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Spatio-temporal dynamics of nitrogen use efficiencies in the Chinese food system, 1990–2017

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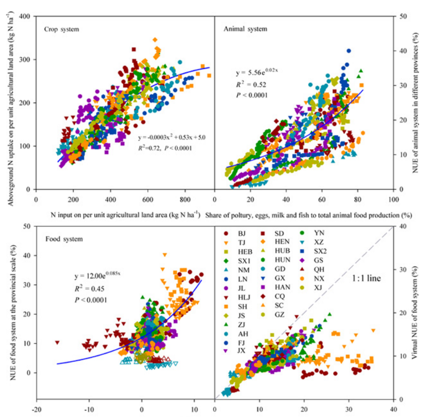
HIGHLIGHTS

- Spatio-temporal dynamics of nitrogen use efficiencies in China were studied.
- N input per unit agricultural land reduced in 24 provinces in 2017 relative to 2010.
- The higher poultry, eggs, milk and fish ratios, the higher NUE of animal system.
- Modestly import is an option for improving NUE of the provincial food systems.
- Virtual NUE is an accurate indicator for provincial food systems' NUE comparisons.

GRAPHICAL ABSTRACT



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ABSTRACT

Understanding the influence factors of nitrogen (N) use efficiencies (NUEs) in different stages of the food system at the provincial scale is critical to achieving cleaner food production while ensuring food security. Nevertheless, they are not well understood. Here we comprehensively analyzed NUE and its influence factors at different stages of the provincial food system. The results showed that per unit agricultural land N input increased by 5–92% in 27 provinces, during 1990–2010, resulting in a low NUE for the crop system when N input per unit agricultural land exceeded about 400 kg N ha^{-1}. This situation has brought some positive changes, as N input decreased by 3–271 kg N ha^{-1} in 77% of the provinces in 2017, relative to that of 2010, but 10 provinces were still over 450 kg N ha^{-1} in 2017. Animal food production is expected to continue to expand because 35% and 68% of provinces' urban and rural households, respectively, were still below the recommended minimum animal food N consumption recommendation in 2017, posing great challenges for reducing environmental N pollution. An exciting result is that the NUE of the animal system can be improved by increasing the share of animal food contributed by poultry, eggs, milk and fish, to align with the diets recommended by the Chinese Nutrition Society. NUEs of the provincial food systems excluding Inner Mongolia, Xinjiang, Qinghai and Tibet, would increase by 13% if the net imported food N increased by 1 kg capita^{-1}. Nevertheless, virtual NUE—including N input for imported food in the calculation of NUE—should be considered for accurate comparison of the NUEs of the provincial food systems, especially in highly urbanized areas, while N input for non-food

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animals should be excluded for accurate evaluation of the NUE in pastoral areas, considering their special production systems and feeding structures.

1. Introduction

Reactive nitrogen (Nr) has played very important roles in sustaining food production for the increasing global population (Tilman et al., 2011). Nevertheless, the disturbed biogeochemical cycle of Nr has exceeded the consumption capacity of the terrestrial ecosystem, resulting in a series of environmental problems and endangering global sustainability (Steffen et al., 2015). What's more, the global food-crop demand for nitrogen (N) is expected to increase by 100–110% relative to 2005 levels, while the global fertilizer N use will increase from 100 Mt to 225–250 Mt, by 2050 (Tilman et al., 2011). The world is facing a great challenge: reducing the environmental effects of Nr while ensuring food security. The latest research indicates that China's food system will consume around 30–45% of global fertilizer use and generate 40% of global Nr losses by 2030 if the current trend continues (Ma et al., 2019). China—more than any other country or region—is a case in point for addressing the more severe challenges in coordinating the role of Nr in food production and environmental protection (West et al., 2014; Powlson et al., 2018; Ma et al., 2019). Hence the Chinese situation has significant global implications for reducing the environmental effects of Nr to the maximum extent possible while ensuring food security (West et al., 2014).

China's food production system operates under large-scale agro-ecological conditions (Chen et al., 2014), producing multiple crops each year in various agricultural regions (Gao et al., 2018a). The mean chemical fertilizer N input ranges from 30 kg N ha^{-1} per year for an average soybean field in Northeastern China to 978 kg N ha^{-1} for greenhouse vegetables in China as a whole (Gao et al., 2018a). Different crops and N application rates have resulted in substantial variation in the N use efficiency (NUE) of crop production, among the different provinces, from 13% in Shanghai to 80% in Heilongjiang, in 1980; and from 12% in Henan to 45% in Tibet, in 2005 (Ma et al., 2012). However, how crop structure and N applied intensities affect the NUE of crop production at the provincial scale has not been well reported. Meanwhile, the NUE of the livestock system has increased from 8 to 9% in 1980–1990 to 14–16% in 2005–2010 (Ma et al., 2012; Bai et al., 2018a), because the percentage of landless industrial livestock production systems showed exponential growth trends between 1978 and 2015, and this approach has higher NUE than do traditional backyard and mixed crop-livestock systems (Bai et al., 2018a). Yet large variations remain among the NUEs of livestock systems in different provinces, from 3% in Sichuan to 12% in Tianjin, in 1980; and from 9% in Gansu to 24% in Liaoning, in 2005 (Ma et al., 2012), because of the differences in animal species and livestock densities per hectare of cultivated land (Bai et al., 2018a). Yet the impact of breeding structure on the NUE of livestock systems at the provincial scale is not well understood.

The NUE of food systems and the N costs of the selected foods could be used to represent the environmental effects of per unit food N consumption (Bleken and Bakken, 1997; Gao et al., 2018b; Ma et al., 2012). The NUE of the Chinese food system as a whole significantly decreased from, 17% to 10%, during 1980 to 2012 (Gao et al., 2018b; Ma et al., 2012). It is now slightly higher than the global mean food-system NUE (9.0%) (Galloway and Cowling, 2002). Yet large variations remain among the NUEs of food systems in different provinces, having dropped significantly from a range of 5–49% to 3–14% between 1980 and 2005, because of the combined effects of changes in the crop and livestock production systems, the food trade and the food consumption system (Ma et al., 2012). There are some indications that modestly increasing food imports from abroad could help improve the NUE of the food system in China (Ma et al., 2019), and we hypothesized that increasing the frequency of food trade would also effect the NUE of the food system at the provincial scale. Yet there has been an appalling lack of concern for the relationship between the food trade and the NUE of the food system, at the provincial scale.

The above N costs are defined as the ratio between an initial investment of N into a food system and the N embodied in food stuffs (Bleken and Bakken, 1997), and it represents the new N input (in kg) to the food system for the delivery of per unit N in food stuffs (Ma et al., 2012; 2014). It has been indicated that the inter-provincial staple crop trading volumes totaled 125 billion tons in China during 2010 and 2012 (Wu et al., 2018). As more and more food is traded among Chinese provinces, the N cost of a food system is not suitable for reflecting the actual environmental N effects of the final food consumption in a province, because the virtual N input or output for the net imported food and feed N at the production stage is not taken into account (Bleken and Bakken, 1997; Gao et al., 2019b; Leach et al., 2012). Hence, we hypothesized that the NUE of the food system and the N cost of food might be either overestimated or underestimated because of food trade inconsistencies at the provincial scale. We have therefore proposed virtual NUE—defined as total food N consumption divided by total new N input to crop and animal production systems—for tracking the actual environmental N effects of total food consumption at the regional scale (Gao et al., 2019b; Huang et al., 2019). The formulation of measures and policies adapted to local conditions for improving the NUE of local food systems requires a comprehensive understanding of the actual effects of the food system on the environment, at different scales (Lin et al., 2016; Verger et al., 2018; Huang et al., 2019). Nevertheless, the virtual NUE of the food system and its correlation with the NUE of the food system at the provincial scale has not been well documented.

This study aimed (i) to analyze the spatio-temporal variations of N inputs to the crop production system during 1990–2017 and their effects on the NUEs of crop systems at the provincial scale; (ii) to explore the controlling factors of NUE in the provincial livestock system and suggest some measures for improving it; (iii) to analyze N costs and the NUEs of the food systems and the correlation between NUE and virtual NUE in all provinces; and (iv) to promote some strategies for improving the NUE at different stages of the provincial food system and make some recommendations for the entire Chinese food system, from the national perspective.

