China's future food demand and its implications for trade and environment

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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 the environmental impacts of China's growing demand on the country itself and on its trading partners. We find that the increasing food demand, especially for livestock products (+16%~+30% across all scenarios), would domestically require 3~12 Mha of additional pasture between 2020 and 2050, resulting -2%~+16% growth in agricultural greenhouse gas (GHG) emissions. The projected 15%~24% reliance on agricultural imports in 2050 would result in 90~175 Mha of agricultural land area and 88~226 Mt CO_2eq $yr^{-1}$of GHG emissions virtually imported to China, which account for 26%~46% and 13%~32% of China's global environmental impacts, respectively. The distribution of the environmental impacts between China and the rest of world would substantially depend on development of trade openness. Thus, to limit the negative environmental impacts of its growing food consumption, besides domestic policies, China needs to also take responsibility in the development of sustainable international trade.

Introduction

China has undergone remarkable social and economic development over the past two decades to become the world's second largest economy. Over the same period, this successful development has led to a large increase in demand for food, especially for livestock products[1,2]. The import value of agricultural products has increased by 78% in constant USD[3] while domestic agricultural value increased by 36% from 2010 to 2018. For soybean products in particular, the reliance on imports increased from 46% to 83%; for ruminant meat from 2% to 17%, and for dairy products from 11% to 24%[2]. 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 (SDG13), air and water pollution (SDG3 & 6), and biodiversity loss (SDG15).

China's domestic crop production increased by 44% between 2000 and 2018. Cropland expansion (4.9 Mha)[5] contributed 7% of production increase, with the remaining 93% came from intensification. As a result, the use of nitrogen fertilizer in China today accounts for 32% of global fertilizer use. Similarly, livestock production also intensified with increased reliance on concentrate feeds[2]. China's agricultural production is now responsible for 13% of global GHG emissions[2]. Air and water pollution have reached 4.2 and 2.7 folds of sustainability thresholds[6,7] defined by PM2.5 and nitrogen discharge, largely due to the agriculture intensification. In addition, irrigation water use in China represents 13% of global water withdrawals, and the efficiency (48%) has a significant room for improvement compared to the levels in Europe and in North America (i.e., 55-71%)[8,9].

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 production[10–12]. The contribution of China's food demand to the challenge of achieving sustainable development of China's trading partners has also been highlighted. For instance, 43% of deforestation emissions due to soybean cultivation in Brazil can be attributed to China's soybean imports in 2017[13]. Also GHG emissions embodied in ruminant products exports to China accounted for 17% of total New Zealand livestock emissions in 2010[14].

China's food demand is projected to keep increasing in the coming decades with further increase in the reliance on food and feed imports[15]. It is therefore necessary to assess the impacts of such growing demand on China's domestic environment as well as the environment of its trading partners to inform sustainable development policies. However, current forward-looking assessments (see Supplementary Methods 1) either focused on local impacts only without considering global market spillovers[14,16,17], covered only a part of the agricultural sector (e.g., bioenergy demand and afforestation[18,19]), or assessed only one or two environmental dimensions[20–22]. Assessments of future trade patterns mostly present trade with a world pool market[23,24], making it hard to track global environmental impacts. An integrated assessment simultaneously analyzing global agricultural markets and China's bilateral trade, land-use competition and associated environmental impacts in detail and presenting for China separately from other regions is still lacking.

Here we provide a comprehensive assessment of the global environmental impacts of China's future food demand 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 four environmental impacts: the use of agricultural land (crop harvested area and pasture); GHG emissions from agriculture, forestry, and other land uses (AFOLU); the use of synthetic nitrogen fertilizer; and irrigation water use. We quantify these environmental impacts using the Global Biosphere Management Model (GLOBIOM, see www.globiom.org), an agricultural and forest sector model which has been extensively used for environmental sustainability analysis of the land based sectors over the last decade[25–29]. For this study, the representation of China's agricultural sector and environmental dynamics was enhanced in the model (see Methods and Supplementary Methods 2 for details). The future development assumed in the projections follows the Shared Socio-economic Pathways (SSP)[30], middle-of-the-road scenario, representing a continuation of current socio-economic and technological trends (the business-as-usual (BAU) scenario). To cover the range of uncertainty in future developments, we also considered two additional socioeconomic scenarios – a Restricted development (RD) scenario and a High development (HD) scenario – and provided a comprehensive sensitivity analysis with respect to the role of individual scenario driver (see Methods for details). This work was 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

In this section, we first consider the respective contribution of domestic production and international trade to satisfying China's future food demand, then we explore the implications for domestic environment, and implications for the environment by major trading partners are assessed afterwards. This section concludes with a thorough analysis of the main drivers of the forward-looking scenarios and their sensitivity analysis.

China's food demand increasingly relies on imports

China's total demand for agricultural products, including food, feed, biofuel or other use, is projected to increase substantially by mid-century (Fig. 1a). This is reflected in a 13% increase in per-capita calorie demand in 2050 BAU scenario relative to 2010 and a 6% increase relative to 2020 (Supplementary Figure 1). Per-capita demand for animal sourced calories is projected to increase three times as fast, by +45% compared to 2010 and +23% compared to 2020. Total demand for ruminant meat and dairy products is projected to almost double, reaching respectively 19 and 68 Mt in 2050. Pig and poultry products drive livestock demand increases, although the increase is projected to level off after 2040, because of a progressively saturated per-capita demand and a projected decrease in population. Nevertheless, they remain 30 Mt higher in 2050 compared to 2010. The increase in the demand for crop products (34%) 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, however, the demand from 2010 to 2020 constituted the major portion (+57%) of the increase. The demand for cereals is projected to increase from 420 Mt in 2010 to 530 Mt in 2050, mainly driven by the increase of cereal feed demand (84%). In terms of other crops, the increase in demand is comparatively slow, only 9% higher than the 2010 level.

