Quantifying the environmental and economic impacts of upcycling low-opportunity-cost feed as animal feed in China: a general equilibrium approach

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Abstract

Upcycling low-opportunity-cost feed products (LCFs), such as food waste and food processing by-products, as animal feed could reduce environmental impacts of livestock production, but rebound effects, where lower feed costs lead to livestock production expansion, may diminish these benefits. Using an integrated environmental-economic model, we assessed the global impacts of upcycling LCFs in China's monogastric livestock production. We found that the upcycling increased monogastric livestock production by 23-36% and raised Chinese economy-wide acidification emissions by 2.5-4.0%. Eutrophication emissions decreased by 0.2% with partial upcycling but increased by 0.2% with full upcycling. Greenhouse gas emissions decreased slightly by 0.5-1.4% through less LCFs in landfills and incinerators, and non-food production contraction. This upcycling accompanying with resource reallocation across the whole economy enhance food security in China without compromising that of its trading partners. Implementing emission taxes to a proper level provides an opportunity to absorb the rebound effects in China and safeguard global food security.

Keywords

circular food system; food waste; food security; environmental impacts; environmental-economic modelling; rebound effects.

1. Introduction

Animal-sourced food (ASF), such as meat, milk, and eggs, is the main contributor to the environmental impacts of food systems. The surge in demand for ASF, driven by population growth, prosperity, and urbanization, [1,2] is expected to double by 2050, especially in developing countries [3]. This surge in livestock production has exacerbated food-feed competition and significantly contributes to the exceedance of the planetary boundaries (PBs) for nitrogen (N), phosphorus (P) and greenhouse gas (GHG) emissions. Currently, 70% of global agricultural land is used for producing animal feed [4], and global livestock production accounts for 13-18% of the total anthropogenic GHG emissions [5], 40% of the ammonia (NH_3) and nitrous oxide (N_2O) emissions [6], and around 24% of N and 55% of P losses to water bodies [7]. It has been shown that the global 1.5°C climate target cannot be achieved without mitigating emissions from food systems [8].

Global food waste has risen from 1.3 to 1.6–2.5 billion tons in recent years despite substantial efforts to reduce food waste [9]. A large proportion of food waste ends up in landfills or incinerators, exacerbating GHG emissions and climate change [10]. Upcycling low-opportunity-cost feed products (LCFs), such as food waste and food processing by-products, as animal feed is, thus, crucial for reducing environmental impacts and building more circular food systems [11], as it offers a pathway to mitigate land-related pressures [12], alleviate the food-feed competition [11], and reduce emissions from food systems and improper food waste disposal [13]. This is because LCFs typically compete less for land and natural resources than human-edible feeding crops [11-13]. Increased utilisation of LCFs as feed may also contribute to achieving Sustainable Development Goals (SDGs), including SDG 2 (zero hunger), SDG 6 (clean water and sanitation), SDG 12 (responsible consumption and production), SDG 13 (climate action), and SDG 15 (life on land) [14].

While many studies acknowledge the environmental benefits of increasing LCFs utilisation as feed, significant gaps remain in the existing literature, particularly in three critical areas. First, previous studies [11-13] employing linear optimization models to evaluate the environmental impacts of this circular transition may have overestimated the environmental benefits by disregarding "rebound effect" (or “Jevons paradox”) [15]. The rebound effect, where lower feed costs lead to livestock production expansion, may diminish the environmental benefits of feeding animals with LCFs.

Second, the “rebound effect” phenomenon has been extensively studied in energy systems [16,17], but its implications in food systems are largely lacking. Although previous studies have explored rebound effects related to a global dietary shift towards plant-based food [18] and halving food loss and waste [19], there is still limited understanding of the rebound effect of upcycling LCFs as animal feed. Third, strateiges to absorb these negative rebound effects resulting from upcycling LCFs as animal feed have not yet been formally explored. Implementing emissions taxes is considered as an effective policy instrument to identify the most cost-effective mitigation pathway for achieving a given emission mitigation target [20-22]. For example, many countries, such as the United states, France, Canada, and New Zealand, have implemented various forms of carbon taxes to mitigate GHG emissions [23]. China has committed to tackling both global environmental challenges, such as reducing GHG emissions through its pledge for carbon neutrality by 2060 under the Paris Agreement [24,25], as well as addressing local environmental pollution, including emissions of acidification and eutrophication pollutants, to meet the reduction targets set in the “14th Five-Year Plan” [26]. It remains unclear by how much rebound effects may influence the expected benefits of upcycling LCFs as animal feed.

In this study, we fill these gaps and contribute to the existing literature by using an integrated environmental-economic modelling approach based on the applied general equilibrium (AGE) models to assess the environmental and economic consequences of upcycling LCFs in China’s monogastric livestock production as feed in a global context. Next, we explore how implementing economy-wide emissions taxes could absorb rebound effects of this upcycling while safeguarding food security. We focused on China for our study because it is the world’s largest animal producer, accounting for 46%, 34%, and 13% of global pork, egg, and poultry meat production in 2018, respectively [27]. Furthermore, 27% of food produced for human consumption are lost or wasted in China [28], implying a great opportunity to upcycle food waste as feed. In addition, the Chinese government has proposed to lower the agricultural product processing loss rate to below 3% by 2035 [29], and to substitute human-edible feed ingredients, such as soybeans and maize, in animal feed with food processing by-products [30]. Thus, we considered two types of LCFs, i.e., food waste (cereal grains waste, vegetables & fruits waste, roots & tubers waste, and oilseeds & pulses waste) and food processing by-products (cereal bran, alcoholic pulp, and oil cakes). We addressed three main research questions. First, how will an increased utilisation of LCFs as feed influence livestock production, food supply, and other sectors in China and its main food and feed trading partners (MTP, including Brazil, the United States, and Canada)? Second, how will an increased utilisation of LCFs influence economy-wide emissions of GHGs, acidification pollutants, and eutrophication pollutants, as well as food security (i.e., average food price, food affordability, population at risk of hunger, and food availability)? Third, how will emission taxes absorb rebound effects of this upcycling while safeguarding food security?

We examined five scenarios: (i) the baseline (S0) scenario represents the economies of China and MTP in 2014; (ii) scenario 1 (S1) involves upcycling partial use of LCFs (54% of food waste and 100% of food processing by-products) as feed for monogastric livestock production in China; (iii) scenario 2 (S2) involves upcycling full use of LCFs (100% of food waste and 100% of food processing by-products) as feed for monogastric livestock production in China; (iv) scenario 3 (S3 = S1 + A modest emission mitigation target) entails implementing economy-wide emission taxes to ensure that emissions of GHGs, acidification pollutants, and eutrophication pollutants in both China and MTP do not exceed their baseline (S0) levels; (v) scenario 4 (S4 = S1 + an ambitious emission mitigation target) entails implementing economy-wide emission taxes to meet China's and MTP's annual GHG mitigation targets under the Intended Nationally Determined Contributions (INDC) of the Paris Agreement [24,25], while also addressing China's emission reduction goals for acidification and eutrophication pollutants in line with the "14th Five-Year Plan" [26]. The levels of upcycling partial and full use of LCFs as animal feed is estimated using calculations from Fang, et al. [12], who determine that the maximum utilisation rate of food waste with high moisture content in China is 54% when cross-provincial transportation of food waste is not allowed. When substituting primary feed (i.e., feeding crops and compound feed) in animal diets with food waste and food processing by-products, we kept the total protein and total energy supplies for per unit of animal output were kept constant in all scenarios. The scenarios mentioned above are further described in Supplementary Table 1.