2. Methods and data

2.1. Description of the study boundaries

The material flow analysis approach, which encompasses the entire food production-consumption chain, was adapted for quantifying N flows of food systems at the provincial scale. The system boundaries followed the geographic boundaries of 31 provinces, and excluded Taiwan, Hong Kong and Macao because of limited data availability. The food system included crop production, animal production, aquatic production, food processing, household consumption (including both urban and rural households), and waste disposal systems (Fig. S1). The crop-production system included 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 almost all of the total area sown in each province. The animal-production system included hogs, breeding sows, dairy cattle, beef cattle, draft cattle, laying hens, broilers, sheep, horses, mules, donkeys, and rabbits. The aquaculture system included both farming and wild-capture fishing, both freshwater and seawater. The food-processing system included storage, transportation, processing, packaging, and retail sectors. We assumed that the difference between food N supply and final consumption by residents was the N stock in some stages and the N loss in food processing, because no Nr loss data were available at these sector levels (Gao et al., 2018b; Ma et al., 2010). The household-consumption system included rural and urban household diets. The division between rural and urban households was based on national statistical information (NBSC, 1991–2018). The waste-disposal system included human and animal excreta, food processing wastes, kitchen wastes and crop residues.

Distinctions were made among N recycled inside the food system, such as seeds, grain feed, manure, crop residues, and food waste. New N imported from outside the provincial food systems includes N from chemical fertilizers (pure N and compound fertilizers), BNF, atmospheric deposition, irrigation, imported animal feed, and fish and seafood from the aquatic systems, and there may also be some N embodied in net imported food in provincial trade. The compound fertilizer N applied to cropland was calculated as the amount of compound fertilizer multiplied by an N content of 30% for compound fertilizer in China as a whole, as has been done in previous studies on N cascade flows in China (Cui et al., 2013; Gu et al., 2015), because of limited data availability at the provincial scale. The net imported food can be calculated by the differences between various foods' supply and supply demands (Gao et al., 2019b; Ma et al., 2012). The calculations of different foods' supply and supply demands were given in the SI text. The net imported food was viewed as the sum of the gaps between the different foods' supply demands, minus the actual food supplies. Then the net imported food N was calculated by different foods' net import rates multiplied by the N content in the corresponding food stuffs.

Mass balances (inputs = outputs + accumulations) (Eq. S1) were used as the basic principle for calculating N input, output, and accumulation values in different sectors of the provincial food systems, if no data were available for determining them directly (Ma et al., 2012; Gu et al., 2015): for example, net imported feed N to the animal-food production system at the provincial scale (see SI for details).

2.2. Data collection

The basic data used in this study—such as population, arable land, fertilizer usage, crop planting area and yields, livestock production and products, per capita food consumption, urbanization rates, etc.—were mainly taken from national statistical yearbooks. The second category of data was the coefficients used for the calculation of N fluxes (Tables S1–15), they were mainly obtained from the literature and can be divided to four sub-categories as described in the SI text–. And some coefficients were calculated based on national statistical information and official survey data. For example, the ratios of the selected foods consumed away from home were calculated based on the published survey data for 9 provinces from the China Health and Nutrition Survey (CHNS) (Table S16). The detailed principle of the ratios of the selected foods consumed away from home for non-sampling provinces was given in the SI text. The ratios of different food supplies and consumptions used the means for China as a whole (Table S13) because no data were available at the provincial scale; these were calculated by the selected foods' consumption divided by corresponding supplies, which were calculated by main crop products in the constructed database in our previous study (Gao et al., 2018b) multiplied by relative proportions of harvested main crop products as foods in China, in different decades (Table S12), and the same method was used as before, for the years from 2013 to 2017.

Per capita food consumption amounts in the home were taken from statistical yearbooks: these included 16 categories (cereal, starches, pulses, vegetables, vegetable oils, fruit, pork, beef and mutton, poultry, milk, eggs, fish, cake, tea and sugar). We then corrected total food consumption by the mean ratios of food consumption away from home in each province as mentioned above (Table S16), and calculated per capita food N consumption by multiplying the amounts of different foods by the corresponding N contents (Tables S4–5).

2.3. Calculations of NUE at different stages of the food system, and N costs

NUE in crop production (NUE_c) and animal production (NUE_a) were defined by the ratio of the crops' aboveground N uptake, the N embodied in the outputs of livestock and poultry, eggs, milk, and fish, and the total N input to crop and animal production systems, respectively (Gao et al., 2018b; Ma et al., 2012):

$$NUE_c = \left(W_{i,g} \times N_{i,g} + W_{i,g} \times r_{i,s/g} \times N_{i,s}\right)/I_c \times 100\%$$

$$NUE_a = W_{i,m\&p} \times N_{i,m\&p}/I_a \times 100\%$$

where $W_{i,g}$, $N_{i,g}$, $r_{i,s/g}$, $N_{i,s}$ represent crop *i*'s grain yield, the N content in crop *i*'s grain, crop *i*'s straw and grain ratio, and the N content in crop *i*'s straw, respectively; I_c represents total N input to the crop system, including chemical fertilizers, BNF, atmospheric N deposition, irrigation, seeds, crop residues, human excrement, animal manure, and food-waste compost applied to fields; $W_{i,m\&p}$ represents animal meat and product *i*'s yield, including pork, beef and mutton, poultry, eggs, milk, fish and rabbit meat; $N_{i,m\&p}$ represents animal meat and product *i*'s N content; I_a represents total N input to the animal system, including grain feed, straw as feed, food process waste as feed, food waste as feed, green fodder, and feed imported from outside.

N cost (NC, kg N kg^{-1} food N) as mentioned above can be calculated using Eq. 3 (below) and it only reflects the apparent N cost (Gao et al., 2019b; Ma et al., 2014). The apparent N cost of the food system is not suitable for reflecting the actual environmental N effects of the final food consumption in a province, as explained above. Virtual N cost of food (VNC, kg N kg^{-1} food N) was therefore proposed for tracking the actual environmental N effect of total food consumption at the regional scale (Gao et al., 2019b; Huang et al., 2019); it can be calculated using Eq. 4, and the virtual NUE of a provincial food system was calculated as 1 divided by the VNC.

$$NC = \left(I_{c+a} + F_{aqu} + F_{wc} + Feed_{Net-imp} + Food_{Net-imp}\right)/FN_{con}$$

$$VNC = \left(I_{c+a} + F_{aqu} + F_{wc} + Feed_{VN-Net-imp} + Food_{VN-imp} - Food_{VN-exp}\right)/FN_{con}$$

where I_{c+a} represents the total new N input to the crop and animal production systems as mentioned above; F_{aqu} represents feed N input to freshwater and seawater aquaculture (its calculation is described in the SI text); F_{wc}, $Feed_{Net-imp}$, $Food_{Net-imp}$, FN_{con}, $Feed_{VN-Net-imp}$, $Food_{VN-imp}$, $Food_{VN-exp}$, $Feed_{VN-Net-imp}$ and $Food_{VN-imp}$ as described in our previous study (Gao et al., 2019b). Many studies have already quantified agricultural products' input-output at the provincial level in China (Li et al., 2013; Hu et al., 2018), but their results included N from manure and/or seed that did not belong to the new N input to the food system, in the calculation of the N cost of food consumption (Ma et al., 2012) and they lacked time series data. Hence, we calculated the new N exported from a province as local plant food N (PN) and animal food N (AN) export rates multiplied by the N cost of locally produced PN and AN in the corresponding year, because no data were available for calculating the detailed N cost of these foods at the provincial scale (Gao et al., 2019b).

2.4. Uncertainty analysis

To minimize the uncertainty of our analysis to the maximum extent possible, we used the associated parameters reported in the literature for the provincial situations for calculating some N inputs, such as atmospheric N deposition, relative partitioning of total straw use, ratios of food consumed away from home, etc. But there are still uncertainties in estimating N inputs and outputs, NUEs at different stages of the food system, and the N cost of food, because of the multiple data sources and complex parameters, as cited in Tables S1 to S16. We set up different uncertainty ranges for different sources' activity data and parameters (see SI for details), and an uncertainty analysis was performed using the error propagation equation of mathematical statistics (Eqs. S2–S3) (IPCC, 2001) (see SI for details). The means and uncertainty ranges are reported in the figures.

