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The environmental and socioeconomic trade-offs of importing crops to meet domestic food demand in China

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Abstract

China increasingly relies on agricultural imports, driven by its rising population and income, as well as dietary shifts. International trade offers an opportunity to relieve pressures on resource depletion and pollution, such as nitrogen (N) pollution, while it poses multiple socioeconomic challenges, such as food availability. To quantify such trade-offs considering the roles of different crop types, we developed a unique crop-specific N budget database and assessed the impacts of the crop trade on multiple sustainability concerns including N pollution caused by crop production, crop land area, independence of food supply, and trade expenditures. We quantified the ‘virtual’ N inputs and harvested areas, which are the amount of N inputs and land resources used in exporting countries for China’s crop import. In addition, we proposed the concepts of ‘alternative’ N inputs and harvested area to quantify the resources needed if imported crops were produced in China. By comparing results from ‘alternative’ and ‘virtual’ concepts, we assessed the role of trade in Chinese crops over the past 30 years (i.e. 1986–2015) in alleviating N pollution and saving cropland in China and the world. Crop imports accounted for 31% of Chinese crop N consumption in 2015, and these crop imports eased the need for an additional cropland area of 62 million ha. It also avoided an N surplus by 56 and 36 Tg (Tg = 10\textsuperscript{12} kg) for China and the world respectively but led to $621 billion crop trade expenditures over the 30 year period. The N pollution damage avoided by crop imports in economic terms was priced at $22 ± 16 billion in 2015, which is lower than the crop trade expenditures but may be surpassed in the future with the development of the Chinese economy. Optimizing a crop trade portfolio can shift domestic production from N-intensive crop production (e.g. maize, fruits, and vegetables) to N-efficient crop production (e.g. soybeans), and consequently mitigate an N surplus by up to 12%. Improving N use efficiency for individual crops can further increase the mitigation potential of N surplus to 30%–50%, but requires technology advancement and policy incentives.

1. Introduction

Feeding the world’s growing population with limited land and resources is one of the great challenges of the 21st century (Godfray et al 2010, Mueller et al 2012). This challenge is particularly pressing in China, which needs to feed 22% of the global population with only 7% of the global cropland (Piao et al 2010). To address this challenge, it is critical to intensify and boost agricultural productivity. Synthetic nitrogen (N) fertilizer has proved to be effective in boosting yield in modern agriculture, and has been intensively used in crop production in China. Unfortunately, more than 60% N has been lost to the environment during crop production (Zhang et al 2015a, Liu et al 2016), causing severe environmental problems, such as air pollution, eutrophication of surface waters, and soil degradation (Gu et al 2017). In 2010, the N loss from cropping...
systems in China was 4–5 Tg for surface and groundwater, and 8 Tg for reactive N gases (NH₃ and NOₓ), which is a more than two-fold increase compared to N pollution in 1980 (Gu et al. 2015, Yu et al. 2019). Meanwhile, income growth in China has driven up the population’s protein consumption from meat. Increasing N inputs into livestock feeds impose greater pressures on both crop production and the environment. Consequently, there is a critical need to meet China’s rising food demands with limited domestic natural resources (e.g. land and water) and minimum N pollution.

Globalization of food trade provides an opportunity to alleviate the pressure of increasing food demands and exacerbating N pollution within an individual country, and potentially for the world. A diverse and carefully designed crop trade portfolio can protect a country against local disruptions and shortfalls in production. Currently, 23% of the food produced for human consumption is traded internationally (D’Odorico et al. 2014). Driven by the spatial heterogeneity of crop productivity, food trade tends to flow from regions with high production efficiency to less efficient regions and consequently decreases the overall environmental impacts of agricultural production (Dalin and Rodriguez-Iturbe 2016). Despite substantial N pollution embedded in international trade, food trade reduces global N inputs by redistributing the production of commodities to regions more efficient in N use (Oita et al. 2016). Therefore, the international food trade could mitigate environmental degradation by coordinating global food supply and demand.

