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Driving forces of nitrogen flows and nitrogen use efficiency of food systems in seven Chinese cities, 1990 to 2015

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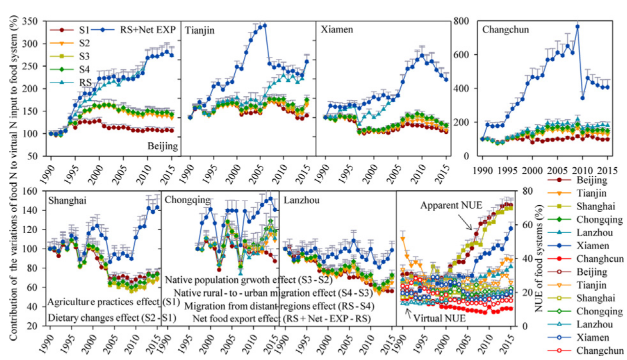
HIGHLIGHTS

- Apparent and virtual N input and driving forces were analyzed in seven cities.
- Virtual N input was significantly higher than apparent N input in majority of cities.
- Food trade has some effect on virtual N cost of cities' food N consumption.
- Migration, dietary changes and agricultural practices are the key drivers of N input.
- Virtual NUE is an accurate indicator for cities' food systems' NUE comparisons.

GRAPHICAL ABSTRACT



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ABSTRACT

The effects of population growth (PG), dietary changes (DC), native rural-to-urban migration (NM), migration from regions distant from the cities (M), and agricultural patterns and practices (AP) on N use in food systems and the food trade, and on apparent and virtual nitrogen (N) and N use efficiencies (NUE), at the city scale, are not well understood. Here we selected seven Chinese cities as the study subjects, analyzed the food trade effects on apparent and virtual N inputs and NUE, and quantified the relative magnitudes of these factors on N inputs to cities' food systems during 1990–2015, by designing several scenarios. Our results show that food-sink cities are relying more and more on external food and feed, but in 2015 they transferred 33.8–74.9% of their N input for food or feed productions to areas outside their boundaries, and the food trade showed different effects on the virtual N cost of food N consumption. Apparent NUEs of food systems were 33.1–74.9% higher than those calculated from virtual N costs in Beijing, Tianjin, Shanghai, Lanzhou and Xiamen in 2015. But in cities that export large amounts of food and feed—for example, Chongqing and Changchun—apparent NUE was underestimated by 4.0–46.4% relative to virtual NUE. Native PG, DC, NM, M, and AP accounted for 1.2–14.1%, −6.6–30.0%, 0.6–8.2%, −7.7–131.0%, and −43.8–12.8%, respectively, of the increase in virtual N inputs associated with cities' food systems in 2015, compared to 1990. Our study concludes that M, DC, and AP changes should be considered for mitigating N input in these Chinese cities, and virtual N exports induced by the food trade should also be included if the city is a net food

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exporter. Selective food trade could help improve the NUE of cities' food systems, and virtual NUE should be used as an indicator, rather than apparent NUE.

1. Introduction

Anthropogenic reactive nitrogen (Nr), mainly from fertilizers for food production, has significantly changed the global Nr cycle (Galloway et al., 2004). The biogeochemical cycle of nitrogen (N) has exceeded planetary limits, resulting in a serious threat to environmental security, and endangering global sustainability (Steffen et al., 2015). The global food-crop demand for N will increase by 100–110% compared with 2005, and global fertilizer N use will increase from 100 Mt to 225–250 Mt, by 2050 (Tilman et al., 2011). In order to ensure food security, the environmental effects of Nr must be reduced to the maximum extent possible—a worldwide challenge.

Global urbanization is driving more people into cities, especially in China, India, Southeast Asia, and Africa (UN-Habitat, 2010; Tilman and Clark, 2014), while at the same time the dietary shift toward a higher proportion of animal food is accelerating. There are indications that rural-to-urban migration over the past three decades has increased China's animal food nitrogen (AN) consumption by an additional 17%; in 2012, a resident registered as living in a city required 0.5 kg more AN yr^{-1} than one living in a rural area (Gao et al., 2018). This shift from low-AN diets to those higher in AN will require more N inputs to the food system (Tilman and Clark, 2014), because the N loss to the environment for delivering a unit of AN is higher than that for delivering a unit of plant food N (PN) (Galloway and Cowling, 2002; Tilman and Clark, 2014). Hence, urbanization has become an engine driving new N inputs to the Chinese food system (Hou et al., 2014; Gao et al., 2018).

Meanwhile, China has undergone the greatest urbanization of its history during the last few decades (Bai et al., 2014). Urbanization prompts large numbers of young people to move into cities, from relatively remote and economically stressed rural areas to more prosperous urban areas, in search of jobs and to make a better life for themselves. The outflow of young people from the countryside has slowed population growth in some areas: for example, the natural population growth rate in Northeast China has been less than 1.0‰ since 2013, significantly lower than the national mean of 5.0‰, and in 2016 it actually turned negative (NBSC, 2017). Unbalanced socioeconomic development, however, has led to large variations in urbanization rates among different Chinese provinces—from 30% in Tibet to 88% in Shanghai in 2016 (NBSC, 2017). Meanwhile, food consumption is also affected by food availability, accessibility, and choice, which in turn may be influenced by geography, demography, disposable income levels, urbanization, marketing, etc. (Kearney, 2010). Hence, food consumption patterns show significant regional differences among Chinese cities (Liu and Cai, 2014), exerting a significant effect on N input to food systems, because the N cost of food varies widely by food type (Galloway and Cowling, 2002; Leach et al., 2012; Gao et al., 2018). Moreover, N use efficiencies (NUE) of food systems for 31 provinces of China dropped significantly between 1980 and 2005, from 5 to 49% to 3–14% (Ma et al., 2012). Yet disturbingly, an understanding of the correlations between the above human socioeconomic activities and the N flows of food systems at the city scale has received little concern, even though it is essential for meeting the challenge of reducing the environmental effects of Nr while ensuring food security. The differing population growth rates, dietary changes, and rural-to-urban migration rates (both from rural areas surrounding cities and from regions distant from the cities), combined with agricultural patterns and practices in different cities, provide an opportunity to analyze and compare the forces driving new N input to Chinese cities' food systems in the emerging economy. China's N input for food must be reduced, because more and more food will be imported into and consumed in cities, as urbanization proceeds.

Nr spatial intensity (NrSI) was developed as an indicator for estimating the intensity of Nr loss to the environment from food systems (including food consumption, housing, etc.) in both agricultural and heavily populated areas on a per area basis within national boundaries, and can indicate the potential for environmental impacts, identifying Nr emission hotspots and informing management recommendations (Liang et al., 2018). However, this benchmark does not reflect the transfer of pollution connected with the virtual N input caused by the net import of food and feed between cities and their surrounding areas (Verger et al., 2018). Cities—especially megacities as net consumers—frequently import food from their neighboring rural areas, thereby driving up resource inputs and environmental costs in the areas where the food and feed is produced (Zhu et al., 2017; Verger et al., 2018). In order to implement regional or national sustainable N management, urbanization effects on external N input for food production in areas surrounding cities must be fully assessed (Lin et al., 2016), and regional coordination must be strengthened, just as carbon mitigation is currently carried out under the Clean Development Mechanism in China (Mi et al., 2016; Meng et al., 2018). Moreover, large changes in production mode and technological level lead to different virtual N factors (VNF) among regions and food products (Leach et al., 2012; Cui et al., 2016; Guo et al., 2017), yet the impacts of cities' food trade on apparent and virtual N inputs and NUEs are not receiving enough attention.

This study aimed (i) to analyze the historical trends in per capita PN and AN consumption and urban-rural differences in this consumption, and the relationships between PN and AN supply and consumption from 1990 to 2015; (ii) to compare apparent and virtual N inputs and NUEs of cities' food systems; (iii) to analyze the impacts of food trade on the N cost of local food consumption; (iv) to estimate the relative contributions of native population growth, dietary changes, rural-to-urban migration (both from rural areas surrounding the cities and from regions distant from the cities) to increases in food N consumption; and (v) to quantify the contributions of the above key drivers to inputs of N through the food system. The findings of this research could be helpful for the formulation of N reduction measures in areas of rapid urbanization.