3. Results and discussion

3.1. Spatio-temporal variations of N application per unit agricultural land area at the provincial scale

Total N input to the Chinese crop system increased from about 31.7 Tg in 1990 to 50.8 Tg in 2014, then declined to 49.5 Tg in 2017 (Fig. S2 and S3), the reasons for this variation were given in the SI text. At the same time, N application rates per unit agricultural land area at the provincial scale showed large spatio-temporal variations among the different provinces (Fig. 1), varying from around 142 kg N ha^{-1} in Heilongjiang to 760 kg N ha^{-1} in Shanghai in 1990, with the relatively high N input mainly distributed in the eastern coastal provinces (Fig. 1a). With the rising demand for food driven by population growth, N input per unit agricultural land area significantly increased, by 5–92% in most of the provinces, during 1990–2010, except for Shanghai, Zhejiang, Jiangxi and Guizhou, where it decreased by 4–24% (Fig. 1a, b, c and S4). Nevertheless, N input per unit agricultural land area decreased by 3–271 kg N ha^{-1} in 2017, relative to that of 2010, in 77% of the provinces (Fig. 1d). Under this situation, N input per unit agricultural land area of the whole Chinese crop system decreased by 41 kg N ha^{-1} in 2017–10% of the total N input per unit cropland in 2010. This decrease was attributed to the long-term effort to implement soil testing and fertilizer recommendation technologies since 2005 as mentioned above (Zhang et al., 2016; Cui et al., 2018), and the effort to achieve zero growth of chemical fertilizer use by 2020 (Gu et al., 2015).

3.2. Impacts of N input on the NUE of crop systems at the provincial scale

With the growth in N application intensities, the NUEs of the crop systems slowly decreased in a majority of the provinces during 1990–2005 (Ma et al., 2012), but they continuously increased after 2005 in the majority of provinces (Fig. S5a), because of the continuous increase in gross grain yield in the past decade (NBSC, 2006–2018) combined with the decrease in per unit area N input (Fig. 1d) from popularizing optimized soil–crop fertilization management technologies (Zhang et al., 2016; Cui et al., 2018). The NUEs of the crop system fell below 50% in most provinces, much lower than that in North America or Europe, where it is typically above 50% (Gu et al., 2015). And we found that the aboveground N uptake had a significantly positive relationship

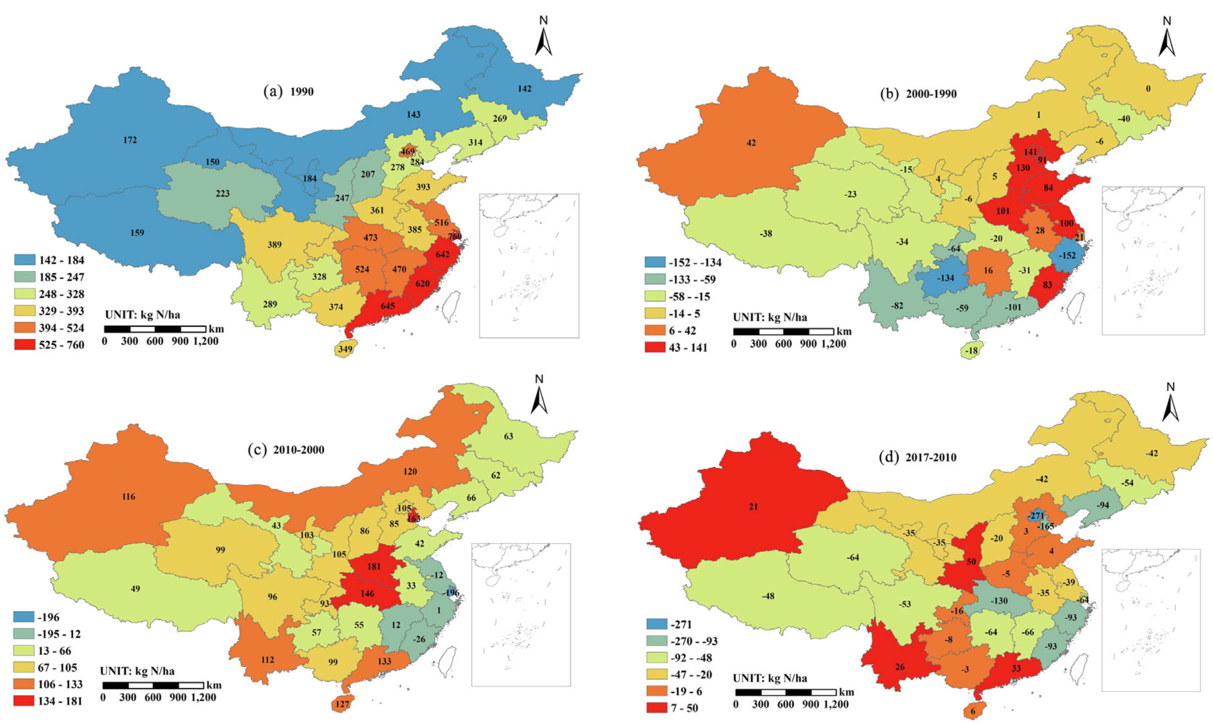

Fig. 1. N application rates per unit agricultural land at the provincial scale in 1990, and the relative changes of N input during 1990–2000, 2000–2010 and 2010–2017, 2000–1990, 2010–2000 and 2017–2010 represents N input in the year before minus the year after.

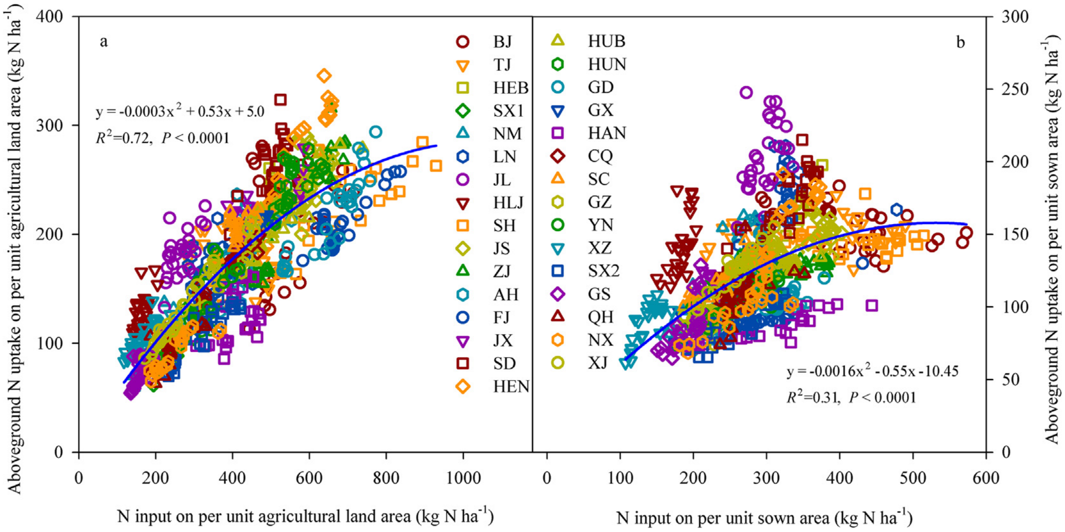

Fig. 2. The correlation of N input per unit agricultural land with aboveground N uptake at the provincial scale. The abbreviations BJ, TJ, HEB, SX1, NM, LN, JL, HLJ, SH, JS, ZJ, AH, FJ, JX, SD, HEN, HUB, HUN, GD, GX, HAN, CQ, SC, GZ, YN, XZ, SX2, GS, QH, NX, and XJ represent Beijing, Tianjin, Hebei, Shanxi, Inner Mongolia, Liaoning, Jilin, Heilongjiang, Shanghai, Jiangsu, Zhejiang, Anhui, Fujian, Jiangxi, Shandong, Henan, Hubei, Hunan, Guangdong, Guangxi, Hainan, Chongqing, Sichuan, Guizhou, Yunnan, Tibet, Shaanxi, Gansu, Qinghai, Ningxia, and Xinjiang provinces, respectively. These abbreviations are also used in subsequent figures.

with N input on per unit agricultural land area (P < 0.0001) (Fig. 2a), but the more N input, the lower the NUE of the crop system (the slope of the fit curve between aboveground N uptake and N input on per unit agricultural land area) when N input per unit agricultural land area exceeded about 400 kg N ha^{-1} at the provincial scale. Chinese cropping systems usually receive excessive amounts of N fertilizer, about two to even tens times greater than the actual crop demand (Ju et al., 2009; Nayak et al., 2015). Hence, improvements in the NUE of the crop system should first focus on decreasing the N input per unit agricultural land area at the provincial scale by popularizing advanced production technologies, as mentioned above (Zhang et al., 2016; Cui et al., 2018), especially for 10 provinces with an N input per unit agricultural land over 450 kg N ha^{-1} in 2017 (Fig. S4).

