et al (2019)(He *et al* 2018).

Food type (standardized)	Intake (g/day/capita)	Consumption (g/day/capita)
apple	9.7	15.4
banana	0.8	1.2
citrus	2.0	3.1
pear	1.1	1.7
grape	0.4	0.7
Other fruits	12.4	19.6
vegetables	237.5	370.4
Pig meat	143.2	180.7
Poultry meat	11.8	14.9
beef	3.4	4.3
Goat and sheep	0.4	0.5
Egg products	20.9	26.4
Dairy product	15.8	17.7
Aquatic products	14.8	23.0

rice	140.6	225.9
wheat	99.5	159.8
maize	3.8	6.2
potato	20.4	39.4
Sweet potato	1.5	3.0
Other coarse pulse	6.8	11.0
groundnut	0.7	0.9
Other nuts	0.1	0.1
Rapeseed oil	24.4	29.6
Soybean	9.4	11.5

Table S3 Ratios of food loss occurred during different stages of food production and consumption for various food groups.

Food groups	Agricultural production	Post-harvest handling & storage	Processing& packaging	Distribution	Consumption	Left*
cereals	2.0%	10.0%	10.0%	2.0%	20.0%	62.2%
roots & tubers	20.0%	7.0%	15.0%	9.0%	10.0%	51.8%
oilseeds & pulses	6.0%	3.0%	5.0%	1.0%	4.0%	82.3%
fruits & vegetables	10.0%	8.0%	2.0%	8.0%	15.0%	63.5%
meat	2.9%	0.6%	5.0%	6.0%	8.0%	79.3%

fish & seafood	15.0%	2.0%	6.0%	11.0%	8.0%	64.1%
milk	3.5%	1.0%	1.2%	0.5%	5.0%	89.2%
others	0.0%	0.0%	0.0%	0.0%	0.0%	100.0%

Left=(1-Loss_Agricultural production)(1-Loss_ Post-harvest and storage)*(1-Loss_Food processing and packaging)*(1-Loss_Food distribution)*(1-Loss_Food consumption)

Table S4 the key nutrient and its content of the representative food items defined in Chinese Dietary Guidelines.

Food groups	Standardized item	Key nutrient for standardizing	Content of key nutrient per item	Energy supply
Cereals and pulses		Carbohydrate	80g/100g	267-360kcal/100g
tubers	Potato	Carbohydrate	20g/100g	80-113kcal/100g
Livestock and poultry	Lean pork	Energy	140kcal/100g	140kcal/100g
Eggs and egg products	Chicken egg	Protein	14g/100g	130-200kcal/100g
Soybean and soybean products	Soybean	Protein	35g/100g	260-400kcal/100g
Dairy products	Uncondensed milk	Protein	3g/100g	22-55 kcal/100g
Aquatic products	Fish	Energy	100kcal/100g	100kcal/100g
Nuts	Sunflower seed	Fat	50g/100g	400-550kcal/100g
Fruits	Apple	Energy	50kcal/100g	50kcal/100g

Vegetables	Normalized only based on the edible part	15-35 kcal/100g

Table S5 China's national total food consumption (including non-edible portion and food waste) by food type in *Baseline* and in four dietary change scenarios as a ratio of baseline. The second column displays national total consumption (unit: gram) of various food products in 2011 representative of current diet. The third, fourth, fifth and sixth column displays the ratio of food consumption in dietary change scenarios with respect to baseline consumption respectively. China's national population of 1303366480 persons are used.

Food types	*Baseline* national consumption by food type (g/day)	The ratio of consumption in scenarios to consumption in *Baseline*			
		US	*SRRM*	*CDG*	*EAT*
apple	2.64E+10	3.44	1.00	8.53	5.06
banana	1.56E+09	18.02	1.00	28.11	14.56
citrus	5.49E+09	12.51	1.00	22.50	13.02
pear	3.15E+09	1.83	1.00	28.34	15.86
grape	1.29E+09	11.31	1.00	36.47	19.19
other fruits	4.51E+10	2.98	1.00	6.80	4.34
vegetables	5.17E+11	0.46	1.00	2.17	1.24
pig	1.22E+11	0.19	0.00	0.35	0.23