We project that the increasing demand would largely be satisfied by increasing domestic production (+25% for cereals, +33% for pig and poultry products, +62% for ruminant meat, and +38% for dairy products, see second row of Fig. 1a). However, the reliance on imports is also projected to increase. The share of imports in total demand is projected to increase from 7% to 20% for ruminant meat, from 12% to 20% for dairy products, and from 54% to 70% 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. Currently, the pig farming industry in China is influenced by African Swine Fever, causing a 22% deviation from the statistics in 2020. We find that these temporary fluctuations will not have substantial impact on long term projections (see quantitative validation in Supplementary Methods 3 and Supplementary Figure 2-6).

The patterns of bilateral trade are projected to change in the future. As shown in Fig. 1b, China's imports of soybean products account for 35% of the global soybean trade with 45 Mt total imports in 2010, and major trade partners are Brazil and USA which each export similar amounts of soybean to China (18 Mt). In 2050, China is projected to account for 46% of global soybean trade, and the import quantity is projected to reach 126 Mt. But the bilateral trade pattern (53% import from Brazil and 37% import from USA) would differ from that in 2010, which is in line with current status. Imports of dairy products originate mainly from New Zealand (2.7 Mt or 40% of total import) and the European Union (1.0 Mt or 20% of total import) in 2010. By 2050, China is projected to import an additional 8.0 Mt of dairy products, and its share in global trade would increase from 13% to 20%. New Zealand remains the major dairy exporter accounting for 71% of China's dairy imports in 2050.

Environmental impacts of Chinese food demand

In response to the projected increase in China's food demand between 2010 and 2050, the domestic and imported agricultural land is projected to expand by 25 and 63 Mha, respectively (Fig. 2a). Compared with our projections for 2020, the projected increase of imported agricultural land area (21 Mha) would also significantly higher than that brought into production domestically (6 Mha) until 2050. In 2050, agricultural imports are projected to represent 41 and 77 Mha of crop harvested area and pasture, respectively (Supplementary Figure 9a). The increase in virtual crop harvested area imports between 2010 and 2050 is 15 Mha, while the domestic crop harvested area remains at the same level. The increase in imported crop harvested area is mainly due to soybean (77%), rapeseed (7.9%) and wheat (3.9%). For pasture, the increase in virtually imported land is 49 Mha between 2010 and 2050, which is twice the domestic increase (26 Mha).

In 2050, the increase in domestic GHG emissions from agricultural production (104 Mt CO_2eq yr^{-1}, Fig. 2b), mostly from livestock sector, would be fully compensated by the carbon sink from China's ambitious afforestation programs (205 Mt CO_2eq yr^{-1}, see Supplementary Figure 10 for detailed information on land transition patterns). This means that net domestic GHG emissions from the AFOLU sector in 2050 (628 Mt CO_2eq yr^{-1}) would be lower than the levels in 2010 (809 Mt CO_2eq yr^{-1}). We also estimate that China will be responsible for 123 Mt CO_2eq yr^{-1} of virtually imported GHG emissions in 2050. A total of 86% of these trade-embedded emissions would be due to the imports of livestock products. Imports of ruminant meat, dairy, and oil products would create 85, 18, and 12 Mt CO_2eq yr^{-1} of direct GHG emissions, respectively. Agricultural imports would also lead to large emissions from deforestation globally (23 Mt CO_2eq yr^{-1} in 2050, see Supplementary Figure 11). 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. It is worth noting that GHG emissions related to China's afforestation programs are included in total AFOLU sector emissions for completeness. However, for consistency, they should not be included when comparing with imported effects for consistency, because for imported land-use change emissions, only deforestation emissions were considered.

Increased domestic production requires more inputs and resources: we project a 17% increase in nitrogen fertilizer use and additional 25 km^3 of irrigation water use in the peak period (2030) in China (Fig. 2c and 2d). Because China's major import crop, soybean, does not require much nitrogen and irrigation, the virtually imported fertilizer N and water from trade partners would be less than 9.0% of overall consumption (Supplementary Figure 9c and 9d), but still higher than the present level.

Environmental challenges for China's main trade partners

Most of China's virtual crop-related trade impact (crop harvested area, nitrogen fertilizer, and water use) occurs in a few countries with large agricultural sectors, mainly Brazil, the United States and Canada (Fig. 3). Oil crops are highly traded. For instance, China is projected to import 66 Mt of soybean from Brazil in 2050, which would account for 40% of Brazil's soybean production, occupying 16 Mha of crop area, and using 0.7 Mt nitrogen fertilizer. Virtual water trade occurs mainly with the USA, 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, Australia is projected to export 0.3 Mt of bovine meat to China, which would occupy 14 Mha of pasture in 2050. In comparison, the USA export even higher amount of bovine meat to China (0.5 Mt) but at the expense of 4.0 Mha of pasture in 2050. Because the intensive grain-based ruminant systems are dominant in the USA and pasture productivity there is higher than in Australia. With respect to the imports of total virtual GHG emissions, Brazil, New Zealand, and Australia carry the main burden, with 30, 21 and 20 Mt CO_2eq yr^{-1}, respectively. Bovine meat export accounts for 77% of virtual trade in GHG emissions from Brazil to China. For Australia, 5.7 Mt CO_2eq yr^{-1} from deforestation emissions and 10 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 8% of the Brazil's total AFOLU emissions. In the case of New Zealand, GHG emissions embodied in exports to China, all due to ruminant products, would account for 33% of the country's total AFOLU emissions in 2050.

