The environmental consequences of a weakening US-China crop trade relationship

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Abstract

Consideration of tariffs on China's imports of US agricultural products have focused on economic impacts, while environmental consequences have received less attention. Tariffs could alter farming practices in both countries and spill over to the rest of the world, resulting in unexpected environmental consequences through crop portfolio changes. Using a global computable general equilibrium model, we show here that China's tariffs pose unexpected increases in nitrogen and phosphorus pollution and water extraction in the US, as farmers shift from soybeans to more pollution-causing crops, costing the US approximately an additional \$88~608 million for nitrogen mitigation. China's diverted demands to South America would reduce South American nutrient pollution and water uses, although it may add additional pressures on deforestation. On a global scale, trade policies could help to reduce nutrient pollution and water source depletion by promoting crop production where it is most efficient in nutrient and water use.

Introduction

The volatile US-China trade relationships have ignited a fierce debate on the economic relationships of the two countries. As one of the US top agricultural markets, China consumed \$19.6 billion US agricultural exports in 2017[1]. Of these US agricultural exports, 86% are crop products, of which 74% are soybeans[2]. China's retaliation on US agriculture has directly hit US farmers' incomes[3–6]. In 2018, China's retaliation led to a sudden drop in China's total agricultural imports from the US by about \$10 billion[7], especially soybeans[2]. However, little attention has been paid to the environmental consequences of such uncertain trade relationships, which will, in turn, affect the socioeconomic welfare for farming communities and downwind and downstream communities and ecosystems.

The weakening agricultural US-China trade relationship potentially has profound environmental impacts on the two countries and the world, given the current impacts of crop production. Agriculture demands over 60% - 80% of the consumptive water in both countries[8,9]. Agriculture is also a dominant nutrient polluter. Nitrogen (N) and phosphorus (P) applications to croplands in excess of crop demands are mostly lost to the environment, eventually leaching to water bodies causing eutrophication or emitted to the atmosphere as gases that alter climate[10–15] and impact human health[16,17]. Potential production shifts among crops with different resource use efficiency (RUE) are likely to change the national and local regional nutrient pollution and irrigation water demands.

RUE of crop production varies among crops and differentiates across countries, due to different production costs, practices, and technologies[18]. International trade enables the reallocation of crop production from the resource-inefficient regions (e.g., China) to the resource-efficient areas (e.g., US)[19,20]. Through crop imports, China avoided \$22 ± 16 billion N pollution damage costs in 2015[21].

Contrarily, reducing excessive N pollution was estimated to cost US about $157.1 billion per year[22] – much higher than $68 billion the US total crop export revenues in 2017[2]. Consequent crop shifts due to US-China trade tensions could also spill over to the rest of the world through trade flows– opening up opportunities for South America to step in and significantly increase its shares in China's market[3,23], potentially boosting its fertilizer uses and cropland expansion. Therefore, it is critical to evaluate the global environmental consequences of the potential US-China crop trade tensions, especially the impacts on irrigation water use and nutrient (i.e., N and P) pollution driven by crop shifts, beyond economic welfare changes[24].

To evaluate trade policy impacts on economic activities, Computable General Equilibrium (CGE) models have been widely used with well-considered market-mediated responses captured in production and consumption functions from all sectors[25–27,23]. However, few CGE studies have investigated the impacts of trade on both water consumption and nutrient pollution, which are major environmental impacts of crop production. In contrast, previous studies on linkages between trade and nutrient pollution usually neglected the market-mediated responses, ignored the different irrigation water and nutrient fertilizer use intensity among different crop types[28–30], and overlooked subnational impacts.

In this paper, we first characterized the environmental impacts of China-US crop trade by using N surplus and P surplus (see detailed definition in Methods) to evaluate excessive N and P lost to the environment and by using blue water consumption – the surface and ground water used for irrigation – to assess irrigation water use[18,21]. Then, we employed an extended version of Global Trade Analysis Project (GTAP) model – GTAP-BIO (See Methods and Supplementary Information (SI) S.1)[31–33], a CGE model to evaluate the static global and national long-term environmental impacts resulted from an example scenario of crop portfolio changes driven by

China's tariff increases on US agricultural products. Finally, we downscaled the environmental impacts to a gridded level using gridded crop distribution data and the modeled crop production shifts by agro-ecological zones (AEZ) to investigate the potential heterogonous impacts of trade policies on subnational scales (Fig. 1).

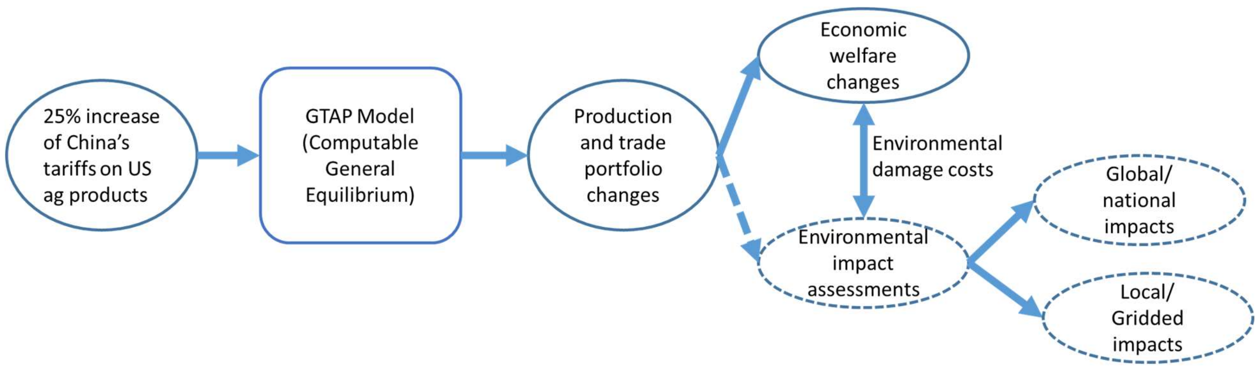

Fig. 1 Schematic flow of environmental impact evaluation structure. The square represents the model. Ovals with solid lines and solid arrows represent direct inputs and outputs from the model. Ovals with dashed lines and dashed arrows are calculations based on model output and region- and crop-specific environmental impact database. Economic welfare changes and environmental impact changes can be compared through environmental damage costs, calculated as the monetary values of environmental degradation.

