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Driving forces and impacts of food system nitrogen flows in China, 1990 to 2012

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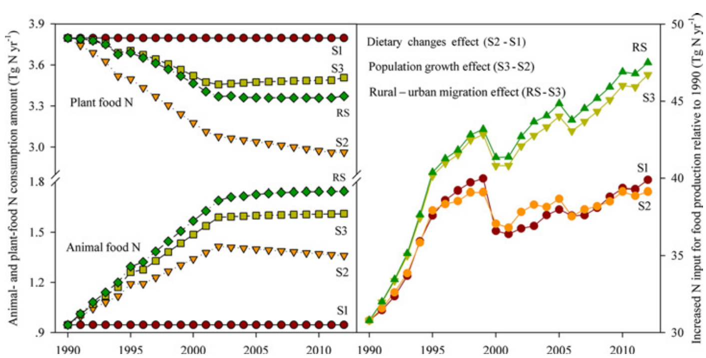
HIGHLIGHTS

- Urban-rural difference in per capita plant- and animal-food N (AN) was studied.
- One demands 0.3–0.5 kg more AN yr^{-1} when he moved to cities from a rural area.
- 17% increased AN was caused by rural-urban migration between 1990 and 2012.
- Dietary changes no significant contribution to N increase over the past two decades.
- Urban food-sourced N losses to soil and water bodies are 3 folds of rural area.

GRAPHICAL ABSTRACT



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ABSTRACT

Food nitrogen (N), which includes animal-food (AN) and plant-food N (PN), has been driven by population growth (PG), dietary changes associated with income growth (DC) and rural-urban migration (M) over the past three decades, and these changes combined with their N cost, have caused some effect on N use in China's food system. Although there is an increasing literature on food N and its environmental impacts in China, the relative magnitude of these driving forces are not well understood. Here we first quantify the differences in per capita AN and PN consumption in urban and rural areas and their impacts on N input to the food system during 1990–2012, and then quantify the relative contributions of DC, PG and M in the overall N change during this period. Our results show that a resident registered as living in city required 0.5 kg more AN yr^{-1} and 0.5 kg less PN yr^{-1} than one living in a rural area, in 2012. DC, PG and M accounted for 52%, 31% and 17% of the total AN increase, respectively. These three factors caused 46% of the increased N use for food production over the past two decades. Another 54% was mainly caused by the declining in N use efficiencies of the food system. Food-sourced N loss intensity in urban and rural areas were 502 and 162 kg N hm^{-2} in 2012, a three-fold difference due to the increasing amount and a linear rural-urban flow of N input, and inadequate N recovery via solid waste and wastewater treatment in cities. Our study highlights China is facing higher risks of environmental N pollution with urbanization, because of the high demand for AN and higher food-sourced N loss intensity in urban than in rural areas.

1. Introduction

Nitrogen (N) is an essential and irreplaceable element, which can sustain food production and global population after it is converted

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into reactive N (Nr) species (Galloway et al., 2004). But the imbalances and overuses of N have caused large Nr losses to the environment, resulting in a cascade of negative effects on natural resources and environmental quality, including soil acidification, eutrophication of aquatic systems, coastal dead zones, biodiversity loss, stratospheric ozone depletion, and an enhanced greenhouse effect as well as human health effects, all of which have become the focus of research in areas with large populations and intensive agriculture (Liu and Diamond, 2005; Schlesinger, 2009; Robertson and Vitousek, 2009; Sutton et al., 2011). An understanding of the correlation between human socioeconomic activities and N flows at local, regional, national, and global scales is essential to improving the N use efficiency (NUE) and to balancing food production with the aim of minimizing damages to environmental systems (EPA, 2012; Cui et al., 2013).

At the global scale, the Nr rate has already transgressed the planetary boundaries and seriously affected global sustainability (Steffen et al., 2015; Liu et al., 2015), and this trend may continue in the coming decades, driven by the rapidly increasing world human population and the concomitant increase in prosperity (Tilman et al., 2011). The human diet is transitioning to higher animal-food consumption, especially in emerging economies (Tilman et al., 2001), and the share of animal-food N (AN) to total food N is higher in urban diets than in rural ones (FAO, 2013; Tilman and Clark, 2014). These dietary changes are expected to continue, as some 2 billion more people move into cities in the wave of global urbanization, especially in China, India, southeast Asia and Africa (UN-Habitat, 2010), and the shift from traditional diets to those higher in AN will cause serious health and environmental problems (Tilman and Clark, 2014).

As the largest Nr producer and consumer, the most populous country in the world, and the one undergoing the greatest urbanization during the last few decades (Yang, 2013; X.M. Bai et al., 2014), China is an interesting case that demonstrates how population growth, dietary changes, rural-urban migration, and N management practices can affect long-term trends in N use and loss from the food production and consumption system, in an emerging economy. In crop production, large quantities of synthetic-fertilizer N have been used to increase crop yields, with the total amount reaching 30.6 Tg by 2012, up from 17.5 Tg in 1990 (NBSC, 2013). If current trends continue, total Nr input to China by 2050 will be more than double the 48.0 Tg in 2010 (Gu et al., 2015), accounting for 33–40% of the global increase in fertilizer N demand, from 2005 to 2050, driven by the increase in food demand (Tilman et al., 2011).

Furthermore, the continuing Chinese dietary shift to animal foods, which has driven the rapid intensification and specialization of animal-food production, has significantly increased the demand for animal feed (Chen et al., 2014), and caused serious environmental pollution because of the poor management of animal wastes (Ma et al., 2012; Hou et al., 2014). All these pressures, taken together, will increase the total demand for AN, and consequently the amount of N imported into the Chinese food system.

Rapid urbanization is indeed the most critical component of the increase in agricultural N demand, in China. Most of the high-AN food is consumed in urban settings (Lin et al., 2013; Hou et al., 2014). Yet the higher the urbanization rate, the lower the efficiencies of nutrient recycling, because an urban ecosystem is a combination of high nutrient density fluxes and disrupted N cycling (Grimm et al., 2008; Lin et al., 2013; Ma et al., 2014). The blocked recycling of N from cities to rural areas will cause large amounts of N to be stranded in urban environments after consumption (Marzluff et al., 2008). Thus, the rapid urbanization in developing countries will cause even more severe resource and environmental problems than have been previously suffered in developed countries (Lin and Grimm, 2015). One limitation to assessing and solving Nr-driven problems is the scarcity of detailed analyses on N stock and N losses to the environment in both urban and rural settings in China.A substantial literature has been undertaken to analyze the dietary transition per capita (Wei et al., 2008; Cui et al., 2016), quantify the budgets of N at the national scale in China (Ti et al., 2011; Cui et al., 2013; Gu et al., 2015) and the inputs or flows of N in the Chinese food chain system (Ma et al., 2012; Hou et al., 2014; Cui et al., 2016). However, these researches have mainly focused on analyzing the historical trends in per capita food consumption (Cui et al., 2016), and on quantifying N balances, losses at the national scale (Cui et al., 2013; Gu et al., 2015; Cui et al., 2016), nutrient use efficiency, and nutrient cycling in the different food chain sectors: crop and animal production, food processing, and food consumption (Ma et al., 2012). There is a lack of research quantifying the relative impacts of the specific drivers behind the changes in budgets and flows of N, or any comparison of N losses to the environment from food systems in urban and rural areas during the period in which China has experienced rapid urbanization. Although Hou et al. (2014) found that the food N consumption in China's urban settings increased about fivefold from 1980 to 2010, while decreasing in rural settings after the 1990s, they neglected to further analyze the impacts of population growth, dietary changes and the changes in food N consumption in urban and rural areas on the N input to the Chinese food system, and ignored the effect of rural-urban migration on food N consumption and N input.

This study aimed (i) to analyze the historical trends in per capita food N consumption and the differences in per capita food N consumption between urban and rural areas from 1990 to 2012; (ii) to estimate the changes in AN and PN consumption, driven by population growth, dietary changes associate with income growth and rural-urban migration; (iii) to quantify the contributions of the above key drivers to inputs of N through the food system; and (iv) to compare the intensities of food-sourced N losses to the environment in urban and rural areas.

