The environmental sustainability challenge of future food demand in China and its major trading partners

Hao Zhao[1,2,3], Petr Havlík[3]*, Jinfeng Chang[3,4], Van Dijk Michiel[3,5], Hugo Valin[3], Charlotte Janssens[3,6], Lin Ma[1]*, Zhaohai Bai[1], Mario Herrero[7], Pete Smith[8], Michael Obersteiner[3]

1, Key Laboratory of Agricultural Water Resources, Hebei Key Laboratory of Soil Ecology, Center for Agricultural Resources Research, Institute of Genetic and Developmental Biology, Chinese Academy of Sciences, 286 Huaizhong Road, Shijiazhuang 050021, Hebei, China.

2, University of Chinese Academy of Sciences, 19 A Yuquan Road, Shijingshan District, Beijing 100049, China.

3, International Institute for Applied System Analysis (IIASA), Schlossplatz 1, Laxenburg A-2361, Austria.

4, College of Environmental and Resource Sciences, Zhejiang University, 310058, Hangzhou, China.

5, Wageningen Economic Research, Wageningen University and Research, Prinses Beatrixlaan 582 – 528, 2595, BM, The Hague, The Netherlands.

6, Department of Earth and Environmental Sciences, University of Leuven (KU Leuven), Heverlee, Belgium.

7, Commonwealth Scientific and Industrial Research Organisation, St. Lucia, 4067 QLD, Australia.

8, Institute of Biological and Environmental Sciences, University of Aberdeen, 23 St Machar Drive, Aberdeen AB24 3UU, UK.

* Corresponding authors: havlikpt@iiasa.ac.at, malin1979@sjziam.ac.cn

Abstract

Satisfying China's food demand without harming the environment is one of the greatest sustainability challenges for the coming decades. Here we provide a comprehensive forward-looking assessment of environmental impacts of China's growing demand domestically and for its trading partners. We find that the projected 41% increase in livestock product consumption by 2050, would domestically require additional 55 Mha of cropland devoted to producing animal feed, and 38 Mha of pasture expansion, and would lead to a 33% increase in agricultural greenhouse gas emissions. Imports of agricultural products are projected to expand by 144%, almost doubling the reliance of China on imports from 10% in 2010 to 18% in 2050. As a result, more additional agricultural land would be virtually imported to China (49 Mha) than brought into production within China (44 Mha). Thus, to limit negative environmental impacts of its growing food consumption, besides domestic policies, China needs to take more responsibility in the development of sustainable international trade.

Introduction

China has undergone remarkable economic and social development over the past two decades to become the world's second largest economy. This successful development has led to a large increase in demand for food (14%), especially for livestock products (39%)[1,2]. At the same time, according to the Ministry of Agriculture in China, the import value of agricultural products has increased by 78% in constant USD since 2010[3] while domestic agricultural value increased by 36% from 2010 to 2018. For soybean products in particular, the reliance on imports increased from 47% to 89%; for ruminant meat from 2% to 10%, and for dairy products from 11% to 24%, between 2000 and 2015. The increasing demand also presents a great challenge to achieving the Sustainable Development Goals (SDGs)[4] in China and worldwide as the agricultural sector is a key contributor to greenhouse gas (GHG) emissions[5] (SDG13), air and water pollution[6,7] (SDG3 & 6), and biodiversity loss[8] (SDG15).

To supply adequate food, between 2000 and 2018, China's domestic crop production quantity increased by 44%. Cropland expansion (4.9 Mha)[9] contributed 7% of production increase, with the remaining 93% coming from intensification. Similarly, livestock production intensified, with increased reliance on concentrate feeds[2]. As a consequence, the use of nitrogen fertilizer in China today accounts for 32% of global fertilizer use, and China's agricultural production is responsible for 13% of global GHG emissions[2]. Air and water pollution have reached 4.2 and 2.7 folds of sustainability thresholds[10,11] defined by PM2.5 and nitrogen discharge. Additionally, irrigation water use represents 13% of global water withdrawals, and the efficiency (48%) has a significant room for improvement compared to 55-71% water use efficiency in Europe and North America[12,13].

Expanding imports are contributing to environmental pressure in exporting countries. Recent studies showed that displacement of resource use and environmental damage through international trade in the recent past represented a substantial share of the environmental impacts of domestic food consumption[14–17]. The contribution of China's historical food demand to the challenge of achieving sustainable development by China's trading partners has also been highlighted. For instance, 43% of deforestation emission due to soy plantation in Brazil can be allocated to China's soybean imports[18]. GHG emissions embodied in ruminant products exports to China accounted for 17% of total New Zealand livestock emissions in 2010[19].

While past environmental impacts of China's food demand are well understood, the projected future rising demand and the increasing reliance on trade[20] call for forward-looking studies to inform sustainable development policies in China and its trading partners. Existing studies focusing on China have looked either exclusively at the agricultural market variables[21], considered only domestic environmental impacts[22], or covered just a limited number of environmental dimensions when dealing with China's demand impacts on the rest of the world[19,23]. None of the forward-looking studies considered the ongoing socio-economic developments in the exporting regions, and thus only partial assessment of the sustainability challenge outside of China has been achieved.

Here we provide a comprehensive assessment of the global environmental impacts of China's future demand for food by 2030, the milestone in the UN 2030 Agenda, and up to 2050. The environmental impacts are assessed domestically, and in terms of virtual environmental trade flows with China's economic partners, looking at: the use of agricultural land (harvested crop area and pasture); GHG emissions from agriculture, forestry, and other land uses (AFOLU); the use of synthetic nitrogen fertilizer; and irrigation water use. The future scenarios rely on the Shared Socio-economic Pathways (SSP) framework[24,25]. This study focuses on the business-as-usual (BAU) scenario, representing a continuation of current socio-economic and technological trends, which corresponds to SSP2. However, to cover the range of uncertainty in future socioeconomic developments, we also consider two additional scenarios representing alternative futures to BAU: a Restricted development (RD) scenario characterized by limited technological progress and growing trade barriers; and a High development (HD) scenario with fast economic growth, flourishing consumption, and integrated international trade (see Methods for details). We quantify the impact of the scenarios using the Global Biosphere Management Model (GLOBIOM, see www.globiom.org), an agricultural and forest sector model which has been over the last decade extensively used for environmental sustainability analysis of the land based sectors[26–30], and has been enhanced and validated for this study with an upgraded representation of China's agricultural sector and land use dynamics, see Methods and Supplementary Notes 1-2 for details. The work has been conducted as part of the Food, Agriculture, Biodiversity, Land, and Energy (FABLE) Consortium of country teams that develop integrated pathways towards sustainable land-use and food systems[31].

Results

China's demand for agricultural products is projected to rely increasingly on imports

China's total demand for agricultural products, including food, feed, biofuel or other use, is projected to increase substantially by mid-century (Figure 1a). This is reflected in a 25% increase in per-capita calorie demand in 2050 relative to 2010 in the BAU scenario (Supplementary Fig. 1). Per-capita demand of animal sourced calories is projected to increase almost twice as fast as for total calories, by +43% compared to 2010. Total demand for ruminant meat and dairy products is projected to almost double, reaching respectively 19 and 71 Mt in 2050. Pig and poultry products drive livestock demand increases, although their demand is projected to level off after 2040 because of a progressively saturated per-capita demand and a projected decrease in the population. Nevertheless, they remain 30 Mt (31%) higher in 2050 compared to 2010. The 59% increase in the demand for crop products is projected to be driven mainly by the additional feed requirements. In particular, the demand for oil crops is projected to expand twofold compared to 2010 and reach 200 Mt in 2050. The demand for cereals is projected to increase from 416 Mt in 2010 to 575 Mt in 2050, and 66% of this increase is due to the additional feed demand. In terms of other crops, the increase in demand is comparatively slow, only 12% higher than the 2010 level.