However, heavily relying on food supply from a global trade system may expose a country to risks in the international market and lead to unintended environmental consequences for both the importing and exporting countries (Sun et al. 2018). The risks in the international market include the uncertainty and instability caused by climate change, regional policy, food price and other factors (Fader et al. 2013). For example, in 2010, Russia experienced a 130 year record heat wave, causing a significant yield drop in its grain crop. The Russian government imposed an export ban to protect local consumers and local farmers, which pushed up world grain prices and caused a 1.6% poverty increase in Pakistan over this ban period (Welton 2011). In this case, the extreme weather and changes in trade policies from exporting countries led to international price spikes which negatively affected lower income or food-deficit regions, and may have even caused social unrest (Trostle 2011). Additionally, with an increasingly connected global market for agricultural products, crop consumption and production may occur in distant countries. The consumption in importing countries may increase without appreciating the constraints of limited natural resources or pollution, while the exporting countries suffer intensive environmental burdens, such as soil N mining, loss of soil fertility, N₂O emissions, GHG emission, air and water pollution, and loss of biodiversity (Galloway et al. 2008, Schipanski and Bennett 2012). For example, deforestation in the Amazon is related to the massive cultivation of soybean exports to China (Martinelli et al. 2012, Lassaletta et al. 2014b). The eutrophication of the Gulf of Mexico in the US is largely caused by the maize-soybean rotation in the corn-belt area of the Mississippi river basin, where 49% of the soybean and 14% of the maize produced are destined for export (McLellan et al. 2015).

Consequently, it is critical to understand and quantify the trade-offs of using international trade as one of the strategies for resolving food demand and environmental challenges. Most existing research efforts have focused on the aggregate N contents and N pollution embedded in traded crop products but have not been able to reflect the distinct roles that each crop plays in N pollution and food security during production and trade (Lassaletta et al. 2014a, Oita et al. 2016, Shi et al. 2016). For example, Oita et al. (2016) treated the ‘agriculture’ sector as a whole in their multi-region input–output database when estimating the embedded N emission along the supply chain through the linkages between N emissions at the crop production stage and monetary transaction data among regions and sectors. The consideration of crop mix is important yet challenging—different crop types have different impacts on the food supply and nutrient pollution, but the environmental impact of crop-specific data is limited (Lassaletta et al. 2014a). Using crop-specific fertilizer application data for 27 countries and regions, Shi et al. (2016) evaluated the impact of trade on reducing N pollution in China and increasing N pollution in exporting countries, but did not assess the overall impacts on global N pollution or other sustainability concerns such as land use. The lack of systematic approaches for assessing the impacts of trade on sustainability has been preventing us from understanding the synergies and trade-offs among different environmental and socioeconomic concerns related to trade and crop production.

In this study, we aimed to use historical records of crop production and trade to shed light on sustainability policies for balancing food demand, crop production, trade expenditure, and the environmental degradation associated with food production in China. In particular, leveraging the recent advances in crop–specific N budget data (Zhang et al. 2015a), we, (1) developed methodologies to quantify the N pollution associated with crop imports and domestic crop production over 1986–2015, (2) assessed the historical impacts of the crop trade on sustainability from environmental, social, and economic perspectives, and (3) investigated the alternative strategies of meeting both crop demands and environmental targets in China over this historical period. This study is among the few to quantitatively evaluate the role of crop trading, considering both the socioeconomic and environmental impacts, and is one of the pioneer studies to
consider the impact of crop mixes in an import portfolio and domestic production.

2. Data and methods

2.1. N budget and crop trade flows

Following methodologies developed for the Global Database of N Budget in Crop Production (GDNBCP, Zhang et al 2015a), we updated the N budget database for the period of 1986–2015 based on crop production and fertilizer records from the Food and Agricultural Organization of the United Nations (FAOSTAT, http://fao.org/faostat/en/#data/TM, accessed on 13 April, 2018). The N budget database counts N inputs and N outputs of the crop production system by country and crop type. The N inputs (Tg N) comprise all N sources used by crops—fertilizer, livestock manure, symbiotic fixation and atmospheric N deposition, and N outputs (Tg N) are dominated by the harvested N embedded in crop products. Each crop’s N content was derived from Bouwman et al (2005), Lassaletta et al (2014b) and Zhang et al (2015a).

The difference between the N inputs and N outputs is defined as the N surplus (Tg N), indicating the amount of N could be potentially lost to the environment (Bouwman et al 2013). N use efficiency (NUE), the N output-input ratio, reflects the transfer efficiency of N in a crop production system (Lassaletta et al 2014a). N yield (kg N ha⁻¹) is the N output of each ha of harvested crop, reflecting the N productivity in cropland.