poultry for meat	2.25E+10	3.24	1.00	0.75	0.47
cattle for beef	6.19E+09	8.09	0.00	1.28	0.37
goat and sheep	6.70E+08	1.19	0.00	1.88	0.50
chicken for eggs	4.14E+10	0.96	1.00	2.21	0.38
cattle for milk	2.73E+10	16.66	1.00	16.74	0.96
aquatic products	4.53E+10	0.92	1.00	4.78	2.50
rice	4.76E+11	0.13	1.00	0.54	1.00
wheat	2.88E+11	1.26	1.00	0.49	0.85
maize	1.26E+10	7.26	1.00	6.66	1.61
potato	6.40E+10	2.21	1.00	3.32	0.76
sweet potato	3.56E+09	0.05	1.00	4.84	0.97
other coarse pulse	2.81E+10	4.10	1.00	6.52	2.26
other tubers	6.13E+09	1.27	1.00	7.68	1.37
groundnut	1.59E+09	5.54	1.00	6.75	16.51
other nuts	7.92E+08	14.42	1.00	85.22	193.05
rapeseed oil	3.67E+10	1.00	1.00	0.83	0.81
soy products*	2.81E+10	1.92	5.80	2.16	6.95
soybean**	1.50E+10	1.92	5.80	2.16	6.95

* include various products that are made of soybean, e.g. bean curd, tofu, etc.

** soybean content in soy products

Table S6 Food production in four dietary change scenarios as a ratio of baseline production. This ratio equals to food consumption in scenario as a ratio of baseline production for all food types except for animal feed crops.

Food types	The ratio of production in scenarios to production under *Baseline*			
	US	*SRRM*	*CDG*	*EAT*
apple	3.44	1.00	8.53	5.06
banana	18.02	1.00	28.11	14.56
citrus	12.51	1.00	22.50	13.02
pear	1.83	1.00	28.34	15.86
grape	11.31	1.00	36.47	19.19
other fruits	2.98	1.00	6.80	4.34
vegetables	0.55	0.93	1.93	0.41
pig	0.19	0.00	0.35	0.23
poultry for meat	3.24	1.00	0.75	0.47
cattle for beef	8.09	0.00	1.28	0.37
goat and sheep	1.19	0.00	1.88	0.50
chicken for eggs	0.96	1.00	2.21	0.38
cattle for milk	16.66	1.00	16.74	0.96

aquatic products	0.92	1.00	4.78	2.50
rice	0.28	0.93	0.58	0.94
wheat	1.19	0.81	0.50	0.72
maize	1.31	0.43	0.94	0.54
potato	2.21	1.00	3.32	0.76
sweet potato	0.05	1.00	4.84	0.97
other coarse pulse	4.10	1.00	6.52	2.26
other tubers	1.27	1.00	7.68	1.37
groundnut	5.54	1.00	6.75	16.51
other nuts	14.42	1.00	85.22	193.05
rapeseed oil	1.00	1.00	0.83	0.81
soy products	1.05	0.75	0.69	0.92

Table S7 Domestic food supply and food utilization (including animal feed, human food and others) for animal feed crops in China in 2011, data from FAO's Food Balance Sheet (FBS).

Crop	Supply	Utilization		
	Net import as a percentage of domestic production	Animal feed as a percentage of supply	Human food as a percentage of supply	Other utilizations as a percentage of supply
Rice	0.39%	8.6%	82.0%	9.3%
Wheat	2.09%	22.8%	75.1%	2.1%

Maize	2.84%	66.9%	5.3%	27.8%
Soybean	376.69%	73.6%	7.9%	18.4%
Vegetables	8%	9%	83%	8%

Table S8: Monthly mean NH_3 concentrations (ppbv) at 918hPa observed by AIRS and simulated by WRF-Chem and GEOS-Chem with NH_3 emissions by Zhang et al (2018) and with MIX NH_3 emissions (Kang et al (2016)(Kang *et al* 2016); Huang et al (2017) (Huang *et al* 2012)).