The environmental impacts of existing US-China crop trade

In the past, producing the volumes of crops demanded by China exacerbated US environmental pressure, but potentially relieved the environmental stress for China and the world. In 2014-2016, on average, to produce the crops exported to China, the US devoted an additional 12 million ha of harvested area, demanded 4 trillion L more irrigation water, and added 0.5 Tg N and 0.007 Tg P pollution (Fig. 2). In contrast, if China's crop imports from the US were produced in China domestically with its current technology and management practices, assuming adequate resources, suitable climates, and soil conditions, it would require nearly 20 million ha harvested area, 9 trillion

L irrigation water, and lead to 1.7 Tg N and 0.3 Tg P. Such drastic shifts in the associated environmental stress would consequently produce a net increased global burden of environmental impacts, including a net additional exploitation of 8 million ha of harvested area and 5 trillion L of irrigation water, and further losses of 1.2 Tg N, and 0.3 Tg P to the environment (See Methods for calculations).

Hypothetically, if China's crop imports from the US were entirely substituted by other regions of the world, the environmental impacts of the production would vary among regions due to the different production efficiencies and available resources (Fig. 2). For example, to produce the same volume of crops demanded by China, Brazil would require additional 2 million hectares (ha) of harvested area and would induce an additional 0.5 Tg and 0.22 Tg N and P surplus, respectively, compared to the US, but would substantially reduce irrigation water demands. Other South American countries (SoAmer) would be a less polluting alternative to the US given that region's more efficient N and irrigation water use, but higher costs and limited resources available may impede it from replacing the crop supply from the US entirely[23,34,35]. Overall, this comparison suggests that the current production reallocations from China to the US achieved through international trade has relieved environmental burdens for China and the world, due to China's relatively low efficiencies in both irrigation water and fertilizer use compared to the US and the rest of the world. Admittedly, this comparison with Brazil and SoAmer demonstrates an extreme case of production reallocation from the US to the rest of the world, and such reallocation is likely to be buffered by market-mediated responses with considerations of biophysical limits, but it demonstrates the general direction of changes in environmental impacts in the context of weakening China and US crop trade.

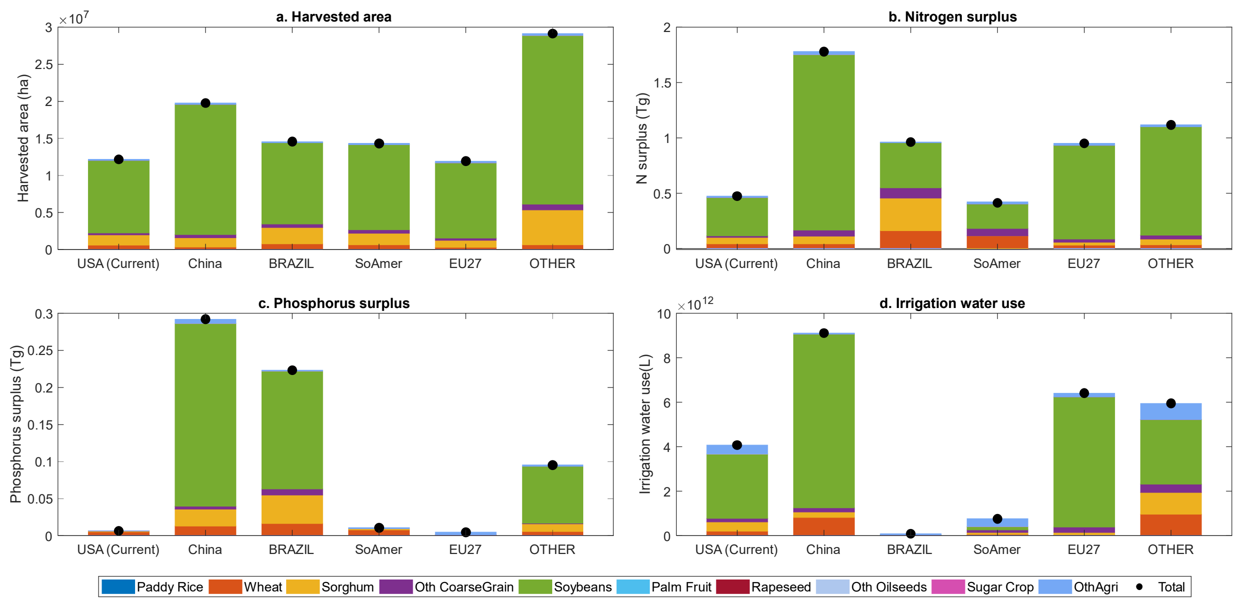

Fig. 2 Required harvested area (ha) (a), induced N surplus (Tg) (b), P surplus (Tg) (c), and irrigation water uses (L) (d) for producing the crop portfolio China imported from the US average 2014-2016 at each region's current resource use efficiency levels, given adequate resources and suitable climate and soil conditions. Black dots denote the total environmental impacts that are possibly generated. Each colored bar indicates the environmental impact of each crop type. Only the US bars show the environmental impacts that actually happened. Other regions' bars are the potential environmental impacts that would have happened if those regions grew the same crops to meet China's demands. For example, in the N surplus graph, the US bar shows the US actually generated 0.5 N surplus when producing the crops exported to China. China's bar in the N surplus group denotes that China would have generated 1.4 Tg N surplus if its entire imported crop portfolio from the US had been produced domestically. Similarly, if those crops had been grown in Brazil, the potential burdens are shown by the bars for Brazil.