2. Study area and methodology

2.1. Description of the selected cities and study boundaries

We selected seven typical cites: Shanghai, Chongqing, Xiamen, Lanzhou, and Changchun, Beijing and Tianjin, as the study subjects. These cities are located in East, Southwest, South, Northwest, Northeast, and North China, respectively (Fig. S1). Beijing, Tianjin, Shanghai, and Chongqing are municipalities directly under the central government; Xiamen is a vice-provincial city and special economic zone, as well as an important central city along the southeastern coast; Lanzhou and Changchun are two provincial capital cities located in relatively remote areas. The urbanization rates were 86.5%, 82.9%, 87.9%, 62.9%, 88.9%, 70.8%, and 61.0% for the above cities, respectively, in 2015: all higher than the mean 56.1% for China overall (NBSC, 2016). These cities' urbanization trends represent two different stages of a stretched-out S-shaped curve (Northam, 1979) and show four different types of urbanization processes (Fig. S2) (see SI for details).

The material flow analysis approach, which is defined as the entire food production-consumption chain, was adapted for quantifying N flows in city-scale food systems. The system boundaries followed the geographic boundaries of the studied cities. The food system was divided into four categories (Fig. S3): crop production, animal production, aquatic production, and household consumption (including both urban and rural households). The crop-production category included 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 almost all of the total area sown in each city. The animal-production category included 12 types of animals (hogs, breeding sows, dairy cattle, beef cattle, draft cattle, laying hens, broilers, sheep, horses, mules, donkeys, and rabbits). The aquaculture category included both farming and fishing, both freshwater and seawater.

The household-consumption category included rural and urban household diets. The division between rural and urban households and the quantification of native rural-to-urban migration were based on statistical information provided by the cities' governments. The rural-to-urban migration can be further separated into migrants who have registered as living in cities and have become true urbanites, and migrants who are living in cities but have not registered as urban residents (Gao et al., 2018) (see SI for details).

New N imported from outside the cities' food systems included N from chemical fertilizers, biological N_2 fixation (BNF), atmospheric deposition, irrigation, imported animal feed, and fish and seafood from aquatic systems, and there may have been some N embodied in net imported food in provincial trade. The net imported food can be calculated by the differences between a given food's supply and consumption amounts (Ma et al., 2012; Gao et al., 2018). We calculated different plant food supplies in the studied cities using the yields of main crop products multiplied by the relative proportions of harvested crop products in China as a whole; estimated different animal food supplies using the directly reported animal products in the statistical yearbooks; and calculated different foods' consumption amounts by multiplying per capita food consumption in urban and rural households by the corresponding city populations.

Mass balances were used as the basic principle (inputs = outputs + accumulations) (Eq. (S1)) for calculating N input, output, and accumulation values in different sectors of the cities' food systems, if no data were available for determining them directly (Ma et al., 2012; Gu et al., 2015; Cui et al., 2016): for example, 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, per capita food consumption, urbanization rates, and the shifting population in the cities—were mainly taken from cities' statistical yearbooks and bulletins. The second category of data is the coefficients used for the calculation of N fluxes, such as the contents of N in harvested products and foods, the relative proportions of harvested crop products over their uses, the rate of BNF, atmospheric N deposition rates, livestock excrement, the ratio of food consumed away from home, and the VNF of selected foods, etc., in China (Tables S1–S13); such information was mainly obtained from the literature.

Per capita food consumption amounts in the home were taken from statistical yearbooks of the studied cities: these included 16 categories (Table S6, plus vegetable oils). We then corrected total food consumption by the mean ratios of food consumption away from home in China as a whole (Table S6), and calculated per capita food N consumption by multiplying the amounts of different foods by their N contents (Tables S8–S9). In China, large numbers of migrants have moved to cities from rural areas, and many of them have registered as living in cities and become true urbanites. These new immigrants usually adopt urban dietary habits, whereas migrants who have only lived in cities but never registered there usually do not. And there are many residents whose dietary habits fall somewhere between rural and urban types (Gao et al., 2018). For convenience of calculations, food N consumption by migrants in cities was assumed as the average of the two patterns, assuming the most likely value to be the average of the minimum and maximum values (Huang et al., 2017).

2.3. N cost of different foods and NUEs of food systems in the studied cities

N cost is 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 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 food stuffs (Ma et al., 2012). Ma et al. (2014) used this concept to study the N cost of foods in Beijing using the direct N inputs from fertilizer N application, BNF, irrigation, atmospheric deposition, forage feed, net imported food and feed, and fish, divided by the consumed food N by households; they called it the apparent N cost. In this study, we calculated the apparent N cost of food consumption (NC_{app}, kg N kg^{-1} food N) using Eq. (1). However, with more and more food being imported, the apparent N cost of cities' food systems might not suitable for tracking the actual environmental N effects of the final food consumption in cities, because imported food still requires N to be used in its production process (Leach et al., 2012; Cui et al., 2016). To quantify new N inputs associated with these imported foods, the VNF concept was proposed (Burke et al., 2009; Leach et al., 2012); it is defined as any initial N that is used in the food production process but not embodied in the food. This is used as a proxy for the N footprint per unit of food at the product level (Leach et al., 2012; Cui et al., 2016). In their calculations, we can see that VNF plus 1 equals the N cost of a food in its production process (Cui et al., 2016); we adopted this approach. The mean VNFs of the selected foods, in China as a whole (Table S13), were used for all studied cities because the food imported from outside the cities probably came from different regions, and no data were available on the sources of imported food at the city scale. Furthermore, we calculated the virtual N exported from cities as local PN and AN export rates multiplied by the N cost of locally produced PN and AN, because no data were available for calculating the detailed N cost of these foods at the city scale. We then calculated the virtual N cost of food (NC_{vir}, kg N kg^{-1} food N) for the studied cities using Eq. (2), and the apparent and virtual NUE of cities' food systems by as 1 divided by the apparent and virtual N costs, respectively.

$$NC_{app} = \left(I_{c+a} + F_{aqu} + F_{wc} + Feed_{Net-imp} + Food_{Net-imp}\right)/FN_{con} \quad (1)$$

$$NC_{vir} = \left(I_{c+a} + F_{aqu} + F_{wc} + Feed_{VN-Net-imp} + Food_{VN-imp} - Food_{VN-exp}\right)/FN_{con} \quad (2)$$

where I_{c+a} represents total N input to crop and animal production systems, including N from chemical fertilizers, BNF, atmospheric deposition, irrigation, and forage feed; F_{aqu} represents feed N input in freshwater and seawater culture (its calculation is described in the SI Text); F_{wc} represents N embodied in wild captured fish and seafood from the aquatic system; $Feed_{Net-imp}$ and $Food_{Net-imp}$ represent N embodied in net imported feed and food, respectively; $Feed_{VN-Net-imp}$, $Food_{VN-imp}$ and $Food_{VN-exp}$ represent virtual N input in imported feed and food production processes, and virtual N exported through local food exports, respectively; and FN_{con} represents total food N consumption in cities.

Food trade might also generate some effects on the N cost of food in some cities. In this study, we calculated the N costs of imported food (NC_{imp}, kg N kg^{-1} food N) and locally produced food (NC_{LPF}, kg N

kg^{-1} food N) using Eqs. (3) and (4), respectively, and the combined N cost of imported food plus locally produced food was equal to the virtual N cost of food at the city scale.

$$NC_{imp} = (FN_{i,import} \times (VNF_{i,food} + 1)) / \sum FN_{i,import} \quad (3)$$

$$NC_{LPF} = (I_{c+a} + F_{wc} + Feed_{Net-imp}) / (FN_{con} + Food_{exp} - Food_{imp}) \quad (4)$$

where $FN_{i,import}$ represents food *i*'s imported rate; $VNF_{i,food}$ represents virtual N input in food *i*'s production process (Table S13); and $Food_{exp}$ and $Food_{imp}$ represent net exported and imported food N, respectively.

To quantify the relative magnitudes of forces driving apparent and virtual N inputs to cities' food systems, apparent and virtual N costs of PN ($NC_{app,PN}$ and $NC_{vir,PN}$, kg N kg^{-1} food N) and AN ($NC_{app,AN}$ and $NC_{vir,AN}$, kg N kg^{-1} food N) were calculated as indicators using Eqs. (S2) to (S5) (see SI for details), respectively, because there is a large difference in N costs between PN and AN consumption in China (Ma et al., 2012; Gao et al., 2018).