Alternative futures

Two alternative socioeconomic scenarios, RD (Restricted development) and HD (High development), and their decomposition by individual driver (e.g., population, GDP, diet, productivity, trade), provide insights into the robustness of the BAU results in the context of a wide range of alternative plausible futures and to explain the role of each driver. Domestic impacts are less sensitive to the different scenario assumptions than the trade mediated impacts (Supplementary Figure 13). The imported impacts in both scenarios differ considerably in comparison with that of the BAU in terms of agricultural land and GHG emissions (Fig. 4a), but they represent still substantial impacts on the rest of world. In the RD scenario, the share of imported land and GHG emissions in China's global environmental impacts reach 26% and 15%, respectively, and in the HD scenario those numbers could reach 46% and 31%, respectively. With respect to nitrogen fertilizer and water use, the imported impacts account for less than 10% of global impacts, except for the HD scenario (around 15% of imported share).

Openness of trade is the key determinant to the differences in virtual trade flows, in particular for agricultural land (Fig. 4b). The total China related agricultural land area is +32% higher in the HD TRADE scenario and -20% lower in the RD TRADE scenario compared with the BAU projections. This difference is mostly due to the imported impacts, for example, virtual agricultural land area import in 2050 HD TRADE scenario reaches 288 Mha, which is more than twice the BAU value (132 Mha), whereas restricted trade increases the domestic environmental challenges in China (Supplementary Figure 14). HD TRADE assumption would lead to a decrease in GHG emissions mainly because of increasing imports from low GHG intensity regions compared to China (e.g., EU and USA, see Supplementary Table 2). Environmental impacts are also sensitive to changes in GDP and population growth that varies food consumption. For GDP growth, RD and HD scenarios differ with the BAU projection on GHG emissions by -5% and +11%, respectively. Population change has an opposite effect, resulting in a difference in emissions with the BAU (+4% and -3%). Shifting diets to more livestock consumption (HD DIET) leads to +7% more agricultural land and GHG emissions and +3% more of fertilizer N use. An increase in food waste would also increase 3% fertilizer N and water use as shown in the RD DIET scenario. The impact of changes in productivity (YILD and FEEF) are less pronounced.

As the assessment has been conducted in a global context, understanding the effects from variations in the socio-economic development trajectory in China in comparison to the ROW is also important (See Methods). We find that the assumptions on drivers for China dominate the environmental effects. Thus, changes in the driver assumptions for China only (Supplementary

Figure 15a) result in similar environmental effects in comparison to applying them globally (Fig. 4b). Driver changes in the ROW only (i.e., keep the drives for China same as the BAU) have much less influence on China's global environmental impacts (±3.2% in Supplementary Figure 15b).

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 China's increasing food demand. 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. China's per-capita calorie consumption is projected to increase from 2974 kcal in 2010 to 3376 kcal/day in 2050, where livestock products share increases from 19% to 22%. The projected increase in demand compares well with projections in other studies (Supplementary Table 3). The increasing consumption of ruminant products would require 224 Mha pasture area (59% domestically) and 514 Mt CO_2eq yr^{-1} GHG emission (80% domestically) in 2050. A 10% increase in livestock consumption would result in 7% more land and GHG impacts (Supplementary Figure 16). Therefore, a shift to less meat intensive but more diversified diet with healthy food and a low environmental footprint, such as insects, seaweed and plant based protein substitutes, would bring essential nutrients and reduce the costs for environment[33–35]. Meanwhile, malnourishment needs to be taken into account. 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]. China has recently reiterated, through the voice of its president Xi, its commitment to drastically reduce food waste, which would bring environmental benefits from the 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]. However, the livestock production with high environmental intensities dominates future sustainability outcomes (Supplementary Figure 9), and it might require stronger policy interventions. In 2050, 50 Mha of harvested crop area in China is projected to produce feed for highly productive livestock systems (Supplementary Figure 17). In addition to the local feed produced in China, domestic livestock production relies heavily on imported feed crops contributing to environmental degradation and GHG emissions also domestically. 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[38]. Developing marginal land to produce feed and reconnecting livestock production with land should represent a priority.

Our projected livestock production allocation within China follows the current patterns and thus does not have substantial impact on the future country-level environmental outcomes. But in reality, because of the heterogeneity of China, spatial allocation may have a substantial effect which can lead to divergent environmental impacts[39]. Careful spatial planning is therefore necessary to exploit the environmental efficiency potentials to facilitate sustainable development. Increasing ruminant productivity, is another promising way for reducing environmental pressure, since China still has large productivity gaps compared to developed countries (Supplementary Figure 18). We also find that assumptions about livestock feed efficiency change in the ROW have an important impact on the agricultural land and GHG emissions footprint of Chinese consumption (FEEF in Supplementary Figure 15b). China could thus reduce its footprint also by promoting productivity improvement in its trading partners.

Sourcing agricultural imports sustainably. Imported environmental impacts vary considerably not only depending on the openness of trade but also depending on the country of origin. For instance, milk related GHG emissions intensity of the EU 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 2), as shown also by other studies[40]. Our results show that increasing openness of trade (HD TRADE scenario) without accompanying measures can lead to both positive as well as negative impacts on the environment. Higher dairy imports from EU and bovine meat from USA would lead to less GHG emissions relative to BAU scenario, however this scenario would also lead to increased beef imports from Latin American countries where land footprints are high (Supplementary Discussion 2). 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[41]. 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[42]. The environmental performance of specific supply chains is promoted, among others, by certification schemes such as "Zero Deforestation" beef[43] or "Fairtrade" labelling[44]. 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 promoting adoption of environmentally friendly production systems in exporting countries by favoring imports of products from certified supply chains and, in general, by enforcing respect of ambitious environmental standards by its trading partners.

In summary, 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. Design of such policies will require models with high spatial resolution recognizing the heterogeneity of production conditions as well as environmental impacts in a country of the size of China. Although the role of international trade is a buffer to shocks on the domestic market, in addition to satisfying part of food demand as a stable source, potential consequences of global short-term events will need to be considered. These important aspects would however go beyond the scope of our study.