The national and global impacts of weakening trade relationship

Considering market-mediated responses to the changing trade relationships and biophysical limits, we used the GTAP-BIO model to examine a case of a 25% tariff increase, given the existing production and policy situation in 2016, in five groups of US agricultural products – soybeans, other coarse grain (primarily corn), wheat, sorghum, and beef (See Methods section and SI Section S.1 for details). China's tariff increase on US agriculture would potentially increase prices of US agricultural products in China's domestic market, which would lower China's demands for US agricultural products, eventually hindering US farmers' income and discouraging them from producing relevant crops. Given the fact that over 70% of China's crop imports from the US are soybeans, domestic soybean prices in the US are mostly affected. Soybean production in the US is thus predominantly depressed. Crops that are less traded with China are less affected by China's tariff increase. US farmers would switch to these less-traded crops, such as other coarse grains, wheat, other oilseeds, and other agricultural products (othAgri). Relatively lower-priced soybeans from Brazil and SoAmer countries, due to the absence of tariffs, would incentivize China's soybean imports from these countries. China's rising demands increase the income of soybean farmers and motivate soybean expansion in Brazil and SoAmer countries, adding pressures to their cropland expansion. A reduction in the global supply of soybeans, which is one of the cheapest oilseed products, would spur China's oilseed production in general. However, with intensified agricultural production and limited harvested area expansion capability, China would experience limited changes in crop portfolio and harvested area. As soybean is the major protein source for ruminant animals[23,36], tariffs on beef could further disincentivize soybean production.

Shifts in crop portfolios are accompanied by environmental impact changes such as N surplus, P surplus, and irrigation water uses. Although the total harvested area in the US would contract, its total N surplus (measured by kg N) would increase as soybean, an N-fixing crop with relatively high nitrogen use efficiency (NUE) and low N surplus rate (measured by kg N ha^{-1}), shifts to other crops (Fig. 3: the reduction in N surplus due to soybean contraction in the US is less than additional N surplus generated by the expansion of other crops with higher intensity in N loss – i.e., higher N surplus rate). In contrast, Brazil and SoAmer countries would alleviate their N surplus thanks to substitutions of soybean for other crops. Despite the N surplus increase in the US and the N surplus decline in Brazil caused by a shift in where soybeans are grown to meet demands from China, these soybean-driven surplus changes offset each other globally. However, global N surplus would significantly benefit from the contractions of N-inefficient crops due to soybean expansion in South America, such as other coarse grain, wheat, sugar crops, and othAgri, and their increase in the US, where they can be grown more efficiently and with less N surplus to the environment.

The contraction of the harvested area in the US would not lead to the saving of irrigation water. The crops with increasing production in the US either replace soybean with higher water demand on the same land (e.g., corn in the Other Coarse Grain category) or expand in the region with high irrigation demand (e.g., Other Oilseeds). Although it is unlikely that farmers will invest significantly for irrigation equipment given a short-term policy or market shock, it is possible for farmers to increase irrigation water use on the land already equipped with irrigation infrastructure, to shift crop type (e.g., from soybean-corn rotation to continuous corn), and perhaps even to invest in new equipment as the trade tension becomes a norm in the context of growing tensions between US and China. Therefore, the reduction of irrigation water use for soybean in the US is outweighed by the increase of irrigation water use for other crops (Fig. 3d). Similar patterns are observed on the global scale: global irrigation water demand would increase because the benefits of irrigation water savings from shifts in soybeans production (i.e., shifts from the US to Brazil and SoAmer) would be offset by the increasing irrigation water use in non-soybean oilseed expansion in water-scarce regions.

Trade-offs and synergies, however, also exist within each region across different types of environmental impacts. The expansion of soybean, an N-fixing plant that is relatively more efficient than many other crops, would be a "pro" for Brazilian N surplus but a "con" for its P surplus. Brazilian soybean is intensively produced in areas with highly-weathered and phosphorus-poor soil. Brazilian soybean production thus requires higher P fertilizer inputs than soybeans produced in most temperate regions[37]. With similar PUE levels, P surplus increase due to soybean expansion is higher than the P surplus decline driven by other crop's contraction – resulting in a net P surplus gain in Brazil. Although most of this P surplus is currently retained in Brazilian soils, the accumulated P in Brazilian soils could eventually reach saturation and pollute water bodies[38]. In addition, the increased demand for P fertilizer in Brazil may exceed domestic supplies of rock P reserves. In contrast to Brazil, the US will suffer from an increase in both P and N surplus as the production shifts from soybean to more fertilizer-use intensive crop types; while SoAmer would experience alleviation in both P and N surplus due to the shifts opposite to the US. Global P surplus would further aggravate as soybeans are expanded in P use inefficient Brazil.

Overall, the weakening US-China agricultural trade relationship would worsen the US environmental conditions in both nutrient pollution and water resource depletion. The projected aggravating US environmental conditions are also reinforced in terms of environmental impact intensity rate per ha of harvested area and resource use efficiency (total outputs per unit of inputs). All three measurement metrics show that SoAmer would be the largest "winner" in all three types of environmental impacts. Brazil would alleviate its N surplus and irrigation water use through crop mix changes but face an aggravating P surplus issue. China would experience limited environmental improvements. Globally, trade-offs exist between N surplus reduction and increases in P surplus and irrigation water use.