2. Methods and data

2.1. Description of the Chinese food system

The material flow analysis approach was adapted for quantifying the flows of N in the Chinese food system, which is defined as the entire food production–consumption chain, including the recycling of wastes from food production and consumption. The system boundaries followed the geographic boundaries of China, and excluded Taiwan, Hong Kong and Macao because of limited data availability. In this study, the food system was divided into five categories (Fig. 1): crop production, animal-food production, food processing, household consumption (including both urban and rural households), and waste disposal. The crop-production category includes 19 crops (rice, wheat, maize, millet, sorghum, other cereals, beans, potatoes, peanuts, canola, sesame, cotton, flax, sugarcane, sugar beets, tobacco, fruit trees, vegetables, and green fodder). These crops accounted for >95% of the total area sown in China (NBSC, 2013). The animal-production category included 12 animals (hogs, sows, dairy cattle, beef cattle, draft cattle, laying hens, broilers, sheep, horses, mules, donkeys, and rabbits); fish and seafood were viewed as N input from other systems to the food system. The food-processing category included storage, transportation, processing, packaging, and retail sectors. We supposed that the difference between food N supply and final consumption by residents was the N stock in some stages and N loss in food processing, because we lacked information on Nr loss data at these sector levels (Ma et al., 2010). The household-consumption category included rural and urban household diets. The division between rural and urban households was based on national statistical information (NBSC, 1991–2013), and we further distinguished migrants who were living in cities but not registered in the urban population, and quantified rural-urban migration (Fig. S1, see SI for details). Here the imported and exported foods were included in the calculation of household food consumption. The waste-disposal category included human and animal excreta, food processing wastes, kitchen wastes, crop residues and sludge. Distinctions were made among: new N imported from outside the food system via chemical fertilizers, biological N_2 fixation (BNF), atmospheric deposition, irrigation,

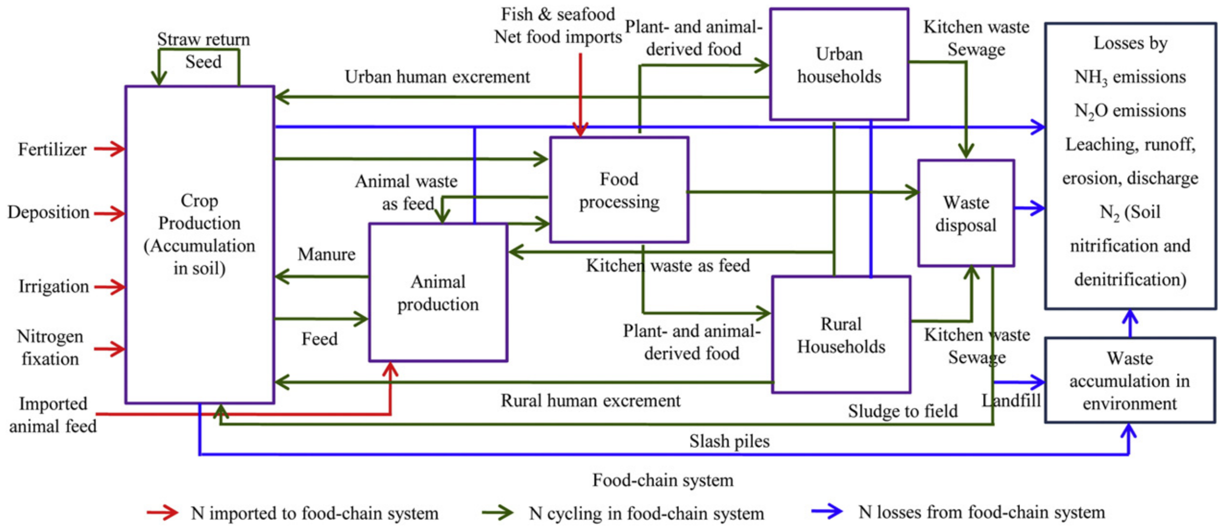

Fig. 1. Detailed food-chain system of China, including crop production, animal production, food processing, household consumption (divided into urban and rural areas) and waste disposal. Arrows in the food-chain system represent N flows. The dotted box on the right-hand side represents N losses to various environments. Net import is the difference between import and export.

imported animal feed, fish and seafood from the aquatic system, net imported food from international trade, and N recycled inside the food system, such as grain feed, manure, crop residues, slash and food wastes, and sludge applied to fields.

We further constructed a simply coupled urban and rural N flow model of the Chinese food system based on the principles of that system, including rural, urban and environmental systems (atmosphere, soil and water bodies) (Fig. S2), in order to quantify and compare the food-sourced N losses to the environment, in both urban and rural areas (see SI for details).

Mass balance calculations as a basic principle (inputs = outputs + accumulations) were adopted for the calculations of N input, output and accumulations in different sectors of the entire food system, if no data were available for calculating them directly (Ma et al., 2012; Gu et al., 2015; Cui et al., 2016): for example, N accumulation in cropland soil and feed N imported to the animal-food production system (see SI for details).

2.2. Data collection

The basic data used in this study, such as population, fertilizer usage, crop yields and plant area, livestock production, and urbanization rates, were mainly taken from China's statistical yearbooks and bulletins (NBSC, 1991–2013); the import and export data were separately collected from the China Customs Statistical Yearbook (CCA, 1991–1995) and the Chinese Agricultural Statistical Yearbook (MOA, 1996–2013). The second category of data is coefficients used for the calculation of N fluxes, such as the ratio of crop straw to grain, seeding rate, relative partitioning of harvested main crop products over its various uses, and the relative partitioning of foodstuff over its various uses in different times, the contents of N in harvested products and foods, the rate of BNF, denitrification, and livestock excrement, and sewage treatment rates in urban and rural areas (Tables S1–S15); such information was mainly obtained from the literature. Atmospheric deposition was regarded as new N imported into the food system. Previous studies have determined that about one fourth of the nitrogen oxides (NO_x) and ammonia (NH_3) emissions are deposited on croplands (Liu and Zhang, 2009; Shi et al., 2015). In this study, we used this parameter and the data on NO_x from multiple pollutants in the WIOD database (Timmer et al., 2015), and NH_3 emissions calculated in our study, to calculate the amount of N deposited onto cropland in a year.

2.3. Per capita food and food N consumption

Per capita habitual food intake figures for urban and rural residents in 1982, 1992, 2002 and 2012 were taken from Zhai et al. (2005) and NHFPC (2015): these include rice, flour, other cereals, beans, potatoes, soy products, vegetable oils, vegetables, fruit, pork, beef and mutton, poultry, milk, eggs and fish (16 categories) (Table S16). We estimated per capita habitual food intake for the non-sampling years during the period 1990–2012, by linear interpolation between every two adjacent intervals of the surveys. We then estimated per capita food consumption by urban and rural residents combined with the ratio of kitchen wastes (Table S11), and calculated per capita food N consumption by multiplying the amounts of different foods by their N contents (Tables S7–S8). In China, where large numbers of migrants have moved to cities from rural areas, in search of jobs and to make a better life for themselves, many of them have registered as living in cities and become true urbanites, diets difference between natives and new immigrants in urban areas may disappear with the rural-urban migration, whereas others have only lived in cities but never registered there, and their dietary patterns do not immediately change to the urban type, and in fact there are many residents whose dietary habits fall somewhere between these two types, rural and urban. For convenience of calculations, the food N consumption by migrants in cities was assumed as the average of the two patterns, according to the principle of the most likely value being the average of the minimum and maximum values (Huang et al., 2017).