2.4. Scenario design for analyzing driving forces

Population growth, dietary changes, rural-to-urban migration, agricultural patterns and practices, and the NUE of the food system have been reported to have significant effects on total PN and AN consumption and N use in China's food system over the past three decades (Ma et al., 2012; Hou et al., 2014; Gao et al., 2018). However, urban population growth includes both increases in the local population and immigration from regions both surrounding and distant from the cities (see SI for details). These factors will have varying impacts on PN and AN consumption, and on N inputs to food systems at the city scale. We quantified the relative impacts of all these factors on the city-scale AN and PN consumptions, by setting up different scenarios (Table 1). Then the relative magnitude of these driving forces to the new N input to cities' food systems were calculated using the variations of AN and PN (Fig. S4) driven by the different factors multiplied by the calculated apparent and virtual N costs of the consumed AN and PN in each city. We also added net food exported as one of the forces driving N input for food production, because some cities—such as Chongqing and Changchun—still provide food for their surrounding areas. The changes of apparent and virtual N inputs under scenario S1 are mainly caused by agricultural patterns and practices: arable land use changes (Gu et al., 2019), and N management practices in crop and animal production (Gao et al., 2018). We set food N consumption, apparent N and virtual N inputs in 1990 as our benchmark—100%, and quantified the percentage variation of food N consumption and apparent and virtual N inputs in a given year, relative to 1990. Thus we could ascertain the relative importance of each driver to the total variations in the full scenario (RS).

2.5. Uncertainty analysis

There are uncertainties in estimating N input, food N consumption, and N inputs driven by food consumption, etc., because of the multiple data sources and complex parameters, as cited in Tables S1 to S13. 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 propagation equation of mathematical statistics (IPCC, 2000). The means and uncertainty ranges are reported in the figures.

3. Results and discussion

3.1. Historical trends of per capita food N consumption in urban and rural areas in the studied cities

Per capita food N consumption differed greatly among the urban areas of the studied cities (Fig. 1), varying from 3.1 to 4.5 kg N yr^{-1} in 1990 to 3.8–5.3 kg N yr^{-1} in 2015. These results fell into the ranges of 3.0–5.0 kg N yr^{-1} for China's urban households as a whole in 1990 (Wei et al., 2008; Cui et al., 2016; Gao et al., 2018) and 3.8–5.6 kg N yr^{-1} between 2005 and 2012 (Ma et al., 2010; Gao et al., 2018). Meanwhile, the proportion of AN to total food N in urban areas varied significantly; in 1990, it ranged from 30.2% in Changchun to 44.6% in Shanghai, but by 2015 the ranges had increased to 37.0% in Lanzhou and 46.9–56.0% in Beijing, Shanghai, Chongqing, and Xiamen. These trends were similar to the proportion of food N contributed by animals in China's urban areas, which increased from 35% to 41%, between 1990 and 2012 (Gao et al., 2018). There are indications that rising incomes and urbanization are driving an increased demand for per capita food N consumption both in China and at the global scale (Tilman et al., 2011; Gu et al., 2013). The values of food N contributed by animals in the studied cities' urban areas, except for Lanzhou, are now higher than the global mean value of 39% but still far below the 60–80% found in developed countries (FAO, 2013).

Per capita food N consumption in the rural areas surrounding the studied cities has moved in the opposite direction, except for Chongqing in the three years prior to 2016, varying from 4.1 to 5.4 kg N yr^{-1} in 1990 to 3.6–5.1 kg N yr^{-1} in 2015, following the decreasing trend of household diets in China's rural areas (Cui et al., 2016; Gao et al., 2018). At the same time, per capita PN consumption in Changchun's rural areas reached a high of 5.4 kg N yr^{-1} in 1990, and was close to 5.8 kg N yr^{-1} in Changchun as a whole in 1993 (Zhang et al., 2017). This represents an increase of 32%, or around 1.3 kg PN yr^{-1}, relative to Beijing, Lanzhou, and Xiamen in 1990 (see SI for details). Per capita food N consumption in the studied cities was slightly higher than the 4.0–4.3 kg N yr^{-1} for China's rural areas as a whole in 1990 (Wei et al., 2008; Gao et al., 2018), but dropped to 3.6–3.8 kg N yr^{-1} between 2005 and 2012 (Ma et al., 2010; Gao et al., 2018). In 2015, the proportion of food N contributed by animals in rural areas of the studied cities varied from 21.1% in Changchun to 56.9% in Xiamen. Chongqing (27.9%),

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

Code	Scenario description	Driving-force effect calculation	
S1	Urban and rural populations maintained at 1990 level, diets unchanged since 1990, and no migration into urban area	$S1_i$[a]$/S1_{1990}$	Agricultural practices effect (only for N input)
S2	Urban and rural dietary changes based on S1	$S2_i/S1_{1990}$	Dietary changes effect (S2 − S1)
S3	Native population growth based on S2	$S3_i/S1_{1990}$	Native population growth effect (S3 − S2)
S4	Migration into urban area from surrounding rural area of the same city based on S3	$S4_i/S1_{1990}$	Native rural-to-urban migration effect (S4 − S3)
RS	Actual situation: migration into city from outside based on S4	$RS_i/S1_{1990}$	Migration from distant regions effect (RS − S4)
RS + Net EXP	Net food export from cities based on RS	RS + Net $EXP_i/S1_{1990}$	Net food export effect (RS − RS + Net EXP)

[a] i represents all the years included in our study.