We project that the increasing demand would largely be satisfied by increasing domestic production (+37% for cereals, +33% for pig and poultry products, +71% for ruminant meat, and +42% for dairy products, see Figure 1b). However, the reliance on imports is also projected to increase. The share of imports in total demand is projected to increase from 5% to 12% for ruminant meat, from 12% to 18% for dairy products, and from 62% to 75% for oil crops (mostly soybean), between 2010 and 2050. Pig and poultry products rely little on imports, but significant imports of oil crops are required for feed. These projections are in line with the observed trends (Supplementary Fig. 2-6 in SI).

The patterns of bilateral trade are projected to change in the future only slowly. As shown in Figure 1c, China's imports of soybean products account for 38% of the global soybean trade with 42 Mt total imports in 2010, and major trade partners are Brazil and USA which each export similar amounts of soybean to China. In 2050, China is projected to account for around 50% of global soybean trade, and the import quantity is projected to reach 124 Mt, but the bilateral trade pattern would remain similar to that in 2010. Imports of dairy products originate mainly from New Zealand (45%) and the European Union (22%) in 2010. In 2050, China is projected to import additional 6.6 Mt of dairy products, but its share in global trade would remain stable at 14%. Compared to 2010, the share of New Zealand in China's dairy imports reaches 62%.

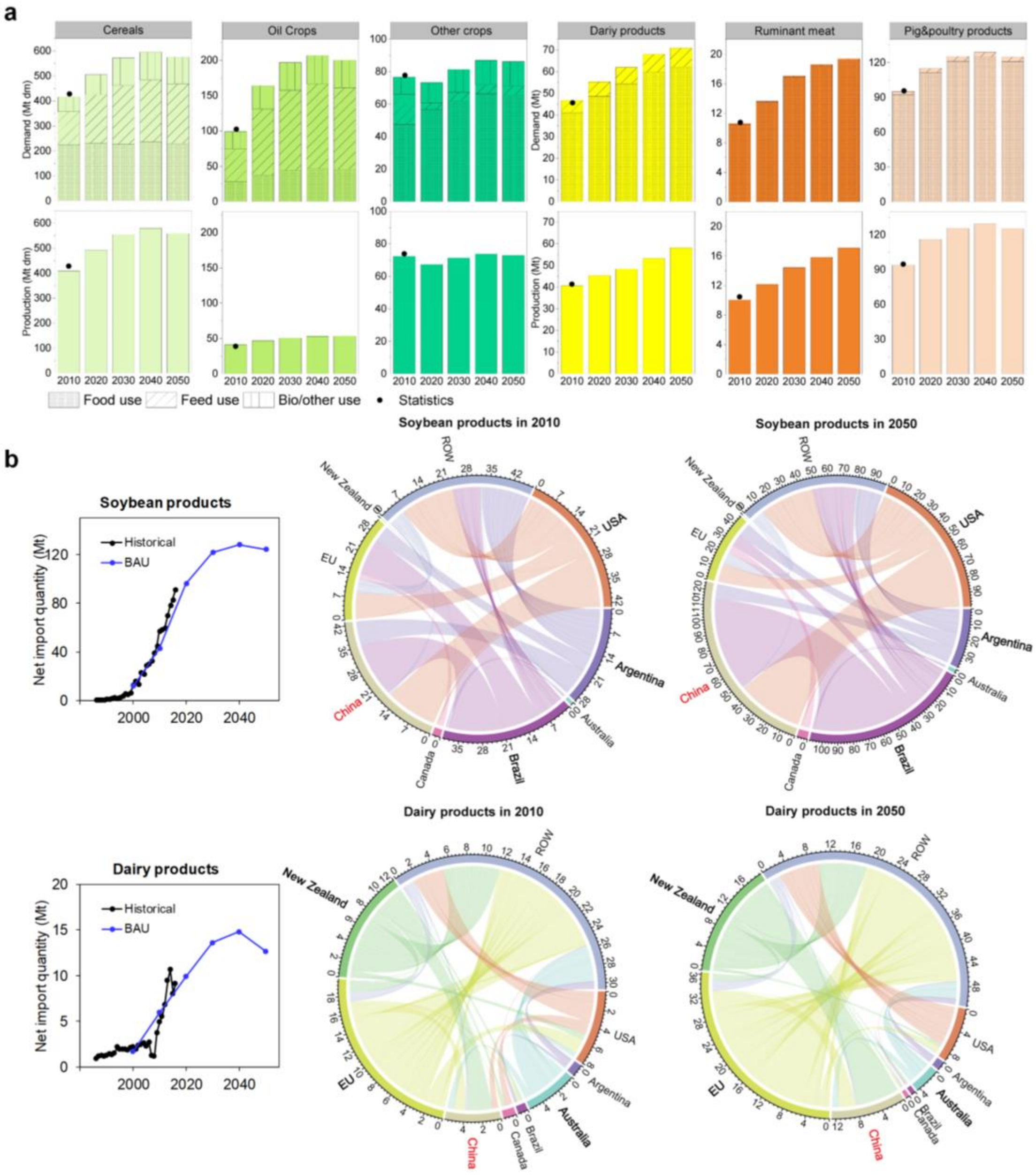

Figure 1. Trends in the demand, production, and trade of agricultural products in China under the BAU scenario. a, The first row is demand for food, feed, and biofuel/other use are differentiated in shapes; the second row represents domestic production of agricultural products. The dots in the subplots show historical data averaged for the period 2009–2011. For detailed results for individual

product categories and for all the scenarios see Supplementary Table 1. b, The plots on the left show the trends of net import quantity for dairy and soybean products (See Supplementary Fig. 7,8 for more commodities). The circular plots in the centre and on the right side represent the bilateral trade between China and its major partners in 2010 and 2050, respectively. Each arrow represents the amount of products coming from the exporting region (with the same color as the exporting region) to the importing region.

Environmental impacts of Chinese food demand inside and outside of China

In response to the projected increase in China's food demand, between 2010 and 2050, agricultural land use is projected to expand by 44 and 49 Mha inside and outside of China, respectively (Figure 2a). This result would mean a substantial rebalancing compared to 2030, where 38 Mha would be brought into production domestically, and almost twice as much additional agricultural land, 80 Mha, would be virtually imported. In 2050, agricultural imports are projected to represent 44 and 56 Mha of crop area and pasture, respectively (Supplementary Fig. 9a). For crop area, the increase of virtual land import between 2010 and 2050 is 19 Mha, which is more than twice the domestic area increase. The imported crop area increases are mainly due to soybean (68%), rapeseed (16%), and wheat (5%) import. For pasture, the increase of virtual land import between 2010 and 2050 is 29 Mha, which is lower than the domestic area increase (37 Mha).

In 2050, the increase in domestic GHG emissions from agricultural production (239 Mt CO_2eq yr^{-1}) , mostly from livestock sector, would be compensated by the carbon sink from China's ambitious afforestation programs (242 Mt CO_2eq yr^{-1}, see Supplementary Fig. 9b). Thus, net domestic GHG emissions from AFOLU sector by 2050 (820Mt CO_2eq yr^{-1}) would be similar to levels in 2010. We also estimate that China should be responsible for 115 Mt CO_2eq yr^{-1} of virtually imported GHG emissions in 2050. A total of 61% of these trade-embedded emissions would be due to the imports of livestock products. Imports of ruminant meat, dairy, and oil products would create 51, 16, and 14 Mt CO_2eq yr^{-1} of direct GHG emissions, respectively. Agricultural imports would also lead globally to large emissions from deforestation (208 Mt CO_2eq yr^{-1} in 2030, see Supplementary Fig. 10). As the demand for imports levels off after 2030, deforestation in exporting regions decreases, and the changes in deforestation emissions embodied in trade to China become negative by 2050.