2.4. Calculations of the N cost of Chinese food

The N cost is defined as the ratio between total new N input into the food system and the N in foodstuffs (Bleken and Bakken, 1997), and it can be interpreted as the amount (in kg) of new N input to the food system for the delivery of 1.0 kg N in the food entering households (Ma et al., 2012). In this study, N costs of PN (NC_{PN}, kg N kg^{-1} PN), AN (NC_{AN}, kg N kg^{-1} AN) and total food N (NC_f, kg N kg^{-1} food N) were used as indicators for analyzing the new N imported into the Chinese food system, driven by the consumed AN and PN. They were estimated as follows:

$$NC_f = N_{(\text{fertilizers+BNF+irrigation+deposition+imported animal feed+fish\&seafood+net imported N} - \text{N other use})} / (PN_{consumed} + AN_{consumed} + PN_{exported} + AN_{exported} - PN_{imported} - AN_{imported}) \quad (1)$$

$$NC_{PN} = (N_{grain\ for\ food} \times NC_g)/(PN_{consumed} + PN_{exported} - PN_{imported}) \quad (2)$$

$$NC_{AN} = (N_{grain\ as\ feed} \times NC_g + N_{imported\ animal\ feed} + N_{fish\&seafood})/(AN_{consumed} + AN_{exported} - AN_{imported}) \quad (3)$$

where $N_{(fertilizers\ +\ BNF\ +\ irrigation\ +\ deposition\ +\ imported\ animal\ feed\ +\ fish\ \&\ seafood\ +\ net\ imported\ N\ -\ N\ other\ use)}$ represents total new N input to the food system minus the N used for other uses; $N_{grain\ for\ food}$ represents the grain used for plant food production; NC_g is the N cost of the harvested grain N (defined as the amount of new N input to the crop production system for the delivery of 1.0 kg N in the grain), calculated using the new N input to crop production system divided by the harvested N in grain, because harvested grain was used for rations, feed and other uses (Table S6). $PN_{consumed}$, $AN_{consumed}$, $PN_{exported}$, $AN_{exported}$, $PN_{imported}$, and $AN_{imported}$ represent the consumed, exported, and imported PN and AN, respectively.

2.5. Driving forces of the national food N consumption and N input to the Chinese food system

The impacts of population growth, dietary changes associate with income growth, and rural-urban migration on the national AN and PN consumption was quantified by setting up different scenarios (Table 1). The impacts of these factors on N input to the Chinese food system was calculated by the variation of AN and PN, driven by the three factors multiplied by the N costs of AN and PN consumption.

2.6. Uncertainty analysis

There are uncertainties in estimating N input, food N consumption, N inputs driven by food consumption and N losses to the environment, etc. driven by the multiple activity data sources and complex parameters, as cited in Tables S1 to S16. We set up different uncertainty ranges for these activity data and parameters (see SI for details). And an uncertainty analysis was performed using the error transfer formula of mathematical statistics. The means and uncertainty ranges are reported.

3. Results and discussion

3.1. Historical trends in N inputs from outside the food system

Between 1990 and 2012, total inputs of N from outside into the food system in China increased by 23.2 Tg (1 Tg = 10^{12} g), from 31.9 to 55.0 Tg (Fig. 2). The amount of N from outside imported to the food system was 48.8 Tg in 2005 (Ma et al., 2012), at which year the total input of N from outside the food system was 50.7 Tg in our study, and we included 1.2 Tg more N from fish and seafood. The main inputs were chemical fertilizers, atmospheric deposition, BNF, imported animal feed and net imported N associated with food. Among these, the imported animal feed N increased from 6.9 to 12.7 Tg during 1990–2003, then decreased to 9.6 Tg in 2012. The variation originates from the marked increase in soybeans imported for vegetable oil after 2003 (MOA, 1991–2015). Soybean cake, a byproduct of soybean oil, was mainly used as animal feed; it was one of the internally recycled N sources from within the food processing system. Additionally, a portion of the N input to cropland was used for producing grains for other uses, and this increased from 1.2 to 7.3 Tg between 1990 and 2012. Hence, the total N inputs from outside the food system associated with food production increased by 16.7 Tg from 1990 to 2012.

3.2. Urban-rural differences in per capita food and food N consumption, and dietary-pattern changes

Distinct differences exist between urban and rural household diets (Fig. 3a, b). Per capita food consumption gradually decreased from approximately 440 to 357 kg yr^{-1} in urban households during 1990–2012, during which time per capita food consumption gradually decreased from about 425 to 360 kg yr^{-1} in rural households. Among these foods, the proportions of animal food consumed in urban areas increased from 17.2% to 25.8% during 1990–2002, and then decreased to 21.5% by 2012, while it increased from 6.0% to 14.4% in rural areas. These data indicate that diets in all areas are shifting to a higher animal food rate in China (Ma et al., 2012; Gu et al., 2015), but that per capita food consumption has shifted toward a more healthful diet in recent years in urban areas, concurrent with social and economic development, and an increase in citizens' health awareness (NHFPC, 2015).

Per capita food N consumption varied from 4.7 to 3.8 kg yr^{-1} and 4.0 to 3.8 kg yr^{-1} in urban and rural areas, respectively, during 1990–2012 (Fig. 3c, d), during which time the food N consumption for migrants who were living unregistered in cities ranged from 4.3 to 3.8 kg yr^{-1}. The results of this study in 1990 were close to the results of 5.0 and 4.3 kg yr^{-1} for urban and rural areas, respectively, in 1992, from Wei et al. (2008), who used the data on per capita habitual food intake reported by the National Health and Family Planning Commission of China (Zhai et al., 2005). And a finding similar to the results of 4.1 to 3.7 kg yr^{-1} in rural regions from 1990 to 2009, was reported by Cui et al. (2016), who used the data from the China Statistical Yearbook, but the per capita N consumption in his study showed a reverse trend to our results, changing from 3.0 to 3.9 kg for urban regions between 1990 and 2009. All the above per capita food N consumption results were lower than the 5.0 kg N yr^{-1} in China between 1980 and 2008, the 5.5 kg N yr^{-1} in 2010 (Gu et al., 2013, 2015), and 5.6 kg N yr^{-1} in China's urban households in 2005, but Ma et al. (2010) found results of 3.6 kg yr^{-1} in China's rural households in 2005; their calculations were mainly based on the data on China's urban household food purchases and rural household food consumption from the China Statistical Yearbook, and were checked using the data from the National Health and Family Planning Commission of China (Gu et al., 2013, 2015; Ma et al., 2010). The above comparisons illustrate that the sources of the data on the per capita food consumption is an all-important factor for the calculation of per capita food N consumption. The data on the China's urban household per capita food purchases and rural household per capita food consumption may be different when using the definition from the national bureau of statistics of China, which may have some difficulty distinguishing between food supply/consumption data for urban and rural residents and per capita actual food intake. In the present study, we used the value of per capita daily food intake by urban and rural residents, with the units of g food d^{-1} (Zhai et al., 2005; Wei et al., 2008; NHFPC, 2015), which perhaps comes the closest to the actual food consumption in urban and rural households.

Table 1
Description of scenarios and calculations of driving force effects.

Code	Scenario description	–	
S1	Urban and rural populations maintained at level of 1990, dietary patterns unchanged since 1990 and no population shift to cities	$S1_{2012}$-$S1_{1990}$	N management practices effect (this affects only N input)
S2	Urban and rural dietary changes based on S1	S2-S1	Dietary changes effect
S3	Population growth based on S2	S3-S2	Population growth effect
RS	Actual situation, population growth, dietary changes and population shift to cities	RS-S3	Rural-urban migration effect

Table 2
N losses to the environment from the food system in rural and urban areas in 1990 and 2012 (in Tg N). Codes refer to the N loss from the food system to the environment, shown in Fig. S1.

Item	Code	Rural		Urban	
		1990	2012	1990	2012
To atmosphere		16.4(5.1)[a]	24.1(7.8)	0.3(0.01)	1.1(0.5)
NH_3, N_2O emissions and animal digestive gas[b]	(1)	3.3	5.2	–[c]	–
NH_3 emissions, N_2O/N_2 from nitrification and denitrification, nitrogen gas from burning straw	(2)	12.4(5.1)	18.3(7.8)	–	–
NH_3 emissions from excrement and nitrogen gas from human bodies	(3)/(6)	0.7	0.6	0.3	0.4
Nitrogen gas from wastes, sewage treatment and sludge burning	(4)/(5)	0.0	0.01	0.01(0.01)	0.5(0.5)
Total N loss to atmosphere in China (2010)[d]		31.0–50.2			
The food system's contribution (%)		33.3–53.9	48.0–77.8		
To soil		2.8	4.2	0.7	1.3
Soil accumulation, and straw pile sets	(7)	2.7	4.1	–	–
Kitchen waste pile sets and landfills in rural areas	(8)	0.08	0.05	–	–
Sludge landfills in rural areas	(9)	0.0	<0.05	–	–
Food processing wastes	(10)	–	–	0.6	1.0
Kitchen waste pile sets, landfills and sludge landfills in urban areas	(11)	–	–	0.1	0.3
Total N loss to soil in China (2010)[c]		22.8			
The food system's contribution (%)		15.4	24.2		
To water bodies		8.7	14.1	0.4	1.2
Soil runoff, leaching and erosion	(12)	5.5	8.3	–	–
Animal waste erosion and leaching	(13)	3.0	5.4	–	–
Direct sewage discharge	(14)/(18)	0.2	0.4	0.4	0.4
Tail water discharge	(15)/(17)	0.0	0.02	0.02	0.8
Food industry wastewater	(16)	–	–	–	<0.1
Total N loss to water bodies in China (2010)[c]		21.8–23.6			
The food system's contribution (%)		38.6–41.7	64.8–70.2		
Intensity to soil and water bodies (kg N hm^{-2})[e]		115.0	162.1	855.7	502.3