2. Materials and Methods

2.1 The integrated environmental-economic model and database

The integrated environmental-economic model based on an AGE framework has been widely used to identify the optimal solution towards greater sustainability and enable efficient allocation of resources in the economy under social welfare maximisation [31-35]. For this study, we developed a global comparative static AGE model, a modified version of an integrated environmental-economic model, [36-39] and improved the representation of food-related (crop and livestock) sectors and associated non-food (compound feed, food processing by-products, nitrogen and phosphorous fertiliser, food waste treatment, and non-food) sectors (see Fig. 1). While the static model has limitations in short-term policy analysis, it minimises assumptions and uncertainties about future economic conditions by not considering technological and resource changes over time, allowing us to isolate the impact of feeding China’s monogastric livestock with low-opportunity-cost feed products (LCFs). Our model distinguished two regions: China and its main food and feed trading partners (MTP, including Brazil, the United States, and Canada). These partners accounted for more than 75% of China's total trade volume related to food and feed in 2014. Our reference year is 2014, which represents the latest available year for data for the Global Trade Analysis Project (GTAP) database. Our model is solved using the general algebraic modelling system (GAMS) software package [40].

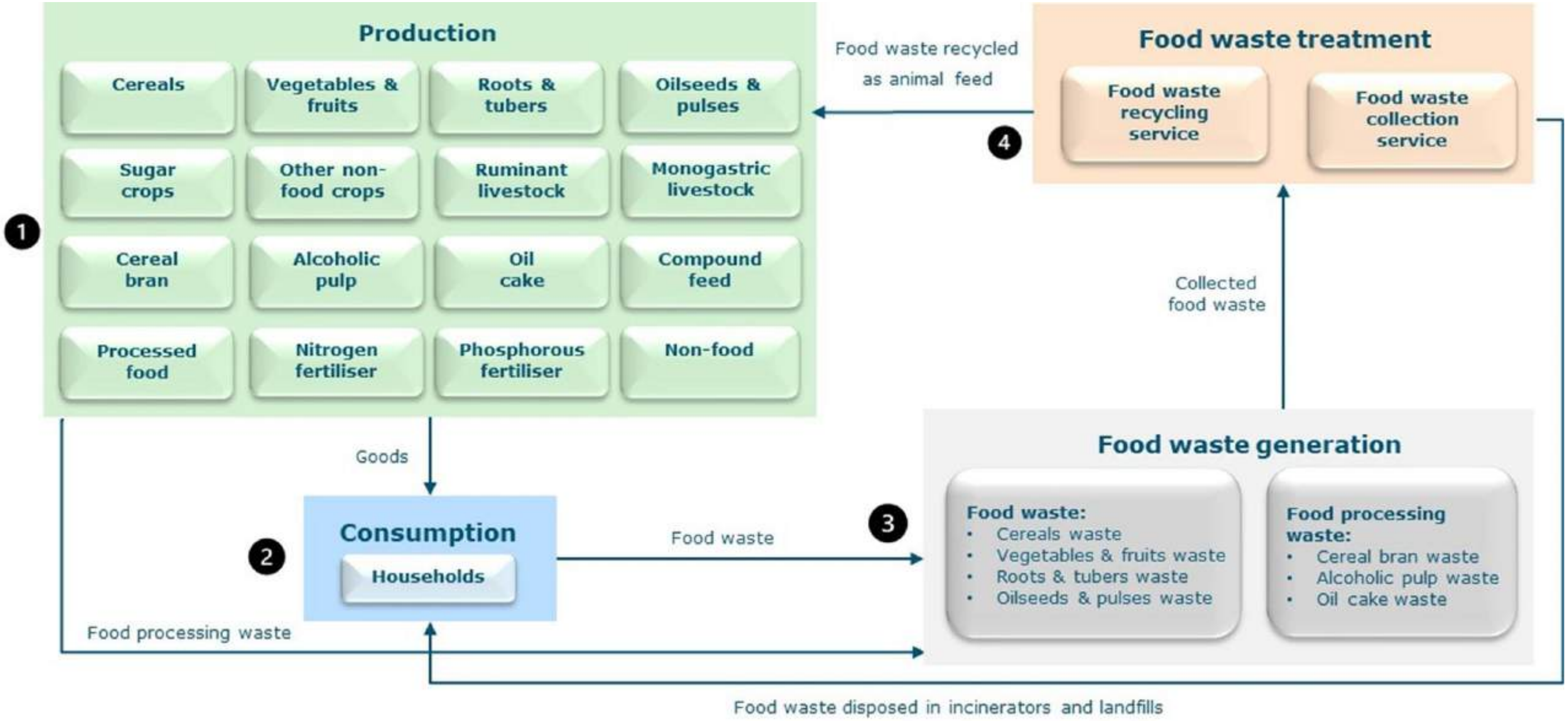

Fig. 1 | Representation of the economy in China in the applied general equilibrium (AGE) framework with food waste and food processing waste. The framework includes four parts: (1) Production; (2) Consumption; (3) Food waste generation; (4) Food waste treatment. The generated food waste is sent either to the 'food waste recycling service' sector or the 'food waste collection service' sector. The food waste recycling service sector recycles food waste as feed for monogastric livestock production. The food waste collection service sector collects food waste for landfill and incineration. The consumer price of food includes both the market price of food and the cost of collecting food waste. Livestock producers bear the cost of recycling food waste as feed. Detailed information is presented in Methods and Supplementary Information.

Modelling circularity in livestock production requires a detailed representation of biophysical flows to consider nutritional balances and livestock feeding constraints of increasing the utilisation of food waste as feed in monogastric livestock production. Following Gatto, et al. [41], we converted dollar-based quantities to physical quantities (Tg) to allow the tracing of biophysical flows through the global economy. GTAP version 10 database [42] was used to calibrate our AGE model and provide dollar-based quantities. We designed a sectoral aggregation scheme comprising 16 sectors (see Appendix Table 1) from the original GTAP database to produce social accounting matrices (SAM) (see Appendix Tables 2-3) in our study. Data on physical quantities (see Supplementary Table 2) of crop and livestock production was obtained from FAO [27]. Feed production was extracted from "Feed" in the FAO food balance sheet. Grass from natural grassland was derived from Miao and Zhang [43]. We only included grass from natural grassland where ruminant livestock is grazing for feed, and grass from remaining grassland was excluded. Data on the trade shares matrix was calculated from the data from the UN Comtrade Database [44].

Livestock categories were aggregated into two sectors, i.e., monogastric livestock (including pigs, broilers, and laying hens) and ruminant livestock (including dairy cattle, other cattle, and sheep & goats). Furthermore, the inclusion of animal-specific dietary constraints in our model allowed us to calculate the nutritional balance (crude protein and digestible energy), feed conversion ratios (FCR, the ratio of fresh feed inputs to live weight gain), and edible feed conversion ratio (eFCR, the amount of human-edible feedstuffs, i.e., feeding crops and compound feed, used for per unit of live weight gain) [45] for each livestock sector. First, we obtained the physical quantities (Tg) of feed protein and energy required to produce the output of livestock. Then, the composition of total feed supplied to each livestock sector is specified. When substituting primary feed (i.e., feeding crops and compound feed) in animal diets with food waste and food processing by-products, we kept the total protein and total energy supplies for per unit of animal output were kept constant in all scenarios. Our FCRs for ruminant livestock are slightly different from FCRs in the literature, as we did not fully account for hay, crop residues, and roughage-like by-products, but this bias did not affect the impacts of feeding food waste and food processing by-products to monogastric livestock. Further model details, nutritional balance, and detailed composition of animals' diets are available in the Supplementary Information (SI).