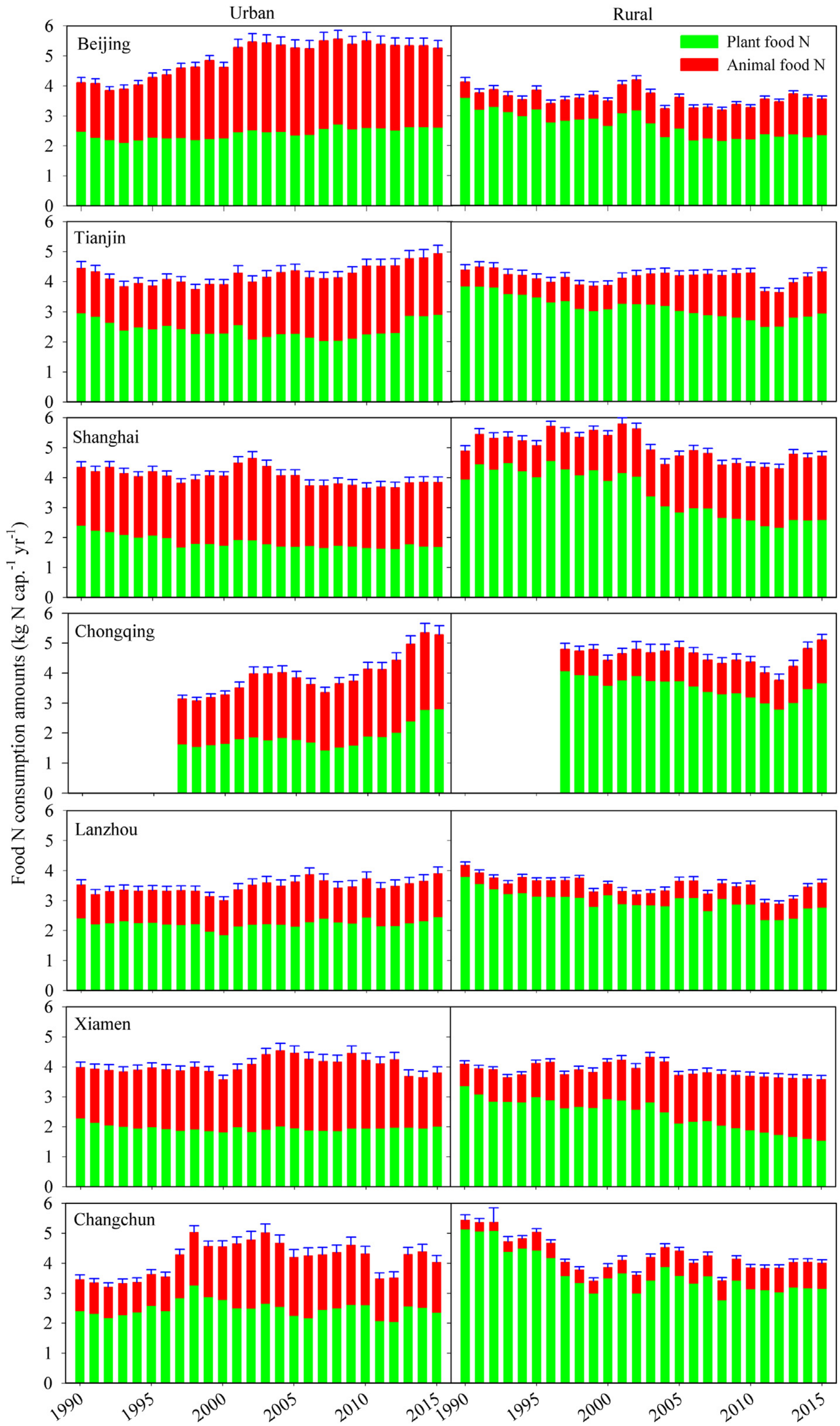

Fig. 1. Per capita food N consumption in urban and rural households, 1990–2015. Error bars represent uncertainty ranges of per capita food N consumption.

Lanzhou (22.8%) and Changchun (21.1%) fell into the means of 14.0% to 28.0% for China's rural areas as a whole during 1990 to 2012 (Gao et al., 2018). Values for Beijing and Tianjin (33.0%) and Shanghai (45.3%) were significantly higher than for China's rural areas as a whole, most likely because some rural areas in the studied cities were located in metropolitan areas, provincial capitals, or special economic zones, and the development and income levels of these areas are relatively higher than for rural areas near more ordinary cities.

The share of food N contributed by animals was higher in urban areas than in rural areas—trends in line with the global values as well as in China (Tilman and Clark, 2014; Gao et al., 2018). These differences indicate that, in the majority of the studied cities, when someone moves from a rural area to an urban area, they will consume 0.6–1.5 kg more AN yr^{-1} and 0.3–0.9 kg less PN yr^{-1}. The exception is for PN in Beijing and Xiamen in recent years (Fig. 1).

In 2015, urban household diets in Shanghai, Lanzhou, Xiamen and Changchun, and rural ones in Beijing, Lanzhou, Xiamen and Changchun came close to meeting the low-energy standards of 3.5 kg N $cap.^{-1}$ yr^{-1} (Fig. S5), calculated using the data on per capita food consumption from the 'Dietary Guidelines for Chinese Residents' (CNS, 2016). The remainder of the diets came close to the recommended intermediate-energy standards of 4.3 kg N $cap.^{-1}$ yr^{-1}(CNS, 2016). These data indicate that China still needs to further increase per capita food N consumption to reach the recommended intermediate-energy standards. However, per capita AN consumption in urban areas of the studied cities did reach the intermediate-energy standards, except for Lanzhou, which was still below even the low-energy standards. Clearly, China's urbanites should control their AN consumption, which has been rising with socioeconomic development; urbanites need to be encouraged to eat less meat, to mitigate the N inputs for food production (Gao et al., 2018; Gu et al., 2019). We also found that there has been a large gap between the actual food structure and the recommended one; for example, the recommended food N contributed by animals should come mainly from fish, milk and eggs (Zhang et al., 2019), but in actuality, poultry, beef and mutton, and pork are the main sources of AN (Fig. S5). Pork plus beef and mutton accounted for 35.1–48.4% and 32.0–60.6% of total AN consumption, respectively, in the urban and rural areas of the studied cities. Hence, China as a whole faces another big challenge: to convert the food structure to the recommended dietary one. Recent studies have indicated that this conversion has a major role to play, in mitigating N input for food production and reducing environmental N pollution in China (Gu et al., 2019; Zhang et al., 2019), because fish, milk, eggs and poultry have higher NUE and lower environmental N footprints than those of pork, beef and mutton production (Leach et al., 2012; Guo et al., 2017).

3.2. Total PN and AN supplies and consumption in the studied cities

From 1990 to 2015, total PN supply showed different trends in different cities (Fig. 2a): Beijing, Tianjin, Shanghai, and Xiamen declined in 2015, compared to 1990, because of the shrinking arable land area (Burney et al., 2010; Fan et al., 2012). Chongqing showed a relatively stable level from 1997 to 2005, but it increased after 2005 because of the increases in crop planting area and per unit area yield (NBSC, 2016). Meanwhile, PN consumption increased in Beijing, Tianjin, Shanghai, Xiamen and Lanzhou (Fig. 2c), because of the large numbers of immigrants in these cities. PN consumption in Changchun remained relatively stable, because the increased PN consumption attributable to the large numbers of rural-to-urban immigrants was offset by the decreased PN consumption caused by the per capita N consumption reduction and population decrease in the rural areas of Changchun. PN consumption in Chongqing slowly decreased from 1990 to 2012, because of the dietary conversion from high PN in rural households to low PN in urban households, during the rural-to-urban migration.

During the study period, the changes in the AN supply were more pronounced than those in the PN supply (Fig. 2b). In Beijing, Tianjin, and Shanghai, they significantly increased after 1990, a trend dominated by the increased demand of AN consumption caused by population growth and dietary shifts to more AN in both urban and rural areas, but they decreased around 2005 and remained stable up to 2015. The decrease of AN supply around 2005 resulted from the avian influenza outbreak in China in 2005–2007. AN supply increased faster in Chongqing and Changchun than in other cities (Fig. 2d), but for different reasons: in Chongqing it was driven by the increased demand for AN consumption, but in Changchun it resulted mainly from increased animal food exports from the city. The sharply decreased AN supply in Changchun in 2010 was caused by increases in the costs of agricultural production, the animal influenza outbreak, and the heavy floods and

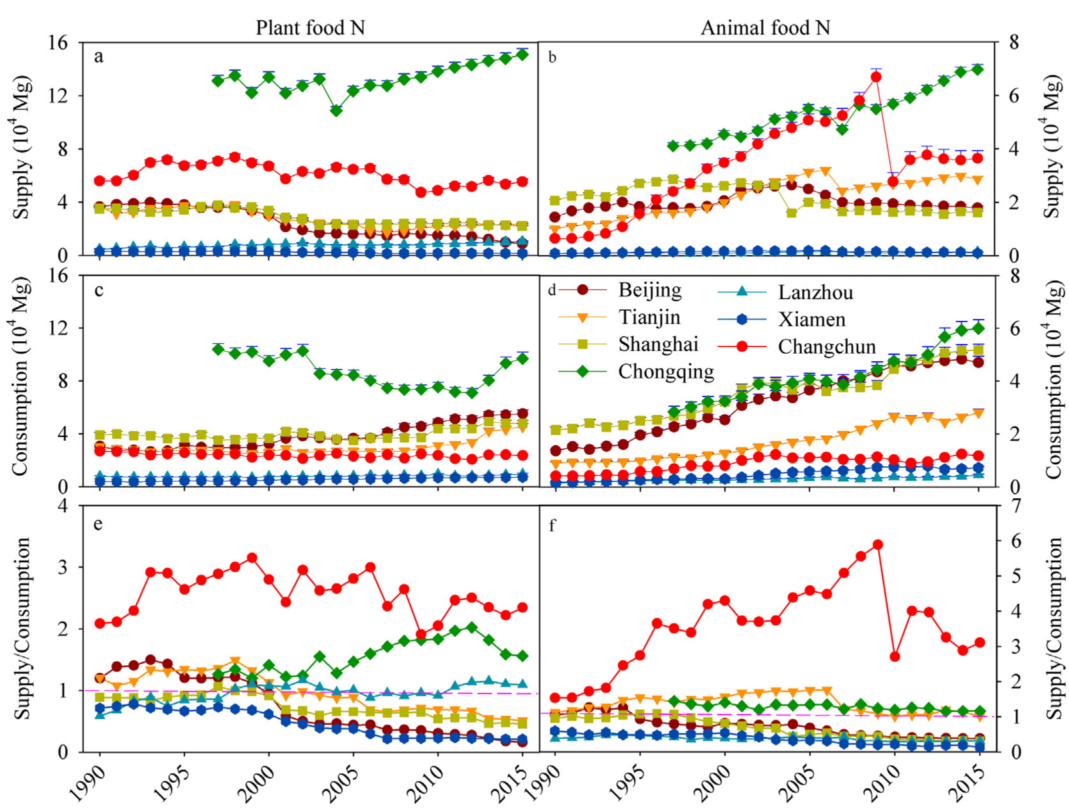

Fig. 2. Plant- and animal-food N supplies and consumption in different cities, 1990–2015.

water-logging disaster that occurred in Jilin province in 2009 (Zhang et al., 2017).