[a] Numbers between brackets refer to molecular N_2 from soil denitrification and sewage treatment in N emissions to atmosphere.
[b] Animal digestive gas refers to the emissions of N_2 and ammonia from animals' digestive processes, account about 7% of the feed intake by animal (Wang, 1997; Shi, 2014).
[c] No data available.
[d] Cited from Cui et al. (2013) and Gu et al. (2015).
[e] Calculated by total N loss to soil divided by built-up rural area plus (agricultural land – gardens – grassland – woodlands – freshwater aquaculture areas) in rural areas and divided by built-up urban area in urban areas, based on data from China Statistical Yearbook (NBSC, 2015), China Urban Construction Statistic Yearbook (MHURDC, 2011, 2014a) and China Urban-Rural Construction Statistic Yearbook (MHURDC, 2014b).

Among per capita food N consumption values, the share of food N contributed by animals in China's urban area increased from 34.8% to 46.0% during 1990–2002, then it fell to 41.3% in 2012, while at the same time, it increased from 14.4% to 28.4% in rural areas. For migrants, the share of AN increased from 25.6% to 34.2% under our assumptions. The fraction of food N from animal sources increased from 38.4% to 43.4% in urban areas and from 16.3% to 28.0% in rural areas, between 1992 and 2002 (Wei et al., 2008). Gu et al. (2013) reported that the fraction of food N from animal sources increased from 10% to 30% in China between 1980 and 2008. These results illustrate that diets are shifting to a higher AN rate in China (Gu et al., 2015; Cui et al., 2016). The values of the share of food N contributed by animals in China's urban areas are now higher than the global mean value of 39%, but in rural areas, approximately 10% below the global mean. The shares of food N contributed by animals in China's urban and rural areas are still far below the values of 60–80% in the developed countries (FAO, 2013). There are some indications that rising incomes and urbanization are driving a global dietary transformation, from traditional largely vegetarian diets to diets higher in animal sources of food (Tilman and Clark, 2014). China's household diets are consistent with this global trend correlated with socioeconomic development (increases in both per capita GDP and the urbanization level) (Hou et al., 2014; Gu et al., 2013, 2015). China's ongoing transition from a rural to an urban population has been a major driver affecting nutrients consumed per capita and the nature of human diets toward those richer in animal products in recent years (Hou et al., 2014). This shift translates to a further increase in AN consumption that

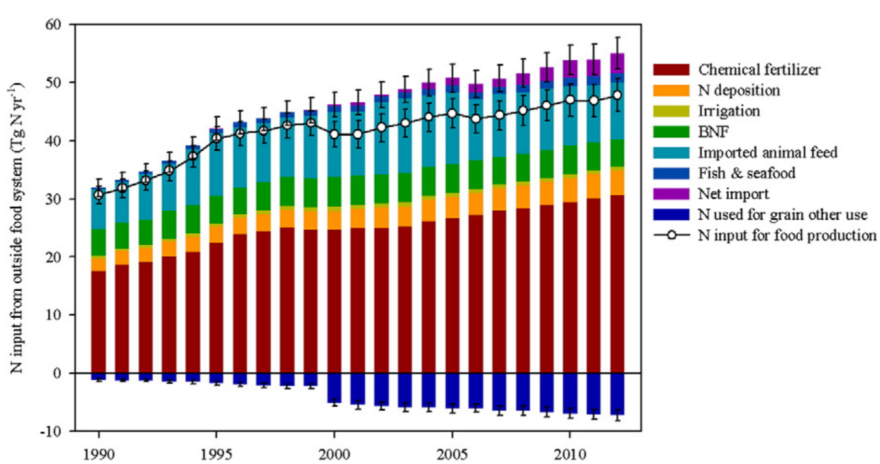

Fig. 2. Historical trends in N input to Chinese food system, during 1990–2012. Error bars represent uncertainty ranges of total N input to food system, N input for food production and N used for grain other use.

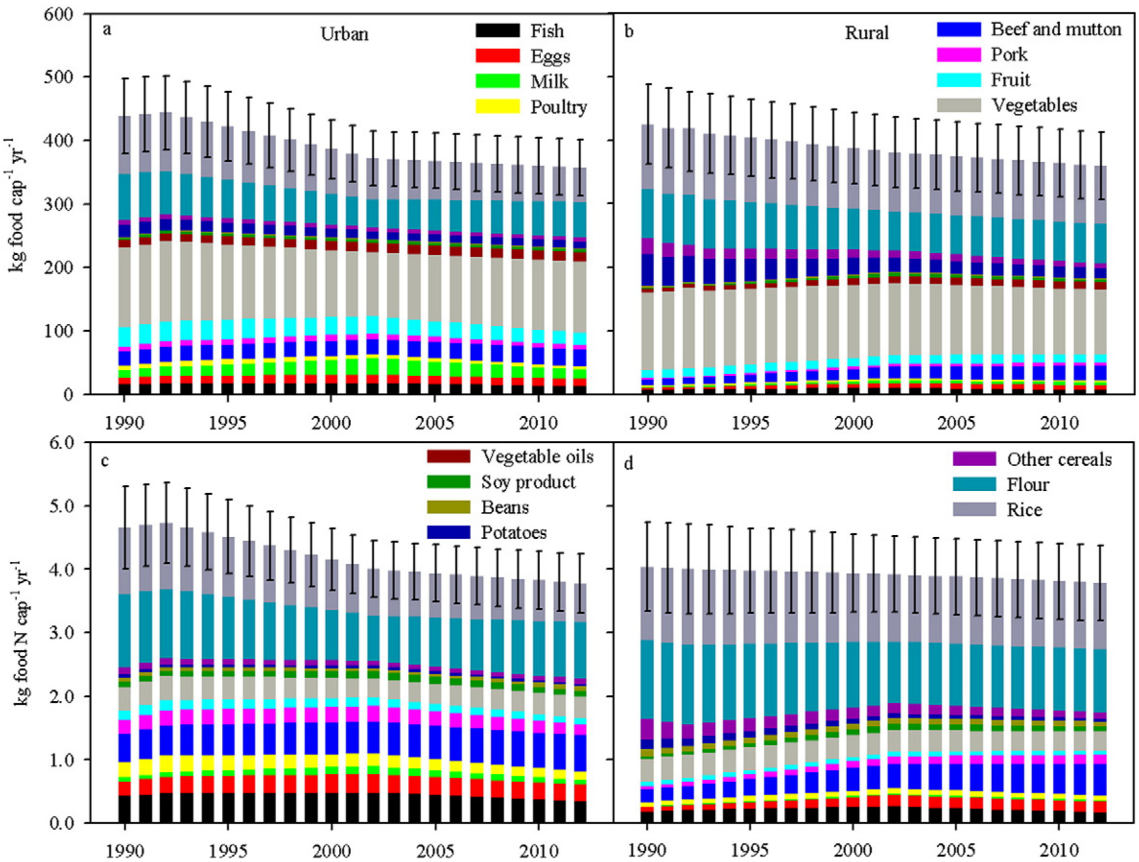

Fig. 3. Per capita food and food N consumption in urban (a, c) and rural (b, d) households, during 1990–2012. Error bars represent uncertainty ranges of per capita total food and food N consumption.

can be expected down the road with rapid economic development and urbanization, because of the low share of food N contributed by animals in rural areas and from migrants in urban areas, and the large gap of the share of food N contributed by animals in urban versus rural areas in China.