2.2 Modelling food waste and food processing waste

In this study, we considered two types of LCFs, i.e., food waste and food processing by-products. Food waste was considered a local resource within China, while food processing by-products could be traded between China and MTP. Food waste refers to discarded food products during distribution and consumption. We only considered plant-sourced food waste because animal-sourced food waste may pose a risk of pathogen transfer, including foot-and-mouth and classical swine fever [46]. Food waste was quantified separately for each type of food product using data on food consumption and China-specific food loss and waste fractions [28] following the FAO methodology [47]. Four types of food waste were distinguished, including cereal grains waste, vegetables & fruits waste, roots & tubers waste, and oilseeds & pulses waste. Food processing by-products refer to by-products produced during the food processing stage, including cereal bran, alcoholic pulp (including distiller's grains from maize ethanol production, brewer's grains from barley beer production, and distiller's grains from liquor production), and oil cakes (including soybean cake and other oil cakes). Food processing by-products were estimated from the consumption of food products and specific technical conversion factors [48]. The total amounts of food waste and food processing by-products and their current use as animal feed and discarded biomass (i.e., landfill and incineration) for China in S0 are presented in Supplementary Table 4.

Our model incorporated two food waste-related sectors, i.e., "food waste collection service" and "food waste recycling service" (Figure 1). The food waste recycling service sector recycles food waste as feed for monogastric livestock production. The food waste collection service sector collects food waste for landfill and incineration. Waste collection, treatment and disposal activities were included in the 'Waste and water (wtr)' sector in the GTAP database. Food waste generation was added as a margin commodity, similar to how GTAP treated transport costs following Peterson [49]. Thus, the consumer price of food includes both the market price of food and the cost of collecting food waste. Consumers allocate their income to both the consumption of goods and food waste collection services, but they derive utility solely from the consumption of goods. In terms of recycling food waste as feed, monogastric livestock production bears the associated cost. By multiplying the quantity of food waste with the price of food waste treatment, we can calculate the value of food waste generation. Physical quantities and prices of food waste recycling service and food waste collection service in China were presented in Supplementary Tables 4-5.

2.3 Environmental impact assessment

Three main environmental impacts of food systems were distinguished, i.e., global warming potential (GWP, caused by greenhouse gas (GHG) emissions, including carbon dioxide(CO_2), methane (CH_4), and nitrous oxide (N_2O) emissions; converted to CO_2 equivalents), acidification potential (AP, caused by pollutants leading to acidification, including ammonia (NH_3), nitrogen oxides (NO_x), and sulphur dioxide (SO_2) emissions; converted to NH_3 equivalents), and eutrophication potential (EP, caused by pollutants leading to eutrophication, including N and P losses; converted to N equivalents). The conversion factors for GWP, AP, and EP were derived from Goedkoop, et al. [50]. Data on CO_2, CH_4, and N_2O emissions were obtained from the Climate Analysis Indicators Tool (CAIT) [51]. All GHG emissions calculations in our model follow the IPCC Tier 2 approach [52]. We derived NH_3, NO_x, and SO_2 emissions from Liu, et al. [53], Huang, et al. [54], and Dahiya, et al. [55], respectively. We considered NO_x emissions from energy use only, as agriculture's contribution to NO_x emissions is generally small ($\leqslant 2\%$). We used the global eutrophication database of food and non-food provided by Hamilton, et al. [7] to obtain data on N and P losses to water bodies.

The total emissions of GHGs, acidification pollutants, and eutrophication poluutants for the food and non-food sectors in the base year were estimated first. Then, we allocated the total emissions to specific sectors according to the shares of emissions per sector in total emissions to unify the emission data from different years. Detailed information about emissions sources across sectors is provided in Appendix Table 4. The sector-level emissions as well as the US dollar-based emission intensities of GHGs (t CO_2 equivalents million USD^{-1}), acidification pollutants (t NH_3 equivalents million USD^{-1}), and eutrophication pollutants (t N equivalents million USD^{-1}) are presented in Appendix Tables 5-10. We attributed the environmental impacts between the main (e.g., cereal flour) and joint products (e.g., cereal bran) according to their relative economic values (see Supplementary Table 6).

Two types of land use, i.e., cropland and pastureland, were distinguished. We updated the GTAP data on crop harvested areas using the FAO [27] database. Pastureland was defined as areas where ruminant grazing occurs. We derived nitrogen and phosphorous fertiliser use by crop types and countries from Ludemann, et al. [56].

2.4 Food security indicators

he FAO [57] defines food security as encompassing four key dimensions: availability (adequate food supply), access (sufficient resources to obtain food), utilisation (nutritious and safe diets), and stability (consistent access to food over time). We focused on the first two dimensions. First, food availability is defined as 'calories per capita per day available for consumption'. ‘Population at risk of hunger’ refers to the portion of people experiencing dietary energy (calorie) deprivation lasting more than a year following the FAO-based approach [58]. This approach has been widely used in agricultural economic models to evaluate the risk of food insecurity [21,59,60]. In essence, the population at risk of hunger is determined by multiplying the prevalence of undernourishment (PoU) by the total population and is based on dietary energy availability calculated by our model. It is assumed that there is no risk of hunger for high-income countries; consequently, the population at risk of hunger is not applied to the United States and Canada [21,59,60]. Second, the access dimension is tied to people’s purchasing power, which depends on food prices, dietary habits, and income trends [61]. We calculated the average food (including primary food products and processed food) price, and estimated changes in food affordability by subtracting changes in the average wage across the whole economy from fluctuations in cereal prices.

2.5 Definition of scenarios

To estimate the impacts of increased utilisation of LCFs as animal feed on food security and the environment, we examined five scenarios, including one baseline (S0) scenario representing the economies of China and MTP in 2014, two scenarios involving increased utilisation of LCFs as animal feed, and two scenarios with utilisation of LCFs as animal feed combined with emission mitigation measures. We implemented economy-wide emission taxes under the partial use of LCFs as animal feed (scenario S1), considering the perishability and collection challenges of food waste, as well as the reduced availability of food waste for feed in accordance with SDG 12.3 ("halving food waste") [14]. The latter four scenarios were compared to the 2014 baseline (S0) scenario. The scenarios are further described below and in Supplementary Table 1.

2.5.1 S1 - Partial use of LCFs as feed

Scenario S1 investigated the impacts of upcycling partial LCFs as feed (54% of food waste and 100% of food processing by-products for monogastric livestock). Cross-provincial transportation of food waste was not allowed in S1, which limits the maximum utilisation rate of food waste with high moisture content to 54% in China, according to Fang, et al. [12].

2.5.2 S2 - Full use of LCFs as feed

Scenario S2 analysed the impacts of upcycling sull LCFs as feed (100% of food waste and 100% of food processing by-products for monogastric livestock). Cross-provincial transportation of food waste was allowed in S2 because we assumed that new technology will become available for processing food waste with high moisture content. Economies of scale in food waste recycling were considered in S2; a 1% increase in recycled waste resulted in only a 0.078% rise in recycling costs [62]. Thus, as production scales up, marginal costs decrease and then stabilise.