The ratios of supply to consumption of PN and AN reflect the self-sufficiency rates of the cities (Fig. 2e, f). Before 2000, most of studied cities had high self-sufficiency rates, even though some PN and AN was imported from Beijing, Tianjin, Chongqing and Changchun. However, with population growth and decreases in arable land area, highly urbanized cities demanded more and more food imported from outside. Changchun, however, always maintained high ratios of supply to consumption, making it's a large net exporter of both PN and AN. Lanzhou was basically PN self-sufficient during our study period, but now it has to import 70% of its AN. We can conclude that with the competition for arable land and the large migrations into cities that accompany rapid urbanization, more and more cities are relying on food imported from outside their geographic boundaries. There are some indications that virtual land, water, and carbon flow are growing, in the inter-provincial trade of staple crops in China (Wu et al., 2018). As a consumer metropolis, Beijing, for example, is typically a net importer of carbon flow (a net consumer), with 76% of its consumption-based carbon emissions coming from outside its geographic boundary (Meng et al., 2018), and other cities are also moving in this direction. Our results show that in 2015, if only apparent N inputs were considered, N inputs were underestimated by 33.8–74.9% in Beijing, Tianjin, Shanghai and Xiamen. Conversely, N inputs to the food systems were overestimated by 4.0–46.4% in Chongqing and Changchun, if only apparent N inputs were considered, because these two cites still export large amounts of food (Fig. S6). Hence, apparent N inputs to cities' food systems might not accurately reflect the transfer of pollution connected with virtual N input caused by the net import of food and livestock feed between different countries or regions (Cui et al., 2016; Verger et al., 2018). Virtual N inputs should therefore be included when quantifying the N demand of food systems at the city scale, especially in highly urbanized areas that rely on their surrounding areas to produce their food (Burke et al., 2009; Leach et al., 2012).

3.3. Apparent and virtual N costs of food, and different NUEs of food systems in the studied cities

Virtual N costs in the studied cities fell into the range of 2.4 kg N kg^{-1} food N in Tianjin to 10.1 kg N kg^{-1} food N in Xiamen, in 1990, and from 3.9 kg N kg^{-1} food N in Tianjin to 6.6 kg N kg^{-1} food N in Changchun, in 2015 (Fig. 3). Virtual N cost can be used for tracing the actual environmental effects of N associated with cities' consumed food production; this is the actual N cost of Chinese food, as has been reported in several different literatures. This cost increased dramatically, from around 6.0 kg N kg^{-1} food N during 1980–1990 to 9.9–11.0 kg N kg^{-1} food N during 2005–2012 (Ma et al., 2012; Gao et al., 2018). The virtual N cost at the city scale is relatively low compared to the actual N cost of Chinese food, and is close to the estimates for the world as a whole (Galloway and Cowling, 2002; Pierer et al., 2014). However, virtual N cost at the city scale might be underestimated because the gaps between food supply and final consumption have been neglected, since no data have been available on this information at the city scale. The gaps between food supply and consumption were in the range of 13.3–35.2% during 1990 to 2009 (Cui et al., 2016). Moreover, we can see that virtual N costs were significantly higher than the corresponding apparent N costs, by 45.2–298.7%, in Beijing, Tianjin, Shanghai, Lanzhou and Xiamen, in 2015, while the virtual N costs were lower than the apparent N costs, by 25.2–40.2%, in Chongqing and Changchun, between 1990 and 2015.

NUE is equal to 1 divided by the N cost (Galloway and Cowling, 2002; Ma et al., 2012). The apparent NUEs of food systems showed a wide range, from 10.3% in Changchun to 71.6% in Beijing, in 2015, while the virtual NUEs of food systems showed a narrower range, from 15.1% in Changchun to 25.6% in Tianjin (Fig. 3). Most city governments like to see the apparent NUEs of food systems, because these appear to reflect high efficiencies of N use in food production, especially for highly urbanized cities such as Beijing, Tianjin, Shanghai and Xiamen, but the higher apparent NUEs actually mean that more N is imported from outside the cities in the form of final edible food. The apparent NUE of a food system is also misleading for cities that export food and feed; for example, the NUEs of food systems calculated from apparent N cost were underestimated by 4.0–46.4% in Chongqing and Changchun. The virtual NUE of a food system should therefore be used to evaluate the environmental losses of N over the entire life cycle of a consumed food at the city scale.

3.4. N cost of imported food and locally produced food, and their combined effect on virtual N cost in the studied cities

We studied the N costs of imported food and locally produced food, and their combined effect on the virtual N cost in the studied cities (Fig. 4). This graph clearly shows that the N costs of the imported food, the locally produced food, and the total of both, showed significant differences among the cities. The N cost of imported food was significantly higher than that of locally produced food in Beijing, Tianjin, Shanghai and Changchun in the initial stage of our study, because during that time they mainly imported fruits, pork, beef and mutton, which had high N costs (Fig. S7). If a city's N cost of imported food is higher than that of locally produced food, its virtual N cost will be increasingly elevated as the ratio of food imports rises, as has happened in Beijing, Tianjin, and Shanghai. There are indications that the

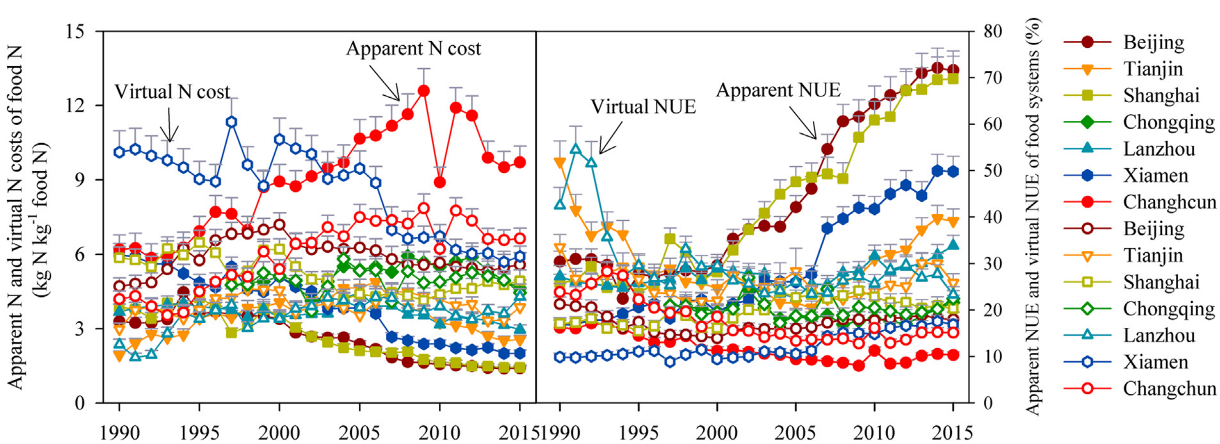

Fig. 3. Apparent and virtual N costs and NUEs of food systems in different cities, 1990–2015.