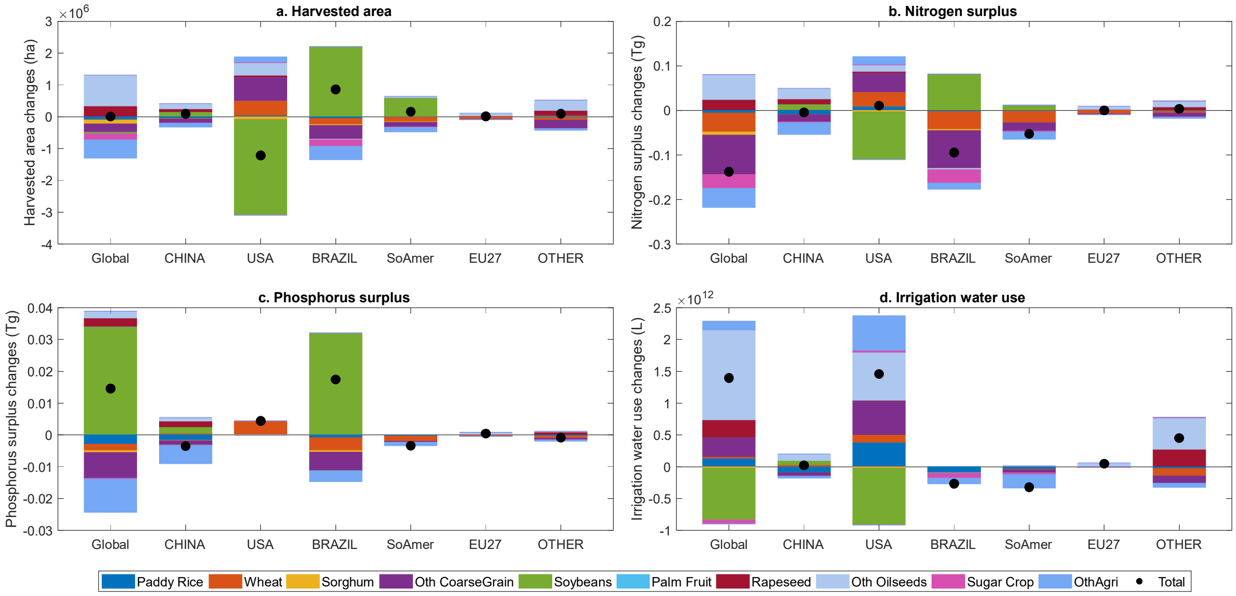

Fig. 3 Harvested area changes (ha) (a), total nitrogen surplus changes (Tg) (b), total phosphorus surplus changes (Tg) (c), and total blue water use changes (L) (d) by region and crop type. Total changes are denoted by black dots, which are further decomposed into contributions from changes in each crop type represented by colored bars. The leftmost bar summarizes total global changes as well as the contributions from each crop type.

Local (subnational) hotspots with additional environmental stress

Heterogeneous distributions of crops and varying crop mixes and RUE in crop production at subnational level make the environmental impacts of changes in trade relationships on a local scale diverge from the aggregate changes on a national scale (Fig. 4), leading to spatial trade-offs and synergies within each environmental impact and across different environmental impacts. To investigate the heterogeneous impacts on a subnational scale, we downscaled the modeling results from GTAP for each AEZ to 30 by 30 arc mins grids by assuming 1) the expansion of the harvested area for a crop type only happens in the grid or region with the production for the same crop type in 2014-2016; 2) the changes in harvested area for each crop at each grid cell is the same as the changes in its corresponding crop type in each AEZ within a country (See Methods for details). The downscaling approach has been applied in multiple studies[39–41], and the downscaled results represent one of the most likely changes in crop distribution on a subnational scale under the trade-tension. We focus the investigation on the subnational environmental impacts on per hectare of harvested area basis – intensities of N surplus, P surplus, and irrigation water use, which are direct results of crop portfolio changes regardless of changes in total agricultural activities.

The 25% tariff increase would lead to the contraction of soybean production mainly in US soybean/corn belt where the production of other coarse grain (primarily corn) expands – resulting in a reduction in total N surplus but an increase in N surplus intensity for this region (Fig. S7). The expansion of other coarse grain is much less than the reduction in soybean production, leading to the decline of harvested area for the region. However, since other coarse grain has higher N surplus intensity than soybean, the intensity of N loss increases on the remaining cropland (Fig. 4). The expansion of wheat would focus on northern and western regions of the Midwestern US, accompanied with increasing P surplus intensity. The expansion of other oilseeds production would locate in Southern regions, with minimal impacts on N surplus but higher requirements for irrigation water.

In contrast to the contraction in the US, soybean would expand in South America mainly in Brazilian central-west and south regions and Argentinian northeastern regions. While inserting pressures on land use changes in these areas, the expansion of soybean production may relieve N surplus intensity and irrigation water use intensity the cost of replacing wheat, other coarse grains, sugarcane, and OthAgri production. Consistent with national scale analysis, the Brazilian soybean production area may experience aggravated P surplus intensity but SoAmer would benefit from P surplus intensity due to different types of soils.

Considering all three environmental impacts together (i.e., the intensity of N surplus, P surplus, and irrigation water use), the increased tariffs would lead to the worsening of one or more environmental impacts in most regions (Fig. 4d). The region with reduced environmental stress mainly concentrates in Southeast and Northeast China where soybeans and rapeseeds expand at the cost of other resource-intensive crops, and Argentina where soybean production is incentivized. The rest of China is dominated by aggravating irrigation water use intensity, while some regions would face increased N surplus (green areas in Fig. 4d China) and P surplus intensity as well (green and orange areas in Fig. 4d China). Most regions in Midwestern, Southern, and Northeastern US are dominated by increases in N surplus and irrigation water intensity (green areas in Fig. 4d US), as part of soybean production shifts to other crops with more intensive N surplus and/or irrigation water use. Northern parts of the US West have modest concerns on intensifying nutrient surplus, and southern areas of the US West have slight intensification in irrigation water use. The Brazilian Amazon region faces the situation of intensified nutrient losses and irrigation water uses of existing agricultural practices– as a result of a reduction in resource use efficient crops in crop mixes. The rest of Brazil is dominated by intensified P fertilizer use and P losses in soybean intensive areas. Uncertainties in the elasticities will not alter the directions of the changes at local levels. Rather, it may amplify the changes –improved areas will benefit more and degraded areas will be further aggravated (See SI Section S.4 and S.7). Subnational assessments in this analysis provide directional guidance of potential changes.