2.5.3 S3 - S1 + A modest emission mitigation target

Economy-wide and uniform emission taxes were implemented across all sectors (crop, livestock, and non-food) at the regional level to achieve a modest emission mitigation target, assuming that emissions of GHGs, acidification pollutants, and eutrophication pollutants in both China and MTP do not exceed their baseline (S0) levels. For a given emission mitigation target for each type of pollutant, the AGE model can endogenously determine the emission taxes for various pollutants (expressed in \$ per ton of CO_2 equivalents, \$ per ton of NH_3 equivalents, and \$ per ton of N equivalents). This approach is commonly used in the literature [21,22,60,63] and allows to identify the most cost-effective mitigation pathway for achieving a given emission mitigation target.

2.5.4 S4 - S1 + An ambitious emission mitigation target

Economy-wide and uniform emission taxes were implemented across all sectors (crop, livestock, and non-food) at the regional level to achieve an ambitious emission mitigation target, assuming that emissions of GHGs, acidification pollutants, and eutrophication pollutants remain within the emission thresholds set by China's and the MTP's annual GHG mitigation targets under the Intended Nationally Determined Contributions (INDC) of the Paris Agreement [24,25], as well as China's emission reduction goals for acidification and eutrophication pollutants in line with the "14th Five-Year Plan" [26].

3. Results

3.1 Rebound effects of livestock production expansion and its knock-on effects on other commodities.

China produced about 104 Tg of monogastric livestock products (pork: 57 Tg; poultry meat: 18 Tg; egg: 29 Tg) and 53 Tg of ruminant livestock products (milk: 42 Tg; beef: 6 Tg; lamb: 4 Tg) in 2014. We estimated that 226 Tg food waste (equivalent to 54 Tg in dry matter; 7 Tg in crude protein; 690 billion MJ in energy) and 163 Tg food processing by-products (equivalent to 139 Tg in dry matter; 49 Tg in crude protein; 1907 billion MJ in energy) was available in China in 2014, but only 39% of the food waste and 51% of the food processing by-products were recycled as feed, with the remainder disposed in landfills and incinerators (Supplementary Tables 3-4). The limited use of food waste for feed production in China is primarily due to the early stage of industrialization of recycling food waste as feed, which currently has a low processing capacity [64]. Despite being protein-rich, food processing by-products, such as unprocessed oil cakes, contain anti-nutritional factors that hinder protein absorption by animals. Although fermentation can effectively eliminate these anti-nutritional factors and enhance digestion and growth performance [65], its limited adoption in China leads to a large amount of these by-products being discarded in landfills or incinerators.

Unlike previous studies that considered recycling LCFs as feed to be costless [11-13], we modelled an increasing cost of more recycled LCFs as feed born by monogastric livestock producers and a decreasing cost of less LCFs in landfills and incinerators covered by consumers. We demonstrated that upcycling 54-100% of food waste and 100% of food processing by-products as feed in scenarios S1 and S2 increased the share of food waste and food processing by-products used as feed within the total feed use by 10-14% in dry matter (Supplementary Fig. 2). The upcycling increased the supply of feed protein by 27-40% (14-21 Tg) and feed energy by 26-39% (883-1318 billion MJ), and reduced total feed cost (i.e., feeding crops, compound feed, food waste, and by-products) for per unit of monogastric livestock production by 2.1-3.0%. This led to a 23-36% (24-37 Tg) increase in monogastric livestock production in S1 and S2 (Fig. 2b). This shift signifies a transition for China from a net importer of monogastric livestock, importing 1% (1.2 Tg) of output in the baseline (S0), to an exporting nation, with 18-25% (24-37 Tg) of output being exported (Fig. 2e). Ruminant livestock production decreased by 3% (2 Tg) as the expansion of monogastric livestock reduced the availability of feeding crops and compound feed to ruminant livestock (Fig. 2b). To meet domestic demand, ruminant livestock imports rose from 1% (0.5 Tg) of output in the baseline (S0) to 4% (2 Tg) (Fig. 2e).

Expanded monogastric livestock production raised the demand for primary feed (i.e., feed crops and compound feed), which suprisingly outweighed the reduction in primary feed use by substituting it with food waste and food processing by-products. The overall feed demand for both monogastric and ruminant livestock increased by 17-34% (116-236 Tg) due to a 33-67% (118-238 Tg) rise in feed demand for monogastric livestock (Fig. 3b). The upcycling increased the feed conversion ratio (FCR, the ratio of fresh feed inputs to live weight gain) for monogastric livestock by 0.22-0.62 kg kg^{-1}, but decreased the edible feed conversion ratio (eFCR, the amount of human-edible feedstuffs, i.e., feeding crops and compound feed, used for per unit of live weight gain) by 0.11-0.19 kg kg^{-1}, indicating its reduced reliance on human-edible feedstuffs (Supplementary Fig. 3a). Since feeding crops and compound feed account for only 12% of ruminant feed (compared to 88% from grass, see Supplementary Fig. 4d), the upcycling had a minor impact on ruminant production and its FCR and eFCR (Supplementary Fig. 3b). The growing demand for crop used as animal feed increased reliance on crop imports, with the import share rising from 11% (146 Tg) in the baseline (S0) to 15–19% (184–236 Tg) (Fig. 2d), considering that the total crop production declined by 1.2-4.4% (15-57 Tg) (Fig. 2a). However, the crop cultivated area expanded by 0.6-13% (1-24 Mha) (Fig. 3a). Detailed impacts on crop production structure, as well as the use of N and P fertilisers, were explicitly presented in Supplementary Results.

Adjustments in crop and livestock production also had knock-on effects beyond the agricultural sectors in the broader economy, thus influenced sectoral employment, gross domestic product (GDP), and household welfare (a measure of economic well-being in US dollars). We observed that the 27-43% (11.5-18.4 million people) increase in employment in monogastric livestock production was largely a transfer from the non-food sector (i.e., industries and services; detailed in Appendix Table 1) (Supplementary Fig. 7a,c). The non-food sector experienced a slight relative output decline of 1.0-1.4% (Supplementary Fig. 8a,c) and the largest absolute loss of 28-41 billion US dollars (USD, 2014 constant price) (Supplementary Fig. 9a). In contrast, N and P fertiliser production surged by 35-36% (13.7-14.0 Tg) and 20-59% (3.5-10.1 Tg) (Fig. 2c), respectively, due to rising demand and decreased production costs, as the shrinking non-food sector made key inputs more available to fertiliser production. As a consequence, China became an exporter of N fertiliser (11.8-12.7 Tg) and P fertiliser (3.1-9.3 Tg) (Fig. 2f). The absolute value of fertiliser output rose by 5.4-7.0 billion USD (Supplementary Fig. 9a), which compensated less than one-fifth of the total output decrease of the non-food sector. The economic losses in the crop and non-food sectors were largely offset by the expansion of the monogastric livestock and fertiliser sectors (Supplementary Fig. 9a). The overall impact on China's economy was a 0.02-0.07% (0.8-2.6 billion USD) decrease in GDP (Supplementary Fig. 11) and a slight positive impacts on household welfare (0.18-0.32%) (Supplementary Fig. 12).