Increased domestic production requires more inputs and resources: we project a 17% increase in nitrogen use and additional 43 km^3 of irrigation water use in the peak period (2030) in China (Figure 2c and 2d). While the virtually imported nitrogen and water from trade partners would be less than 9% of overall consumption (Supplementary Fig. 9c and 9d), but still higher than present. This is because China's major import crop, soybean, does not require much nitrogen fertilization or irrigation.

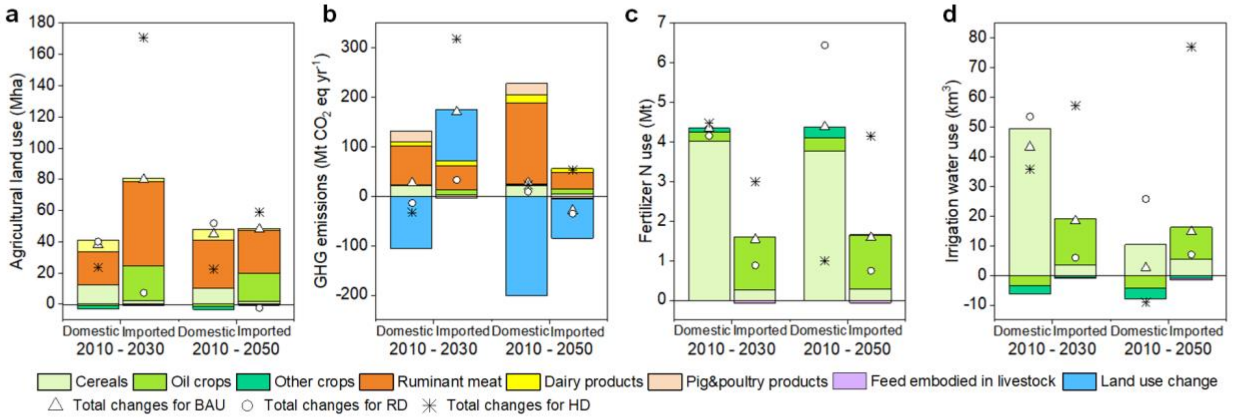

Figure 2. Difference in domestic and imported environmental impacts compared to 2010 in agricultural land use (a), GHG emissions (b), nitrogen use (c), and irrigation water use (d). The stacked bars represent the decomposed effects by product for the BAU scenario, and the markers represent the total effects for the three scenarios. Detailed environmental impacts of the two alternatives scenarios can be found in Supplementary Fig. 10-12. For imported land use change emissions, only deforestation emissions have been considered, for further details on the calculation of the virtual trade flows, see Methods.

Environmental sustainability challenges for China's main trade partners

Most of China's virtual crop-related trade impact (land area, nitrogen, and water use) occurs in a few countries with large agricultural sectors, mainly Brazil, Canada, and the United States. Oil crops are highly traded. For instance, China is projected to import 66 Mt of soybean from Brazil in 2050, which would account for 39% of Brazil's soybean production, occupy 15.7 Mha of crop area, and use 0.7 Mt nitrogen. Virtual water trade occurs mainly with the United States, where irrigation is widely used to produce cereals and oilseeds. Not only crop products, but also crops embodied as feed in livestock product exports to China, represent additional environmental pressure. In New Zealand, 15% of nitrogen use and irrigation water use can be attributed to feed use for livestock products exported to China.

The intensity of trade in terms of embodied pasture area depends on the prevalent livestock production system[32]. For example, in 2050 Australia is projected to export 0.3 Mt of bovine meat to China, which would occupy 14 Mha of pasture. In comparison, the United States would export a similar amount of ruminant meat to China (0.28 Mt) but at the expense of 2.1 Mha of pasture. Because the intensive grain-based beef systems are dominant in the United States and pasture productivity there is higher than in Australia. With respect to the imports of total virtual GHG emissions import, Brazil and Australia carry the main burden, with 27.8 and 23.3 Mt CO_2eq yr^{-1}, respectively. Beef export accounts for 73% of virtual trade in GHG emissions from Brazil to China. And for Australia, 6.9 Mt CO_2eq yr^{-1} from deforestation emissions and 14.8 Mt CO_2eq yr^{-1} from ruminant production can be allocated to exports to China. Although the virtual trade in GHG emissions is highest in Brazil, it represents only 5.2% of the Brazil's total AFOLU emissions. In the case of New Zealand, GHG emissions embodied in exports to China (18.4 Mt CO_2eq yr^{-1}, all due to ruminant products) would account for 28% of the country's total AFOLU emissions in 2050.

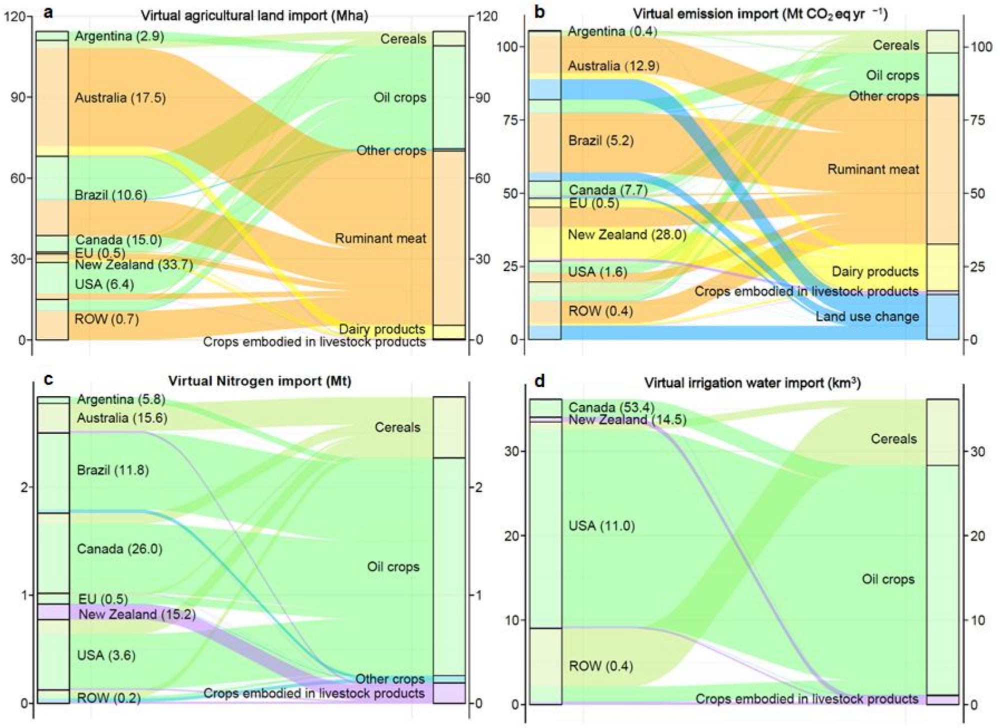

Figure 3. Virtual trade flows of environmental impacts due to China's agricultural imports in terms of the agricultural land use (a), GHG emissions (b), nitrogen use (c), and irrigation water use (d) for the major trading partners and the rest of the world (ROW). The impacts are for 2050 under the business-as-usual scenario. The environmental impacts on the exporting regions are shown on the left and the source of environmental impacts by commodity is on the right. The numbers in

brackets represent the impacts due to exports to China as a share of the total environmental impacts of domestic production in the exporting regions. For example, virtual agricultural area imports by China from Argentina account for 2.9% of Argentina's total agricultural area use.