We constructed the global N bilateral trade flows using the N contents of primary crops and bilateral trade volumes of food products over 1986–2015. The bilateral trade data was from the FAOSTAT database (http://fao.org/faostat/en/#data/TM, accessed on 20 August, 2018). We aggregated the bilateral trade data (including crops and food products) into the primary crop equivalent quantities of 159 crops, using the conversion factor from Dalin et al (2017). We assumed China’s crop consumption to be the sum of each crop’s domestic production and net imports (imports minus exports). Crop consumption is originally estimated for all 159 crops defined in FAOSTAT. We grouped the 159 crops into 11 crop types (wheat, rice, maize, other cereals, soybean, oil palm, other oilseeds, fiber crops, sugar crops, fruits/vegetables, and other crops) according to the International Fertilizer Industry Association’s report on fertilizer use by crop (Heffer 2009).

2.2. The impacts of the crop trade on sustainability

We assessed the impacts of the crop trade on sustainability from environmental, social, and economic aspects, considering N pollution and associated economic cost, land use pressure, stability of food supply, and trade expenditure.

2.2.1. Environmental impacts: ‘virtual’ and ‘alternative’ N and harvested area

The environmental impacts of the crop trade were assessed for exporting countries, importing countries, and the world. Importing crop products has great potential for alleviating land use pressure, water shortages, and N pollution in importing countries but adding those environmental stresses to their exporting partners. So far, the concept of ‘virtual’ water has been broadly used to evaluate the impacts of trade on water use in exporting countries (Allan 1998). Specifically, ‘virtual’ water refers to the amount of embedded water associated with production across international borders as a result of trade, revealing the water costs behind the traded product (Hoekstra and Hung 2002). We borrowed the ‘virtual’ water concept and defined ‘virtual’ N inputs, N surplus and harvested area for each importer as the N inputs, N surplus and harvested area incurred in producing traded crops to its exporting partners.

As opposed to the ‘virtual’ concepts, we proposed concepts of ‘alternative’ N inputs, N surplus and harvested area to evaluate the potential N inputs, N surplus and harvested area required if all imported crops were produced domestically (figure 1). Taking an ‘alternative’ harvested area as an example, this counterfactual concept aims to assess how much the crop imports have potentially relieved the land use pressure for crop production in an importing country, and it does not suggest that the importing country actually has that amount of land available. We estimate the ‘alternative’ harvested area (or N inputs and N surplus) by assuming the additional production will be at the current yield and NUE levels at the importing country (figure 1). Admittedly, producing the imported crops domestically is likely to further intensify N use or expand cropland on marginal land, and consequently leads to lower NUE and N yield level (Qiang et al 2013, Zhang et al 2016, Ali et al 2017). Therefore, the ‘alternative’ N and harvested area estimated against the current domestic level is rather conservative, at least in the short term.

The difference between ‘alternative’ and ‘virtual’ harvested area evaluates the impacts of the crop trade for the world as a whole. The difference is caused by the different yield levels in importing and exporting countries. When an ‘alternative’ harvested area is higher than a ‘virtual’ harvested area, the associated trade link is saving land for both the importing country and the world. Similarly, a higher ‘alternative’ N inputs and N surplus suggests higher NUE in exporting countries, and consequently indicates the associated trade link is saving N inputs and reducing N pollution for both the importing country and the world.

2.2.2. Social impacts: food independence and self-sufficiency ratio (SSR)

Importing crop products may alleviate domestic environmental pressure, but may also increase a
country’s dependency on imports and may consequently threaten food supply, which is critical for social stability and sustainability (Suweis et al. 2015). To measure the impact of trade on food availability, we used the SSR, a food import dependency indicator. It is frequently used to characterize a country’s social capacity to provide stable supplies to meet its domestic food demands (Luan et al. 2013). Specifically, we calculated SSR for nitrogen (N SSRcr yr) as N in domestically produced crop cr divided by N in consumed crop cr in year yr for China (Gu et al. 2017). Although there is no ideal level for SSR, 85% has been considered as a bench mark: countries with SSR higher than 85% are considered to be ‘self-sufficiently safe’, countries with SSR lower than 85% may be exposed to more risks outside of the country’s judiciary (Clapp 2017).