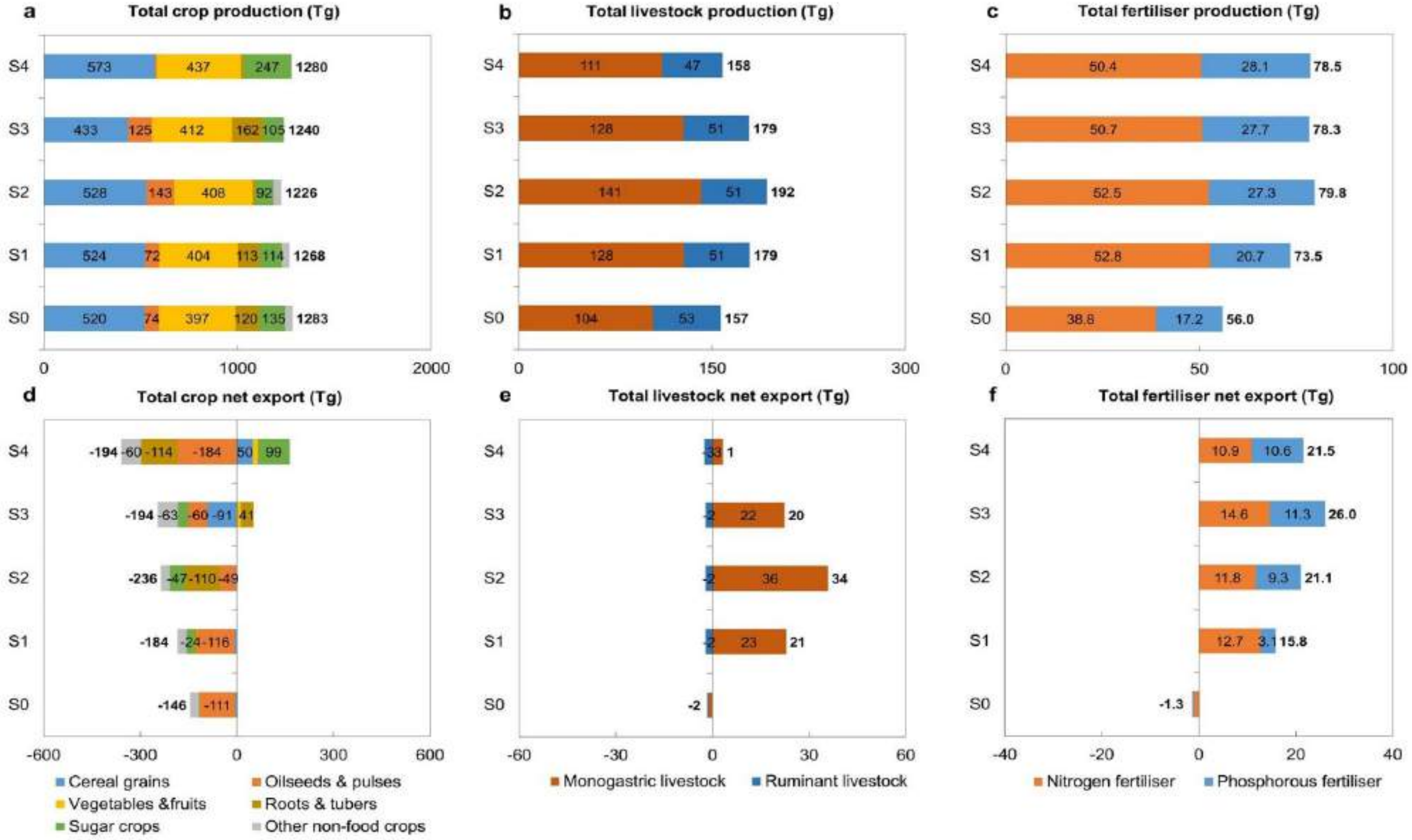

Fig. 2 | Impacts of upcycling low–opportunity–cost feed products (LCFs) in China's monogastric livestock as feed on domestic production and net export of total crop, livestock, and fertiliser. Total (**a**) crop, (**b**) livestock, and (**c**) fertiliser production (Tg) in scenarios. Total (**d**) crop, (**e**) livestock, and (**f**) fertiliser net export (Tg) in scenarios. Total crop production exclude food waste and food processing by-products used by "food waste recycling service" and "food waste collection service" sectors (see Supplementary Table 4 for detailed data). Definitions of scenarios (S1 - 'Partial use of LCFs as feed'; S2 - 'Full use of LCFs as feed'; S3 - 'S1 + A modest emission mitigation target'; S4 - 'S1 + An ambitious emission mitigation target') are described in Table 1.

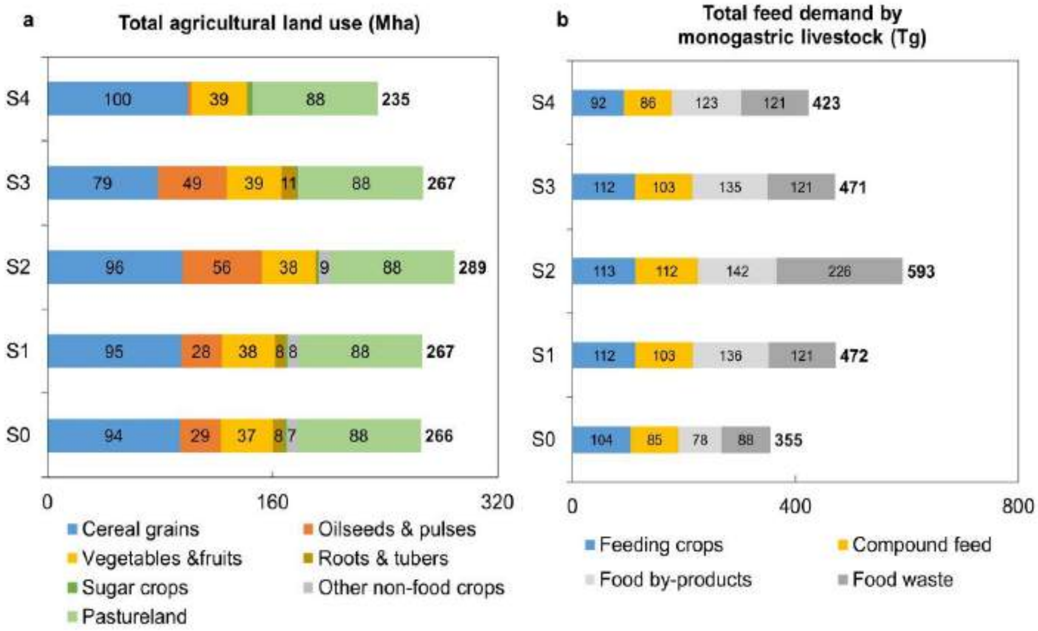

Fig. 3 | Impacts of upcycling low–opportunity–cost feed products (LCFs) in China's monogastric livestock as feed on domestic total agricultural land use and feed demand. (**a**) Total agricultural land use (crop harvested area and pastureland) (Mha) and (**b**) feed demand by monogastric livestock (Tg) in scenarios. Definitions of scenarios (S1 - 'Partial use of LCFs as feed'; S2 - 'Full use of LCFs as feed'; S3 - 'S1 + A modest emission mitigation target'; S4 - 'S1 + An ambitious emission mitigation target') are described in Table 1.