We also analyzed the relationship between aboveground N uptake and N input on per unit sown area, to explore the effect of crop structure on the NUEs of crop systems at the provincial scale (Fig. 2b). The highest NUE of the crop system appeared in Heilongjiang (Fig. S5a), where it reached 64–91% in our study period, because maize and legume crops accounted for about 52–71% of the total crop sown area, and these two have higher NUEs than other crops (Cui et al., 2018; Gao et al., 2018a). In addition, Heilongjiang has adopted higher mechanization and advanced management practices, and has large-scale farms, in contrast to the smallholder farms in other regions, and these factors could help improve crop yields and the NUE of crop production (Ju et al., 2016; Zhang et al., 2016). Jilin and Liaoning also had high NUE values, >60% in most years, under the same N input (around 300 kg N ha^{-1}) as other regions, because maize accounted for about 52–69% and 35–61% of the total crop sown area in the two sites, respectively, and the area hosts relatively large-scale farms compared to the smallholder farms in other regions (Ju et al., 2016). The highest N input on per unit sown area exceeded 500 kg N ha^{-1} in Beijing and Shanghai in some years (Fig. 2b), where the NUE of crop production fell below 30%, because they produce larger proportions of vegetables and fruits (NBSC, 1991–2018), which have higher N input and lower efficiencies than other crops (Cui et al., 2016; Guo et al., 2017). The lowest NUE of all the crop systems—around 25%—appeared in Hainan in recent years (Fig. S5a), because of the vegetable- and fruit-sown areas accounted for > 40% of the total crop sown area, after 2005. Clearly, the NUE of the crop system could be improved by increasing maize and legume cultivation in some provinces, in addition to reducing per unit area input. It was indicated that, increasing maize and legume crop planting not only contributes to improving the NUE of crop systems in low-efficiency areas, but also matches the trend of increasing requirement for maize in animal feeds as the demand for animal protein increases (Chen et al., 2014) and the expected decrease in rice demand with diets change (Gao et al., 2019a), as economic development proceeds and more soybean demand is driven by the recommended diets (Bai et al., 2018b).

3.3. Breeding structure effects on the NUE of the animal system at the provincial scale

China is facing great challenges for reducing environmental N pollution, because animal food consumption is expected to continue to expand, because 35% and 68% of provinces' urban and rural households, respectively, were still below the recommended minimum animal food N consumption in 2017 (Fig. S6). An exciting result is that we found that the NUE of the animal food production system is significantly positively correlated with the share of poultry, eggs, milk and fish in the total animal food supply at the provincial scale (P < 0.001) (Fig. 3). This is consistent with the policy of encouraging people to eat more poultry, eggs, milk and fish, to replace high N-cost foods such as pork, beef and mutton (Leach et al., 2012; Zhang et al., 2019; Gu et al., 2019), and to achieve the recommended diets in the CHNS (CHNS, 2016). The lowest NUE for the animal systems appeared in Guizhou, Yunnan, Qinghai, Gansu, and Tibet (Fig. S5b), because beef, mutton, donkey meat, and horsemeat together accounted for 38–71%, 30–67%, 55–74%, 76–87%, and 71–90% of the total animal products in these provinces, respectively. The production of these meats has higher N costs than those of poultry, eggs, milk and fish (Leach et al., 2012; Pierer et al., 2014; Gu et al., 2019).

Under the same proportion of poultry, eggs, milk and fish in their total animal food production as Ningxia, Inner Mongolia, and Heilongjiang, the NUEs of animal food production systems in Jiangsu, Zhejiang, and Fujian exceeded 20% in most years, and the highest value was close to 40% in Fujian, because fish production accounted for 18–34%, 37–72%, 41–62% of total animal food production, respectively, in the three coastal provinces, and fish

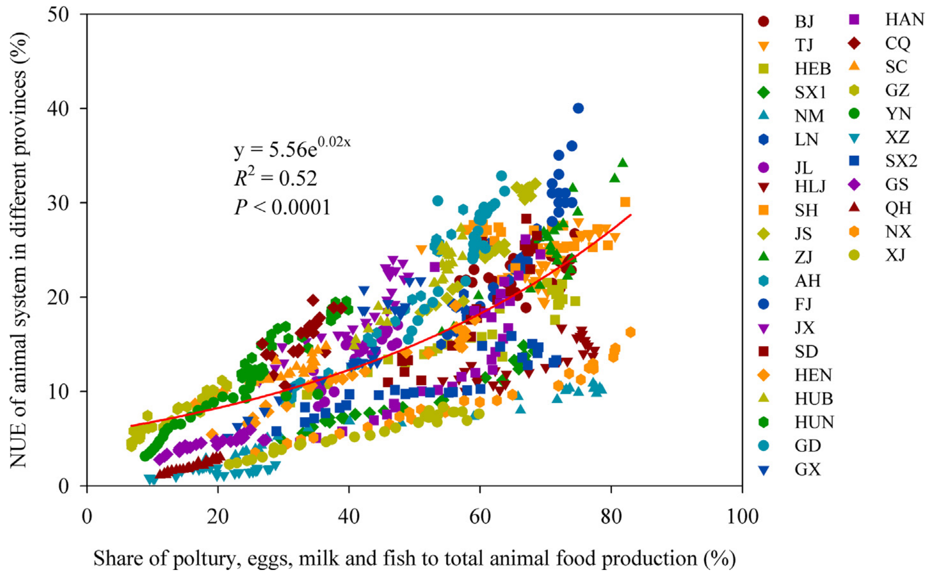

Fig. 3. Effect of the share of poultry, eggs, milk and fish in the total animal food production, on the NUE of animal systems, at the provincial scale.

production has relatively low N cost than those of beef, pork or poultry in China as a whole per unit N than other animal products (Guo et al., 2017; Liang et al., 2016). That's why some studies have advocated that local governments should offer incentives for livestock farmers to switch from producing beef and pork to producing milk, chicken and fish (Zhang et al., 2019; Gu et al., 2019). However, in the future, milk production will have to increase if per capita milk consumption rises to the recommended amount, and this in turn will accelerate environmental N losses from the dairy sector if production efficiencies are not improved; Soy milk is being promoted as a substitute for animal milk, given its similar nutritional value and low environmental N effects compared to animal milk (Bai et al., 2018b), and more soybean sowing can help to improve the NUE of a crop system, as mentioned above. In addition, the increases in fish consumption could help improve the NUE of the animal system relative to the same amount of increases in pork, beef and poultry consumption, but increasing fish from wild capture is not worth advocating because China is already suffering from over-exploitation of marine resources (Blomeyer et al., 2012), China's aquaculture sector is destined to diminish wild fish stocks worldwide (Cao et al., 2015), and the increased fish production might further accelerate the already serious water eutrophication in China (see SI for details). Hence, several strategies might be pursued, as alternative ways of meeting this fish demand: rather than exporting fish from some low-efficiency regions, increase trade from provinces where production efficiency could be improved; this could mitigate the pressure from increasing domestic demand. In addition, fish imports from Oceania, South and North America and the EU, which have relatively higher fish production efficiencies than China, could be modestly increased (Blomeyer et al., 2012; Cui et al., 2016).