From 1990 to 2012, per capita AN consumption changed from 1.6 to 1.8 kg N yr^{-1}, and then back to 1.6 kg N yr^{-1}; from 0.6 to 1.1 kg N yr^{-1}; and from 1.1 to 1.3 kg N yr^{-1}; for urban residents, rural residents and migrants, respectively, and their per capita PN consumptions, from 3.0 to 2.2, from 3.4 to 2.7 and from 3.2 to 2.5 kg N yr^{-1}, respectively. The differences in per capita food N consumption between urban and rural residents indicate that, per capita, residents will consume 0.5 kg more AN yr^{-1} and 0.5 kg less PN yr^{-1} when someone registered as living in a rural area moves to an urban area. In 2012, the per capita AN and PN consumptions in urban areas come close to the superior energy standards of 1.7 kg AN yr^{-1} and 2.3 kg PN yr^{-1}, calculated using the data on per capita food consumption from the 'Dietary Guidelines for Chinese Residents' (CNS, 2011). In rural areas the per capita AN consumption was just over the low energy standard of 1.0 kg AN yr^{-1}, but the per capita PN consumption exceeded the superior energy standard. Diets in rural area are expected to change to the recommended moderate standard of 1.4 kg AN yr^{-1} with the improvement of income and living standards, and the per capita AN consumption can be expected to increase with rural-urban migration and household registrations changing to urban. There is some opportunity to mitigate the increase in demand for AN by guiding urban diet changes to lower AN consumption, and rural diet changes to the recommended AN consumption, in 'Dietary Guidelines for Chinese Residents' (CNS, 2011). This measure was suggested popularly in countries where the diets are high rich in AN or rapid developing toward to more AN (Sutton et al., 2011; Leach et al., 2012; Cui et al., 2013; Gu et al., 2015).

There is still some question about the differences in dietary patterns between new immigrants and natives in urban areas. Many studies have directly used urban populations multiplied by per capita food N consumption in urban households, for calculating the consumption of food (Ma et al., 2010; Hou et al., 2014; Cui et al., 2016), without making a distinction between the original and newly registered urban residents, because of the unavailability of statistical data for the latter, in China; on the other hand, the investigations of food consumption have been for the average urban resident (including both original and newly registered residents) in the survey year (NHFPC, 2015). Hence, we believe that the diets will be transformed to the urban pattern from the rural one, with the change in household registration. For the migrants who moved to an urban area from a rural area but did not register, their diet can be assumed as the mean of urban and rural residents, as mentioned in the Methods and data section, so that per capita consumption will be 0.2 kg AN yr^{-1} more and 0.2 kg PN yr^{-1} less for someone who is registered as living in rural area and then moves to an urban area. This group has become a floating population over the past two decades of urbanization. The large number of immigrants moving to urban areas from rural areas, whether or not they are registered as living in cities, will consume more AN and less PN than when they were living in a rural area, and the increase in AN equals the PN reduction. Hence rural-urban migration will raise the inputs of N to the Chinese food production system, because the N cost of AN production is higher than that of PN (Galloway and Cowling, 2002; Xue and Landis, 2010).

3.3. Plant- and animal-food N consumption

In the present study, we estimated the total food N consumption in China by separately calculating AN and PN consumption amounts by urban residents ($Urban_{AN}$, $Urban_{PN}$), rural residents ($Rural_{AN}$, $Rural_{PN}$)

and migrants in urban areas ($Migrants_{AN}$, $Migrants_{PN}$) between 1990 and 2012 (Fig. 4). This study differs from other similar studies on estimated food N consumption in China by summing food N consumption by urban and rural residents (Ma et al., 2010; Cui et al., 2016). The results showed that the total food N consumption changed from 4.7 ± 0.6 to 5.1 ± 0.5 Tg yr^{-1}, higher than the results of 3.9 Tg food N yr^{-1} in 1990 reported by Cui et al. (2016), and closer to the 4.4 Tg food N yr^{-1} in 2005 reported by Ma et al. (2013), and the 4.6 Tg N yr^{-1} in 2009 reported by Cui et al. (2016), but far below the 7.6 Tg food N yr^{-1} in 2010 reported by Gu et al. (2015), whose results included approximately 1.0 Tg food N yr^{-1} from other sub-systems, but did not include food from cropland and livestock systems, and, moreover, calculated cropland food based entirely on grains. In our study, total PN decreased from 3.8 to 3.4 Tg yr^{-1}, at an average rate of 0.02 Tg yr^{-1} between 1990 and 2012, during which time total AN increased from 0.9 to 1.7 Tg yr^{-1}, at an average rate of 0.04 Tg yr^{-1}. Clearly, more food N will be consumed, with higher AN consumption driven by population growth, urbanization and dietary changes.

From 1990 to 2012, the proportions of $Urban_{PN}$, $Rural_{PN}$ and $Migrants_{PN}$ to total PN consumption increased, from 19.0% to 31.4%, 75.5% to 51.5%, and 5.4% to 17.0%, respectively, and $Urban_{AN}$, $Rural_{AN}$ and $Migrants_{AN}$ from 40.9% to 42.8%, 51.8% to 39.6%, and 7.4% to 17.6%, respectively. These increases indicate that more and more food N was consumed in urban areas with the large rural-urban migration; this is known as "nutrient urbanization" (Lin et al., 2013). In addition, rural residents and migrants in urban areas will cause more AN consumption, as levels of income and urbanization increase, because they still account for a large part of the total AN consumption in 2012 and their diets exhibit a large potential to change toward more AN consumption, following a trend similar to urban household diets. Furthermore, household diet differences between urbanites and migrants in urban areas should be considered in calculating national food N consumption, because food consumption by migrants occupies a growing proportion of the urbanization effects—a trend that was overlooked in similar studies (Ma et al., 2010; Gu et al., 2015; Cui et al., 2016).

3.4. N cost of Chinese food

Different N inputs from outside the food system, coupled with different amounts of food N consumption, will result in different environmental costs per unit of food N consumption. The N cost is defined as the ratio between total new N input into food system and the N in food stuffs (Bleken and Bakken, 1997). It also reflects the amount (in kg) of new N input to the food system for the delivery of 1.0 kg N in the food entering households (Ma et al., 2012). N costs of the Chinese food N, PN, and AN consumption were calculated in this study (Fig. 5). The results showed that, the N cost of the Chinese food N increased dramatically from 6.3 to 9.9 kg N kg^{-1} between 1990 and 2012. Hence, for the delivery of 1 kg of food N to the consumption stage, food production processes used about 10 kg of N and lost 9 kg N to the environment, in 2012. The results of this study showed the same trend as the N cost of Chinese food during slightly different study periods, increasing from 6.0 to 11.0 kg N kg^{-1} during 1980–2005 (Ma et al., 2012) and from 8.3 to 10.5 kg N kg^{-1} during 1990–2009 (Cui et al., 2016). The large difference in the N cost of food in 1990 between our study and the results from Cui et al. (2016), can be attributed to the omitted N used for grains for other uses, and the import and export of food included in our later research. The N cost of Chinese food is relatively high compared to the estimates for the world as a whole (Galloway and Cowling, 2002; Pierer et al., 2014). The increase in the N cost of Chinese food may originate from the 12% growth in new N input for harvested grain, which was used for plant-food production during 1990–2012, even though the PN supply only increased by 6%. As a result, N recovery efficiency in grain production decreased to approximately 20%, much lower than the global average of 33% (Raun and Johnson, 1999; Ma et al., 2012). Another cause of the increase in N cost of Chinese food is the increased gap between PN supply and final consumption, from 8.9% in 1990 to 28.3% in 2012, which indicates that China has not been making full use of plant food production, possibly because some food stocked in the retail stage may spoil or expire before it is put out for sale, because these sectors have not kept up with the rapid development of urbanization (Chen et al., 2006; Ma et al., 2014), coupled with the increase in consumer fastidiousness and rejection of lower-quality food with the rise in the standard of living (Ma et al., 2012). And more food is consumed in restaurants—where more food N is wasted—instead of at home in urban area (Cheng et al., 2012; Ma et al., 2014). The increase in the consumption of animal-derived food (Fig. 4) is an additional cause of the increase in N cost of Chinese food.