3.2 Asymmetric impacts of upcycling low-opportunity-cost feed as animal feed on global environmental sustainability and food security.

We found that the 23-36% (24-37 Tg) expansion in monogastric livestock production in scenarios S1 and S2 increased Chinese economy-wide emissions of acidification polluants by 2.5-4.0% (0.83-1.36 Tg NH_3-eq) (Fig. 4b), and eutrophication pollutants by ±0.2% (±0.02 Tg N-eq) (Fig. 4c). The 0.5-1.4% (56-163 Tg CO_2-eq) decease in economy-wide GHG emissions was dominated by less LCFs in landfills and incinerators (119-222 Tg CO_2-eq), along with non-food production contraction (98-145 Tg CO_2-eq) (Fig. 4a). China's main food and feed trading partners (MTP, including Brazil, the United States, and Canada) experienced a reduction in economy-wide emissions of GHGs by 1.1-1.3% (85-102 Tg CO_2-eq), acidification pollutants by 8-13% (1.13-1.80 Tg NH_3-eq), and eutrophication pollutants by 2.5-4.0% (0.14-0.22 Tg N-eq). These environmental benefits for MTP arose from a reduction in their domestic livestock and fertiliser production, as China shifted from a net importer to an exporter of livestock products and fertilisers (Fig. 2e,f).

For assessing food security, we used four indicators covering two dimensions. Two indicators for food availability, i.e., dietary calorie availability and the population at risk of hunger. Two indicators for food access, i.e., cereals affordability for labour force and the average food (including primary food products and processed food) price. Our findings suggested that upcycling accompanying with resource reallocation across the whole economy enhance food security in China without compromising that of its trading partners.In addition, the reduced cost of food waste collection for landfill and incineration enabled consumers in China to allocate more of their income to food consumption. Since the cost of food waste collection for landfill and incineration was quite small in the baseline (S0), the impact of reduced collection costs had only a modest positive effect on most food security indicators. Globally, the average food price declined by 0.1-0.2% (Fig. 5a,e). In China, dietary calorie availability increased by 0.16-0.32% (5.2-10.3 kcal capita^{-1} day^{-1}), and the population at risk of hunger, representing 17% of the global population at risk of hunger, decreased by 1.6-3.2% (2.2-4.5 million people) (Fig. 5c,d). Cereals affordability for labour force increased by 0.29-0.47% (Fig. 5b), as a result of a rise in the average wage across the Chinese economy (0.13-0.22%) (Supplementary Fig. 5) and a decrease in cereals price (0.16-0.26%) (Supplementary Fig. 15).

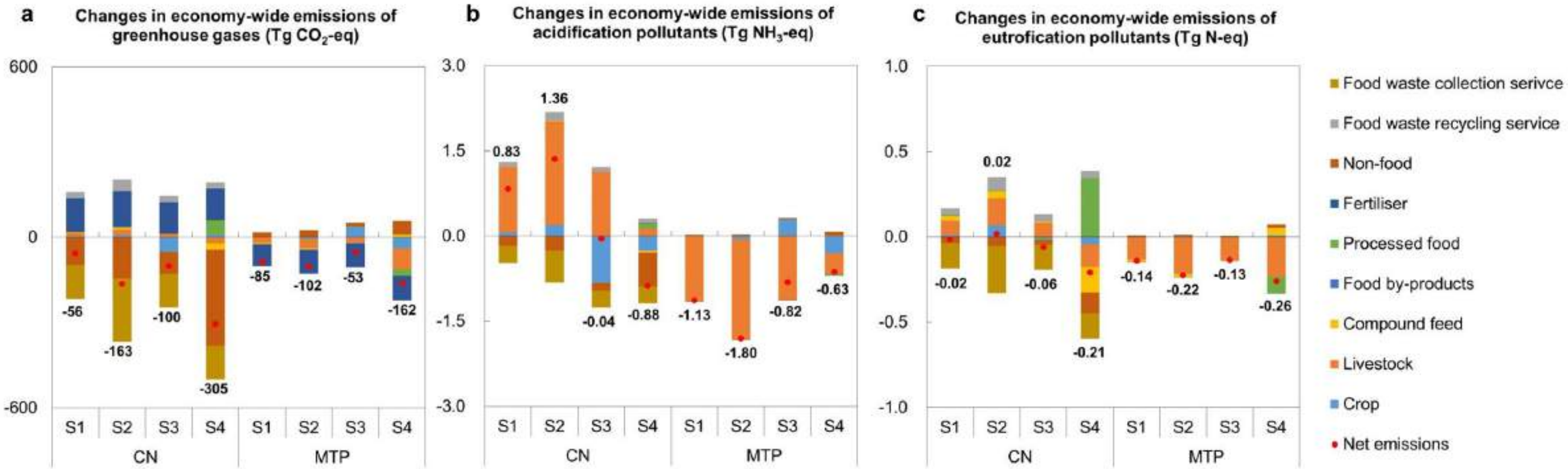

Fig. 4 | Impacts of upcycling low–opportunity–cost feed products (LCFs) in China's monogastric livestock as feed on economy-wide emissions in China (CN) and China's main food and feed trading partners (MTP). Changes in (**a**) economy-wide emissions of greenhouse gases (Tg CO_2-eq), (**b**) acidification pollutants (Tg NH_3-eq), and (**c**) eutrophication pollutants (Tg N-eq) in China and MTP in scenarios with respect to the baseline (S0). MTP includes Brazil, the United States, and Canada. Definitions of scenarios (S1 - 'Partial use of LCFs as feed'; S2 - 'Full use of LCFs as feed'; S3 - 'S1 + A modest emission mitigation target'; S4 - 'S1 + An ambitious emission mitigation target') are described in Table 1.

3.3 Absorbing rebound effects in China through upcycling low-opportunity-cost feed as animal feed and implementing emission taxes.

We assessed the impacts of implementing economy-wide emission taxes to achieve two emission mitigation targets under the partial use of LCFs as animal feed (scenario S1), considering the perishability and collection challenges of food waste, as well as the reduced availability of food waste for feed in accordance with SDG 12.3 ("halving food waste") [14]. Scenario S3 aimed at decreasing emissions of GHGs, acidification pollutants, and eutrophication pollutants in both China and MTP to below baseline (S0) levels. Scenario S4 aimed at achieving China's and MTP's annual GHG mitigation targets under the Intended Nationally Determined Contributions (INDC) of the Paris Agreement [24,25], while also addressing China's emission reduction goals for acidification and eutrophication pollutants in line with the "14th Five-Year Plan" [26].

A modest mitigation target of S3 could absorb the rebound effects of upcycling LCFs as feed in China (Fig. 4) and safeguard global food security. Changes in food security indicators under S3 were nearly identical to those in S1 (Fig. 5). This is due to the implementation of a low tax rate on emissions of acidification pollutants (3 \$ ton^{-1} NH_3-eq) in China. The reduction in emissions of all pollutants in S3 was mainly attributed to a decrease in total crop production compared to S1 (Fig. 2a; Fig 4), which reduced emissions of GHGs by 51 Tg CO_2-eq, acidification pollutants by 0.82 Tg NH_3-eq, and eutrophication pollutants by 0.01 Tg N-eq (Supplementary Fig. 14a,b,c). Livestock production also slightly decreased in scenario S3 (Fig. 2b). However, P fertiliser production increased by 40% (7 Tg) while N fertiliser production decreased by 6% (2 Tg) compared to S1 (Fig. 2c). As a result, emissions increased in MTP compared to S1 (Fig. 4) due to a shift of emission-intensive production from China to MTP. Nonetheless, emissions of all pollutants in MTP still remained below baseline (S0) levels.