2.2.3. Economic impacts: trade expenditures and environmental damage costs (EDCs)
Importing instead of producing crops domestically could increase a country’s trade expenditure, but potentially reduce the adverse environmental impacts associated with domestic production, and consequently avoid the cost of those environmental impacts to society. To assess such trade-offs, we estimate the EDC associated with domestic crop production based on a society’s willingness-to-pay to reduce reactive N compounds (N2O, NOx, NH3, NO3−) in the environment (Gu et al. 2012, Kanter et al. 2015). Damage cost for each reactive N compound is obtained from (Kanter et al. 2015, Zhang et al. 2015b). Please see supplementary material section S1, available online at stacks.iop.org/ERL/14/094021/mmedia for more details in EDC description and calculation.

2.3. Evaluating the role of trade in improving sustainability
Adjusting crop portfolios in trade and domestic production can potentially reduce N pollution while meeting other sustainability requirements such as food availability. To assess the potential of this opportunity, we designed algorithms to optimize the crop production portfolio in China. Optimization will minimize the N pollution resulting from domestic crop production under various constraints, whilst considering the sustainability requirements and feasibility of crop mix shift. The N pollution level was evaluated by the N surplus.

\[
\text{Minimize} \quad \text{N surplus} = \sum_{\text{crop}} N \text{ yield}_{\text{cr}} \times \left( \frac{1}{\text{NUE}_{\text{cr}}} - 1 \right) \times h_{\text{cr}},
\]

(1)

where \( h_{\text{cr}} \) represents the harvested area in each crop type (cr) that is subject to change. \( N \text{ yield}_{\text{cr}} \) and \( \text{NUE}_{\text{cr}} \) represent the nitrogen content and nitrogen use efficiency of each crop type, respectively.
suitable climate and soil conditions for oil palm is no more than the current level due to the fact that two other constraints: shifts in a crop production portfolio. Therefore, we set Soil characteristic and climate conditions limit the are N yield and NUE for crop cr. The optimization of the crop production portfolio considered the following four categories of constraint:

1. Crop N consumption (N Cons cr): Local production and net imports can meet the current N consumption level of each crop type. We assume that historical stock changes are negligible

\[ N_{yield_{cr}} + h_{cr} + NetImports_{cr} \geq N_{Cons_{cr}} \]  \hspace{1cm} (2)

2. Land use: The sum of the harvested area of all crop types is equal to the current harvested area level (\( \sum HArea_{cr} \)), suggesting no additional land use change for crop production

\[ \sum h_{cr} = \sum HArea_{cr} \]  \hspace{1cm} (3)

Soil characteristic and climate conditions limit the shifts in a crop production portfolio. Therefore, we set two other constraints: (a) the harvested area for oil palm is no more than the current level due to the fact that suitable climate and soil conditions for oil palm plantations in China are limited (Basiron 2007)

\[ h_{oilPalm} \leq HArea_{oilPalm} \]  \hspace{1cm} (4)

(b) Most rice cultivation cannot be used for other crop types due to the flooding irrigation management system used in China. Only 5% of cropland (i.e. rainfed rice cropland) can be switched to other crops (Peng et al 2009)

\[ h_{rice} \geq 0.95 \times HArea_{rice} \]  \hspace{1cm} (5)

3. Food availability: N SSR in wheat and rice is not less than 85%

\[ N_{yield_{cr}} \times h_{cr} \geq 0.85 \times N_{Cons_{cr}} \]  \hspace{1cm} (6)

4. Net import expenditures: China’s total net import expenditure on crops is not more than a certain level of total trade expenditure budgets (please see scenario designs for more details)

\[ \sum [(N_{Cons_{cr}} - N_{yield_{cr}} \times h_{cr}) \times CropPrice_{cr}] \leq Trade\ Expenditure\ budgets. \]  \hspace{1cm} (7)

Besides adjusting crop portfolios in trade and domestic production, improving crop N management (i.e. increasing crop NUE) is another major strategy in reducing N pollution in China. To evaluate and compare the potential of the two major strategies in reducing N pollution, we designed nine scenarios (3 x 3, see table 1), considering three levels of N management practices and three possible trade budgets on China’s net import expenditure (equation (7)). The three levels of N management practices were represented by setting China’s NUE at its current NUE level, the US level, and the global average level. We propose three levels of China’s net import expenditure constraints to evaluate the role of monetary budgets in N pollution reduction: (1) a conservative constraint where China would not spend beyond its current crop trade expenditure level, (2) an ambitious constraint where China’s import budget would be equivalent to 2% of its GDP—the share that the majority of countries would spend on crop imports, and (3) an extreme constraint where China would spend 6% of its GDP on trade imports (5% of countries spend more than 6% of their GDP on trade imports. Please see supplementary material section S2 for more details). We compared each of these scenarios with the baseline scenario where China’s crop production portfolio, NUE, and N yield were at their current levels (average of 2011–2015).