Alternative futures

The alternative socioeconomic scenarios presented here, RD and HD, attempt to map the range of alternative plausible futures. Both of them still have large environmental impacts either inside or outside of China (Figure 4). The total environmental sustainability challenges would be higher in the High Development (HD) scenario because faster economic development would lead to more food demand in China, 3774 kcal/cap/day in 2050, even higher than USA today. The sole exception is that the HD scenario projects the lowest overall nitrogen use (31.8 Mt) in 2050, because of faster technological development and higher imports of products from nitrogen use efficient regions (Supplementary Fig. 11). Domestic impacts are less sensitive to the different scenario assumptions than the trade mediated impacts (Figure 2 and Supplementary Fig. 12). The domestic use of nitrogen and water in RD scenario is respectively 7% and 6% higher than in the BAU scenario in 2050, due to the assumed slower technological progress. Flexibility of trade is the key contributor to the differences in virtual trade flows shown by the alternative scenarios, Figure 2. The HD scenario, projects larger imports and thus larger trade impacts outside of China than the other scenarios, for example, virtual land import in 2030 reaches 237 Mha, which is twice as much as in BAU. In the RD scenario, the limited international trade would lead to increased domestic production, and the most pronounced impacts on environmental sustainability in China across the considered scenarios. These results illustrate that the distribution of the environmental impacts between China and the rest of the world will substantially depend on socio-economic developments, and in particular, on trade policies and the level of integration of China in global agricultural markets.

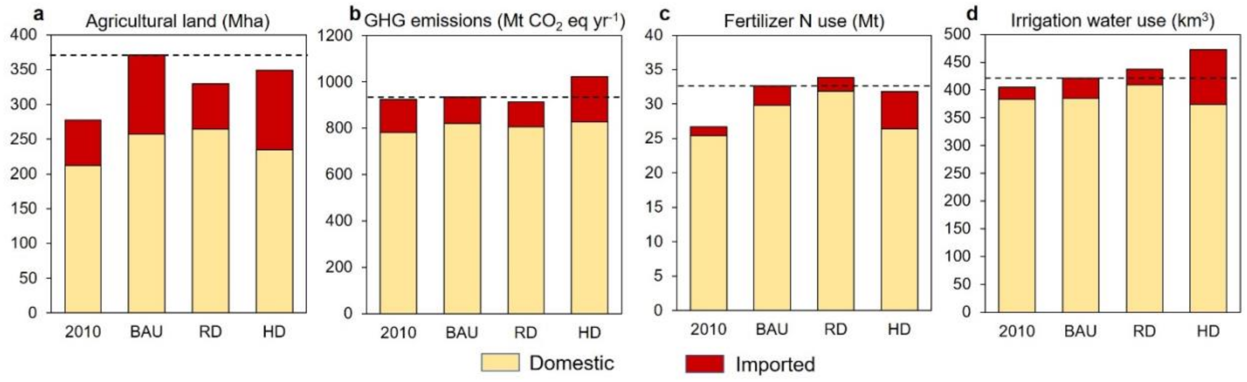

Figure 4. Comparison of China's overall environmental impacts in 2010 and 2050 for all scenarios in terms of agricultural land use (a), GHG emissions (b), nitrogen use (c), and irrigation water use (d). BAU is the baseline scenario, RD and HD are alternative scenarios.

Discussion

Our study based on a well-established global model with thorough validation for China and its bilateral trade flows provides a medium to long-term perspective on the potential global environmental impacts of the China's increasing food demand. While we took a global long-term perspective, and thus had to make abstraction from intra-China regional specificities as well as very specific bilateral trade policy scenarios, or even shocks, this study allowed to derive robust conclusions in terms of the growing role of China in the global sustainability. The results have far reaching implications for China's policies related to food demand, production systems and environmental and resources management, as well as international trade.

There is potential to reduce meat consumption. Following continuous economic growth in China, per-capita calorie demand is projected to increase from 2891 kcal in 2010 to 3603 kcal/day in 2050, and livestock products share increase from 19% to 22%. These levels of demand would be similar to the demand of the United States or European countries today (around 3500-3700 kcal/capita/day[2]). This projected livestock product demand increase compares well with projections in other studies (Supplementary Table 2). The large projected increase in ruminant meat consumption (94% in BAU) would lead to a large increase in pasture area (72 Mha). In addition, the consumption of feed crops would increase by 82% by 2050. Therefore, shifting to a diet that involves lower levels of meat consumption, as it starts to occur in several developed countries, would be an efficient way to reduce the negative impacts on environmental sustainability[33,34]. A shift to less meat intensive but more diversified diet, with higher share of fruits and vegetables, would bring also health[35]. However, changing diets may be a challenge for emerging markets, especially for consumers in China, as currently there is a lack of awareness of the link between meat consumption, health and environmental sustainability[36]. So, the government will need to primarily promote consumers' awareness of sustainable diets and provide incentives to farmers to diversify their production towards highly nutritious products, meanwhile, malnourishment needs to be taken account. China has recently reiterated, through the voice of its president Xi, its commitment to drastically reduce food waste, which could bring environmental benefits from consumer side.

Sustainable livestock production is imperative. Integrated, long-term, and large-scale investments have been made in sustainability programs in China, which have had a considerable positive impact on the promotion of cropland quality, grassland ecological protection and biodiversity conservation[37–39]. However, the livestock issue is still unresolved, and it might require stronger policy interventions. In 2050, 43% of total cropland in China is projected to produce feed concentrates for highly productive livestock systems (Supplementary Fig. 13). In addition to the local feed produced in China, domestic livestock production also relies heavily on imported feed crops contributing to environmental degradation and GHG emissions. For instance, the large amount of imported feeds results in additional manure that could become a source of pollutants because of the disconnection between animal and crop production[40]. Besides reconnecting livestock production with land, increasing ruminant productivity is a promising way of reducing environmental pressure, since China still has large yield gaps compared to developed countries (Supplementary Fig. 14). Our projected production allocation within China is following the current patterns and thus does not have substantial impact on the average country-level environmental outcomes. But in reality, because of the heterogeneity of China, spatial allocation will have a substantial effect which can lead to divergent environmental impacts[41,42]. Careful spatial planning is therefore necessary to exploit the environmental efficiency potentials to facilitate sustainable development.

Sourcing agricultural imports sustainably. Environmental impacts of China's imports largely depend on the country of origin. For instance, milk related GHG emissions intensity in the European Union is 0.9 kg CO_2eq per kg of product, whereas in New Zealand it is 1.4 kg CO_2eq per kg (Supplementary Table 3), as shown also by other studies[43]. However, milk imported from the European Union only accounts for 24% of China's total milk imports while New Zealand represent 62% of these imports. Also the past ban on soybean imports from the US raised concerns about potential substitution with imports from Brazil and the related impacts on deforestation in the Amazon[44]. The environmental considerations need to be taken into account next to economic efficiency and political sensitivities when designing China's trade policies to avoid unintended environmental consequences.