3.4. N costs of food and NUE of food systems at the provincial scale

N costs of food in China as a whole increased from 6 kg N kg^{-1} food N to 10 kg N kg^{-1} food N during 1990 to 2012, then declined to about 9 kg N kg^{-1} food N in 2017 (Fig. 4a). This shows a trend similar to the N cost of Chinese food as a whole, which increased dramatically from around 6 kg N kg^{-1} food N during 1980–1990 to 10–11 kg N kg^{-1} food N during 2005–2012 (Gao et al., 2018b; Ma et al., 2012), close to the global mean N cost value of 11 kg N kg^{-1} food N (Galloway and Cowling, 2002). At the provincial scale, N costs showed large imbalances, varying from 2 kg N kg^{-1} food N to 53 kg N kg^{-1} food N in 1990, and from 3 to 27 kg N kg^{-1} food N in 2017. Lower N costs, ranging from 2 to 6 kg N kg^{-1} food N, appeared in Beijing, Tianjin, Shanghai, Zhejiang, and Guangdong, because large amounts of N were imported in the form of food in highly urbanized areas, and higher NUEs of the animal systems in Zhejiang, and Guangdong were caused by the relatively high shares of poultry, eggs, milk and fish, in the total animal food production (Fig. 3). Relatively higher N costs than that of the global mean value appeared in pasture, semi-agricultural and semi-pastoral areas (Inner Mongolia, Xinjiang, Ningxia, Qinghai, and Tibet), which can be attributed to the proportions of cows, horses, mules, donkeys, and sheep to the total livestock and poultry; these were 23–53%, 28–71%, 18–39%, 69–90% and 87–99%, respectively, in the five sites. These percentages caused the low animal system NUE as mentioned above, and the N used for grain feed and embodied in pasture were included in the N imported to the food system, but the meat from horses, mules, and donkeys was excluded from the household consumption figures (Gao et al., 2019b; Ma et al., 2012). Hence, the N cost might be overestimated in some provinces where horses, donkeys, and mules are used mainly as draft animals and their meat is excluded from food consumption (Gao et al., 2019b; Ma et al., 2014). In that case, the N input for feeding these animals should be excluded from the calculations of N cost of the food system, in future studies.

N cost can easily be transformed into the NUEs of food systems, which can be helpful for comparing the efficiencies of different regions (Galloway and Cowling, 2002; Ma et al., 2012). During the study period, the overall trend of NUE in the Chinese food system was downward, dropping from 16% in 1990 to around 10% in 2017 (Fig. 4b), consistent with the trend of 16% to 9% during 1980 to 2005 (Ma et al., 2012), only slightly higher than the mean NUE of the global food system, 9% (Galloway and Cowling, 2002). The latest study indicated that the NUE of the Chinese food system as a whole would decrease to 8% in 2030 if current trends continue (Ma et al., 2019). Among the provinces studied here, the NUE of the food system varied from about 2% to 40% in 1990, and from 4% to 34% in 2017 (Fig. 4b). The relatively low NUE values in the food system (<6%) appeared in the high N cost areas as mentioned above. The relatively high NUE of the food system (>20%) appeared in highly urbanized areas—Beijing, Tianjin, Shanghai and Zhejiang, because these regions depend more and more on direct food

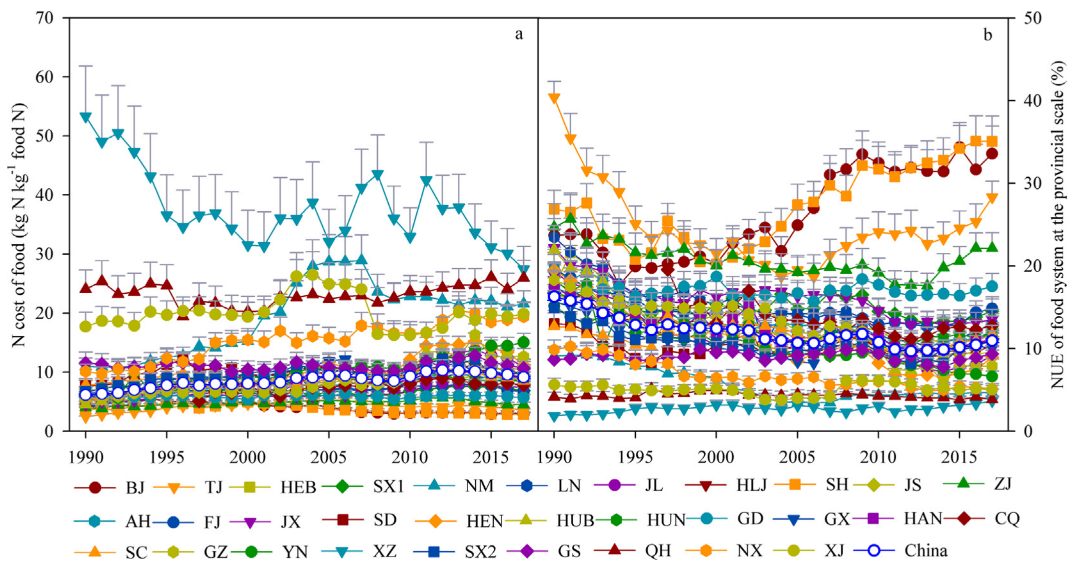

Fig. 4. N costs and the NUEs of provincial food systems.

imports from outside as they rapidly urbanize (Gao et al., 2019b; Ma et al., 2014). However, the above NUEs of the food systems neglected the virtual N inputs caused by the imported food in its production stage (Burker et al., 2009). This NUE could be viewed as apparent NUE (Gao et al., 2019b; Ma et al., 2014), and it might be overestimated in the areas where large amounts of food and feed are imported from outside, or it might be underestimated in areas where large amounts of food and feed are exported to outside.

3.5. Impacts of food N imports on new N applied and the NUE of the food system at the per capita scale

There are indications that the demand for N fertilizer in China will decrease by about 10% compared with that in the business-as-usual (BAU) scenario, if it is assumed that food and feed imports will increase by 10% by 2030 (Ma et al., 2019). We therefore hypothesized that the net imported food N will generate some effects on the above-mentioned new N imported from outside the provincial food systems. The correlation of the net imported food N with new N imported from outside and the NUE of the provincial food system at the per capita scale were analyzed, because there is no comparability among provinces because of the different population volumes. The result clearly shown that, per capita new N applied in the food system showed a significantly negative linear correlation ($P < 0.0001$) with per capita net imported food N at the provincial scale, excluding Inner Mongolia, Xinjiang, Qinghai and Tibet (Fig. 5a). The relatively higher per capita new N applied in the food system of these four provinces might be attributable to the combined effects of the high N cost of food production caused by their special production systems and feeding structures, and the relatively low population densities of those two areas. It has been shown that the mean new N applied in the food system is about 42 kg cap.$^{-1}$ when no food is imported or exported, the N cost of food is 8–11 kg N kg^{-1} food N based on 4–5 kg food N cap.$^{-1}$ consumed per year (Gao et al., 2018b; Gu et al., 2015), closes to the values of N costs as mentioned above, and the new N applied in provincial food systems will decrease by about 5 kg when 1 kg food N is imported from outside, at the per capita scale. In addition, we found that the NUEs of the provincial food systems have an exponential growth correlation with per capita net imported food N, which was also excluded the above four provinces (Fig. 5b). The NUEs of provincial food systems will increase 13% when the net imported food N increases by 1 kg cap.$^{-1}$. These results indicate that the higher the amount of food N imported, the lower the per capita new N applied in the food system, and therefore the relatively high NUE of the food system; examples include Beijing, Tianjin, and Shanghai (Fig. 5b). In contrast, the higher the amount of food exported, the more the N input to the food system; examples include Shandong, Henan, and Heilongjiang, which are all located in major crop production areas (Gao et al., 2018a). Although the highest food N exports were in Heilongjiang, the per capita new N applied in the food system there was significantly lower than those of Inner Mongolia, Xinjiang, Qinghai or Tibet, because Heilongjiang mainly exports cereal products and milk, and these have the highest crop system NUE in China (Fig. S5) (Ma et al., 2012), while milk has a relatively low N cost compared to beef and mutton (Cui et al., 2016). Hence, any attempts to improve the NUE of a food system should focus not only on improving the NUEs of the different food production stages (Oenema et al., 2014; Cui et al., 2018; Bai et al., 2018a), but also on modestly increasing food or feed imports from abroad, in order to ease the pressure on natural resources and mitigate the environmental effects of food production in China (Ma et al., 2019).