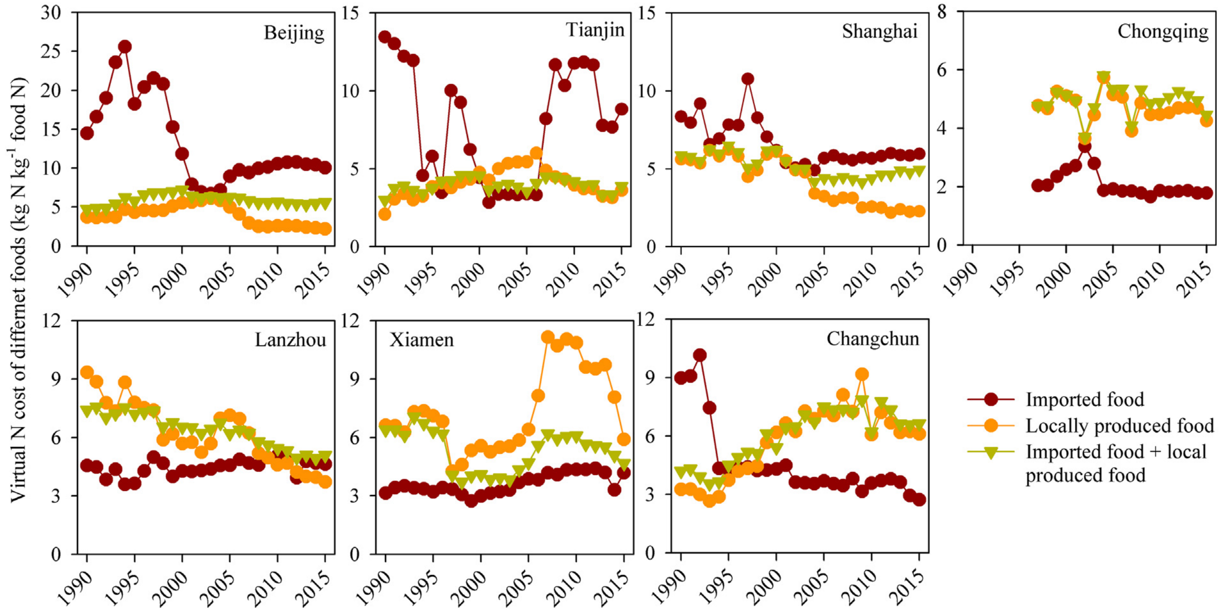

Fig. 4. N costs of imported food, locally produced food, and imported food plus locally produced food.

developed or higher-efficiency regions play a major role in driving raw material consumption growth through changes in their trade structures, as they increasingly shift production to less material-efficient input suppliers (Plank et al., 2018). Other cities, however, can reduce the virtual N costs and improve their actual NUEs, by increasing the ratio of food imports, as has happened in Lanzhou and Xiamen, because the N cost of locally produced food in these two cities has been significantly higher than the costs of imported food (see SI for details). Our findings indicate that for food-importing cities, virtual N cost might be lowered by choosing foods with low N cost in their regions of origin. In China, for example, the imported virtual N trade fluxes significantly decreased compared to actual imported N fluxes with China's increasing food trade, because we mainly imported legume crops (mostly soybeans and oilcrops) with relatively low N costs, while at the same time exporting large amounts of vegetables, fruits, and meat, which have higher N costs. Hence, to a large extent, the recent trend of food trade structure in China has increased its territorial environmental burden of Nr pollution (Cui et al., 2016), and China should therefore change its current food trade structure to reduce the risks of Nr pollution associated with food systems, as should cities that rely heavily on food imports. In food-exporting cities, such as Chongqing and Changchun, however, virtual N cost is completely dominated by the N cost of locally produced food. For this type of city, the NUE of the food system can be improved with better N management of local crop and livestock

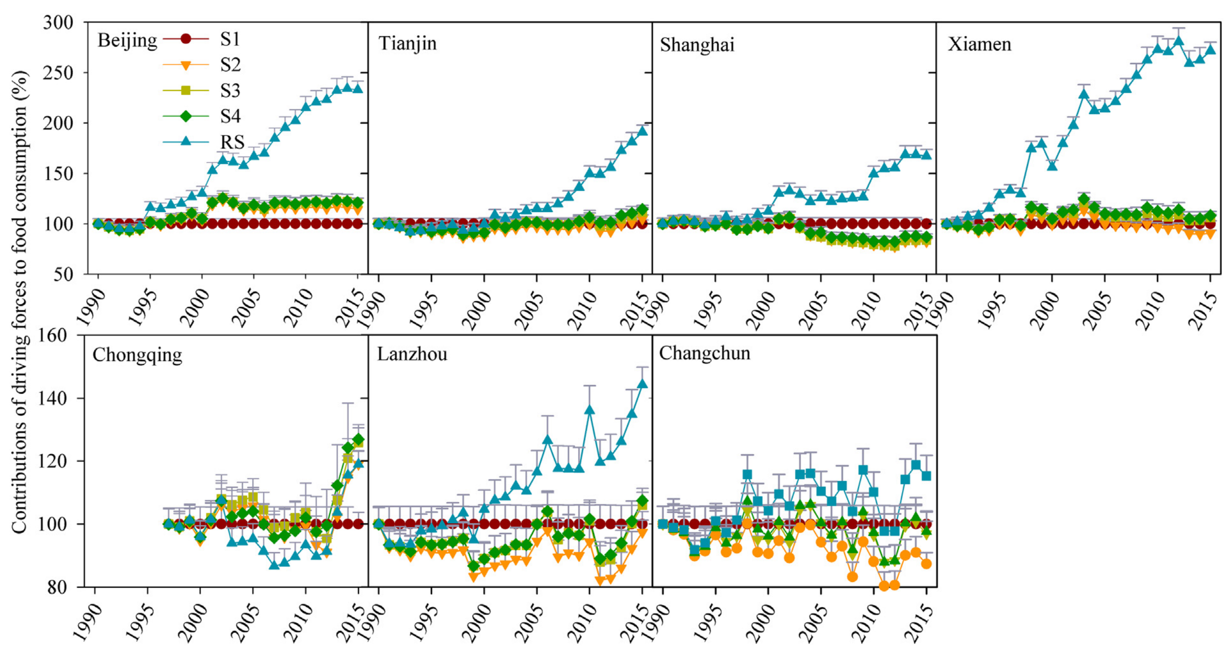

Fig. 5. Relative changes in total food N consumption in different cities, driven by dietary changes, native population growth, native rural-to-urban migration, and migration from regions distant from the cities, relative to 1990, 1990–2015. S1, S2, S3, S4 and RS are explained in Table 1. Error bars represent the uncertainty range of each value.

production (Ma et al., 2014). This is especially true for Changchun, because it is exporting more and more vegetables, and beef and mutton —foods whose production has a high N cost (Leach et al., 2012; Guo et al., 2017).

3.5. Driving forces of food N consumption in the studied cities

We have separately quantified the relative contributions of dietary changes, native population growth, native rural-to-urban migration, migration from regions distant from the cities, and agricultural patterns and practices, to total food N consumption, by looking at several scenarios (Fig. 5). The results clearly showed that, as would be expected, dietary changes dominated total food N consumption increases of 14.4–19.0% in Beijing and Chongqing (S2–S1), because per capita food N consumption increased in these two cities in 2015, relative to 1990 (Fig. 1). However, the dietary-change effect disappeared in Beijing after 2000, because per capita food N consumption was relatively stable in this city, between 2001 and 2015 (Fig. 1). At the same time, dietary changes dominated by total food N consumption decreased by 17.8%, 2.6% and 12.7% in Shanghai, Lanzhou, and Changchun, respectively, because the decreases in PN consumption were higher than the increases in AN consumption (Fig. S4). And this effect also disappeared in Shanghai after 2005, for the same reason as in Beijing. However, the dietary changes had almost no impact on total food N consumption in Tianjin or Xiamen, because the AN consumption increase was equal to the PN decrease (Fig. S4). Relative to dietary changes, native population growth showed a smaller contribution to food N consumption increase in the studied cities (S3–S2), varying from 1.5% in Shanghai to 15.2% in Xiamen, while native rural-to-urban migration contributed less than 3.2% of the increase in total food N consumption, based on the S3 scenario (S4–S3), because native rural-to-urban migration is very low in highly urbanized areas. The rural-to-urban migration from regions distant from the cities, however, played an important role in promoting food N consumption in the studied cities (RS–S4) (except that in Chongqing it decreased by 8.0% between 1990 and 2015), contributing 17.7–163.3% of the increased food N consumption in 2015, relative to 1990, based on scenario S4. It is well known that population growth and dietary changes have been the main driving forces for the changes in food N consumption throughout the world (Galloway et al., 2004; Tilman and Clark, 2014; Gu et al., 2015), and rural-to-urban migration could result in an additional increase in food N consumption in China (Gao et al., 2018). Our results indicated that in the process of urbanization, migration from regions distant from the cities has become the largest driver of the increase in food N consumption, even more than dietary changes, native population growth and native rural-to-urban migration (except for Chongqing because its population is moving outward). The combined effects of the new National Urbanization Plan (Bai et al., 2014) and the newly implemented two-child policy in China will likely be a higher number of people migrating to metropolitan areas and more prosperous regions, such as Beijing, Tianjin, Shanghai, and Xiamen, from the secluded and economically backward areas, such as Chongqing and Changchun, and this migration will result in more and more demand for food N in these destination cities. This shift might cause serious N pollution in cities, not only because per capita AN consumption is higher in most urban households than it is in rural ones (Cui et al., 2016; Gao et al., 2018), but also because the N from cities that is not recycled into rural areas will cause large amounts of N to be stranded in urban environments after consumption (Marzluff et al., 2008; Grimm et al., 2008; Zhuet al., 2017; Yu et al., 2017).