An ambitious emission mitigation target of S4 counteracted the rebound effects further and achieved a further emission reduction, but could pose a risk to food security, as the average global food price increased by 9.4% (Fig. 5a,e) and cereals affordability for labour force decreased by 20% in China (Fig. 5b) and by 15% in MTP (Fig. 5f). The negative impact on food security in China and MTP was a result of the higher tax rates on emissions in both regions (5 \$ ton^{-1} CO_2-eq , 788 \$ ton^{-1} NH_3-eq, and 6969 \$ ton^{-1} N-eq in China; 2.5 \$ ton^{-1} CO_2-eq in MTP). Food availability in MTP decreased by 3.3% (108 kcal capita^{-1} day^{-1}), while in China, it increased by 3.6% (116 kcal capita^{-1} day^{-1}) (Fig. 5d,h). The latter was a result of consumers transitioning from ruminant-sourced food to less expensive plant and monogastric-sourced food in China (Supplemntary Fig. 16c). Consequently, the population at risk of hunger in MTP increased by 346% (18.3 million people), but declined in China by 36% (50.4 million people) (Fig. 5 c,g). The 2.6% reduction in total GHG emissions (305 Tg CO_2-eq) and the 2.5% decrease in emissions of acidification pollutants (0.88 Tg NH_3-eq) in China in S4 were largely driven by the non-food production contraction compared to S1 (Fig. 4a,b). The 2.0% reduction in total emissions of eutrophication pollutants (0.21 Tg N-eq) (Fig. 4c) in China was mainly the result of shifting from ruminant to monogastric livestock production (Supplementary Fig. 14f). For MTP, the 2.0% reduction in total GHG emissions (162 Tg CO_2-eq) was largely attributed to reductions in total crop and livestock production (Fig. 4a). Meanwhile, emissions of acidification and eutrophication pollutants decreased both by 5% in MTP (Fig. 4b,c).

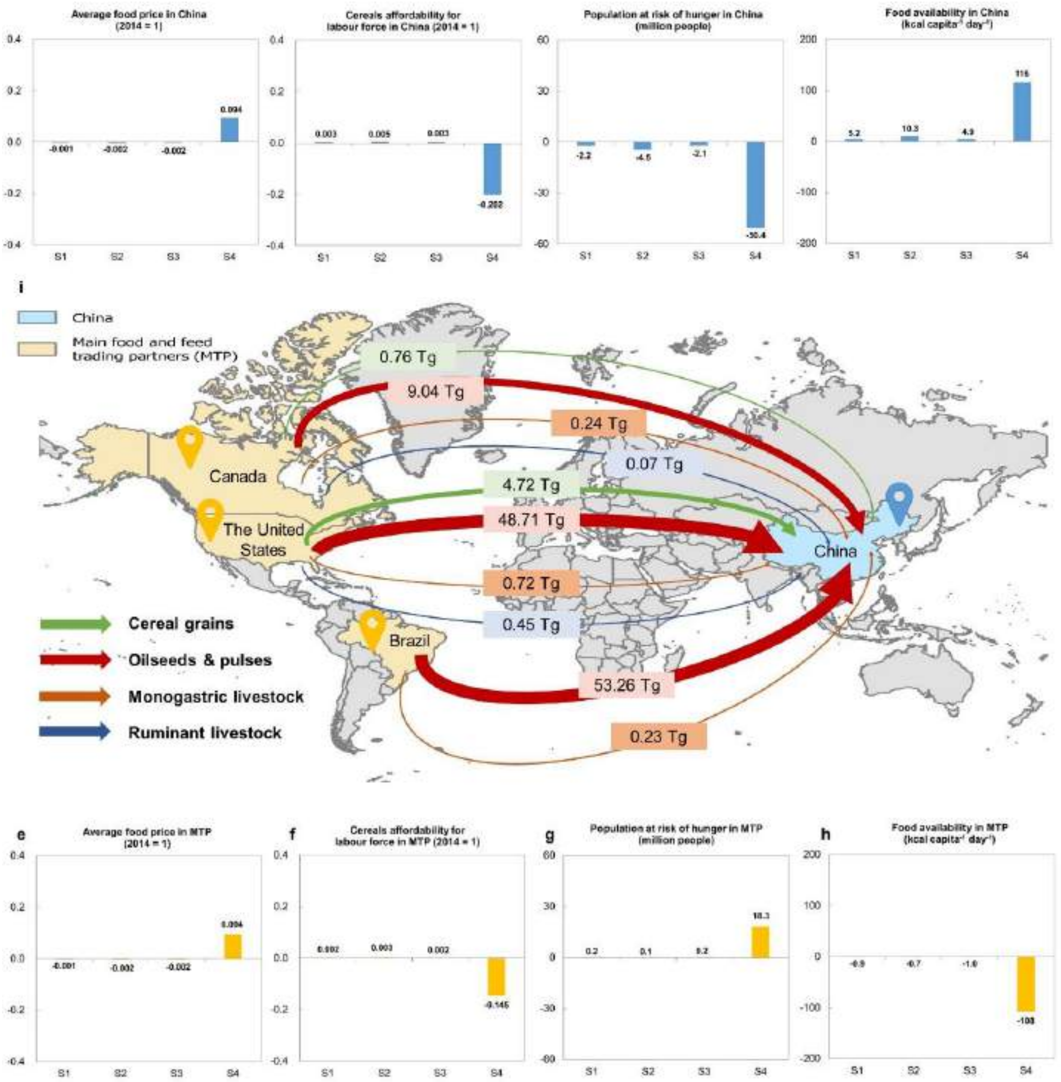

Fig. 5 | Impacts of upcycling low–opportunity–cost feed products (LCFs) in monogastric livestock as feed on food security indicators in China (CN) and China's main food and feed trading partners (MTP). Changes in (**a**) average food (including primary food products and processed food) price, (**b**) cereals affordability for labour force, (**c**) population at risk of hunger (million people; S0 = 140.7 million people), and (**d**) food availability (kcal capita^{-1} day^{-1}) in China in scenarios with respect to the baseline (S0). Changes in (**e**) average food (including primary food products and processed food) price, (**f**) cereals affordability for labour force, (**g**) population at risk of hunger (million people; S0 = 5.3 million people), and (**d**) food availability (kcal capita^{-1} day^{-1}) in MTP in scenarios with respect to the baseline (S0). (**i**) Net imports (Tg) of main food and feed products from MTP to China in the baseline (S0). MTP includes Brazil, the United States, and Canada. According to the FAO approach, it is assumed that there is no risk of hunger for high-income countries; consequently, the population at risk of hunger is not applied to the United States and Canada [21,59,60]. Definitions of scenarios (S1 - 'Partial use of LCFs as feed'; S2 - 'Full use of LCFs as feed'; S3 - 'S1 + A modest emission mitigation target'; S4 - 'S1 + An ambitious emission mitigation target') are described in Table 1. Credit: World Countries base map, Esri (https://hub.arcgis.com/datasets/esri::world-countries/about).