3.6. Virtual NUE of food systems at the provincial scale

The range of virtual NUE—from 1% to 19% in 1990 and from 2% to 12% in 2017—became smaller than that of the apparent NUE (Fig. 6a). And we found that most of the ratios between NUE and virtual NUE were below the 1-to-1 line (Fig. 6b). The more the virtual NUE deviates from the 1-to-1 line, the higher the virtual N imported. If virtual N input to the imported food production process was not taken into account, the NUE of the food system would be significantly overestimated, by 83%–416%, 111%–232%, 75%–302%, 31%–137% and 28%–103% in Beijing, Tianjin, Shanghai, Zhejiang and Guangdong, respectively. Hence, virtual N input should receive more concern when quantifying the N demand of the food system at the provincial scale, especially in highly urbanized areas (Burker et al., 2009; Gao et al., 2019b); this can help achieve a more accurate evaluation of the NUE of provincial food systems and contribute to improving the NUE of the Chinese food system as a whole.

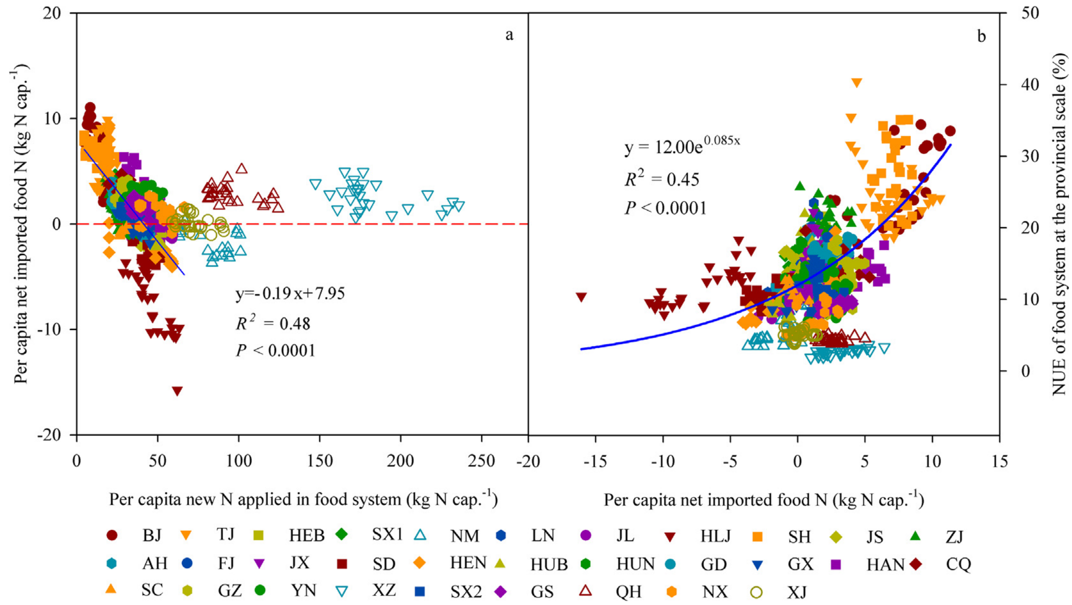

Fig. 5. Relationship of the net imported food N with new N applied in food system at per capita scale, and its effect on NUE of food system (Blank dots were not included in the correlation analysis).

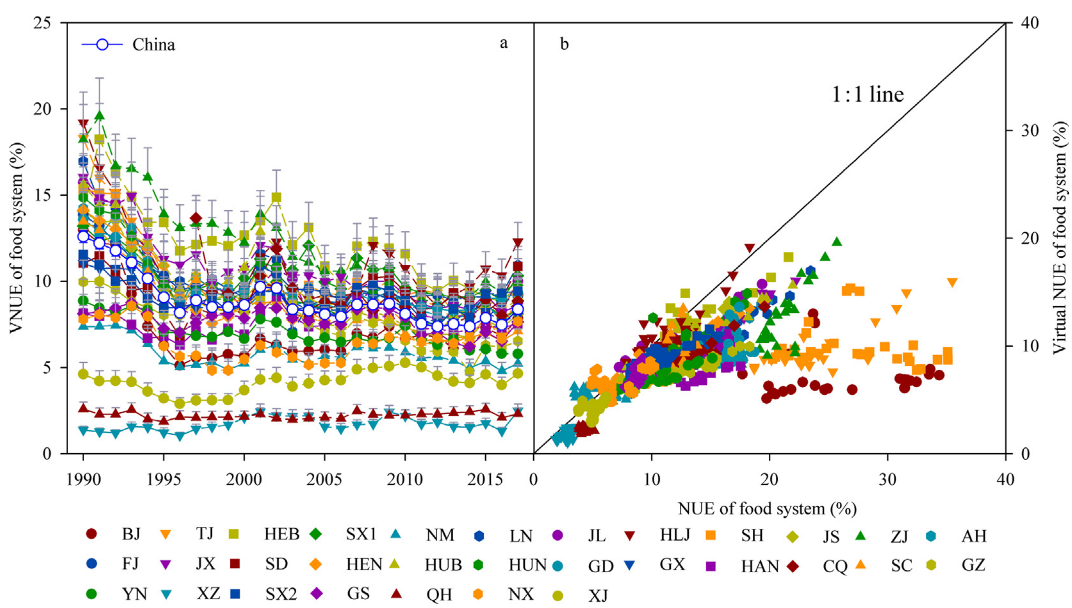

Fig. 6. Virtual N use efficiencies of food system and its comparison with NUE of food system at the provincial scale.

3.7. Limitations and future research

There are uncertainties in estimating some Nr fluxes, for example, per capita food consumption at home in 31 provinces were collected from China's statistical data, and the ratios of the selected foods consumed away from home in the nine surveyed provinces were used as described above. But food preferences based on religion (Kearney, 2010) and the increasing popularity of door-to-door food delivery service in Chinese cities (Song et al., 2018) would both have some effect on the frequency of dining out. Hence, using China's statistical data and CHNS data (see SI for details) is a reasonable method, because these two influences were also covered in randomized investigation samples. While age, gender, education, and some other parameters might also have a bearing on per capita food consumption at home and the ratio of food consumption away from home, nevertheless, none of these parameters has been directly reported.

In addition, we estimated per capita food consumed away from home for the non-sampling years during the period 1990–2017, by linear interpolation between each two adjacent intervals of the surveys, a frequently used simple trend projection method (Li, 2007). At the same time, China has adopted an anti-corruption policy in the past few years, and this might contribute to a decrease in dining out, but we did not consider this factor in the calculations because no related data were available. Hence the impacts of religion, the anti-corruption policy, and take-out food consumption on food consumption N and the NUE of the food system are worth further study.

N cost is an important indicator for analyzing the new N input to a food system when per unit N embodied in food stuffs was consumed in or exported from another region (Bleken and Bakken, 1997; Ma et al., 2012). To the best of our knowledge, although many studies have already quantified the agricultural N input-output at the provincial level in China (Li et al., 2013; Hu et al., 2018), there is still a lack of data on the N costs of the selected foods at the provincial scale. The N costs of the selected foods at the provincial scale are required in order to accurately quantify new N input to a regional food system, as food trade becomes more and more frequent, and it is very important for implementing regional or national sustainable N management.

Harvested crop products and animal foods will always be used as food and feed or others (Ma et al., 2012), or lost in food processing and food preparation (Leach et al., 2012; Guo et al., 2017), and some food stocked in the retail stage may spoil or expire before it is put out for sale (Chen et al., 2006). It was found that the food loss rates over the entire supply chain were 19%, 20–30%, and 5–15% for grains, vegetables and fruits, and eggs, respectively (Liu et al., 2013), and the total food N loss ratio was 13–35% in China as a whole from 1990 to 2009 (Cui et al., 2013). Hence, the ratios of the selected foods' consumptions compared to supplies need to be considered when evaluating the new N input to the food system, as these values are driven by the final food consumption. In this study we used the ratios of different foods' consumptions to supplies for China as a whole, because no data were available at the provincial scale. We assumed that the uncertainty driven by this parameter was ±50%, but it would take <6% uncertainty in the final N costs and NUEs of the food systems for all provinces in recent years (assuming that the ratios of different food consumptions and supplies are constant) to produce this effect, because the net imported food N accounts for only a small proportion of total N input to the provincial food system (except for the highly urbanized areas—Beijing, Tianjin and Shanghai, where more and more food is imported from outside (Gao et al., 2019b; Ma et al., 2014). Hence, using the mean of the ratios of different food consumptions and supplies for China as a whole at the provincial scale would have only a small effect on the N costs in our study, except for Beijing, Tianjin and Shanghai.