### Table 1. The settings of scenarios for NUE and trade expenditures.

| NUE                        | Net import expenditures     |
|---------------------------|-----------------------------|
| (A) At the current level  | (1) Conservative: keep constant |
| (B) At global average level | (2) Ambitious: 2% GDP |
| (C) At the US level       | (3) Extreme: 6% GDP         |
| A1                        | A2                          | A3 |
| B1                        | B2                          | B3 |
| C1                        | C2                          | C3 |

3. Results

3.1. Historical shifts of crop consumption, production, and imports

Crop consumption in China has grown dramatically from 8.3 Tg N in 1986 to 20 Tg N in 2015 (figure 2(a)). Despite most of the crop consumption being met by domestic production, China’s reliance on imports has been increasing with an emphasis on N-rich soybean imports (figure 2(b)). In 2015, the share of imported crop N in China’s crop N consumption reached 31% (6 Tg N). In contrast, crop N exports to other countries were less than 2% of total crop N consumption.

China’s crop mixes in domestic production and import portfolios are different and have been changing over time. The consumption of cereal crops (including wheat, rice, maize, and other cereals) was steady at 6–7 Tg N yr\(^{-1}\) from 1986 to 2003 and increased after 2003 (figure 2(c)). However, their proportion of N consumption has declined from 73% to 50%. This reduction was a result of more protein requirements in feed formulations (e.g. soybeans) and consumption expansion in cash crops (e.g. fruits/vegetables). Domestic N consumption from fruits/
vegetables has increased more than four times from 0.3 Tg N in 1986 to 1.6 Tg N in 2015 and was mainly met by domestic production (figure 2(d)). As the soybean boom continued, the soybean share in China’s import portfolio rose after 1995 (figure 2(b)). In 2015, China only produced 0.7 Tg N soybeans domestically, and about 89% of soybean consumption was sourced from overseas.

3.2. Environmental impacts of trade: N pollution and crop land use
To meet China’s import demands, the rest of the world devoted 8 Tg N yr$^{-1}$ N input and 44 million ha land in 2015, leading to 2 Tg N yr$^{-1}$ N lost to the environment (figure 3, ‘alternative’ N input and N surplus and harvested area). But, if China had to produce the imported crop domestically, 12 Tg N yr$^{-1}$ N input and 62 million ha land would be required and 6 Tg N yr$^{-1}$ would be lost to the environment, aggravating the environment and land use burden in China (figure 3, ‘alternative’ N input, N surplus and harvested area). The additional land demand is equivalent to 22% of grassland or 25% of forest land (Kong 2014), which is unlikely to be met within China. During 1986–2015, China’s crop imports helped to avoid an additional 56 Tg N surplus in China and relieved the global N surplus by 36 Tg (the difference between ‘alternative’ and ‘virtual’ N surplus). In particular, in recent years (2011–2015), soybean imports from the US, Brazil, and Argentina saved 38 million ha of land in China, accounting for 21% of the total harvested area in China and saving 14 million ha of global cropland. Notably, the rising ‘alternative’ and ‘virtual’ N surpluses and harvested areas in recent decades emphasize the importance of China’s crop imports in easing the environmental pressure on China and the world.

The much higher ‘alternative’ N surplus than ‘virtual’ N surplus is caused by the lower NUE in China than the global average level (figure 4(a)). For most crop types, NUE in China was lower than the global weighted average, which was similar to or lower than the NUE of imported crops in China. For instance, the NUE of soybeans in China was 55%, far behind the world average (80%). Likewise, the NUE of imported soybeans in China (79%) was also much higher than China’s domestic NUE. The imported fruits/vegetables also had higher NUE (22%) than the global weighted average and domestic production,
although it was relatively low compared to other crops. Overall, the NUE of total imported crop products in China over 1986–2015, varying from 53% to 88%, was much higher than that domestically produced in China (figure S1(a)). Such historical pattern provides evidence for the comparative advantage theory in global trade, which suggests commodities are usually produced in input-efficient and cost-effective regions and redistributed to less-efficient regions through international trade (Kastner et al 2014).