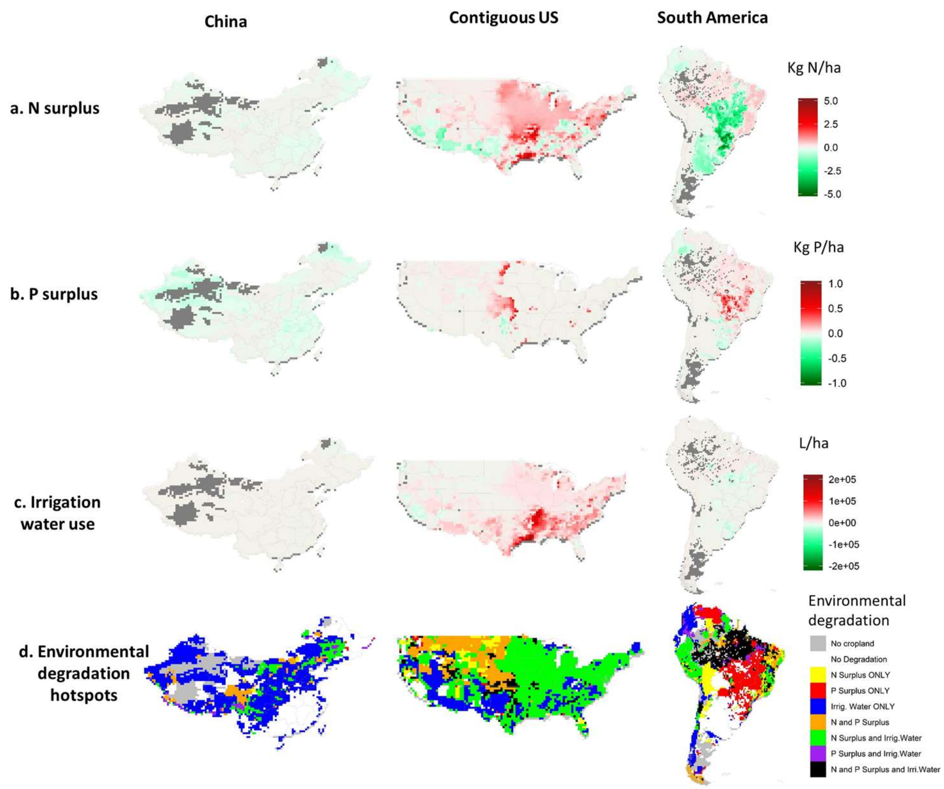

Fig. 4 N surplus, P surplus, and irrigation water use intensity changes across different regions in China, Contiguous US, and South America. Panel a-c present gridded N surplus changes in kg N/ha, P surplus changes in kg P/ha, and irrigation water use changes L/ha due to China's potential trade policy on the US agricultural products. Red grids mean increases in environmental degradation, the green grids represent decreases in environmental degradation, and the grey grids represent no cropland. Combining the three environmental impacts, the lower panel shows the hotspots of increased environmental degradation. The yellow area shows the regions that will only experience an increase in N surplus. The red area represents hotspots with a P surplus increase only. The blue areas are those that will experience rising irrigation water demands. The orange area indicates the regions with both N and P surplus increase; the green areas with both N surplus and irrigation water increases; the purple areas with both P surplus and irrigation water increases; and the black areas with environmental degradation from all three types of environmental impacts.

Discussion

Implications for future policy

The environmental consequences of crop trade need to be factored into the decision-making on trade policy. The current effectiveness of trade policies is primarily evaluated based on the economic welfare of society, while the environmental impacts have often been ignored. To compare environmental impacts with economic impacts, economic damage costs (EDC) – the willingness to pay to avoid potential societal damages driven by environmental degradation resulted from unbalanced ecological equilibrium [42,43]– have often been employed to convert environmental degradation (e.g., N surplus) to monetary values. With a 25% tariff increase imposed by China to the US crop groups in this analysis, the US would lose $2.2 billion dollars total economic welfare, and the US would pay additional $88~608 million to alleviate its additional surplus N, accounting for up to 16% of the US economic welfare reduction, let alone additional EDC for P surplus and water stress alleviation[42,43] (See SI S.4.3 for details). Economic costs for environmental consequences are non-negligible.

Although nutrient pollution and irrigation water consumption are primarily localized, international agricultural trade globalizes these environmental impacts, not only within the partner countries, but also through spillover impacts. The assessments of environmental consequences of international trade across various spatial scales provide critical information for facilitating international conversations on mitigating nutrient pollution and relieving water shortages. While efforts have been made by the global society to collectively address climate change, much less has been done to coordinate the efforts in mitigating nutrient pollution and water scarcity, which also have global scale impacts (e.g., N_2O emission contributes to climate change and ozone depletion). The across-scale environmental impact evaluations in this analysis offer policy-makers a new lens in trade policy negotiations and international collaborations for addressing global environmental challenges.

The varying impacts of trade policy on subnational scales highlight the need for improving the capacity of policy-makers in projecting environmental degradation hotspots and developing strategies to alleviate the negative impacts accordingly. Our analysis provides among the first attempts to downscale the impacts of trade policy change on a gridded level, highlighting the regions that will be negatively affected by the increasing tariff and their major environmental concerns. Although the downscaling approach has uncertainties, the results demonstrate the complex trade-offs among regions and environmental impacts and demonstrate the need to develop strategies to minimize the negative social-environmental outcomes of the trade policy changes. While the effects of international trade discussed here are important, it also needs to be recognized that the sustainability of agricultural production is affected by many socioeconomic and ecological drivers beyond trade, and improvements in agricultural technologies and practices are still fundamental for achieving sustainable agriculture.

Limitations

Current trade negotiations are underway, and actual tariffs imposed on US agricultural products are subject to sudden change during the process of the negotiation and are beyond the scope of this manuscript. However, as China has been increasingly viewed as a threat to the US's economic and political power in the world, China is likely to seek options to reduce her reliance on national food security on agricultural imports from the US, which aligns with the trade scenario examined in this study.