Methods

This section presents the integrated modelling approach adopted, model developments for enhanced representation of China, and model validation. Then the scenario design and the methodology used for sensitivity analysis are introduced. Virtual trade flows calculation is finally described.

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[27], GHG emissions[29], land use change and related biodiversity impacts[45]. The model is particularly suitable for forward-looking assessment of environmental impacts embodied in trade because of its bilateral trade representation[28]. 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[46].

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[47]. 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[48,49]. For forest parameters, GLOBIOM relies on the outputs of a dynamic forest management model, the Global Forest Model (G4M)[50]. Grassland productivity is obtained by combining results from EPIC and CENTURY[25,51]. Livestock production systems are parameterized with the global database developed in Herrero et al[52]. A detailed overview of data sources for the environmental indicators used in this study is presented in Supplementary Methods 4.

GLOBIOM represents international trade through net bilateral trade flows, which allow only one direction of trade flow between two regions. To simulate trade, GLOBIOM uses the Enke–Samuelson–Takayama–Judge spatial equilibrium approach, assuming homogeneous goods (imported and domestic products are the same)[53]. Thereby, GLOBIOM represents international trade through net bilateral trade flows, which allows only one direction of trade flow between two regions. And region 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. Compared with other trade assumptions (e.g., Armington, trade can occur in both directions and gross trade is represented), this trade specification allows for new trade flow creation (no observation in the base year) in response to future prices changes. As China is the largest importer for agricultural products and many countries strengthen cooperation in promoting trade with China, this approach is more appropriate for this study. Data on bilateral trade in the base year are from the BACI database[54], and data on tariffs between different countries and commodities are from the MAcMap-HS6 database[55]. Additional information about the model can be found on 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 Chinese agriculture, we included mechanisms mimicking relevant policies in place. One of the key drivers of the land use in China is afforestation policies initiated in the 1990s. They already led to afforestation of 53 Mha at the cost of cropland, pasture and other land (i.e., unmanaged grass/shrubland, non-

/sparsely vegetation). 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 enforced the self-sufficiency in three major cereal crops of 95% under the baseline scenario in line with the current trade policies. Supplementary Methods 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 2000–2020. FAOSTAT data and Chinese national statistical data until 2019/2020, as well as the OECD-FAO Agricultural Outlook projections for China until 2029 (http://www.agri-outlook.org/data) were then used to validate the model behavior (Supplementary Figure 2-7). The validation focused on the following key variables - crop yield, crop area, per capita food consumption, total demand, production, and trade. The performance of the model for the very recent past has been quantitatively documented in Supplementary Methods 3. We also provide the interpretation of mismatches caused by recent pandemic outbreaks.

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, transport costs, and a nonlinear trade expansion cost 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 until 2020 were calibrated to match the recent FAO trade matrix statistics[2] by manipulating the elasticities and slopes in the trade cost equations. The bilateral trade validation of major commodities is shown in Supplementary Figure 7. Calibration work also benefited from feedback by seven country teams of the FABLE Consortium.

Scenario design. The aim of this 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 complemented this scenario with two variants with contrasted assumptions on future drivers and decompose those 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[30]. These pathways represent the backbone of the climate related scenario analysis within IPCC[56] and have recently been used also for forward-looking biodiversity assessment in the context of IPBES[57]. 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[58], and our scenarios can cover these large uncertainties.

A business-as-usual scenario (BAU) following SSP2[59] that mostly continues recent trends in consumption and technological developments was used as baseline in this study. The two alternative scenarios including (1) the Restricted development (RD) scenario following SSP3 assumption[60] where the population in China increases faster, and growth in the GDP is slower, which leads to lower total food demand, in particular for lower demand for livestock products compared to BAU. In this scenario, international trade becomes more restricted and fragmented, reflecting lower international cooperation. And (2) the High development (HD) scenario follows SSP5 assumption[61] and orients 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 economic growth in per capita (see https://tntcat.iiasa.ac.at/SspDb), which are primary drivers for diet shifts and agricultural productivity changes.

As the food demand patterns has been aggregated at country level, income per capita drives changes in food diets[62]. Food prices are also important drivers for food consumption patterns changes, and are determined by demand price elasticities of food products[63]. The crop yield trends are estimated based on estimation of correlation between yield and scenario-specific GDP growth assumed in the SSPs[64]. In addition, re-allocation of cropland and shift of crop systems endogenously modelled also affect crop yield. For livestock systems, technical change is applied through exogenous assumption on feed conversion efficiencies estimated based on historical trends for the BAU scenario and differentiated for the alternative scenarios based on the average projected crop yield growth[65,66]. Trade assumption is one of the key differences among scenarios. Elasticity or slope of trade costs are varied depending on whether trade flow is observed in the base year or not. The trade liberalization or restrictiveness[28] across scenarios reflecting infrastructure, non-tariff trade barriers and regional factors changes determine elasticities (slopes) are multiplied or divided by 10. More information on GLOBIOM trade specification can be found in Janssens et al.[28]. The values of key scenario drivers for China are provided in Supplementary Table 5 and detailed description of alternative results can be found in Supplementary Discussion 1.