We estimated the N cost of Chinese PN and AN production, to calculate the impacts of population growth, dietary changes and rural-urban migration on PN and AN consumption and their effects on new N input from outside the Chinese food system. The results showed that the N cost of PN consumption increased from 5.5 to 7.6 kg N kg^{-1} between 1990 and 2012, but that it declined significantly in the early 2000s, because the parameters of the relative proportions of harvested main crop products used as food fell after 2001 (Table S6). The N costs of the PN consumption in China were higher than the 3.0, 4.3 and 3.8 kg N kg^{-1} in Austria, the U.S., and the U.S./European hybrid factors, respectively (Leach et al., 2012; Pierer et al., 2014). During this same time period, the N cost of AN consumption increased from 9.0 to 13.6 kg N kg^{-1},

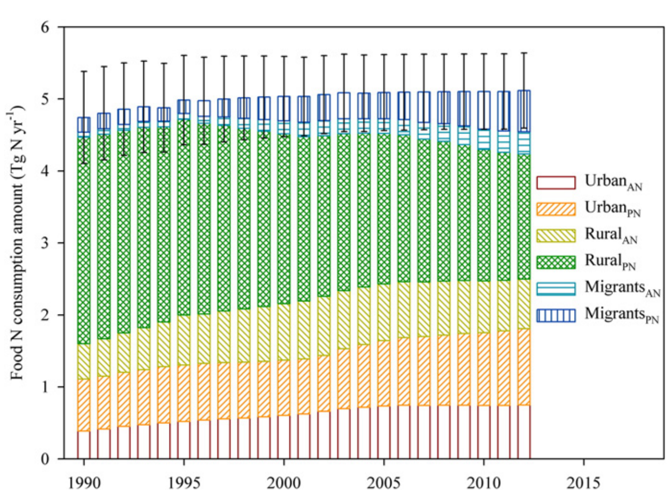

Fig. 4. National PN and AN consumption by urban and rural residents and migrants in urban areas, during 1990–2012. Error bars represent uncertainty range of the national food N consumption.

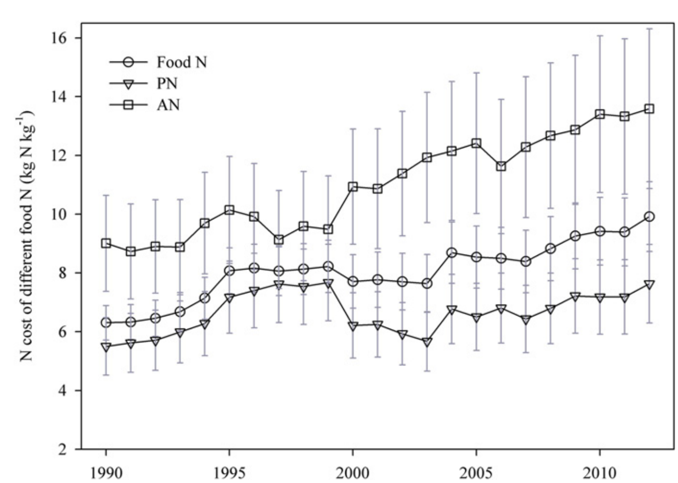

Fig. 5. N cost of different Chinese food, during 1990–2012. Error bars represent uncertainty range of each value.

far higher than the 4.8, 5.6 and 5.3 kg N kg^{-1} in Austria, the U.S., and the U.S./European hybrid factors, respectively (Leach et al., 2012; Pierer et al., 2014). The reason for the high N cost of AN originates from the new N input to the animal-production system, which increased 139%, while the consumed AN only increased 84.5%. The N cost of the AN was 1.8 times the N cost of PN in 2012; hence, the continually increasing demand for AN will lead to more N input from outside to the Chinese food system, with population growth, dietary changes and rural-urban migration.

3.5. Effects of dietary changes, population growth and rural-urban migration on food N consumption

We have estimated the separate impacts of dietary changes associate with income growth, population growth and rural-urban migration on AN and PN consumption, by assuming different scenarios (Fig. 6). The results clearly showed that, as would be expected, dietary changes and population growth were the two main driving forces of the increase in total AN consumption between 1990 and 2012, by which time they accounted for 52% and 31%, respectively, of the increased AN in 2012 compared to 1990, but their contribution dropped in the early 2000s because of China's slower population growth and dietary changes. National population increased by 8–14% and 5–7% in the 1990s and the 2000s, respectively, and per capita AN consumption increased at the rate of 0.02 kg yr^{-1} in the 1990s but showed an appreciably decreasing trend in the early 2000s in urban areas. In rural areas, per capita AN consumption increased at the rate of 0.04 kg yr^{-1} in the 1990s and 0.01 kg yr^{-1} after 2000. It is well known that population growth and dietary changes have been the main driving forces for increasing AN consumption around the world (Galloway et al., 2004; Tilman and Clark, 2014; Gu et al., 2015). However, in the present study, we found that rural-urban migration contributed higher and higher proportions of the increasing total AN consumption in China, which increased from 7.2% to 16.8% between 1990 and 2012, primarily because of the urban-rural differences in per capita AN consumption and urbanization; these may be the main cause of the unexplained residual increase in total inputs of N through the food system, in China (Hou et al., 2014). Urban population could rise to 1.0 billion in China in the next two decades if the current trend continues, and the new National Urbanization Plan of China projects the urban population fraction to rise by 1.0% a year, and to reach 60% by 2020 (X.M. Bai et al., 2014). Combined with the implementation of the two-child policy in China, the higher number of people migrating to cities from rural regions will result in more demand for AN.

From 1990 to 2012, dietary changes dominated the decrease in total PN consumption, which decreased by approximately 0.8 Tg PN in 2012 compared to 1990, during which time rural-urban migration resulted in an additional 0.1 Tg PN yr^{-1} decrease. However, population growth caused an additional 0.5 Tg PN yr^{-1} increase that only offset 56.2% of the decreases in PN consumption caused by dietary changes and rural-urban migration. The three factors eventually resulted in a 0.4 Tg PN yr^{-1} decrease in 2012 compared to 1990. As can be seen from Fig. 3, the per capita PN consumption of urban residents has been relatively stable in recent years, but it keeps dropping in rural areas and urban migrant populations. The projected PN consumption will continue to decline, because rural residents and migrants in urban areas still accounted for approximately 70% of the total PN consumption in 2012, and their dietary patterns will shift to lower PN consumption, to align with the dietary pattern in urban areas, with their increasing levels of income and rural-urban migration. The decrease in PN consumption is dominated by the reduction in rice consumption, as the per capita rice consumption in rural areas was higher (37.9 kg yr^{-1}) than that in urban areas, in 2012 (Fig. 3).

3.6. Driving forces of the N input to the Chinese food system

The separate impacts of dietary changes, population growth and rural-urban migration on AN and PN consumption, coupled with the N cost of AN and PN, could be the driving forces of the N input to Chinese food system (Fig. 7a). The results showed that, as would be expected, the effects of dietary changes, population growth and rural-urban migration on AN consumption were the main driving forces of the increased 14.2 Tg N input to the Chinese food system for animal-derived food production; they accounted for 5.6 (39.4%) (S2-S1), 3.4 (23.9%) (S3-S2) and 1.8 (12.7%) (RS-S3) Tg N increases compared to the S1 scenario in 2012, respectively. Hou et al. (2014) studied the contributions of changes in national population and food production and consumption per capita to changes in total N inputs, and found that the changes in population and food production and consumption per capita were the main driving forces of the increase in N input to the livestock system; they contributed to 17–38% and 59–75% of the changes in total N input to the Chinese livestock system between 1990 and the early 2000s, respectively. However, their N input to the livestock system includes N both from outside and internal to the food system, but in our study we only considered the N from outside the food system, and these other studies ignored the impacts of the changes in AN production efficiencies and the net import of AN on input, a factor that resulted in a

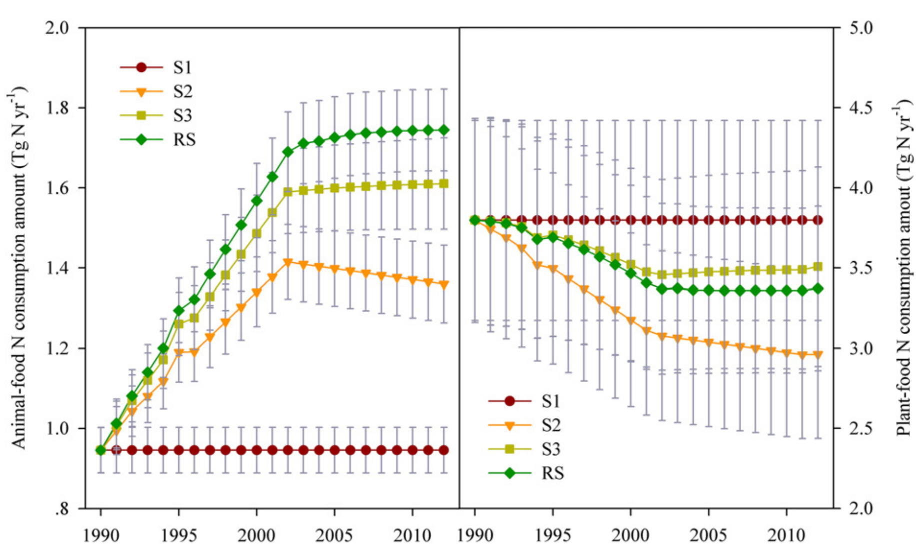

Fig. 6. The changes in PN and AN consumption, driven by population growth, dietary changes, and rural-urban migration relative to 1990, during 1990–2012. S1, S2, S3 and RS are explained in Table 1. Error bars represent uncertainty range of each value.