Environmental consequences resulted from trade policies are not limited to nutrient pollution and irrigation water use. Although we found that US-China agricultural trade tension could relieve N surplus and irrigation water use in South America, additional pressures on cropland expansion could potentially lead to pastureland contraction and deforestation, aggravating climate changes. Induced crop mix and land cover changes could also affect biodiversity. Land use changes and biodiversity losses are not direct results from crop mix changes and thus beyond the scope of this analysis.

This analysis contributes to the current environmental impact evaluations by considering the market mediated responses. In the GTAP model, elasticities govern substitutions among substitutable commodities in industrial manufacturing, and private consumptions, and government purchases, substitutions among imported crops from different origins, and land transformations of different crop types and land covers. This analysis adopts standard GTAP elasticities based on GTAP v.9[44]. However, there is evidence showing soybeans from different origins could be treated as homogenous in the global market[45,46]. As the most-traded crop between the US and China, increased elasticities in soybean imports could amplify the environmental impacts in this analysis[45]. Additionally, flexible land transformations across different crop types could possibly exaggerate environmental impacts in this analysis. Therefore, we also conducted a sensitivity test by assuming soybeans from different origins are identical, and farmers' can easily switch among different crop types if climate and soil conditions permit. This test provides the potential maximum environmental impacts that China's tariff retaliation on the US agricultural products could insert. We find that the direction of impacts remains the same, but the magnitude of changes could be 2-4 times higher than the standard scenario. Please see SI S.7 for more details. Despite the uncertainties embedded in the elasticities, the direction of changes in crop production due to the trade tension would provide solid guidance in policymaking. In this analysis, we assume the fertilizer application rate by each crop type in each region remains constant, which could also be subject to change as fertilizer costs are affected by market-mediated price signals. However, with technology unchanged, market-response variations in nutrient surplus rate may not be very different from this analysis.

Gridded level analyses are based on linear extrapolation from 2000 gridded crop production data to the average of 2014-2016 gridded harvested area data according to each crop's national growth rate. The result could be more accurate with more recent gridded crop production data becoming available.

Conclusions

This analysis is among the first to evaluate the impacts of crop trade policies on the environment (including N and P pollution and irrigation water use) from subnational to global scales due to shifts in crop-mix. We found that environmental consequences driven by the US-China crop trade tension are considerable and could spill over to other countries, especially Brazil and other South American countries through international trade. Trade-offs and synergies exist across different countries for each type of environmental impact and among different environmental impacts within each country. Overall, however, modeled effects of tariffs resulted in increased global environmental impacts. Additionally, spatial synergies and trade-offs occur within each country due to heterogeneous distributions of crop production. Environmental degradation hotspots were identified with one or more types of environmental impact. Future trade policies should consider environmental consequences in addition to the economic outcomes, as the indirect costs to the society through environmental degradation may be comparable to the economic outcomes.

Methods

GTAP Model

The global trade analysis project (GTAP) model is a global multi-regional and multi-sectoral model built upon the Computable General Equilibrium (CGE) theories. Compared to environmental models, the GTAP model captures market-mediated responses through economic theories of supply, demand, trade, and macroeconomic equilibrium relationships. It well considers behaviors from different economic entities, producers, consumers, and government. Decisions of all these entities are driven by relative price levers that are potentially changed by external interventions, such as policies in this analysis. Bilateral trade flows link all economies, and changes in one economy will affect its partners and thus spillover to the globe. GTAP model follows the Armington assumption that the same imported crop is differentiated based on its origns[47]. Imported and domestic crops are substitutable in private, industrial and government consumptions.

In this analysis, we adopt a modified version of the GTAP extension – GTAP-BIO[31]. The version that we use consists of six regions: the US, EU27, Brazil, China, other South America (S_o_Amer), and other regions (OTHER). Land endowments in the model are categorized into 18 agro-ecological zones (AEZ). Harvested area changes driven by market-mediated responses can be tracked by AEZ in each region with a consideration of physical limits – crops can only expand at areas where they were historically grown. The model has ten crop types: paddy rice, wheat, sorghum, corn and other coarse grains, soybeans, palm fruit, rapeseeds, other oilseeds, sugar crops, and other agricultural products (othAgri). The model considers four types of land cover: cropland, pastureland, forest, and unmanaged land. Four types of land can be transformed into each other to some extent with total land within each region/AEZ remaining constant. The model also takes account of multiple cropping practices, and the latest version of the model is reported in Taheripour et al.[32]. Besides land endowments, labor and capital endowments are also considered in producers' decisions. A detailed description of the GTAP model and GTAP-BIO is available in the SI section S.1.

Crop-specific environmental impacts by region

This analysis considers three environmental impacts: N surplus, P surplus, and irrigation water use. N surplus and P surplus, the excessive N and P lost to the environment, are defined as the differences between N (or P) inputs and N (or P) outputs (equation (1)). We use N surplus and P surplus as proxies of nutrient pollution to the environment from agricultural activities. Total N surplus and P surplus are measured in kg.

$$\text{N (or P) Surplus (kg)} = \text{N (or P) Inputs (kg)} - \text{N (or P) Outputs (kg)} \tag{1}$$

N (or P) surplus intensity are N (or P) surplus (kg) per hectare (ha) of harvested area (equation (2)).

$$\text{N (or P) Surplus Intensity } (kg\ ha^{-1}) == \frac{\text{N (or P) Surplus (kg)}}{\text{Harvested Area (ha)}} \tag{2}$$

N (or P) use efficiency (NUE or PUE) are defined as the ratio between N (or P) outputs and N (or P) inputs (equation (3)).

$$\text{NUE (or PUE)} = \frac{\text{N (or P) Outputs (kg)}}{\text{N (or P) Inputs (kg)}} \tag{3}$$

Irrigation water use refers to blue water – the surface and ground water used for irrigation[41,48]. Total irrigation water use is measured in liter (L). Irrigation water intensity rate is total irrigation water uses (L) per ha of harvested area. Irrigation water use efficiency is total product outputs (kg) per L of irrigation water.