Considering that our assumptions of future changes (i.e., BAU, RD, HD scenarios) are based on a set of drivers (demographic and economic development, dietary preferences, agricultural productivity growth, and international trade policies), we conducted a sensitivity analysis in which the impact of individual elements in the RD and HD scenarios is decomposed following the approach by Stehfest et al.[67]. The decomposition was implemented at the (1) global level, (2) rest of the world (ROW) and (3) China level only. This makes it possible to assess the individual impact of the above-mentioned. Demographic development (population, POP) mainly affects future demand volumes adjusted by price effects. Economic development (gross domestic production, GDP) affects income and associated food demand. Dietary preference (DIET) presents differences in dietary patterns between scenarios. Regarding to this dimension, diet shifts and food waste are both included. Crop productivity (YILD) is characterized by a different speed of technological changes. Livestock feeds conversion efficiency (FEEF), is another key component on the supply side, determining future livestock productivity. Trade development (TRADE) represents the level of integration among global regions. The detailed results of the sensitivity analysis are presented in Supplementary Discussion 1 and Supplementary Figure 14-16.

Calculating virtual trade flows in environmental impacts. Virtual trade flows refer to resources or pollution embodied in international trade. We focus our analysis about four environmental aspects (land, GHG, irrigation water, and nitrogen) on seven major trading partners of 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 dominate the overall trade pattern of China, we allocated the export impacts into domestic production side. To calculate trade impact, we assume the same environmental intensity of products for domestic consumption and for export in a country. This is the assumption commonly used in many previous studies on virtual trade in water[68], land[69], GHG[10] and nitrogen[70]. The environmental intensity in a resource for a specific product *P* in exporting regions *R* and specific year *T* is defined as:

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

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

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

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

Where $BilateralT_{R,P,T}$ is the net bilateral trade quantity (Mt) of product P exported to China from region R in year T. $PROD_{R,P,T}$ is total production (Mt) of product P of exporting region R in year in the year T. $AREA_{R,P,T}$ is total harvested area (Mha) of product P in exporting region R.

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 (Mt), and $Water_{R,P,T}$ represents irrigation water use (km^3) for product P of exporting region R in year T. 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 (Mt CO_2 eq yr^{-1}). 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[71] while rice CH_4 based on FAOSTAT average emission factors (http://www.fao.org/fao-stat/en/#data/GR). For livestock products, we used emissions intensity parameters for CH_4 from enteric fermentation, and CH_4 and N_2O from manure management, manure dropped on pastures, rangelands and paddocks, and from the global livestock production systems database[52].

To calculate emissions from deforestation, we rely on a top-down indirect allocation approach[72]. We first determined forest losses in exporting regions based on the G4M model calculations[50], and then attributed the deforestation attributable to cropland and pasture expansion based on Curtis et al.[73]. 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 expansion are then calculated based on the following equations:

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

$$, \forall\ \Delta Crop\_area_{R,P,T} > 0 \qquad (7)$$

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

$$, \forall\ \Delta Pasture_{R,P,T} > 0 \qquad (8)$$

where $Deforemis\_crop_{R,T}$ and $Deforemis\_live_{R,T}$ are deforestation emissions (Mt CO_2 eq yr^{-1}) 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 specific feed requirements of the regional livestock production specific feed requirements from Herrero et al[52]. We calculated the total feed use and the related domestic environmental impacts for different livestock products and the related domestic environmental impacts and allocated them proportionally based on the 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 global 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 main data supporting the results of this study can be found in Supplementary Information and other relevant data are available in the IIASA DARE repository (https://dare.iiasa.ac.at/126/).

Code availability

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

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Author contributions

H.Z., P.H. and L.M. designed the study. H.Z., J.C., P.H., M.v.D. and H.V. contributed the data analysis. H.Z., J.C. and P.H. 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.

Figure captions:

Fig. 1. Trends in demand, production, and trade of agricultural products in China under the BAU scenario. (a), Demand and production patterns. The demand is further decomposed into food, feed, and biofuel/other use (the first row), while the second row represents domestic production of agricultural products. The dots show the historical data from FAOSTAT[2] averaged for the period 2009–2011 for 2010, and the most recent data for 2020 from OECD-FAO Agricultural Outlook[74]. And error bars represent the ranges of RD and HD results. Detailed results for individual product categories see Supplementary Table 1. (b), The plots on the left show the trends of net import quantity for dairy and soybean products (See Supplementary Figure 7 and 8 for more commodities and scenarios). The circular plots in the center and on the right represent the bilateral trade between China and its major partners in 2010 and 2050, respectively. Each arrow represents the volume of products coming from the exporting region to the importing region and has the same color as the exporting region.

Fig. 2. Projected changes in the domestic and imported environmental impacts between 2010 and 2030/2050 for agricultural land (crop harvested area and pasture) (a), GHG emissions (b), nitrogen fertilizer use (c), and irrigation water use (d). The stacked bars represent the decomposed effects by different agricultural products from the BAU scenario, and the markers represent the total effects from the three scenarios (BAU, RD and HD). Detailed environmental impacts from the two alternatives scenarios (RD and HD) can be found in Supplementary Figure 11-13 and Supplementary Discussion 1. For imported land-use change emissions, only deforestation emissions were considered. See Methods for further details on the calculation of the virtual trade flows.

Fig. 3. Virtual trade flows of environmental impacts due to China's agricultural imports in terms of the agricultural land (crop harvested area and pasture) (a), GHG emissions (b), nitrogen fertilizer 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 BAU scenario. The environmental impacts in the exporting regions are shown on the left, and the sources of environmental impacts by commodity are shown on the right. The numbers in the brackets represent the impacts due to the 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 9.3% of Argentina's total agricultural area use.

Fig. 4. Comparison of the global environmental impacts of China's food demand under different scenarios by 2050. (a), Environmental impacts in terms of agricultural land (crop harvested area and pasture), GHG emissions, nitrogen use, and irrigation water use in the BAU and two alternative scenarios (RD and HD). (b) The sensitivity of global environmental impacts to changes in six key drivers. The sensitivity is presented as the relative change of environmental impacts compared to the BAU level due to the changes in the individual key drivers implemented globally. The six key drivers are population (POP), economic development (expressed as GDP), consumption preference (DIET), crop productivity growth (YILD), livestock productivity growth (FEEF), and the level of trade integration (TRADE). See Methods section "Scenario design" for details on the implementation of the sensitivity tests.