Consistent with the comparative advantage theory, self-sufficient crops in China (e.g. rice and wheat) have a higher N yield than at the global level, while those import-dependent crops (e.g. soybeans) have a lower N yield (figure 4(b)). For example, China’s N yield for rice and wheat were 108 and 98 kg N ha⁻¹ yr⁻¹ respectively during 2011–2015, much higher than their global counterparts at 73 and 63 kg N ha⁻¹ yr⁻¹. However, China’s N yield for soybeans was at an evidently lower level (110 kg N ha⁻¹ yr⁻¹), much lower than that of the global weighted average (154 kg N ha⁻¹ yr⁻¹) and imported soybeans (171 kg N ha⁻¹ yr⁻¹). Driven by the NUE and N yield heterogeneity among different countries, crop imports can potentially meet a given crop demand while alleviating the pressure of N pollution and land use for importing countries and even the world (Mueller et al 2017, Zhang 2017).

3.3. Socioeconomic impacts of trade: trade expenditures and food independence

However, relief in the environment and land use burden came at a price: China has become increasingly dependent on crop imports, and trade expenditure on

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**Figure 3.** The ‘virtual’ and ‘alternative’ (a) N inputs, (b) N surplus and (c) harvested area in China.

**Figure 4.** NUE and N yield for different crops and regions around the world. (a) NUE and (b) N yield in each crop type were averaged for the period of 2011–2015.
crops has climbed from $2.2 billion in 1986 to $67 billion in 2015. This sharp increase in crop trade expenditure in China was mostly driven by the expansion of soybean imports, which avoided the environmental damage caused by additional domestic production. Based on the current level of income and the willingness to pay for N alleviation in China, the N pollution damage avoided by importing crops (i.e. ‘alternative’ N surplus) was priced at $22 ± 16 billion in 2015, lower than the price paid for imported crops (figure 5). We acknowledge that other socioeconomic benefits could be added by producing crops domestically, and the estimates of EDC for N have high uncertainty. As China’s income and the willingness to pay grows towards the US level (red line), the EDC which has been avoided (blue line) will rise and even surpass trade expenditure, indicating stronger socioeconomic incentives for importing rather than producing domestically.

Even though crop imports can alleviate domestic environmental pollution and resource pressures, high dependence on crop imports could increase a country’s food supply risks. China swung between net importer and net exporter of crop products before 2003, with N SSR higher than 90%; but its N SSR started to decrease slowly after 2003 and declined to 70% in 2015 (figure S2). The overall N SSR decline was mostly driven by the increasing imports of soybean and oil palm products, the SSR of which dropped to 12% and 35% respectively. In 2015, 96% of soybean imports were from the US, Brazil and Argentina, and the majority (>99%) of oil palm product imports were from Malaysia (2.3 million kg N) and Indonesia (1.4 million kg N). Relying on a crop supply from only a handful of countries potentially makes China’s food supply, even socioeconomic stability, more vulnerable to changes in trade policies and regional climate conditions.

3.4. The potential of trade in improving sustainability
Overall, importing crops for meeting domestic demand in China has exhibited strong trade-offs among various sustainability concerns, and those trade-offs are affected by crop portfolios in domestic production and imports. To supply 1 kg crop N in China, different crop types require different levels of trade expenditures for import, or different N inputs and land uses for domestic production, leading to different levels of N pollution and associated EDC (figure S3). In addition, cereal crops, which meet a large portions of the human need for protein and energy, are usually more critical to food security than other crop types. Therefore, given crop demands, optimizing crop portfolios in domestic production or imports can help achieve various environmental and socioeconomic targets related to sustainability.