NH_3 concentration at 918hPa (ppbv) over AIRS mask		January of 2012	July of 2012
Satellite	AIRS observations before smoothed by using two-dimensional penalized least squares (Warner et al. (2017) (Warner *et al* 2016))	1.57	5.76
Model	WRF-Chem simulation using Zhang et al (2018) NH_3	0.62	6.35
	WRF-Chem simulation using MIX NH_3	0.96	2.98
	GEOS-Chem simulation using Zhang et al (2018) NH_3	0.53	4.61
	GEOS-Chem simulation using MIX NH_3	0.51	2.48

Table S9: Evaluation of modeled daily SNA concentrations in Beijing against observations provided by (Chen et al, 2015) in January and July of 2012.

Variables	Pairs of data	Obs (ug/m^3)	Model (ug/m^3)	Mean bias (ug/m^3)	R	IOA*
SO_4 (January)	29	10.27	10.94	0.66	0.69	0.78
NO_3 (January)	29	8.43	9.96	1.53	0.59	0.67
NH_4 (January)	29	7.05	7.12	0.08	0.68	0.81
SO_4 (July)	29	14.92	14.50	-0.42	0.25	0.52

NO_3 (July)	29	12.16	13.66	1.49	0.15	0.44
NH_4 (July)	29	8.60	9.41	0.81	0.19	0.51

*Index of agreement (IOA) was proposed by Willmott et al 1981 On the validation of models *Phys. Geogr. 2(2) 184-194*

Table S10: Evaluation of modeled hourly $PM_{2.5}$ concentrations in Beijing, Shanghai, Guangzhou and Chengdu against U.S. embassy observations in January and July of 2012.

$PM_{2.5}$	Pairs of data	Obs (ug/m^3)	Model (ug/m^3)	Mean bias (ug/m^3)	R	IOA[1]
Beijing (January)	670	118.92	112.07	-6.85	0.72	0.81
Shanghai (January)	740	64.44	62.28	-2.16	0.50	0.70
Guangzhou (January)	82	80.16	66.53	-13.62	0.29	0.49
Beijing (July)	688	80.65	70.16	-10.49	0.52	0.69
Shanghai (July)	737	26.73	29.27	2.54	0.81	0.89
Guangzhou (July)	740	29.46	22.97	-6.50	0.46	0.65
Chengdu (July)[2]	729	59.03	55.28	-3.75	0.70	0.84

[1]Index of agreement (IOA) was proposed by Willmott et al 1981 On the validation of models *Phys. Geogr. 2(2) 184-194*

[2] A comparison for Chengdu in January is not available due to too many invalid observations there.

Table S11 Consumption-based life-cycle GHG emissions associated with national food consumption per day by food type in *Baseline* and four dietary scenarios.

Food types	Life-cycle GHG emissions (kg CO_2-eq/day)	GHG emissions in scenario/GHG emissions in *Baseline*			
	Baseline	US	SRRM	CDG	EAT
apple	3.52E+06	3.44	1.00	8.53	4.13
banana	5.69E+05	18.52	1.00	27.61	16.26
citrus	7.59E+05	12.64	1.00	22.12	14.14
pear	4.19E+05	1.84	1.00	28.34	17.79
grape	1.72E+05	11.41	1.00	36.47	15.16
other fruits	6.82E+06	2.62	1.00	6.87	4.39
vegetables	3.38E+08	0.46	1.00	1.92	1.30
pig	6.34E+08	0.16	0.00	0.35	0.23
poultry for meat	8.54E+07	3.17	1.00	0.75	0.48
cattle for beef	1.19E+08	7.92	0.00	1.27	0.37
goat and sheep	7.55E+06	1.21	0.00	1.88	0.46
chicken for eggs	1.04E+08	0.95	1.00	2.21	0.38
cattle for milk	3.13E+07	20.40	1.00	16.74	0.96

aquatic products	1.56E+08	0.90	1.00	4.74	3.77
rice	7.87E+08	0.10	1.00	0.54	0.95
wheat	1.78E+08	1.39	1.00	0.50	0.83
maize	4.59E+06	5.89	1.00	6.66	1.62
potato	1.18E+07	2.22	1.00	3.32	0.84
sweet potato	652848	0.05	1.00	4.83	1.05
other coarse pulse	1.75E+07	3.59	1.00	6.59	2.08
other tubers	1.12E+06	1.30	1.00	7.68	1.07
groundnut	1.70E+06	5.97	1.00	6.75	15.22
other nuts	8.50E+05	61.88	1.00	85.21	188.11
rapeseed oil	1.52E+08	1.00	1.00	0.83	0.75
soy products	1.43E+07	1.37	5.05	2.17	8.58

Table S12 Consumption-based life-cycle total water footprint associated with national food consumption per day by food type in *Baseline* and four dietary scenarios.