It is also recognized that even within an exporting country, supply chains may widely differ in their environmental impacts[45]. The environmental performance of specific supply chains is promoted, among others, by certification schemes such as "Zero Deforestation" beef[46] or "Fairtrade" labelling[47]. However, the effectiveness of these measures is limited if non-certified production still finds abundant markets. China, as one of the biggest importers, can play a key role in adoption of environmentally friendly production systems in exporting countries by promoting imports of products from certified supply chains and, in general, by enforcing respect of ambitious environmental standards by its trading partners.

Our results show that satisfying China's food demand while achieving environmental sustainability domestically and in exporting regions is likely one of the biggest challenges of the coming decades. Carefully designed policies across the whole of China's food system, including consumers, producers, and international trade, are necessary to ensure that future demand can be satisfied without destroying the environment.

Methods

Modeling approach. The quantitative analysis presented in our study relied on the Global Biosphere Management Model (GLOBIOM), a bottom-up partial equilibrium economic model designed to represent the key land use sectors, including crops, livestock, forestry, and bioenergy. GLOBIOM is extensively used for assessment of environmental impacts related to agriculture, such as sustainable water use[28], GHG emissions[30], land use change and related biodiversity impacts[48]. The model is particularly suitable for forward-looking assessment of environmental impacts embodied in trade because of its bilateral trade representation[29]. Finally, the model is flexible enough to allow for a detailed representation of a region of interest, in this case China, while still keeping it embodied in the global modeling framework[49].

The spatial resolution of the supply side relies on simulation units, which are aggregated from 5 to 30 arcmin pixels belonging to the same altitude, slope, and soil class and the same country. For the purpose of this study, they were further aggregated to 2 degrees. Commodity markets and international trade are represented for 37 economic regions in this study. Endogenous adjustments in market prices lead to balance between supply, demand and trade for each product and region. The market equilibrium is found through maximization of the sum of consumer and producer surpluses under constraints, such as land and water use balances. The model is solved with recursive dynamics in 10-year time steps. Main exogenous drivers of forward-looking scenarios in GLOBIOM are population and economic growth, technological change, dietary preferences, and bioenergy demand. Main endogenous variables are market variables, incl. demand, supply, trade, and prices, and environmental variables. such as land and water use, GHG emissions and sinks, nutrient balances.

Data on agricultural regional market variables including demand and production are for the base year harmonized with FAOSTAT (http://www.fao.org/faostat/en/). The spatially explicit land use allocation is initialized for 2000 with GLC2000[50]. The spatially explicit productivity of crops, grasslands, forests, and short-rotation tree plantations is estimated together with related environmental parameters (GHG budgets, nutrient and water balance) at the level of the simulation units. For crops, yields under different management systems are calculated with the biophysical Environmental Policy Integrated Climate (EPIC) model[51,52]. For forest parameters, GLOBIOM relies on the outputs of a dynamic forest management model, the Global Forest Model (G4M)[53]. Grassland productivity is obtained by combining results from EPIC and CENTURY[26,54]. Livestock production systems are parameterized with the global database developed in Herrero et al[55]. A detailed overview of data sources for the environmental indicators used in this study is presented in Supplementary Notes 3.

GLOBIOM represents international trade through net bilateral trade flows between global regions, which allows to directly trace the environmental impacts of demand for imports on exporting regions. To simulate trade, GLOBIOM uses the Enke–Samuelson–Takayama–Judge spatial equilibrium approach, assuming homogeneous goods[56]. The assumption is that a country will only import if its domestic price is greater than the price in the exporting country plus the cost of trade. In equilibrium, the difference in price between the importer and exporter equals the cost of trade. An advantage of this trade specification is that even if there is no trade in the base year, new trade flows can appear in response to future changes in price. Data on bilateral trade in the base year are from the BACI database[57], and data on tariffs between different countries and commodities are from the MAcMap-HS6 database[58]. Additional information about the model can be found in www.globiom.org.

GLOBIOM-China. For this study, we modified the core GLOBIOM model to improve representation of China. To better capture the recent and future trends in China agriculture, we particularly included mechanisms mimicking relevant policies in place. One of the key drivers of the land use in China are afforestation policies initiated in the 1990s. They already led to afforestation of 53 Mha. Considering Chinese consumers' preference for monogastric products and important structural changes in the sector, we calibrated the shift from smallholder to industrial systems for pig and poultry production. Fertilizer use efficiency development was calibrated to represent the "zero chemical fertilizer growth by 2020" policy. We also enforce the self-sufficiency in three major cereal crops of 95% under the baseline scenario in line with the current trade policies. Supplementary Notes 2 and Supplementary Table 4 present the model improvements in further detail.

Model calibration and validation. A careful model calibration was performed for the period 2000–2010. The year 2010 is the last historical point solved in GLOBIOM that we could directly calibrate, because the model runs in 10-year steps. FAOSTAT data and Chinese national statistical data until 2017/2018, as well as the OECD-FAO Agricultural Outlook projections[20] for China until 2027 (http://www.agri-outlook.org/data) were then used to validate the model behavior (Supplementary Fig. 2-6). The validation focused on the following variables: key variables - crop yield, crop area, per capita food demand, total demand, production, and trade.

Bilateral trade calibration is of vital importance for this study. In GLOBIOM, future trade flows are determined by commodity prices, trade costs. Trade costs include tariffs[58], transport costs[59], and a nonlinear trade expansion costs[60] that reflect persistency in trade patterns. Tariffs and transport costs are kept same as base year. The trade expansion costs are used in GLOBIOM to represent the capacity constraints slowing down expansion of trade flows in the short term. They can be regarded as investments necessary to expand trading infrastructure. GLOBIOM allows for appearance of new trade flows, which were not observed in the base year. Exponential function represents the trade cost (1) when trade flows are observed in the base year, for new trade flows a quadratic trade cost function (2) is used:

$$Trade\ cost_t = \frac{\varepsilon}{1+\varepsilon} \times \frac{Tariff + Transport\ cost}{Shipment_{t-1}{}^{1/\varepsilon}} \times Shipment_t^{\frac{1}{\varepsilon}+1} \tag{1}$$

$$Trade\ cost_t = Intercept \times Shipment_t + 0.5 \times slope \times Shipment_t^2 \tag{2}$$

Trade costs in period *t* are calculated with ε and $slope$ reflecting the elasticity of trade costs to traded quantity in the respective equations. The intercept is equal to either the tariff plus transport cost The bilateral trade flows between China and other countries for 2010 were calibrated to match the FAO trade matrix statistics[2] by manipulating the elasticities and intercepts in the trade cost equations. The bilateral trade validation of major commodities is shown in Supplementary Fig. 7. Calibration work also benefited from feedback by seven country teams of the FABLE Consortium.

Scenario design. The aim of our study is to provide medium to long-term ex-ante assessment of a global business-as-usual scenario aligned with current socio-economic trends. We complement this scenario with two variants with contrasted assumptions on future drivers to explore the range of results uncertainty. Development of such scenarios at the global level, with consistency across all sectors and regions, is a non-trivial task. Therefore, we decided to rely on the well-established framework of the Shared Socioeconomic Pathways (SSPs) which provide a set of narratives and quantified drivers designed to analyze global trajectories of future development[61,62]. These pathways represent the backbone of the climate related scenario analysis within IPCC[63] and have recently been used also for forward-looking biodiversity assessment in the context of IPBES[64]. We acknowledge that some outbreaks (like the US-China trade war in 2018, or COVID-19) may cause shocks and obstruct development of trade. However, in general these shocks are short-term disruptions[65], and our scenarios can cover these large uncertainties.