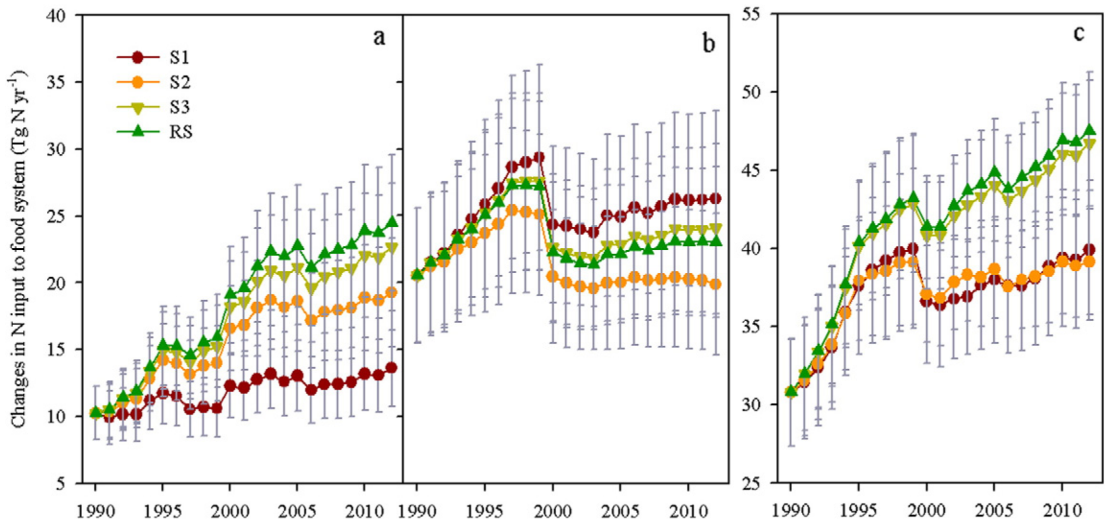

Fig. 7. The changes in N input to Chinese food system, driven by population growth, dietary changes and rural-urban migration relative to 1990, during 1990–2012. a, b, c represents the changes in N input, driven by the variations of AN, PN and AN plus PN relative to 1990, during 1990–2012. S1, S2, S3 and RS are explained in Table 1. Error bars represent uncertainty range of each value.

3.4 Tg N increase for animal-derived food production in the S1 scenario of this study.

Dietary changes and rural-urban migration resulted in 6.4 (S2-S1) and 1.0 (RS-S3) Tg N decreases for PN production, respectively, compared to the S1 scenario in 2012 (Fig. 7b). However, during this time population growth led to an increase of 4.2 Tg N (S3-S2), offsetting 56.8% of the decrease in N for PN production. Together they resulted in a 3.2 Tg N decrease in RS relative to the S1 scenario in 2012. Hou et al. (2014) also found that the changes in population and in food production and consumption per capita were the main driving forces of the increases in N inputs to cropland, accounting for 27–40% and 56–66%, respectively. But our results showed that population growth and dietary changes caused 168% and −257% of the increased N input from outside the food system to cropland, for plant-derived food production. In addition, −42% of the increased N input to cropland for PN production was caused by rural-urban migration. Another 5.7 Tg N increase for PN production in the S1 scenario between 1990 and 2012 originated mainly from the changes in N cost and the net import of PN. Ultimately there was an increase of 2.5 Tg N input from outside the food system input to cropland for PN production in RS in 2012 compared to 1990.

Integrating the impacts of dietary changes, population growth and rural-urban migration on N input to the Chinese food system (showing the influence of these factors on the consumption of AN and PN (Fig. 7c)), we found that the decreased N input (0.7 Tg N in 2012) resulting from lower PN consumption was almost completely offset by the increased N input for more AN consumption, caused by dietary changes (S2-S1). Population growth (S3-S2) and rural-urban migration (RS-S3) drove 7.6 and 0.8 Tg N increases, respectively, compared to S1 in 2012, and together with dietary changes they accounted for 46% of the increased N input to the food system between 1990 and 2012. In other words, to maintain the N cost of the food production based on scenario S1, China's population growth, dietary changes and rural-urban migration created a need for 7.6 Tg increased N to support these changes over the past 20 years. The remaining 54%, or 9.1 Tg, increase in N input was mainly due to the decrease in the NUE of the Chinese food system, from 16% to approximately 9%, between 1980 and 2009 (Ma et al., 2012; Cui et al., 2016). These results indicate that the increased N input to the Chinese food system is not merely due to over-fertilization by Chinese farmers (Ju et al., 2009; Chen et al., 2014), but has been driven by population growth, dietary changes and rural-urban migration. From this perspective, China will face more and more pressure to decrease N input for food production, because these three factors will continue and even increase with the development of China's economy, the new National Urbanization Plan of China and the implementation of the two-child policy. Some scholars have suggested reducing N input and the risks of environmental N pollution by guiding dietary changes to lower consumption of animal-derived food around the world (Sutton et al., 2011; Tilman and Clark, 2014; Oenema et al., 2014), or by substituting beef and mutton using low N cost poultry, pork, fish and seafood (Pierer et al., 2014; Cui et al., 2016), and this might be a suitable solution for the current Chinese population (Ma et al., 2013; Gu et al., 2015). Additionally, there is a large potential for decreasing the N input to the food system by more nitrogen-efficient plant- and animal-food production (Ma et al., 2013; Z.H. Bai et al., 2014; Gu et al., 2015), and such a decrease is crucial for keeping losses of N to the environment as small as possible, both in China and on a global scale (Ma et al., 2013; Tilman et al., 2011; Pierer et al., 2014). Gu et al. (2015) forecast the total N input to China to drop to 31.0 Tg N yr^{-1}, 64% of the level in 2010, whereby total Nr losses in 2050 would be reduced by 48% relative to 2010, by reducing the share of animal-food N to 40% (the level of 2010) by 2050, and improving NUE in each sector, to a level comparable to the current best level worldwide. These goals could be achieved by increasing recycling N rates of livestock and humans from 43% and 23%, to 80% and 50%, respectively, together with improvements in industrial practices (Bouwman et al., 2013).

3.7. N losses to the environment from the food system in rural and urban areas

N losses to the environment (atmosphere, soil and water bodies) from different sectors of the food system in 1990 and 2012 were calculated and separated into rural and urban areas (Table 2). N losses to the atmosphere increased from 16.4 to 24.1 Tg yr^{-1} in rural areas, between 1990 and 2012, mainly from cropland and livestock systems (Ma et al., 2012), during which time N losses to the atmosphere almost tripled, to 1.1 Tg yr^{-1}, in urban areas, mostly from human excrement. With the increases in population and sewage treatment ratios, N losses to the atmosphere from sewage treatment increased from 0.01 to 0.5 Tg yr^{-1} in urban areas from 1990 to 2012. The total N losses to the atmosphere from the food systems in urban and rural areas contributed to approximately 33–54% and 48–78% of the national N loss to the atmosphere in China in 2010, respectively (Cui et al., 2013; Gu et al., 2015).