Food types	Total Water Footprint (TWF) (m^3/day)	Ratio of TWF in scenarios to TWF under Baseline			
	Baseline	US	SRRM	CDG	EAT
apple	2.18E+07	3.44	1.00	8.53	4.66

banana	7.96E+05	18.52	1.00	28.10	13.88
citrus	4.55E+06	12.64	1.00	22.68	14.35
pear	2.85E+06	1.84	1.00	28.34	16.49
grape	4.62E+05	11.41	1.00	36.56	19.18
other fruits	4.91E+07	2.62	1.00	7.00	2.70
vegetables	2.75E+08	0.46	1.00	1.93	3.27
pig	5.79E+08	0.16	0.00	0.35	0.22
poultry for meat	5.45E+07	3.17	1.00	0.87	0.47
cattle for beef	6.60E+07	7.92	0.00	1.26	0.37
goat and sheep	3.25E+06	1.21	0.00	1.78	0.46
chicken for eggs	1.01E+08	0.95	1.00	2.21	0.37
cattle for milk	3.05E+07	20.40	1.00	16.76	1.03
aquatic products	8.57E+07	0.90	1.00	4.84	2.43
rice	5.07E+08	0.10	1.00	0.54	1.11
wheat	3.64E+08	1.39	1.00	0.49	0.86
maize	1.26E+07	5.89	1.00	6.69	1.56
potato	1.54E+07	2.22	1.00	3.32	0.95
sweet potato	937545.7	0.05	1.00	4.84	0.83

other coarse pulse	3.76E+07	3.59	1.00	6.54	2.15
other tubers	1.75E+06	1.30	1.00	7.64	1.29
groundnut	2.80E+06	5.97	1.00	6.75	16.17
other nuts	1.48E+06	61.88	1.00	85.37	210.64
rapeseed oil	1.56E+08	1.00	1.00	0.83	0.85
soy products	9.69E+07	1.37	6.17	2.16	11.20

Table S13 Consumption-based land appropriation associated with national food consumption per day by food type in *Baseline* and four dietary scenarios.

Food types	Land Appropriation (m^2/day)				
	Baseline	US	SRRM	CDG	EAT
apple	2.85E+07	3.44	1.00	8.53	4.92
banana	7.61E+05	18.52	1.00	28.11	14.83
citrus	7.86E+06	12.64	1.00	22.73	14.49
pear	3.82E+06	1.84	1.00	28.34	16.74
grape	1.02E+06	11.41	1.00	36.45	19.54
other fruits	6.62E+07	2.62	1.00	6.97	2.78
vegetables	4.70E+08	0.46	1.00	1.92	1.96
pig	8.79E+08	0.16	0.00	0.35	0.24
poultry for meat	2.89E+08	3.17	1.00	0.75	0.48
cattle for beef	5.76E+07	7.92	0.00	1.30	0.37

goat and sheep	7.60E+06	1.21	0.00	2.04	0.64
chicken for eggs	4.13E+08	0.95	1.00	2.21	0.38
cattle for milk	9.13E+07	20.40	1.00	16.72	0.99
aquatic products	2.76E+08	0.90	1.00	4.83	2.74
rice	5.82E+08	0.10	1.00	0.55	1.03
wheat	7.54E+08	1.39	1.00	0.49	0.83
maize	2.15E+07	5.89	1.00	6.62	1.59
potato	4.29E+07	2.22	1.00	3.32	0.74
sweet potato	2060508	0.05	1.00	4.84	0.91
other coarse pulse	1.16E+08	3.59	1.00	6.56	2.22
other tubers	3.61E+06	1.30	1.00	7.79	1.43
groundnut	4.70E+06	5.97	1.00	6.75	16.35
other nuts	2.08E+06	61.88	1.00	84.55	212.99
rapeseed oil	1.44E+08	1.00	1.00	0.82	1.03
soy products	1.07E+08	1.37	5.48	2.17	6.94

Table S14 Consumption-based land use change carbon emissions associated with national food consumption per day (in kton CO_2-eq/day) by food type in *Baseline* and four dietary scenarios.