SSP2 is a business-as-usual scenario (BAU) that mostly continues recent trends in consumption and technological developments. In SSP3 (RD) scenario, the population in China increases faster, and growth in the GDP is slower, which leads to lower total food demand, as well as lower demand for livestock products compared to BAU. In this scenario, international trade becomes more restricted and fragmented, reflecting lower international cooperation. The SSP5 (HD) scenario is oriented toward high economic growth but limited resource efficiency, leading to inclusive development but at the expense of the environment. International trade expands rapidly in globalized markets in this scenario. All these scenarios make the assumption of a diverse development trajectory of different regions following their GDP and population projections (see https://tntcat.iiasa.ac.at/SspDb), which are primary drivers for diet shifts and productivity changes.

As the food demand patterns has been aggregated in country level, change in income per capita in the baseline drives changes in food diets, such as higher consumption of livestock products[66]. Food prices are also important drivers for food consumption patterns changes, and are determined by demand price elasticities of food products[67]. The crop yield trends are estimated based on estimation of correlation between yield and scenario-specific GDP growth assumed in the SSPs[68]. For livestock systems, technical change is applied through exogenous assumption on feed conversion efficiencies estimated based on historical trends for the SSP2 scenario and differentiated for the alternative scenarios based on the average projected crop yield growth[69,70]. Trade liberalization assumption is one of the key differences among scenarios. Elasticity of trade costs to traded quantity are varied across scenarios depending on the degree of trade liberalization or restrictiveness. More information on GLOBIOM trade specification can be found in Janssens et al.[29]. The values of key scenario drivers for China are provided in Supplementary Table 5.

Calculating virtual trade flows in environmental impacts. Virtual trade flows refer to resources or pollution embodied in international trade. We focus our analysis on seven major exporting regions to China: Argentina, Australia, Brazil, Canada, New Zealand, the United States, and the European Union, which account for more than 80% of the value of China agricultural imports (Supplementary Table 6). With respect to China trade flows, we also calculated the export effects (Supplementary Table 7), however, due to the imports are dominating overall trade pattern of China, we allocated the export impacts into domestic production side. To calculate trade impact, we assume that production for domestic consumption and export have the same domestic environmental impact. This is the assumption commonly used in many previous studies on virtual trade in water[71], land[72], and nitrogen[73]. The environmental intensity in a resource for a specific product P in exporting regions R and specific year T is defined as:

$$Virtual\_area_{R,P,T} = BilateralT_{R,P,T} \times Land\_intensity_{R,P,T} = BilateralT_{R,P,T} \times \frac{AREA_{R,P,T}}{PROD_{R,P,T}} \qquad (3)$$

$$Virtual\_N_{R,P,T} = BilateralT_{R,P,T} \times N\_intensity_{R,P,T} = BilateralT_{R,P,T} \times \frac{N_{input\,R,P,T}}{PROD_{R,P,T}} \qquad (4)$$

$$Virtual\_water_{R,P,T} = BilateralT_{R,P,T} \times Water\_intensity_{R,P,T} = BilateralT_{R,P,T} \times \frac{Water_{R,P,T}}{PROD_{R,P,T}} \qquad (5)$$

$$Virtual\_Agri\_GHG_{R,P,T} = BilateralT_{R,P,T} \times Agri\_GHG\_intensity_{R,P,T} = BilateralT_{R,P,T} \times \frac{Agri\_GHG_{R,P,T}}{PROD_{R,P,T}} \qquad (6)$$

Where $BilateralT_{R,P,T}$ is the bilateral trade quantity of product P exported to China from region R in year T. Bilateral trade volumes are here represented as net flows, as our framework rely on an homogenous good assumption[56] and hence a pair of trading partners will be always trading only in one direction at the same time (see Supplementary Notes 1). $PROD_{R,P,T}$ is in specific year T, total production of product P of exporting region R. $AREA_{R,P,T}$ is total harvested area of product P in exporting region R. The market variables, bilateral trade quantity, land area and production quantities has been estimated based on FAOSTAT data.

Virtual nitrogen (N) and water calculations follow the same logic - see Equation 4 and 5 - where $N_{input\,R,P,T}$ represents synthetic fertilizer use, and $Water_{R,P,T}$ represents irrigation water use for product P of exporting region R. For nitrogen and irrigation water, we used crop-specific resource intensity informed by EPIC model calculations.

Equation 6 was used to calculate virtual agricultural related GHG emissions. Fertilizer nitrous oxide (N_2O) emissions and methane (CH_4) from rice paddies were considered as direct crop related GHG emissions. N_2O was calculated based on N fertilizer consumption and IPCC emission coefficients[74] and rice CH_4 based on FAOSTAT average emission factors (http://www.fao.org/faostat/en/#data/GR). For livestock products, we used emissions intensity parameters for CH_4 and N_2O from enteric fermentation, manure management, manure dropped on pastures, rangelands and paddocks, and manure management from the global livestock production systems database[55]. An overview of the data sources used for these calculations can be found in Supplementary Notes 3.

To calculate emissions from deforestation, we rely on a top-down indirect allocation approach[75]. We first determined forest losses in exporting regions based on the G4M model calculations[53], and then assigned the deforestation attributable to cropland and pasture expansion based on Curtis et al.[76]. Then we allocated the cropland deforestation emissions to individual crops based on their contribution to the total cropland area expansion. The pasture related deforestation was distributed between ruminant products based on the pasture area necessary to cover the grass feed requirements of each livestock production system. Finally, we calculated the share of China's virtual land import within the total area of each agricultural product. The deforestation emissions related to crop or pasture are then calculated based on the following equations:

$$Virtual\_\ deforemission_{R,T} = Deforemis\_crop_{R,T} \times \frac{\Delta Crop\_area_{R,P,T}}{\sum_{P=1}^{P} \Delta Crop\_area_{R,P,T}} \times \frac{Virtual\_Crop\_area_{R,P,T}}{Crop\_area_{R,P,T}}$$

$$, \forall\ \Delta Crop\_area_{R,P,T} > 0 \tag{7}$$

$$Virtual\_deforemission_{R,T} = Deforemis\_live_{R,T} \times \frac{\Delta Pasture_{R,P,T}}{\sum_{P=1}^{P} \Delta Pasture_{R,P,T}} \times \frac{Virtual\ Pasture_{R,P,T}}{Pasture_{R,P,T}}$$

$$, \forall\ \Delta Pasture_{R,P,T} > 0 \tag{8}$$

where $Deforemis\_crop_{R,T}$ and $Deforemis\_live_{R,T}$ are deforestation emission caused by cropland and pasture expansion in region R and year T, respectively; only the expanded area is accounted for in $\Delta Crop\_area_{R,P,T}$; $\frac{Virtual\_Crop\_area_{R,P,T}}{Crop\_area_{R,P,T}}$ indicates the virtual crop area embodied in trade, which is presented in equation (3) and divided by $Crop\_area_{R,P,T}$, to calculate the share of virtual land import. Similarly, deforestation caused by virtual pasture trade can be derived from equation (8).

Environmental impacts due to feed production are included in the virtual trade flows related to livestock products. For this purpose, we used the regional livestock production specific feed requirements from Herrero et al[55]. Then we calculated the total use of feed for different livestock products and the related domestic environmental impacts and allocated them proportionally based on quantities of the bilateral trade to the environmental impacts imported by China. For feed crops embodied in the trade of livestock products, we took into account only locally produced feed. This may lead to minor underestimation of the overall impact of China's imports, but this should remain minor as many livestock products exporters to China are not major feed crop importers.

Data availability

The authors declare that the main data supporting the findings of this study will be made available through a public repository prior to publication. Other relevant data will be made available upon request.