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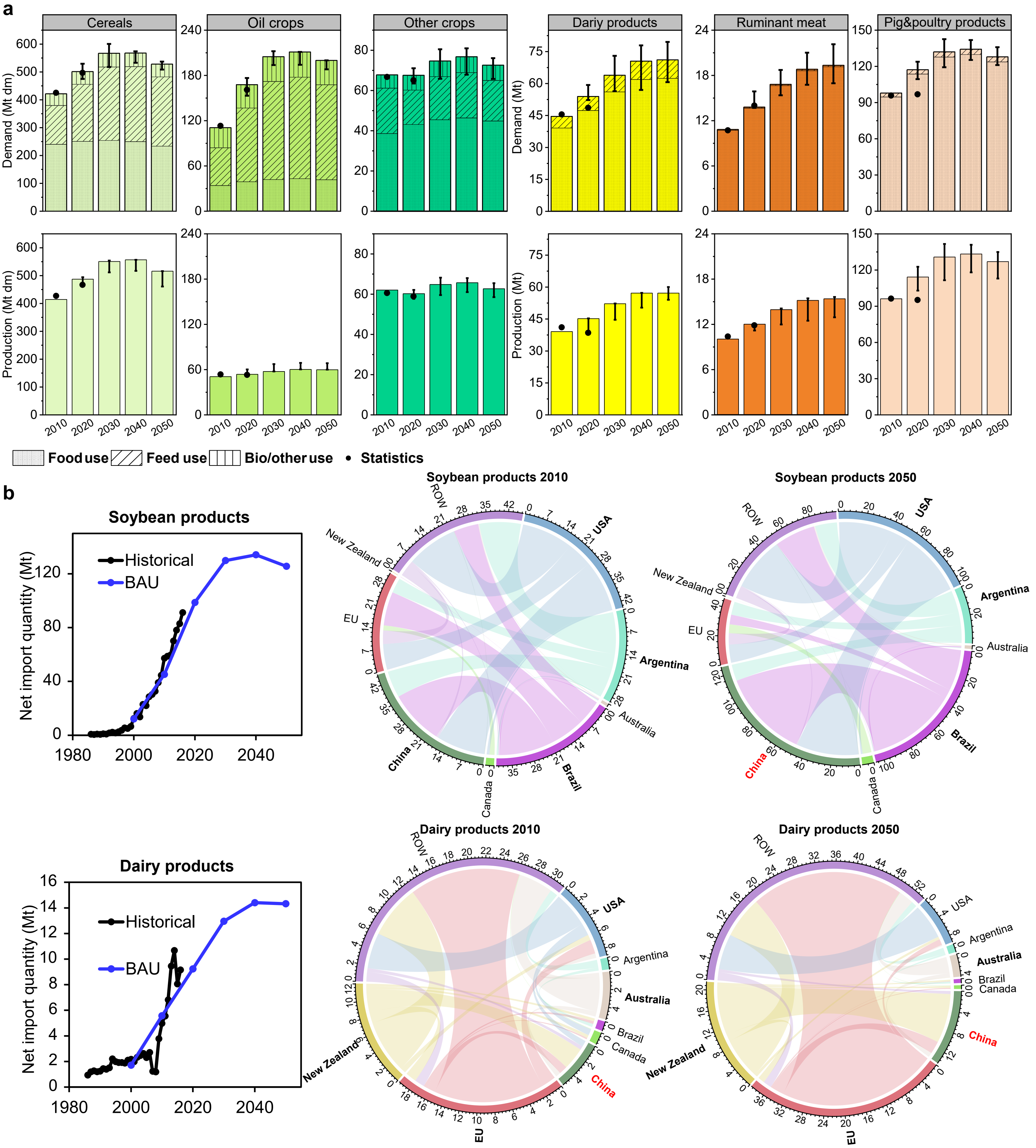