Food types	Baseline	US	SRRM	CDG	EAT

fruits	9.0E+01	4.6E+02	9.0E+01	9.1E+02	5.5E+02
vegettables	3.7E+02	1.7E+02	3.7E+02	7.9E+02	4.5E+02
pork&goat	1.9E+03	4.8E+02	0.0E+00	8.5E+02	4.7E+02
poultry&aquatic	2.4E+02	7.8E+02	2.4E+02	1.8E+02	1.1E+02
dairy&eggs	6.1E+02	3.2E+03	6.1E+02	3.8E+03	3.3E+02
soy	1.6E+02	3.2E+02	9.5E+02	3.6E+02	1.1E+03
grains&pulses	2.2E+03	2.7E+03	2.2E+03	3.7E+03	2.7E+03
nuts&others	1.8E+02	2.5E+02	1.8E+02	2.4E+02	3.7E+02
beef	8.9E+02	7.2E+03	0.0E+00	1.1E+03	3.3E+02
other tubers	5.2E+01	9.8E+01	5.2E+01	2.1E+02	3.5E+01
aquatic	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
Sum	6.68E+03	1.57E+04	4.71E+03	1.22E+04	6.45E+03

Table S15 China's baseline food intake (g/day/capita) in the year 2011 estimated by macro-level statistics (1^{st} column, IMPACT model estimations using a harmonized set of FAO's Food Balance Sheet data to exclude food waste and non-edible portion of food) and by dietary surveys (mapping food intake of a sampled population of ~8000 people to nationwide (2^{nd} column, ***Baseline*** used in this study, excluding food waste and non-edible portion).

Food type	Food intake derived from national statistics (net of food waste	Food intake derived from nutritional surveys (net of food waste and non-edible part, population

	and non-edible part)	re-weighted, adult)
apple	43.2	12.4
banana	3.6	0.7
citrus	8.0	2.8
pear	4.7	1.5
grape	2.0	0.6
Other fruits	53.6	21.1
vegetables	556.0	264
Pig meat	94.0	75
Poultry meat	30.0	13.6
beef	15.0	3.8
Goat and sheep	9.0	0.4
Egg products	47.0	25.1
Dairy product	89.0	16.0
Aquatic products	40.0	23.3
rice	162.0	231.0
wheat	113.0	142.9
maize	12.0	6.5
potato	145.3	25.9

Sweet potato	10.7	1.47
Other coarse pulse	4.0	14.7
groundnut	13.9	1.1
Other nuts	1.1	0.45
Rapeseed oil	17.0	24.4
Soy products	34.7	40.0

Table S16 Relative risk (RR) factors used for public health analysis in this research. RRs are from

		Risk factors				
End-point diseases		red meat intake	Vegetable intake	Fruit intake	Legume intake	Exposure to PM2.5
	mean		0.84	0.95	0.77	RR varies by age group
	low		0.8	0.92	0.65	RR varies by age group
IHD	high		0.88	0.99	0.9	RR varies by age group
Stroke	mean	1.1	0.95	0.77		RR varies by age group

	low	1.05	0.87	0.7		RR varies by age group
	high	1.15	1.03	0.84		RR varies by age group
Other cancers	mean		0.93	0.94		
	low		0.91	0.91		
	high		0.95	0.97		
T2DM	mean	1.15				
	low	1.07				
	high	1.24				
Colon and rectum cancers	mean	1.14	0.93	0.94		
	low	1.04	0.91	0.91		
	high	1.24	0.95	0.97		
COPD						RR varies by age group
Lung cancer	mean		0.93	0.94		RR varies by age group
	low		0.91	0.91		RR varies by age group
	high		0.95	0.97		RR varies by age group

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