Code availability

The authors declare that the code used to present the results in this study is available from the corresponding author upon request.

References

1. He, P., Baiocchi, G., Hubacek, K., Feng, K. & Yu, Y. The environmental impacts of rapidly changing diets and their nutritional quality in China. *Nat. Sustain.* **1**, 122–127 (2018).

2. FAO. FAOSTAT: Statistical database. *FAOSTAT: Statistical database.* http://www.fao.org/faostat/en/ (2019).

3. MOA. China’s import and export of agricultural products. *Ministry of Agriculture, China. China’s import and export of agricultural products [EB/OL]* http://www.moa.gov.cn/ztzl/nybrl/rlxx/201902/t20190201_6171079.htm (2019).

4. United Nations. *Transforming Our World: the 2030 Agenda for Sustainable Development United Nations United Nations Transforming Our World: the 2030 Agenda for Sustainable Development. A/RES/70/1. United Nations* (2015).

5. Lamb, A. *et al.* The potential for land sparing to offset greenhouse gas emissions from agriculture. *Nat. Clim. Chang.* **6**, 488–492 (2016).

6. Parris, K. Impact of agriculture on water pollution in OECD countries: Recent trends and future prospects. *Int. J. Water Resour. Dev.* **27**, 33–52 (2011).

7. Thompson, R. L. *et al.* Acceleration of global N2O emissions seen from two decades of atmospheric inversion. *Nat. Clim. Chang.* **9**, 993–998 (2019).

8. Kok, M. T. J. *et al.* Pathways for agriculture and forestry to contribute to terrestrial biodiversity conservation: A global scenario-study. *Biol. Conserv.* **221**, 137–150 (2018).

9. China Statistics Bureau. China statistical yearbook. *National Bureau of Statistics of China* http://www.stats.gov.cn/english/Statisticaldata/An.

10. Yu, C. *et al.* Managing nitrogen to restore water quality in China. *Nature* **567**, 516–520 (2019).

11. Zhang, Q. *et al.* Drivers of improved PM2.5 air quality in China from 2013 to 2017. *Proc. Natl. Acad. Sci. U. S. A.* **116**, 24463–24469 (2019).

12. FAO. AQUASTAT Main Database. *AQUASTAT Main Database* http://www.fao.org/nr/water/aquastat/data/query/index.html?lang=en (2016).

13. Rohwer, J., Gerten, D. & Lucht, W. Development of functional irrigation types for improved global crop modelling. *PIK Rep.* 1–61 (2007).

14. Caro, D., Lopresti, A., Davis, S. J., Bastianoni, S. & Caldeira, K. CH4 and N2O emissions embodied in international trade of meat. *Environ. Res. Lett.* **9**, (2014).

15. Pendrill, F. *et al.* Agricultural and forestry trade drives large share of tropical deforestation emissions. *Glob. Environ. Chang.* **56**, 1–10 (2019).

16. Chen, B. *et al.* Global land-water nexus: Agricultural land and freshwater use embodied in worldwide supply chains. *Sci. Total Environ.* **613**–**614**, 931–943 (2018).

17. Lathuillière, M. J., Johnson, M. S., Galford, G. L. & Couto, E. G. Environmental footprints show China and Europe's evolving resource appropriation for soybean production in Mato Grosso, Brazil. *Environ. Res. Lett.* **9**, 074001 (2014).

18. CDP Worldwide. *Decoupling china's soy imports from deforestation driven carbon emissions in brazil*. http://resources.trase.earth/documents/issuebriefs/CDP_China_soy_emissions_briefing_FINAL.pdf (2019).

19. Du, Y. *et al.* A global strategy to mitigate the environmental impact of China's ruminant consumption boom. *Nat. Commun.* **9**, 4133 (2018).

20. OECD/FAO. *OECD-FAO Agricultural Outlook 2019-2028*. *OECD Publishing* vol. 52 (OECD, 2019).

21. Sheng, Y. & Song, L. Agricultural production and food consumption in China: A long-term projection. *China Econ. Rev.* **53**, 15–29 (2019).

22. Ma, L. *et al.* Exploring Future Food Provision Scenarios for China. *Environ. Sci. Technol.* **53**, 1385–1393 (2019).

23. Yu, Y., Feng, K., Hubacek, K. & Sun, L. Global Implications of China's Future Food Consumption. *J. Ind. Ecol.* **20**, 593–602 (2016).

24. Fricko, O. *et al.* The marker quantification of the Shared Socioeconomic Pathway 2: A middle-of-the-road scenario for the 21st century. *Glob. Environ. Chang.* **42**, 251–267 (2017).

25. Kriegler, E. *et al.* A new scenario framework for climate change research: the concept of shared climate policy assumptions. *Clim. Change* **122**, 401–414 (2014).

26. Havlík, P. *et al.* Climate change mitigation through livestock system transitions. *Proc. Natl. Acad. Sci.* **111**, 3709–3714 (2014).

27. Hasegawa, T., Havlík, P., Frank, S., Palazzo, A. & Valin, H. Tackling food consumption inequality to fight hunger without pressuring the environment. *Nat. Sustain.* **2**, 826–833 (2019).

28. Pastor, A. V. *et al.* The global nexus of food–trade–water sustaining environmental flows by 2050. *Nat. Sustain.* (2019) doi:10.1038/s41893-019-0287-1.

29. Janssens, C. *et al.* Global hunger and climate change adaptation through international trade. *Nat. Clim. Chang.* **10**, 829–835 (2020).

30. Frank, S. *et al.* Structural change as a key component for agricultural non-CO2 mitigation efforts. *Nat. Commun.* **9**, 1060 (2018).

31. FABLE. *Pathways to Sustainable Land-Use and Food Systems*. www.foodandlandusecoalition.org/fableconsortium. (2019).

32. Robinson, T.P., Thornton P.K., Franceschini, G., Kruska, R.L., Chiozza, F., Notenbaert, A., Cecchi, G., H. & M., Epprecht, M., Fritz, S., You, L., Conchedda, G. & See, L. *Global livestock production systems*. (2011).

33. Godfray, H. C. J. *et al.* Meat consumption, health, and the environment. *Science (80-.).* **361**, eaam5324 (2018).

34. Gerten, D. *et al.* Feeding ten billion people is possible within four terrestrial planetary boundaries. *Nat. Sustain.* **3**, 200–208 (2020).

35. Willett, W. *et al.* Food in the Anthropocene: the EAT–Lancet Commission on healthy diets from sustainable food systems. *Lancet* **393**, 447–492 (2019).

36. Zhong, S. & Chen, J. How Environmental Beliefs Affect Consumer Willingness to Pay for the Greenness Premium of Low-Carbon Agricultural Products in China: Theoretical Model and Survey-based Evidence. *Sustainability* **11**, 592 (2019).

37. Bryan, B. A. *et al.* China's response to a national land-system sustainability emergency. *Nature* **559**, 193–204 (2018).

38. Shi, W., Tao, F. & Liu, J. Changes in quantity and quality of cropland and the implications for grain production in the Huang-Huai-Hai Plain of China. *Food Secur.* **5**, 69–82 (2013).

39. Lü, Y., Ma, Z., Zhang, L., Fu, B. & Gao, G. Redlines for the greening of China. *Environ. Sci. Policy* **33**, 346–353 (2013).

40. Sun, J. *et al.* Importing food damages domestic environment: Evidence from global soybean trade. *Proc. Natl. Acad. Sci.* **115**, 5415–5419 (2018).

41. Zhai, Y. *et al.* Energy and water footprints of cereal production in China. *Resour. Conserv. Recycl.* **164**, 105150 (2021).