The environmental impact database composes 169 crops and 218 regions. We aggregate them into ten crop types and six regions corresponding to the GTAP model. The classification of 169 crops to these ten crop types can be found in SI section S.2. We use an average of 2014-2016 environmental impact data as the base year.

Alternative environmental impacts

Alternative environmental impacts in Fig. 2 refer to the environmental impacts occurred if China's imported crops are entirely produced domestically or in alternative regions' at their resource-use efficiency level by assuming adequate resources and suitable climate and soil conditions at alternative regions *i*. Alternative environmental impact is a hypothetical and counterfactual concept widely used in evaluating the resource use efficiencies of traded crops and the potential environmental stress that foreign demand could impose on local agricultural production[21]. Alternative harvested area (equation (4)), N surplus, and P surplus (equation (5)) follows the calculation in Huang et al.[21], and alternative irrigation water is calculated as imported crops in kg divided by irrigation water use efficiency (kg/L) at alternative regions (equation (6)).

$$Alt\ Harvested\ Area_{i,China}\ (ha) = \frac{N\ (or\ P)\ ImportedCrops_{China}\ (kg)}{N\ (or\ P)\ Yield_i\ (kg\ ha^{-1})} \tag{4}$$

$$Alt\ N\ (or\ P)\ Surplus\ _{i,China}(kg) = N\ (or\ P)\ ImportedCrops_{China}\ (kg\ )(\frac{1}{NUE\ (or\ PUE)} - 1) \tag{5}$$

$$Alt\ Irrigation\ Water\ _{i,China}(kg) = \frac{ImportedCrops\ (kg)}{Irrigation\ Water\ Use\ Efficiency\ (kg/L)} \tag{6}$$

Gridded data analyses

Gridded harvested area data for 175 crops in the year 2000 at 5 arcmins level is obtained from EARTHSTAT[49]. We aggregate 175 crops into 169 crops and aggregate 5 arcmins cells into 30 arcmins grid cells. We linearly extrapolate each crop's harvested area from 2000 to the average of the 2014-2016 level based on their national growth rate[39]. We assume that each crop's harvested area changes within each grid cell are based on the percentage changes of its crop type of the AEZ within each region it belongs to. Harvested area changes and environmental impacts are first evaluated using based on 169 crop types and 218 countries and then aggregate into ten crop types and six regions. In doing so, we try to keep the information losses at the aggregation procedure at minimum.

Data availability

N surplus intensity rate is calculated following the procedure described in Zhang et al. [18]using the data from FAOSTAT[50]. P surplus intensity rate is derived using the methodology described in Tan et al.[51] using the data from FAOSTAT[50]. Irrigation water intensity rate is calculated using the blue water footprint provided by Mekonnen and Hoekstra[41] and yield and harvested area data from FAOSTAT[50]. Gridded crop in distribution maps is downloaded and processed using the database Harvested Area and Yield for 175 Crops year 2000 from EARTHSTAT provided by Monfreda et al.[49]. Global Trade Analyses Project (GTAP) database Version 9 is used in this analysis and updated from the base year 2011 to the year 2016 by Taheripour and Tyner[31].

References

1. FCA. *Major U.S. Agricultural Export Markets: China, Canada, and Mexico*. https://www.fca.gov/template-fca/download/MajorUSAgriculturalExportMarkets.pdf (2018).

2. UN Comtrade. UN Comtrade Database. (2019).

3. Taheripour, F. & Tyner, W. E. Impacts of Possible Chinese 25% Tariff on US Soybeans and Other Agricultural Commodities. *Choices* **33**, 1–7 (2018).

4. Marchant, M. A. & Wang, H. H. Theme Overview: US–China Trade Dispute and Potential Impacts on Agriculture. *Choices* **33**, 1–3 (2018).

5. Chen, B., Villoria, N. B. & Xia, T. Import Protections in China's Grain Market: An Empirical Assessment. (2018).

6. Bakst, D. & Beaumont-Smith, G. Agricultural Trade with China: What's at Stake for American Farmers, Ranchers, and Families. *US Department of Agriculture* (2018).

7. USTR. *The People's Republic of China*. https://ustr.gov/countries-regions/china-mongolia-taiwan/peoples-republic-china (2019).

8. USDA. Irrigation & Water Use. *United States Department of Agriculture, Economic Research Service* https://www.ers.usda.gov/topics/farm-practices-management/irrigation-water-use/ (2019).

9. CWR. China Irrigation - Expanding irrigated land, improving efficiency & spending. *China Water Risk* http://www.chinawaterrisk.org/the-big-picture/china-irrigation/ (2018).

10. FAO. *Water for Sustainable Food and Agriculture: A report produced for the G20 Presidency of Germany*. 27 http://www.fao.org/3/a-i7959e.pdf (2017).

11. Pimentel, D. *et al.* Water resources: agricultural and environmental issues. *BioScience* **54**, 909–918 (2004).

12. Mateo-Sagasta, J., Marjani Zadeh, S. & Turral, H. *Water pollution from agriculture: a global review*. 29 http://www.fao.org/3/a-i7754e.pdf (2017).

13. The Fertilizer Institute. Fertilizer 101: The Big 3 - Nitrogen, Phosphorus and Potassium. (2014).

14. Magdoff, F. & Es, H. van. Management of Nitrogen and Phosphorus. in *Building Soils for Better Crops: Sustainable Soil Management* (Sustainable Agriculture Research & Education, 2009).

15. EPA. *Nutrient Pollution*. https://www.epa.gov/nutrientpollution/issue (2019).

16. Xing, J. *et al.* Quantification of the enhanced effectiveness of NO x control from simultaneous reductions of VOC and NH 3 for reducing air pollution in the Beijing–Tianjin–Hebei region, China. *Atmospheric Chemistry and Physics* **18**, 7799–7814 (2018).