The results from the optimization experiment suggest that shifting 29% of maize, and 37% of fruits/vegetables in domestic production towards soybean and other cereals could simultaneously meet the same crop demands with the current cropland area and reduce 3.4 Tg (12%) of N surplus in China (figure 6, A3 scenario). However, such shifts could lead to a total $467 billion trade expenditure which is equivalent to 6% of GDP per year in China, because fruits/vegetables imports are generally more expensive than soybean imports (table S2). If we restrict the import expenditures (at current level or 2% of GDP), the optimal crop production portfolio will import more maize instead of soybean (A1 and A2 scenario). Keeping current trade expenditure levels constant will allow 40% of maize cropland to be converted to soybean cultivation, supplying half of China’s domestic soybean consumption. However, it only reduces 1–2 Tg (3%–5%) of N surplus in China.
The potential of trade portfolio optimization in mitigating N pollution is less but comparable to the potential of crop N management improvement (i.e. increasing NUE), and is another option for alleviating N pollution in China. With the current crop production portfolio, if China’s NUE in each crop category reaches the global-average, N surplus will be reduced by 25% (from 29 to 22 Tg N). This reduction surpasses the N surplus reductions by optimizing the crop trade portfolio among different import expenditure constraints. Combining the two options will enlarge the mitigation potential. If China can improve its crop NUE to the global-average and optimize its trade portfolio, the N surplus could be reduced by 30%–36% (B1–B3 scenarios). If China’s NUE is further improved to the US level, the optimal crop mix will reduce the N surplus by 44%–50% (C1–C3 scenarios).

4. Discussion

4.1. The role of trade
Our analyses suggest that crop imports can provide opportunities for China to meet domestic crop demands while alleviating pressure on N pollution and crop land use without improving N management practices in domestic production. However, increasing food imports potentially lead to high trade expenditure and exposes the country to greater risks in international markets. Crop portfolios in domestic production and imports have proved to be important in balancing the trade-offs among various concerns, including N pollution, land use change, food supply independency, and economic trade expenditures.

There has been debate as to whether trade has led to higher or lower N pollution in China. Some people agree that importing grain, soybean, and meat products can reduce China’s N pollution and spare domestic cropland (Galloway et al 2007, Oita et al 2016). On the contrary, Sun et al (2018) argues that importing N-fixation crops and converting cropland from soybean to other crops could increase domestic N pollution. The apparent disagreement could be resolved using our analyses of ‘virtual’ and ‘alternative’ concepts for N surplus and harvested areas. To meet a given crop demand, importing instead of producing crops will divert the N surplus associated with crop production to other countries as ‘virtual’ N surplus, and consequently reduce the pressure for N pollution in China. Importing crops from countries with higher NUE (i.e. ‘alternative’ N surplus is higher than ‘virtual’ N surplus) will reduce the N surplus for the world as well. However, if the N pollution level for a given region is of major concern, shifting the crop production portfolio from crop types with high NUE (e.g. soybeans) to those with low NUE (e.g. fruits and vegetables) will result in higher N pollution. Nevertheless, it needs to be noted that such shifts in production portfolio are likely to reduce the ‘virtual’ N surplus, indicating an impact on N pollution outside of the region.

4.2. Implications for China’s trade and agriculture policies
Optimizing trade portfolios provides additional opportunities for N pollution mitigation but requires changes in trade and agriculture policy incentives. For example, our results suggest that producing more soybeans domestically instead of other low NUE crops could cut down N pollution (figure 6). However, producing soybeans in China is less profitable.
compared to those genetically modified organism (GMO) soybean pioneers, such as the US, Brazil and Argentina, where the cost of soybean production is much lower than conventional soybean producers like China (Yao et al. 2018). To address those challenges and incentivize the shifts towards more N-efficient crop production portfolios, the Chinese government may encourage the imports of crops with lower NUE in China (e.g. maize and fruits/vegetables) instead of soybean by levying lower tariffs and removing the logistical barriers for the imports of those crops. Relaxing China’s restrictions on seed imports may reduce the production cost of some crops (e.g. soybean) and make China more competitive compared to major GMO producers, but it may face major concerns around human and environmental safety, as well as ecological risks (Bawa and Anilakumar 2013). In addition to trade policies, investments in technology Research and Development (R&D) in both private and public sectors are critical in order to boost the yield and profitability of soybeans in China.