3.6. Driving forces of the N input to food systems in the studied cities

We quantified the relative contributions of the variations of food N, driven by agricultural practices, dietary changes, native population growth, native rural-to-urban migration, migration from distant regions, and net food exports from cities, to apparent and virtual N inputs to the cities' food systems, relative to 1990, between1990 and 2015 (Fig. 6). The results clearly showed that these factors played different roles in changing both apparent and virtual N inputs to food systems in the studied cities. Virtual N inputs showed larger variations than apparent N inputs (Fig. 6), though, and this trend fits well with the dynamics of food N consumption driven by the various factors (Fig. 5). Under

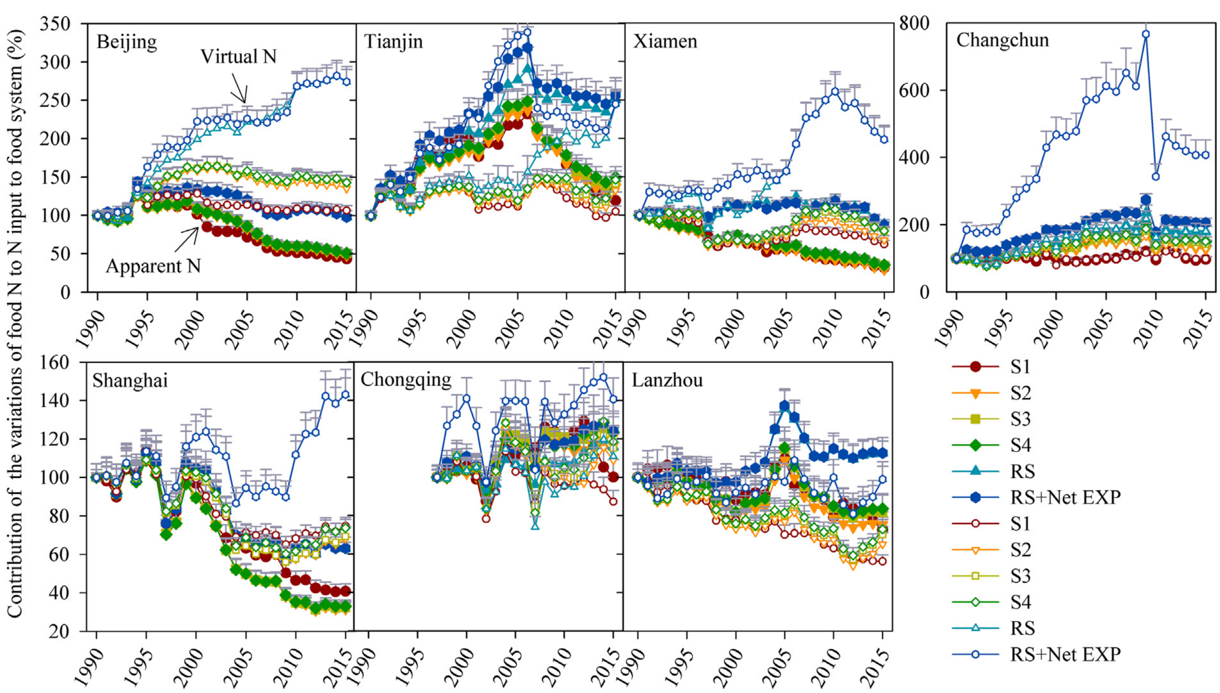

Fig. 6. Relative changes in apparent N and virtual N input to cities' food system, driven by agricultural patterns and practices effect, dietary changes, native population growth, native rural-to-urban migration, migration from distant regions, and net food export, relative to 1990, during 1990–2015. S1, S2, S3, S4, RS and RS + Net EXP are explained in Table 1. Error bars represent the uncertainty range of each value.

scenario S1, virtual N inputs showed opposite trends to apparent N inputs in Beijing, Shanghai, and Xiamen, in the urbanization process, because more and more food and feed is being imported and large amounts of N are needed in the production processes (Leach et al., 2012), indicating that N pollution connected with food production has been transferred to outside the cities (Cui et al., 2016; Verger et al., 2018), just as consumption-based carbon emissions are coming from outside Beijing's geographic boundary, as discussed above (Meng et al., 2018). Under scenario S1, however, virtual N inputs' increase rates significantly decreased relative to apparent N, in Tianjin, Chongqing, and Lanzhou, because Tianjin and Chongqing are net food N exporters, and in Lanzhou maize production increased by 542.0% while imported feed decreased by 63.6%, and the numbers of horses, mules, and donkeys significantly decreased. Although Changchun is a large food exporter (Fig. 2), virtual N inputs' increase rates did not significantly differ from those of apparent N under scenario S1, because Changchun is also a net feed importer, and virtual N inputs for imported feed accounted for 63.0–99.9% of the virtual N inputs induced by exported food. Compared to scenario S1, dietary changes increased virtual N input by 5.4–30.0% in all the studied cities in 2015, relative to 1990, except for Shanghai, where they decreased by 6.6% (S2–S1)—below the increases in N (39.0%) inputs to food systems driven by Chinese dietary changes over the period 1990–2012 (Gao et al., 2018). Relative to scenario S2, native population growth increased virtual N inputs by 1.2–14.1% in 2015 (S3–S2)—below the results of the increases in N inputs to food systems (17.0–40.0%) driven by Chinese population growth over the period 1980–2010 (Hou et al., 2014). Native rural-to-urban migration increased virtual N inputs by 2.7–8.2% compared to scenario S3 (S4–S3), except for Beijing and Xiamen. Migration from distant regions increased virtual N inputs by 26.0–131.0% compared to scenario S4 (RS–S4), except for Chongqing, where they deceased by 7.7% because its population is moving outward. Based on scenario RS, net food exported from cities accounted for 29.8–226.9% of virtual N increases in Chongqing and Changchun in 2015, compared to 1990 (RS + Net EXP–RS), and Tianjin also showed a large virtual N increase induced by food exports—the largest difference compared to RS (198.1%) appearing in 2005. Six factors increased virtual N inputs by 43.1–307.2% in 2015, relative to 1990, except for a fluctuating reduction (0.2–19.0%) in Lanzhou. We can conclude that, at the city scale, migration from distant regions is the largest driving force for apparent and virtual N increases in the majority of cities that are net food importers, followed by dietary changes and agricultural patterns and practices: for example, Beijing, Shanghai, Lanzhou and Xiamen. However, food exported from cities makes the largest contribution to apparent and virtual N increases if the city is a net food exporter, such as has occurred in Tianjin, Chongqing, and Changchun. Hence, migration from distant regions, dietary changes and agricultural patterns and practices are the factors for mitigating apparent and virtual N inputs in Chinese cities, and, if the city is a net food exporter, virtual N input induced by the food trade.