42. Verburg, P. H. & Keulen, H. Van. Exploring changes in the spatial distribution of livestock in China. *Agric. Syst.* **62**, 51–67 (1999).

43. Opio, C., Gerber, P., Mottet, A., Falcucci, A., Tempio, G., MacLeod, M., Vellinga, T., Henderson, B. & Steinfeld, H. *Greenhouse gas emissions from ruminant supply chains*. http://www.fao.org/docrep/018/i3461e/i3461e00.htm (2013).

44. Fuchs, R. *et al.* Why the US–China trade war spells disaster for the Amazon. *Nature* **567**, 451–454 (2019).

45. Soterroni, A. C. *et al.* Expanding the Soy Moratorium to Brazil's Cerrado. *Sci. Adv.* **5**, eaav7336 (2019).

46. le Polain de Waroux, Y. *et al.* The Restructuring of South American Soy and Beef Production and Trade Under Changing Environmental Regulations. *World Dev.* **121**, 188–202 (2019).

47. Acquaye, A. A., Yamoah, F. A. & Feng, K. Int . J . Production Economics An integrated environmental and fairtrade labelling scheme for product supply chains. *Intern. J. Prod. Econ.* **164**, 472–483 (2015).

48. Leclère, D. *et al.* Bending the curve of terrestrial biodiversity needs an integrated strategy. *Nature* **585**, 551–556 (2020).

49. Soterroni, A. C. *et al.* Future environmental and agricultural impacts of Brazil's Forest Code. *Environ. Res. Lett.* **13**, 074021 (2018).

50. Bartholomé, E. & Belward, A. S. GLC2000: a new approach to global land cover mapping from Earth observation data. *Int. J. Remote Sens.* **26**, 1959–1977 (2005).

51. Balkovič, J. *et al.* Global wheat production potentials and management flexibility under the representative concentration pathways. *Glob. Planet. Change* **122**, 107–121 (2014).

52. J. R. Williams, C. A. Jones, J. R. Kiniry & D. A. Spanel. The EPIC Crop Growth Model. *Trans. ASAE* **32**, 0497–0511 (1989).

53. Kindermann, G., McCallum, I., Fritz, S. & Obersteiner, M. A global forest growing stock, biomass and carbon map based on FAO statistics. *Silva Fenn.* **42**, 387–396 (2008).

54. Parton, W. J. *et al.* Observations and modeling of biomass and soil organic matter dynamics for the grassland biome worldwide. *Global Biogeochem. Cycles* **7**, 785–809 (1993).

55. Herrero, M. *et al.* Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. *Proc. Natl. Acad. Sci.* **110**, 20888–20893 (2013).

56. Takayama, T. and Judge, G. G. *Spatial and temporal price allocation models*. (1971).

57. Gaulier, G. & Zignago, S. BACI: International Trade Database at the Product-Level (the 1994-2007 Version). *SSRN Electron. J.* (2010) doi:10.2139/ssrn.1994500.

58. Bouët, A., Decreux, Y., Fontagné, L., Jean, S. & Laborde, D. Assessing applied protection across the world. *Rev. Int. Econ.* **16**, 850–863 (2008).

59. Hummels, D. *Have international transportation costs declined? Purdue University* (2001) doi:10.1257/jep.21.3.131.

60. Baker, J. S. *et al.* Evaluating the effects of climate change on US agricultural systems: Sensitivity to regional impact and trade expansion scenarios. *Environ. Res. Lett.* **13**, (2018).

61. Riahi, K. *et al.* The Shared Socioeconomic Pathways and their energy, land use, and greenhouse gas emissions implications: An overview. *Glob. Environ. Chang.* **42**, 153–168 (2017).

62. O'Neill, B. C. *et al.* The roads ahead: Narratives for shared socioeconomic pathways describing world futures in the 21st century. *Glob. Environ. Chang.* **42**, 169–180 (2017).

63. IPCC. *Climate Change and Land: an IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems*. (2019).

64. Diaz, S. *et al. Summary for policymakers of the global assessment report on biodiversity and ecosystem services*. *Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services - Advance Unedited Version* (2019).

65. Elleby, C., Domínguez, I. P., Adenauer, M. & Genovese, G. Impacts of the COVID-19

Pandemic on the Global Agricultural Markets. *Environ. Resour. Econ.* **76**, 1067–1079 (2020).

66. Valin, H. *et al.* The future of food demand: understanding differences in global economic models. *Agric. Econ.* **45**, 51–67 (2014).

67. Muhammad, A., Seale, J. L., Meade, B. & Regmi, A. International Evidence on Food Consumption Patterns An Update Using 2005 International Comparison Program Data. (2011).

68. van Zeist, W. J. *et al.* Are scenario projections overly optimistic about future yield progress? *Glob. Environ. Chang.* **64**, 102120 (2020).

69. Valin, H. *et al.* Agricultural productivity and greenhouse gas emissions: Trade-offs or synergies between mitigation and food security? *Environ. Res. Lett.* **8**, (2013).

70. Herrero, M., Havlik, P., McIntire, J., Palazzo, A. & Valin, H. African Livestock Futures: Realizing the Potential of Livestock for Food Security, Poverty Reduction and the Environment in Sub-Saharan Africa. Office of the Special Representative of the UN Secretary General for Food Security and Nutrition and the United. 118 (2014).

71. Hoekstra, A. Y. & Hung, P. Q. Globalisation of water resources: International virtual water flows in relation to crop trade. *Glob. Environ. Chang.* **15**, 45–56 (2005).

72. Würtenberger, L., Koellner, T. & Binder, C. R. Virtual land use and agricultural trade: Estimating environmental and socio-economic impacts. *Ecol. Econ.* **57**, 679–697 (2006).

73. Huang, G. *et al.* The environmental and socioeconomic trade-offs of importing crops to meet domestic food demand in China. *Environ. Res. Lett.* **14**, (2019).

74. IPCC. *IPCC Guidelines for National Greenhouse Gas Inventories*. (2006).

75. Sandström, V. *et al.* The role of trade in the greenhouse gas footprints of EU diets. *Glob. Food Sec.* **19**, 48–55 (2018).

76. Curtis, P. G., Slay, C. M., Harris, N. L., Tyukavina, A. & Hansen, M. C. Classifying drivers of global forest loss. *Science (80-.).* **361**, 1108–1111 (2018).

Acknowledgements

This research has received funding from the Gordon and Betty Moore Foundation and Norway's International Climate and Forest Initiative. We also acknowledge the support from UN Sustainable Development Solutions Network (SDSN) – Aline Mosnier, Jordan Poncet, Guido Schmidt-Traub – who initiated this project in the context of FABLE, accompanied it throughout its duration, and provided many invaluable comments. This work is also supported by the National Natural Science Foundation of China, NSFC (31972517); the Youth Innovation Promotion Association, CAS (2019101); Key Laboratory of Agricultural Water Resources, CAS (ZD201802); the Outstanding Young Scientists Project of Natural Science Foundation of Hebei (C2019503054). Finally, we acknowledge the financial support China Scholarship Council (No. 201904910646) and IIASA’s Young Scientists Summer Program for bring collaboration opportunities.

Author contributions

H.Z., P.H. and L.M. designed the study. H.Z., P.H., J.C., V.D.M. and H.V. contributed the data analysis. H.Z., P.H. and J.C. wrote the manuscript with contributions from H.V. and C.J. All authors contributed to the interpretation of the results and commented on the manuscript.

Competing interests

The authors declare no competing interests.