4. Conclusions

We investigated the spatio-temporal dynamics of nitrogen use efficiencies and their controlling factors in different stages of the food system at the Chinese provincial scale during 1990 to 2017. We found that the increased intensities of per unit agricultural land area N input during 1990 to 2010, caused by the demand for food and feed attributable to population growth and dietary changes, has resulted in a low NUE for the crop system when N input per unit agricultural land area exceeded about 400 kg N ha^{-1}, but this situation has brought some positive changes. N input per unit agricultural land area decreased by 3–271 kg N ha^{-1} in 77% of the provinces in 2017, relative to that of 2010. And we found that planting more high N-efficient maize and legume crops can help improve the NUE of a crop system. Fortunately, this finding fits well with the trend of increasing the maize requirement in animal feeds, as the demand for animal protein increases and staple foods (such as rice) decrease in the human diet as economic development proceeds. It also fits with the increasing interest in soy milk, for fulfilling the recommended milk consumption. Although the NUE of the animal system has increased, China faces a great challenge of meeting the higher demand for animal food from the increasingly wealthy population, while simultaneously reducing environmental N pollution. This situation can be improved by increasing the share of animal food contributed by poultry, eggs, milk and fish to align with the recommended diets.

The government and policy makers should attempt to modestly increase net food imports for improving the NUE of the food system. Nevertheless, this indicator might significantly underestimate the environmental N effect of a provincial food system, especially the areas where large amounts of food and feed are imported. Virtual NUE must be considered for the comparisons of NUE at the provincial scale, especially in highly urbanized areas. The high N costs and low NUEs in pastoral regions are mainly driven by their special production system and feeding structure. N input for non-food animals should be excluded from the calculations of N cost and NUE, for an accurate evaluation of the NUE of the food system in these areas. In addition, there is still plenty of work to do, in exploring the effects of food preferences based on religion, take-out food consumption and the anti-corruption policy, on food consumption and the NUE of the food system, and more parameters as mentioned above are required for accurate analysis of the NUE of provincial food systems.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2019.134861.

References

Bai, Z.H., Ma, W.Q., Ma, L., Velthof, G.L., Wei, Z.B., Havlík, P., Oenema, O., Lee, M.R.F., Zhang, F.S., 2018a. China's livestock transition: driving forces, impacts, and consequences. Sci. Adv. 4 (7), eaar8534.

Bai, Z.H., Lee, M.R., Ma, L., Ledgard, S., Oenema, O., Velthof, G.L., Ma, W.Q., Guo, M.C., Zhao, Z., Wei, S., Li, S., Liu, X.J., Havlik, P., Luo, J., Hu, C.S., Zhang, F.S., 2018b. Global environmental costs of China's thirst for milk. Glob. Change Biol. 24 (5), 2198–2211.

Bleken, M.A., Bakken, L.R., 1997. The nitrogen cost of food production: Norwegian society. Ambio 26, 134–142.

Blomeyer, R., Goulding, I., Pauly, D., Sanz, A., Stobberup, K., 2012. The role of China in world fisheries. (European Parliament, Directorate General for Internal Policies, Brussels.

Burker, M., Oleson, K., Cullough, E.M., Gaskell, J., 2009. A global model tracking water, nitrogen, and land inputs and virtual transfers from industrialized meat production and trade. Environ. Model. Assess.14(2), 179–193.

Cao, L., Naylor, R., Henrisksson, P., Leadbitter, D., Metian, M., Troell, M., Zhang, W.B., 2015. Chinas aquaculture and the worlds wild fisheries. Science 347 (6218), 133–135.

Chen, X.K., Wang, C.Y., Diao, W.J., 2006. Designing a modern food circulation system in the Chinese metropolis. Bus. Econ. Adm. 10 (180), 22–27.

Chen, X.P., Cui, Z.L., Fan, M.S., Vitousek, P., Zhao, M., Ma, W.Q., Wang, Z.L., Zhang, W.J., Yan, X.Y., Yang, J.C., Deng, X.P., Gao, Q., Zhang, Q., Guo, S.W., Ren, J., Li, S.Q., Ye, Y.L., Wang, Z.H., Huang, J.L., Tang, Q.Y., Peng, X.L., Zhang, J.W., He, M.R., Zhu, Y.J., Xue, J.Q., Wang, G.L., Wu, L., An, N., Wu, L.Q., Ma, L., Zhang, W.F., Zhang, F.S., 2014. Producing more grain with lower environmental costs. Nature 514, 486–489.

CHNS (Chinese Health and Nutrition Society), 2016. Dietary Guidelines for Chinese Residents. The Tibet People's Publishing House, China.

Cui, S.H., Shi, Y.L., Groffman, P.M., Schlesinger, W.H., Zhu, Y.G., 2013. Centennial-scale analysis of the creation and fate of reactive nitrogen in China (1910–2010). Proc. Natl. Acad. Sci. U.S.A. 110, 2052–2057.

Cui, S.H., Shi, Y.L., Malik, A., Lenzen, M., Gao, B., Huang, W., 2016. A hybrid method for quantifying China's nitrogen footprint during urbanization from 1990 to 2009. Environ. Int. 97, 137–145.

Cui, Z.L., Zhang, H.Y., Chen, X.P., Zhang, C.C., Ma, W.Q., Huang, C.D., Zhang, W.F., Mi, G.H., Miao, Y.X., Li, X.L., Yang, J.C., Wang, Z.H., Ye, Y.L., Guo, S.W., Lu, J.W., Huang, J.L., Lv, S.H., Sun, Y.X., Liu, Y.Y., Peng, X.L., Ren, J., Li, S.Q., Deng, X.P., Shi, X.J., Zhang, Q., Yang, Z.P., Tang, L., Wei, C.Z., Jia, L.L., He, M.R., Tong, Y.A., Tang, Q.Y., Zhong, X.H., Liu, Z.H., Cao, N., Kou, C.L., Ying, H., Jiao, X.Q., Zhang, Q.S., Fan, M.S., Jiang, R.F., Zhang, F.S., Dou, Z.X., 2018. Pursuing sustainable productivity with millions of smallholder farmers. Nature 555 (7696), 363.