However, N pollution mitigation should not solely rely on optimizing crop trade portfolios; improving technologies and management practices in domestic crop production is essential for mitigating N pollution in China and globally. So far, scientists in China have made significant progress in assisting farmers in increasing or maintaining yield for major grains using less fertilizer (Cui et al. 2018); but NUE levels for major grains are still lower in China than the global average. In addition, it is imperative to recognize that NUE improvement in cash crops, especially fruits and vegetables (accounting for 36% of fertilizer use, 41% of N surplus, and continuing to increase), is urgently needed for China to achieve its zero growth target in chemical fertilizer use by 2020. There are still 300 million smallholder farmers who depend on cash crops to increase their income but have little knowledge about efficient fertilizer management. Further improvements in crop NUE requires not only technological advancement and extension services, but also shifts in agricultural policies, which have been prioritizing food security and economic profitability but neglecting their adverse environmental impacts. For example, the government has been subsidizing fertilizers and providing minimum price support for wheat and rice. Those policies may incentivize farmers to achieve high yields by overusing fertilizer or switching to cash crops that require intensive fertilizer use. Therefore, those policies need to be re-evaluated and changed to consider their impacts on aggravating N pollution.

4.3. Impacts of the trade tension between China and the US

The recent trade tension between China and the US imposes an external factor in influencing the crop imports in China. China is the second largest agricultural export destination for the US, and imported up to $18 billion crops from the US in 2015, accounting for one third of China’s total N imports. China’s ‘virtual’ N inputs and harvested area from US-China crop trade are 2.6 Tg and 12 million ha, respectively. On 6 July, 2018, China planned to impose an additional 25% import tariffs on agricultural products, such as soybeans and vegetables from the US. China may substantially decline the imports of agricultural products from the US and potentially reduce the environmental pressures on the US. Meanwhile, China may intensify domestic crop production and import more crops from other countries with lower NUE, which would potentially increase its domestic and global N inputs and N surplus. With the observed or projected shifts in crop production portfolios in China and US, the data and method developed in this paper can be used to analyze the environmental pressure changes driven by the US-China crop trade tension.

4.4. Limitations

In this study, we focused on the trade and production of crops, because crop production accounts for the majority of N fertilizer use, and contributes 74% N pollution from both crop and livestock sectors (Gu et al. 2017). Admittedly, with the increasing demand for food with high protein, China may increase the production and import of meat and dairy products. This will consequently affect the relative importance of crop production to N pollution and will change crop production portfolios. These dynamics warrant further investigation, but are beyond the scope of this study.

Our analyses on the roles of trade in alleviating environmental pressure is based on the assumption that all economic and environmental endowments are at static states, determined by the current situation. However, in reality, crop production expansion requires more resources and affects yield level, NUE, and environmental pollution (Zhang et al. 2015a). Economically, a greater crop supply will influence land rents, agricultural labor, capital costs, and crop prices, etc (Yao et al. 2018). Additionally, shifts in the crop production portfolio may lead to additional economic costs. Such market-mediated responses are worth further investigation using tools such as computable general equilibrium models. Our analyses provide a conservative assessment of the impact of trade on N surplus in importing countries at current technological, environmental, and economic situations.

Finally, this study mainly focuses on N in domestic production and crop imports, one of the macronutrients critical for human survival. However, other macronutrients, as well as micronutrients such as vitamin and mineral, are also critical for healthy diets. Conversion from fruits/vegetables to soybean production in our high trade budget scenarios (A3, B3, and C3) could potentially lead to deficiency in
micronutrients in domestic production, but could be addressed by other approaches such as imports and dietary supplements.

5. Conclusion

Through studying China’s crop production and trade over 1986–2015, we evaluated the impacts of trade on sustainability from environmental (e.g. N pollution and land use), social (e.g. crop self-sufficiency) and economic (e.g. trade expenditure and EDC) perspectives. Our findings show that crop imports can relieve N pollution and land use pressure in China and the world but add environmental burdens to other countries and expose China’s food availability to the risks of the international market. We have also found that the EDCs of N pollution avoided by importing crops in China are less than current trade expenditure, but may reach or surpass it as China’s economy develops.

Optimizing crop import portfolios clearly provides an opportunity to reduce China’s N pollution reduction at the current technology level, but the improving technologies and practices for nutrient management (i.e. improving NUE) is still important for reducing N pollution.

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Data availability statement

The data that support the findings of this study are available from the corresponding author upon reasonable request.

Author Information

Reprints and permissions information is available at www.nature.com/reprints. The authors declare no competing financial interests. Readers are welcome to comment on the online version of the paper. Correspondence and requests for materials should be addressed to XZ (xin.zhang@umces.edu).

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