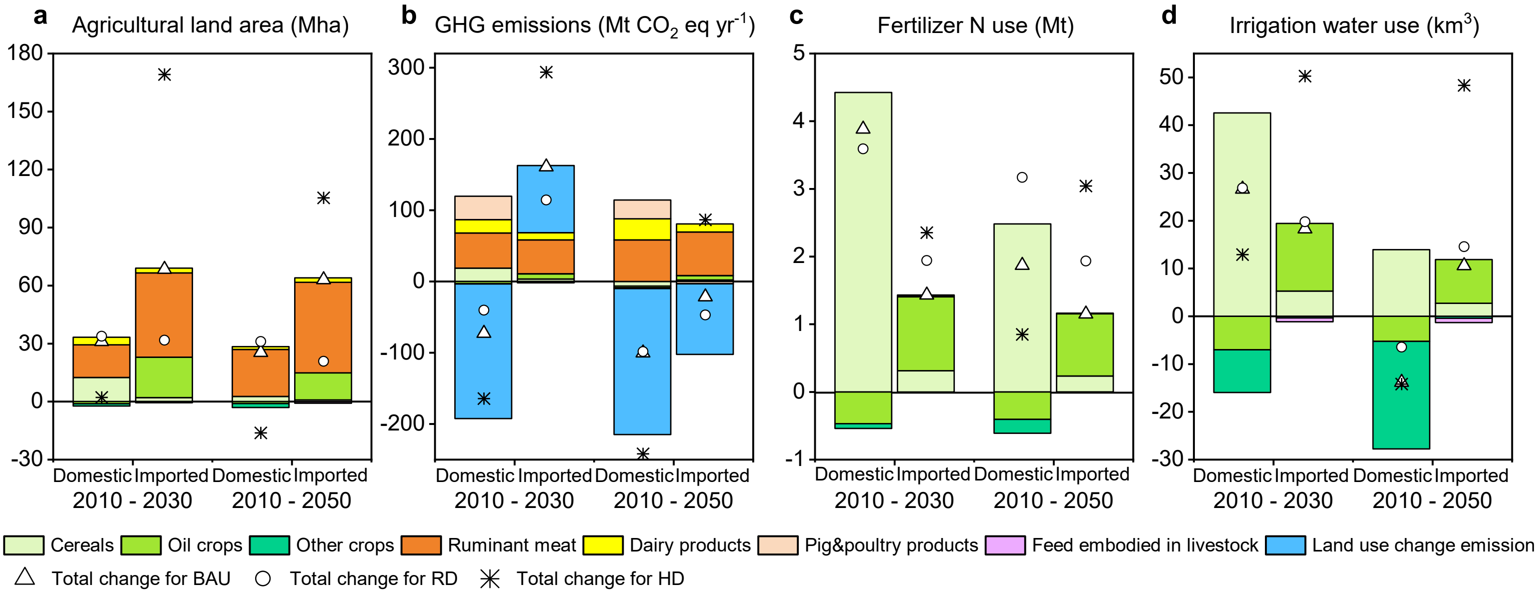
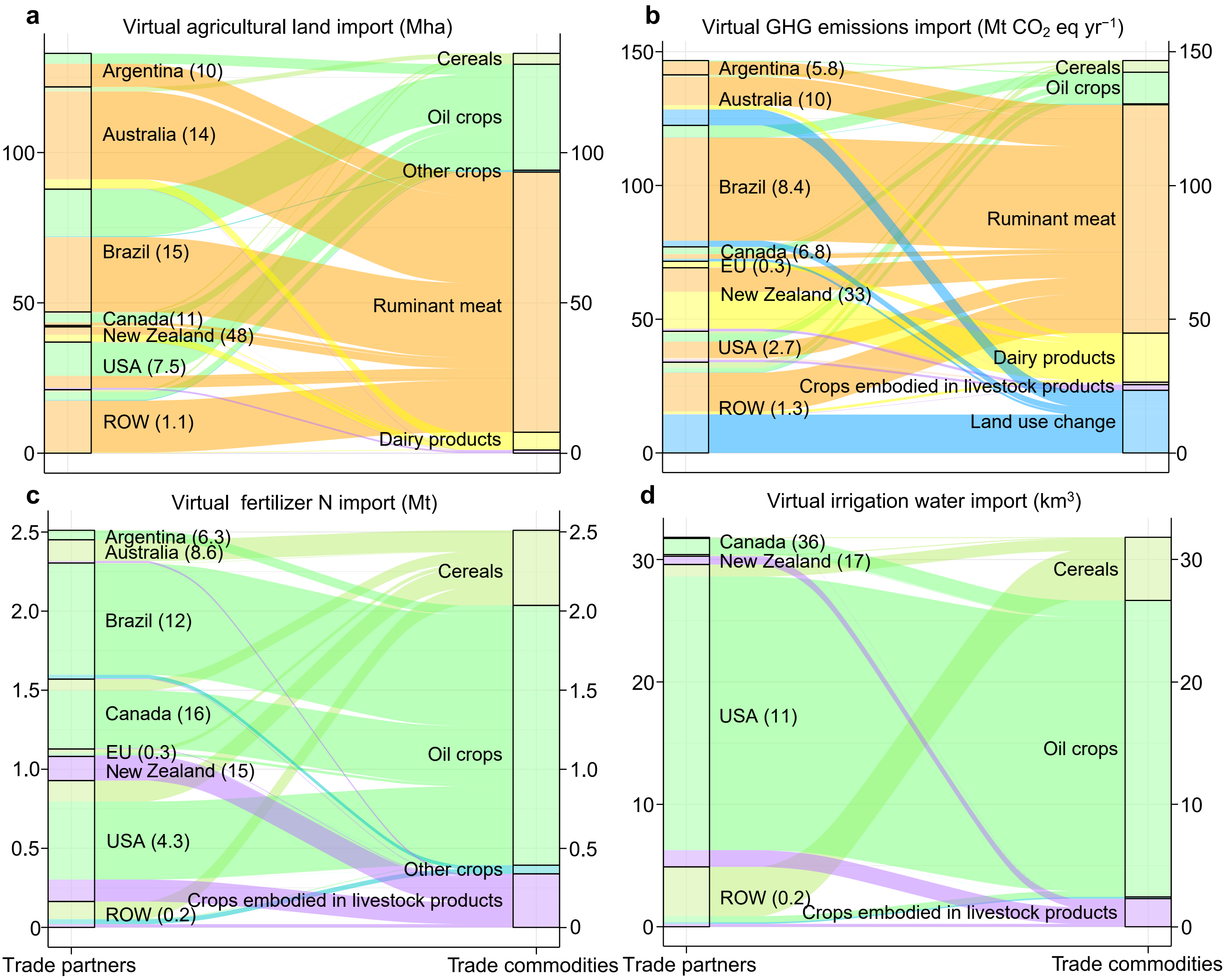
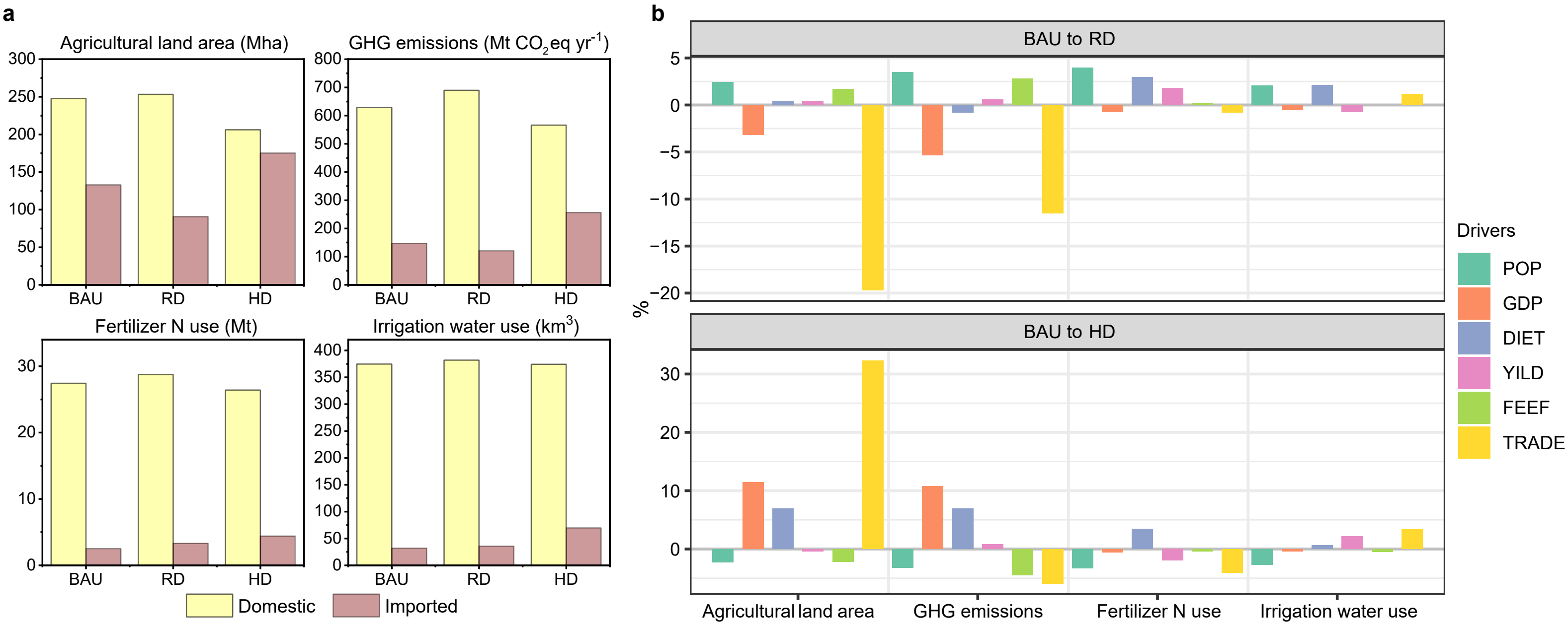