Galloway, J.N., Cowling, E.B., 2002. Reactive nitrogen and the world: 200 years of change. Ambio 31, 64–71.
Gao, B., Huang, T., Ju, X.T., Gu, B.J., Huang, W., Xu, L.L., Rees, R.B., Powlson, D.S., Smith, P., Cui, S.H., 2018a. Chinese cropping systems are a net source of greenhouse gases despite soil carbon sequestration. Glob. Change Biol. 24, 5590–5606.
Gao, B., Huang, Y.F., Huang, W., Shi, Y.L., Bai, X.M., Cui, S.H., 2018b. Driving forces and impacts of food system nitrogen flows in China, 1990 to 2012. Sci. Total Environ. 610–611, 430–441.
Gao, B., Huang, W., Wang, L., Huang, Y.F., Ding, S.P., Cui, S.H., 2019a. Driving forces of nitrogen flows and nitrogen use efficiency of food systems in seven Chinese cities, 1990 to 2015. Sci. Total Environ. 676, 144–154.
Gao, B., Huang, W., Xue, X.B., Hu, Y.C., Huang, Y.F., Wang, L., Ding, S.P., Cui, S.H., 2019b. Comprehensive environmental assessment of potato as staple food policy in China. Int. J. Environ. Res. Pub. Heal. 16, 2700. https://doi.org/10.3390/ijerph16152700.
Gu, B.J., Ju, X.T., Chang, J., Ge, Y., Vitousek, P.M., 2015. Integrated reactive nitrogen budgets and future trends in China. Proc. Natl. Acad. Sci. USA 112 (28), 8792–8797.
Gu, B.J., Zhang, X.L., Bai, X.M., Fu, B.J., Chen, D.L., 2019. Four steps to food security for swelling cities. Nature 566, 31–33.
Guo, M.C., Chen, X.H., Bai, Z.H., Jiang, R.F., Galloway, J.N., Leach, A.M., Cattaneo, L.R., Oenema, O., Ma, L., Zhang, F.S., 2017. How China's nitrogen footprint of food has changed from 1961 to 2010. Environ. Res. Lett. 12, 104006.
Hu, Y.C., Huang, Y.F., Tang, J.X., Gao, B., Yang, M.H., Meng, F.X., Cui, S.H., 2018. Evaluating agricultural grey water footprint with modeled nitrogen emission data. Res. Con. Rec. 138, 64–73.
Huang, W., Gao, B., Huang, Y.F., Zhang, Z.L., Xu, S., Xu, L.L., Cui, S.H., 2019. Transforming nitrogen management of the urban food system in a food-sink city. J. Environ. Manag. 249, 109180.
IPCC, 2001. Good practice guidance and uncertainty management in national greenhouse gas inventories. Published. Japan: IGES.
Ju, X.T., Xing, G.X., Chen, X.P., Zhang, S.L., Zhang, L.J., Liu, X.J., Cui, Z.L., Yin, B., Christie, P., Zhu, Z.L., Zhang, F.S., 2009. Reducing environmental risk by improving N management in intensive Chinese agricultural systems. Proc. Natl. Acad. Sci. USA 106, 3041–3046.
Ju, X.T., Gu, B.J., Wu, Y.Y., Galloway, J.N., 2016. Reducing China's fertilizer use by increasing farm size. Glob. Environ. Change 41, 26–32.
Kearney, J., 2010. Food consumption trends and drivers. Phil. Trans. R. Soc. B 365, 2793–2807.
Leach, A.M., Galloway, J.N., Bleeker, A., Erisman, J.W., Kohn, R., Kitzes, J., 2012. A nitrogen footprint model to help consumers understand their role in nitrogen losses to the environment. Environ. Dev. 1 (1), 40–66.
Li, S.T., He, P., Jin, J.Y., 2013. Nitrogen use efficiency in grain production and the estimated nitrogen input/output balance in China agriculture. J. Sci. Food. Agric. 93, 1191–1197.
Li, Z.M. Study on the development of food consumption and nutrition between urban and rural residents in China. Ph.D. Thesis, Chinese Academic of Agricultural Sciences, Beijing, China, 2007. (in Chinese with English abstract).
Liang, X., Leach, A.M., Galloway, J.N., Gu, B.J., Lam, S.K., Chen, D.L., 2016. Beef and coal are key drivers of Australia's high nitrogen footprint. Sci. Rep. 6, 39644.
Lin, T., Wang, J., Bai, X.M., Zhang, G.Q., Li, X.H., Ge, R.B., Ye, H., 2016. Quantifying and managing food-sourced nutrient metabolism in Chinese cities. Environ. Int. 94, 388–395.
Liu, J.G., Lundqvist, J., Weinberg, J., Gustafsson, J., 2013. Food losses and waste in China and their implication for water and land. Environ. Sci. Technol. 47 (18), 10137–10144.
Ma, L., Bai, Z.H., Ma, W.Q., Guo, M.C., Jiang, R.F., Liu, J.G., Oenema, O., Velthof, G.L., Whitmore, A.P., Crawford, J., Dobermann, A., Schwoob, M., Zhang, F.S., 2019. Exploring future food provision scenarios for China. Environ. Sci. Technol. 53, 1385–1393.
Ma, L., Ma, W.Q., Velthof, G.L., Wang, F.H., Qin, W., Zhang, F.S., Oenema, O., 2010. Modeling nutrient flows in the food chain of China. J. Environ. Qual. 39, 1279–1289.
Ma, L., Velthof, G.L., Qin, W., Zhang, W.F., Liu, Z., Zhang, Y., Wei, J., Lesschen, J.P., Ma, W.Q., Oenema, O., Zhang, F.S., 2012. Nitrogen and phosphorus use efficiencies and losses in the food chain in China at regional scales in 1980 and 2005. Sci. Total Environ. 434, 51–61.
Ma, L., Guo, J.H., Velthof, G.L., Li, Y.M., Chen, Q., Ma, W.Q., Oenema, O., Zhang, F.S., 2014. Impacts of urban expansion on nitrogen and phosphorus flows in the food system of Beijing from 1978 to 2008. Glob. Environ. Chang. 28, 192–204.
Nayak, D., Saetnan, E., Cheng, K., Wang, W., Koslowski, F., Cheng, Y.F., Zhu, W.Y., Wang, K.J., Liu, J.X., Moran, D., Yan, X.Y., Cardenas, L., Newbold, J., Pan, G.X., Lu, Y. L., Smith, P., 2015. Management opportunities to mitigate greenhouse gas emissions from Chinese agriculture. Agric. Ecosyst. Environ. 209, 108–124.
NBSC (National Bureau of Statistics of China), 1991–2018. China Statistical Yearbook. NBSC Press, Beijing, China.
Oenema, O., Ju, X.T., de Klein, C., Alfaro, M., del Prado, A., Lesschen, J.P., Zheng, X.H., Velthof, G., Ma, L., Gao, B., Kroeze, C., Sutton, M., 2014. Reducing nitrous oxide emissions from the global food system. Curr. Opin. Environ. Sustain. 9–10, 55–64.
Pierer, M., Winiwarter, W., Leach, A.M., Galloway, J.N., 2014. The nitrogen footprint of food products and general consumption patterns in Austria. Food Policy 49, 128–136.
Powlson, D., Norse, D., Lu, Y.L., 2018. Agricultural development in China: environmental impacts, sustainability issues and policy implications assessed through China-UK projects under SAIN (UK-China Sustainable Agriculture Innovation Network), 2008–2017.
Song, G., Zhang, H., Duan, H., Xu, M., 2018. Packaging waste from food delivery in China's mega cities. Resour. Conserv. Recycl. 130, 226–227.
Steffen, W., Richardson, K., Rockström, J., CornellS, E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., de Vries, W., de Wit, C.A., Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B., Sörlin, S., 2015. Planetary boundaries: Guiding human development on a changing planet. Science 347. https://doi.org/10.1126/science.1259855.
Tilman, D., Balzer, C., Hill, J., Befort, B.L., 2011. Global food demand and the sustainable intensification of agriculture. Proc. Natl. Acad. Sci. USA 108 (50), 20260–20264.
Verger, Y., Petit, C., Barles, S., Billen, G., Garnier, J., Esculier, F., Maugis, P., 2018. A N, P, C, and water flows metabolism study in a peri-urban territory in France: the case-study of the Saclay plateau. Res. Con. Rec. 137, 200–213.
West, P.C., Gerber, J.S., Engstrom, P.M., Mueller, N.D., Brauman, K.A., Carlson, K.M., Cassidy, E.S., Johnston, M., MacDonald, G.K., Ray, D.K., Siebert, S., 2014. Leverage points for improving global food security and the environment. Science 345 (6194), 325–328.
Wu, S.H., Ben, P.Q., Chen, D.X., Chen, J.H., Tong, G.J., Yuan, Y.J., Xu, B.G., 2018. Virtual land, water, and carbon flow in the inter-province trade of staple crops in China. Res. Con. Rec. 136, 179–186.
Zhang, C.Z., Liu, S., Wu, S.X., Jin, S.Q., Rei, S., Liu, H.B., Gu, B.J., 2019. Rebuilding the linkage between livestock and cropland to mitigate agricultural pollution in China. Res. Con. Rec. 144, 65–73.
Zhang, W.F., Cao, G.X., Li, X.L., Zhang, H.Y., Wang, C., Liu, Q.Q., Chen, X.P., Cui, Z.L., Shen, J.B., Jiang, R.F., Mi, G.H., Miao, Y.X., Zhang, F.S., Dou, Z.X., 2016. Closing yield gaps in China by empowering smallholder farmers. Nature. https://doi.org/10.1038/nature19368.