17. Peel, J. L., Haeuber, R., Garcia, V., Russell, A. G. & Neas, L. Impact of nitrogen and climate change interactions on ambient air pollution and human health. *Biogeochemistry* **114**, 121–134 (2013).

18. Zhang, X. *et al.* Managing nitrogen for sustainable development. *Nature* **528**, 51 (2015).

19. Matsuyama, K. Agricultural productivity, comparative advantage, and economic growth. *Journal of economic theory* **58**, 317–334 (1992).

20. Zhang, X. Biogeochemistry: A plan for efficient use of nitrogen fertilizers. *Nature* **543**, 322 (2017).

21. Huang, G. *et al.* The environmental and socioeconomic trade-offs of importing crops to meet domestic food demand in China. *Environmental Research Letters* (2019).

22. Compton, J. E. *et al.* Ecosystem services altered by human changes in the nitrogen cycle: a new perspective for US decision making. *Ecology letters* **14**, 804–815 (2011).

23. Yao, G., Hertel, T. W. & Taheripour, F. Economic drivers of telecoupling and terrestrial carbon fluxes in the global soybean complex. *Global Environmental Change* **50**, 190–200 (2018).

24. Galloway, J. N. *et al.* International trade in meat: The tip of the pork chop. *AMBIO: A Journal of the Human Environment* **36**, 622–629 (2007).

25. Hertel, T. W. *Global trade analysis: modeling and applications*. (Cambridge university press, 1997).

26. Hertel, T. W. *et al.* Effects of US maize ethanol on global land use and greenhouse gas emissions: estimating market-mediated responses. *BioScience* **60**, 223–231 (2010).

27. Hertel, T. W. & Tyner, W. E. Market-mediated environmental impacts of biofuels. *Global Food Security* **2**, 131–137 (2013).

28. Lassaletta, L. *et al.* Food and feed trade as a driver in the global nitrogen cycle: 50-year trends. *Biogeochemistry* **118**, 225–241 (2014).

29. Oita, A. *et al.* Substantial nitrogen pollution embedded in international trade. *Nature Geoscience* **9**, 111 (2016).

30. Shi, Y., Wu, S., Zhou, S., Wang, C. & Chen, H. International food trade reduces environmental effects of nitrogen pollution in China. *Environmental Science and Pollution Research* **23**, 17370–17379 (2016).

31. Taheripour, F. & Tyner, W. E. Impacts of Possible Chinese Protection on US Soybeans. (2018).

32. Taheripour, F., Cui, H. & Tyner, W. E. An Exploration of agricultural land use change at the intensive and extensive margins: implications for biofuels induced land use change. *Bioenergy and Land Use Change* 19–37 (2017).

33. Taheripour, F., Hertel, T. W., Tyner, W. E., Beckman, J. F. & Birur, D. K. Biofuels and their by-products: Global economic and environmental implications. *Biomass and bioenergy* **34**, 278–289 (2010).

34. Sutton, M. C., Klein, N. & Taylor, G. A Comparative Analysis of Soybean Production Between the United States, Brazil, and Argentina. *Journal of the American Society of Farm Managers and Appraises* (2005).

35. Leibold, K. & Osaki, M. Soybean Production: USA versus BRAZIL. in (Agri benchmark Cash Crop Conference 2009, 2009).

36. Ishler, V. A. & Varga, G. A. *Soybeans and Soybean Byproducts for Dairy Cattle*. https://extension.psu.edu/soybeans-and-soybean-byproducts-for-dairy-cattle (2016).

37. Roy, E. D. *et al.* The phosphorus cost of agricultural intensification in the tropics. *Nature plants* **2**, 16043 (2016).

38. Riskin, S. H. *et al.* The fate of phosphorus fertilizer in Amazon soya bean fields. *Philosophical Transactions of the Royal Society B: Biological Sciences* **368**, 20120154 (2013).

39. Houlton, B. Z. *et al.* A world of cobenefits: Solving the global nitrogen challenge. *Earth's future* (2019).

40. Siebert, S. & Döll, P. Quantifying blue and green virtual water contents in global crop production as well as potential production losses without irrigation. *Journal of Hydrology* **384**, 198–217 (2010).

41. Mekonnen, M. M. & Hoekstra, A. Y. The green, blue and grey water footprint of crops and derived crop products. *Hydrology and Earth System Sciences* **15**, 1577–1600 (2011).

42. Kanter, D. R., Zhang, X. & Mauzerall, D. L. Reducing nitrogen pollution while decreasing farmers' costs and increasing fertilizer industry profits. *Journal of environmental quality* **44**, 325–335 (2015).

43. Johansson, R. C. & Randall, J. Watershed abatement costs for agricultural phosphorus. *Water Resources Research* **39**, (2003).

44. Aguiar, A., Narayanan, B. & McDougall, R. An overview of the GTAP 9 data base. *Journal of Global Economic Analysis* **1**, 181–208 (2016).

45. Yao, G. & Hillberry, R. Structural Gravity Model Estimates of Nested CES Import Demands for Soybeans. (2018).

46. Hillberry, R. H., Anderson, M. A., Balistreri, E. J. & Fox, A. K. Taste parameters as model residuals: assessing the "fit" of an Armington trade model. *Review of International Economics* **13**, 973–984 (2005).

47. Armington, P. S. A theory of demand for products distinguished by place of production. *Staff Papers* **16**, 159–178 (1969).

48. Dalin, C., Wada, Y., Kastner, T. & Puma, M. J. Groundwater depletion embedded in international food trade. *Nature* **543**, 700 (2017).

49. Monfreda, C., Ramankutty, N. & Foley, J. A. Farming the planet: 2. Geographic distribution of crop areas, yields, physiological types, and net primary production in the year 2000. *Global biogeochemical cycles* **22**, (2008).

50. FAO. FAOSTAT statistics database. (2018).

51. Zou, T., Zhang, X. & Davidson, E. A. Improving phosphorus use efficiency in cropland to address future phosphorus challenges and dilemmas in 2050. (2019).