N losses to soil increased by 50% in rural areas, almost all from soil accumulation and straw pile sets in cropland. The amount of N lost to soil in urban areas increased to 1.3 Tg yr^{-1} in 2012 from 0.7 Tg yr^{-1}

in 1990, 77–86% of it from food processing waste, and the rest from kitchen waste piles, landfills and sludge landfills. The total N losses to soil from the food system in urban and rural areas in 1990 and 2012 contributed to approximately 21% and 32%, respectively, of the national N loss to soil in China in 2010 (Cui et al., 2013).

N losses to water bodies in rural areas increased by 62% in 2012 compared to 1990, dominated by the increase in surface runoff, leaching and erosion in cropland, and waste discharges from the livestock system. In urban areas, N losses to water bodies increased from 0.4 to 1.2 Tg yr^{-1}, between 1990 and 2012, almost all from direct sewage discharge, and 33% and 67% were from direct sewage discharge and tail water discharge, respectively, in 2012, because more and more sewage was treated in urban areas. The total N losses to water bodies from the food system in urban and rural areas contributed approximately 39–42% and 65–70%, respectively, of the national N loss to the water bodies in China in 2010 (Cui et al., 2013; Gu et al., 2015).

The amount of N losses to the environment in rural areas was much higher than that in urban areas, but it makes no sense simply to compare such amounts directly. In this study, we compared the combined N emission intensities to soil and water bodies from the Chinese food system separately, for rural and urban areas, but not so for N losses to the atmosphere, because there is no atmospheric boundary between rural and urban areas. The results showed that N emission intensity to soil and water bodies in rural areas increased from 115.0 to 162.1 kg N hm^{-2} between 1990 and 2012, during which time the values in urban areas reach to 855.7 and 502.3 kg N hm^{-2}, respectively. The decreases in the N pollutant intensity in urban areas originated from the rising sewage treatment ratios and N losses to soil and water bodies, which increased by 175% during 1990–2012, while the urban built-up area increased by 287% (MHURDC, 2011, 2014a). However, N emission intensity to soil and water bodies in urban areas was still more than three-fold that in rural areas, indicating higher risks of N pollution in soil and water bodies in urban areas than in rural areas. Moreover, future urbanization will exacerbate the problem of environmental N pollution in China (Lin et al., 2013; Lin and Grimm, 2015). The higher N emission intensity to soil and water bodies in urban areas is due, first, to more and more N imported into cities in the form of food, with increasing urbanization (Lin et al., 2013; Hou et al., 2014), and higher rates of food waste in urban than in rural areas: the same reasons given in the discussion on the gap between PN supply and final consumption above. Second, because urban systems block the N generated by urban anthropogenic activities from re-entering the agro-ecosystem, less N is recycled in urban areas than in rural ones (Table S17) (Wei et al., 2008; Lin et al., 2013). Increasing recycling N rates is one of the important ways of decreasing N input to, and N losses to, the environment, from the food system (Ma et al., 2013; Oenema et al., 2014; Gu et al., 2015); in the city, mitigation measures include adopting strategies to revise the broken N links between rural and urban areas. These measures include, first, increasing the proportion of kitchen waste as feed and compost to replace waste piles and landfills, and, second, as much as possible increasing the application of urban human excrement to fields (Table S14), by establishing effective nutrient recycling mechanisms for the waste N flow back from urban to rural areas, in order to offset the lower recycling utilization of N after food consumption in urban areas (Lin et al., 2013; Ma et al., 2014). Collection and recycling of human urine (which contains 80% of total N excreted by human and accounts for about 75% of the N to domestic wastewater) may be a much better alternative, and it could help to close the N cycle (Larsen and Gujer, 1999; Simha et al., 2017). Gu et al. (2015) indicated that if recycling N rates of human excrement could be increased from 23% to 50%, it would contribute to decreased N input to the urban environment, as discussed above. There is an emerging opportunity for using municipal kitchen waste as compost and human manure as fertilizer, because increasing the recycling of organic N is an effective solution for achieving zero growth of chemical fertilizer use in China by 2020 (DPM, 2015; Gu et al., 2015).

Some changes in agricultural practices could also decrease the use of chemical fertilizers. The increase rate of synthetic N fertilizer application came to approximately 4 times the total annual grain production from 1978 to 2005, resulting in significant decreases in NUE, and large amounts of external N inputs with decreasing NUE have been causing severe environmental degradation since the 1990s (Ju et al., 2009; Guo et al., 2010). For example, the average synthetic N application ranges from 550 to 600 kg of N per hectare for the typical winter wheat–summer maize double-cropping systems in the North China Plain (Cui et al., 2008). Yet 30–60% of the synthetic N input could be saved without sacrificing yields, or the nitrogen partial factor productivity could be increased by 40–46%, maintaining similar synthetic N input while significantly reducing environmental risk, by adopting optimum N management in this crop system (Ju et al., 2009; Chen et al., 2014).

Additionally, there some N to sewage after consumption if they can't be separated and recycled, the environmental N pollution from this part of N can be mitigated by further increasing sewage treatment rates (only 6% and 75% of sewage was treated in rural and urban areas, respectively, in 2012) and enhancing the total N removal rate by adopting new sewage disposal technologies, to cause more N pollutants to be emitted to the atmosphere in the form of pollution-free N_2, to reduce the emissions of ammonia and nitrous oxide. The mean national total N removal rate was only 35–40% in China (Table S15). However, the total N removal rate could reach 70–80% with some disposal technologies (Yang et al., 2010; Bresler, 2012). This illustrates that a large potential exists to improve the N removal rate and decrease water N pollution in China's urban areas, by adopting advanced sewage treatment processes. But removing N to the atmosphere as N_2 is not the best way to avoid N pollution, therefore, the first choice to reduce N pollution in water bodies is increasing recycling N rate of humans excrements.

4. Conclusions

This study presented an integrated assessment of the historical trends and differences in per capita AN and PN consumption between urban and rural areas, and national total AN and PN consumptions, and analyzed the impacts of population growth, dietary changes and rural-urban migration on AN and PN consumption, combining these with the N cost of the Chinese AN and PN, to determine the driving forces of the increase in food production-related N input from outside the food system. Additionally, we compared N losses to the environment from the Chinese food system in urban and rural areas. Our analysis show that people living in urban areas required 0.5 kg more AN yr^{-1} and 0.5 kg less PN yr^{-1} than those living in rural area in 2012, indicating more AN and less PN demand with rural-urban migration, in addition to the dietary changes. Population growth, dietary changes and rural-urban migration were the main driving forces of the increase in AN consumption and the decrease in PN consumption over the past twenty years. The PN reduction more than offset the increase in PN caused by population growth. As a result of the N cost differences between AN and PN, dietary changes showed no significant contribution to the growth in N input to the food system for food production over the past twenty years, while population growth and rural-urban migration together accounted for 50% of the increase. Another increase in N was mainly due to the declining in NUE of the Chinese food system. Our analysis also indicates that China is facing higher risks of environmental N pollution with urbanization, not only because of the higher N input to the food system caused by the additional demand for AN precipitated by population growth and rural-urban migration (and this trend will continue, and even increase, under China's urbanization), but also because of the higher food-sourced N loss intensity to soil and water bodies in urban areas, compared to rural ones. The strategies for reducing the risks of N losses to the environment involve guiding the urban and rural dietary changes to the recommended energy standards, and to mitigate the increase in demand for AN, revising the broken nutrient links by recycling more kitchen waste and human excrement N (especially human urine), moving it from urban to rural areas, and enhancing sewage treatment and total N removal ratio by adopting advanced sewage treatment processes in where N can't be recycled and discharged into wastewater.

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Appendix A. Supplementary data

Detailed descriptions of the calculation of population growth and rural-urban migration in urbanization, coupled urban and rural N flow model of Chinese food system, estimated N accumulation in cropland soils and imported animal feed, description of the uncertainty ranges of activity data and parameters. Table S1–S15, main parameters; Table S16, per capita habitual food intake in 1982, 1992, 2002 and 2012; Table S17, post-consumption N recycling utilization rate of urban and rural areas, 1990 and 2012; Fig. S1, Coupled rural and urban N flow model of Chinese food system, including rural system, urban system and environmental system, arrows in the food system represent N flows; Fig. S2, Urbanization and registered population urbanization rates, population growth and rural-urban migration, 1990–2012. This information is available free of charge via the Internet at http://pubs.acs.org/.

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