4. Conclusions

Apparent and virtual N inputs, N costs and NUEs of food systems, the impacts of food trade on local virtual N cost, and the magnitudes of driving forces of N inputs to food systems in difference types of cities were studied, based on agricultural patterns and practices, dietary changes, native population growth, native rural-to-urban migration, migration from distant regions, and food imported from cities, relative to 1990, during 1990–2015. The results clearly show that agricultural patterns and practices, dietary changes, migration from distant regions and food exports (in food-source cities) were the main driving forces of the changes in food N consumption and N inputs to the food systems in the studied cities. Cities transferred large amounts of N input for food production outside their boundaries, and the N cost of food trade also affects local virtual N cost. Furthermore, virtual NUE is recommended as an indicator for accurately evaluating the actual environmental N effects of cities' food systems for China as a whole. Given the pressure from a growing population and increasing food consumption in cities during urbanization, strategies for reducing the risks of N losses to the environment associated with cities' food systems involve improving the NUE of food production—not only N input in local food production, but also virtual N input for imported food—along with shifting urban household diets toward the lower consumption of pork, beef and mutton, and importing food and feed from high NUE production areas.

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Appendix A. Supplementary data

Includes: detailed descriptions of patterns of urbanization and different types of urbanization processes in the studied Chinese cities; the calculations of rural-to-urban migration, including both migrants who only lived in the city but had never registered there and migrants who had registered as living in the city and had become true urbanites; the estimation of feed N imported from outside the city; the calculations of apparent and virtual N costs of Chinese PN and AN consumption; higher per capita food N consumption in rural households of Changchun than in the other studied cities; higher N cost of local food production in Lanzhou, Xiamen and Changchun than in the other studied cities; and the uncertainty ranges of activity data and parameters (Tables S1–13 show the main parameters). This information is available free of charge via the Internet at http://pubs.acs.org/. Supplementary data to this article can be found online at doi: https://doi.org/10.1016/j.scitotenv.2019.04.136.

References

Bai, X.M., Shi, P.J., Liu, Y.S., 2014. Society: realizing China's urban dream. Nature 509 (7499), 158–160.
Bleken, M.A., Bakken, L.R., 1997. The nitrogen cost of food production: Norwegian society. Ambio 26, 134–142.
Burke, 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.
Burney, J.A., Davis, S.J., Lobell, D.B., 2010. Greenhouse gas mitigation by agricultural intensification. Proc. Natl. Acad. Sci. U. S. A. 107, 12052–12057.
CNS (Chinese Nutrition Society), 2016. Dietary Guidelines for Chinese Residents. The Tibet People's Publishing House, China.
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.
Fan, M.S., Shen, J.B., Yuan, L.X., Jiang, R.F., Chen, X.P., Davies, W.J., Zhang, F.S., 2012. Improving crop productivity and resource use efficiency to ensure food security and environmental quality in China. J. Exp. Bot. 63, 13–24.
FAO (Food and Agriculture Organization of the United Nations), 2013. FAOSTAT: FAO statistical databases. http://faostat.fao.org/default.aspx, Accessed date: 20 March 2013.
Galloway, J.N., Cowling, E.B., 2002. Reactive nitrogen and the world: 200 years of change. Ambio 31, 64–71.
Galloway, J.N., Dentener, F.J., Capone, D.G., Boyer, E.W., Howarth, R.W., Seitzinger, S.P., Asner, G.P., Cleveland, C.C., Green, P.A., Holland, E.A., Karl, D.M., Michaels, A.F., Porter, J.H., Townsend, A.R., Vösmarty, C.J., 2004. Nitrogen cycles: past, present and future. Biogeochemistry 70 (2), 153–226.
Gao, B., Huang, Y.F., Huang, W., Shi, Y.L., Bai, X.M., Cui, S.H., 2018. Driving forces and impacts of food system nitrogen flows in China, 1990 to 2012. Sci. Total Environ. 610–611, 430–441.
Grimm, N.B., Faeth, S.H., Golubiewski, N.E., Redman, C.L., Wu, J., Bai, X., Briggs, J.M., 2008. Global change and the ecology of cities. Science 319 (5864), 756–760.
Gu, B.J., Leach, A.M., Ma, L., Galloway, J.N., Chang, S.X., Ge, Y., Chang, J., 2013. Nitrogen footprint in China: food, energy, and nonfood goods. Environ. Sci. Technol. 47, 9217–9224.
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. U. S. A. 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.

Hou, Y., Ma, L., Gao, Z.L., Wang, F.H., Sims, J.T., Ma, W.Q., Zhang, F.S., 2014. The driving forces for nitrogen and phosphorus flows in the food chain of China, 1980 to 2010. J. Environ. Qual. 42, 962–971.

Huang, W., Huang, Y.F., Lin, S.Z., Chen, Z.H., Bing, G., Cui, S.H., 2017. Changing urban cement metabolism under rapid urbanization—a flow and stock perspective. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2017.01.008.

IPCC, 2000. Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories. Published. IGES, Japan.

Kearney, J., 2010. Food consumption trends and drivers. Philos. 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. J. Environ. Dev. 1 (1), 40–66.

Liang, X., Lam, S.K., Gu, B.J., Galloway, J.N., Leach, A.M., Chen, D.L., 2018. Reactive nitrogen spatial intensity (NrSI): a new indicator for environmental sustainability. Glob. Environ. Chang. 52, 101–107.

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, Y.R., Cai, W., 2014. Comparative study on the consumption structure of urban residents in 31 regions of China (in Chinese). Consu. Econom. 30 (3), 35–41.

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.

Marzluff, J., Shulenberger, E., Endlicher, W., Alberti, M., Bradley, G., Ryan, C., Simon, U., ZumBrunnen, C., Alberti, M., Marzluf, J., Shulenberger, E., Bradley, G., Ryan, C., Zumbrunnen, C., 2008. Integrating humans into ecology: Opportunities and challenges for studying urban ecosystems. Urban Ecology. Springer, USA, pp. 143–158.

Meng, F.X., Liu, G.Y., Hu, Y.C., Su, M.R., Yang, Z.F., 2018. Urban carbon flow and structure analysis in a multi-scales economy. Energy Policy 121, 553–564.

Mi, Z., Zhang, Y., Guan, D., Shan, Y., Liu, Z., Cong, R.G., Yuan, X.C., Wei, Y.M., 2016. Consumption-based emission accounting for Chinese cities. Appl. Energy 184, 1073–1081.

NBSC (National Bureau of Statistics of China), 2016. China Statistical Yearbook. NBSC Press, Beijing, China.

NBSC (National Bureau of Statistics of China), 2017. China Statistical Yearbook. NBSC Press, Beijing, China.

Northam, R.M., 1979. Urban Geography. John Wiley & Sons, New York, p. 66.

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.

Plank, B., Eisenmenger, N., Schaffartzik, A., Wiedenhofer, D., 2018. International trade drives global resource use: a structural decomposition analysis of raw material consumption from 1990—2010. Environ. Sci. Technol. 52, 4190—4198.

Steffen, W., Richardson, K., Rockström, J., Cornell S. 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. doi:https://doi.org/10.1126/science.1259855.

Tilman, D., Clark, M., 2014. Global diets link environmental sustainability and human health. Nature 515, 515–522.

Tilman, D., Balzer, C., Hill, J., Befort, B.L., 2011. Global food demand and the sustainable intensification of agriculture. Proc. Natl. Acad. Sci. U. S. A. 108 (50), 20260–20264.

UN-Habitat, 2010. State of the world's cities 2010/2011-cities for all: bridging the urban divide. http://www.unhabitat.org.

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. Resour. Conserv. Recycl. 137, 200–213.

Wei, J., Ma, L., Lu, G., Ma, W.Q., Li, J.H., Zhao, L., 2008. The influence of urbanization on nitrogen flow and recycling utilization in the food consumption system of China (in Chinese with English abstract). Acta Ecol. Sin. 28 (3), 1013–1024.

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. Resour. Conserv. Recycl. 136, 179–186.

Yu, C.Q., Xiao, Y.C., Ning, S.Q., 2017. Changing patterns of urban-rural nutrient flows in China: driving forces and options. Sci. Bull. 62, 83–91.

Zhang, X.M., Wang, Y., Yan, L., Gao, Q., 2017. The trends and characteristics of nutrition in the food chain in Changchun (in Chinese with English abstract). J. Nat. Resour. 32 (20), 255–265.

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. Resour. Conserv. Recycl. 144, 65–73.

Zhu, Y.G., Reid, B.J., Meharg, A.A., Banwar, S.A., Fu, B.J., 2017. Optimizing Peri-URban ecosystems (PURE) to re-couple urban-rural symbiosis. Sci. Total Environ. 586, 1085–1090.