a
Cereals
Oil crops
Other crops
Dariy products
Ruminant meat
Pig&poultry products
Demand (Mt dm)
Demand (Mt)
Production (Mt dm)
Production (Mt)
2010
2020
2030
2040
2050
Food use
Feed use
Bio/other use
Statistics
b
Soybean products
Net import quantity (Mt)
Historical
BAU
1980
2000
2020
2040
Soybean products 2010
Soybean products 2050
ROW
USA
Argentina
Australia
Brazil
Canada
China
EU
New Zealand
Dairy products
Dairy products 2010
Dairy products 2050


a
Agricultural land area (Mha)
180
150
120
90
60
30
0
-30
Domestic
Imported
Domestic
Imported
2010 - 2030
2010 - 2050
b
GHG emissions (Mt CO_2 eq yr^{-1})
300
200
100
0
-100
-200
c
Fertilizer N use (Mt)
5
4
3
2
1
0
-1
d
Irrigation water use (km^3)
50
40
30
20
10
0
-10
-20
-30
Cereals
Oil crops
Other crops
Ruminant meat
Dairy products
Pig&poultry products
Feed embodied in livestock
Land use change emission
Total change for BAU
Total change for RD
Total change for HD


a
Virtual agricultural land import (Mha)
Argentina (10)
Australia (14)
Brazil (15)
Canada(11)
New Zealand (48)
USA (7.5)
ROW (1.1)
Cereals
Oil crops
Other crops
Ruminant meat
Dairy products
0
50
100
b
Virtual GHG emissions import (Mt CO_2 eq yr^{-1})
Argentina (5.8)
Australia (10)
Brazil (8.4)
Canada (6.8)
EU (0.3)
New Zealand (33)
USA (2.7)
ROW (1.3)
Cereals
Oil crops
Ruminant meat
Dairy products
Crops embodied in livestock products
Land use change
0
50
100
150
c
Virtual fertilizer N import (Mt)
Argentina (6.3)
Australia (8.6)
Brazil (12)
Canada (16)
EU (0.3)
New Zealand (15)
USA (4.3)
ROW (0.2)
Cereals
Oil crops
Other crops
Crops embodied in livestock products
0
0.5
1.0
1.5
2.0
2.5
Trade partners
Trade commodities
d
Virtual irrigation water import (km^3)
Canada (36)
New Zealand (17)
USA (11)
ROW (0.2)
Cereals
Oil crops
Crops embodied in livestock products
0
10
20
30
Trade partners
Trade commodities


a
Agricultural land area (Mha)
0
50
100
150
200
250
300
BAU
RD
HD
GHG emissions (Mt CO$_2$eq yr^{-1})
0
100
200
300
400
500
600
700
800
BAU
RD
HD
Fertilizer N use (Mt)
0
10
20
30
BAU
RD
HD
Irrigation water use (km^3)
0
50
100
150
200
250
300
350
400
BAU
RD
HD
Domestic
Imported
b
BAU to RD
5
0
−5
−10
−15
−20
%
BAU to HD
30
20
10
0
Agricultural land area
GHG emissions
Fertilizer N use
Irrigation water use
Drivers
POP
GDP
DIET
YILD
FEEF
TRADE