4. Discussion

In this study, we explored the possible environmental and economic consequences of upcycling LCFs in China's monogastric livestock production in a global context, and provided possible solutions to absorb the rebound effects in China and safeguard global food security. Our study serves as a step towards bridging monetary AGE models with biophysical and nutritional (e.g. protein and energy) constraints. Our integrated environmental-economic framework complements previous linear optimisation studies [11-13], which overlooked market-mediated responses via the price system by considering both direct and indirect (price-induced) effects of upcycling LCFs as feed. In contrast to previous linear optimisation studies that assume livestock production remains unchanged as long as feed protein and energy are maintained, our modelling framework enables us to capture the indirect "rebound effect" of livestock production expansion induced by lower feed costs and its knock-on effects on other commodities, which may undermine the expected benefits of reducing environmental impacts in the transition to more circular food systems. Furthermore, changes in China's food production structure also had cross-border impacts on its trading partners through international trade.

4.1 The feasibility of upcycling low-opportunity-cost feed as animal feed in China

While upcycling food waste as feed has been shown not to affect livestock productivity [9], to gain acceptance and adoption among livestock producers, food waste protein production must demonstrate its economic competitiveness against conventional feed proteins such as cereals and oilseeds. Upcycling full use of food waste as feed necessitates various investments and policies to support the construction of municipal food waste collection plants to efficiently collet, sanitize, and package food waste for sale to livestock producers as feed [12]. Achieving near-full use of food waste as feed appears feasible in China in the future due to several reasons. The food waste treatment industry (i.e., food waste collection service and food waste recycling service) has seen significant development and expansion in recent years [66]. Reinforced policies on municipal solid waste separation and collection guarantee a stable feed supply for monogastric livestock production [67]. For example, the Chinese government recently launched an action plan to reduce reliance on soybean imports, which includes a key initiative to trial feed production from food waste in 20 cities by 2025

[68]. Additionally, the geographic proximity of industrial livestock farms to municipal food waste collection plants further facilitates the feasibility of upcycling food waste as feed for monogastric livestock production [66].

4.2 Rebound effects may undermine benefits of upcycling low-opportunity-cost feed as animal feed in China

Policymakers focused on reducing the environmental impact of food systems and enhancing food security may find our findings particularly informative, as we unveil the asymmetric impacts of upcycling LCFs as feed on food security and environment sustainability. On the one hand, rebound effects, where lower feed costs lead to a 23-36% (24-37 Tg) expansion in monogastric livestock production, diminish the environmental benefits of upcycling LCFs as feed in China. We observed Chinese economy-wide emissions of acidification and eutrophication polluants increased by2.5-4.0% (0.83-1.36 Tg NH_3-eq) and by ±0.2% (±0.02 Tg N-eq) in scenarios S1 and S2. In contracst, the 0.5-1.4% (56-163 Tg CO_2-eq) decease in economy-wide GHG emissions was dominated by less LCFs in landfills and incinerators (119-222 Tg CO_2-eq), along with non-food production contraction (98-145 Tg CO_2-eq). China’s trading partners obtained environmental benefits through reducing their domestic livestock and fertiliser production, as China shifted from a net importer to an exporter of livestock products and fertilisers. On the other hand, this upcycling accompanying with resource reallocation across the whole economy enhance food security in China without compromising that of its trading partners. Our results echo the findings of Hegwood, et al. [19], who argued that rebound effects could offset more than half of avoided food loss and waste, with reductions in environmental benefits and improvements in food security. Our analysis, thus, enhance the understanding of synergies and trade-offs between economic impacts and multiple environmental stresses associated with upcycling LCFs as feed.

4.3 The need for policymakers to consider the interconnection between food security and environmental sustainability

Our study highlights the need to integrate both food security and environmental sustainability into policy decisions to leverage potential win-win opportunities, especially under the current challenges such as climate change and resource constraints. In essence, policymakers should pay closer attention to the interconnection between food security and environmental sustainability to better leverage potential synergies and minimize trade-offs [69]. The reduction in GHG emissions, coupled with the enhancements in food security, underscores the rationale for policymakers to promote upcycling LCFs as feed. This also aligns with China's recent emphasis on carbon neutrality and food security as leading priorities [70,71]. However, policymakers should remain vigilant regarding indirect effects and spillovers, particularly the unintended increases in emissions of acidification and eutrophication pollutants. We implemented two emission mitigation measures to absorb the rebound effects of upcycling LCFs as feed in China. Our findings revealed that an ambitious emission mitigation target (i.e., emission taxes to meet the Paris Agreement goals and the "14th Five-Year Plan") could counteract rebound effects but risk a 9.4% rise in food prices, threatening global food security. These are confirmed by Hasegawa, et al. [21], who revealed the risk of increased food insecurity under stringent global climate change mitigation policy. Conversely, a modest emission mitigation target (i.e., emission taxes to maintain baseline levels) provides an opportunity to absorb the rebound effects in China and safeguard global food security. Therefore, to avoid unintended negative environmental impacts and achieve the dual dividend of environmental sustainability and food security, it is essential to carefully design and implement tailored, complementary policies and measures rather than relying on a single, one-size-fits-all solution. In China, the responsibility for food security and environmental sustainability often falls to different government agencies, highlighting the pressing need for improved coordination and consistency within the government to effectively tackle these intertwined issues [72]. In addition, a globally coordinated mitigation policy is imperative for respecting the exceedance of the planetary boundaries, as the unilateral environmental policy can lead to 'carbon leakage' by outsourcing the production of emission-intensive goods to countries with lack environmental regulations [39].

Despite the integrated and holistic approach, our study has some limitations that necessitate some follow-up, which are discussed in Supplementary Discussion. While further research is needed, our study provides a starting point by offering an integrated environmental-economic framework to supports policy design aimed at achieving the dual dividend of environmental sustainability and food security. Our analysis holds significant policy implications not only for China, a key global market for food and feed, but also serves as a blueprint for other populous emerging economies striving to achieve a better balance between food security and environmental sustainability with limited agricultural land and growing food demand, thereby resulting in a notable global impact.

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SUPPLEMENTARY INFORMATION

Quantifying the environmental and economic impacts of upcycling low-opportunity-cost feed as animal feed in China: a general equilibrium approach

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Contents

Mathematically, various ways exist to represent applied general equilibrium (AGE) models, according to Ginsburgh and Keyzer [1]. To identify the optimal solution towards greater sustainability and enable the efficient allocation of resources in the economy, we used the welfare format of the AGE models for our analysis. In the supplementary information, we specified the model for our study by explicitly considering producers, consumers, production goods, consumption goods, and intermediate goods. Subsequently, we presented the calibration of our model. Finally, we provided supplementary figures and tables, along with the sectoral aggregation scheme, social accounting matrices, and emissions data for all the regions in our study.

Supplementary Methods

Objective function

The objective function "social welfare (W)" is the weighted sum of the log utility (U_i) of all consumers, according to Zhu and Van Ierland [2].

$$\mathrm{W} = \max \sum_i \alpha_i \log U_i \tag{1}$$

where α_i is the Negishi weight of the representative consumer in each region i (i=China and its main food and feed trading partners (MTP, including Brazil, United States, and Canada)).

Utility function

In our model, the consumer's utility depends on the consumption of rival goods. The utility function is a Cobb-Douglas (C-D) function describing the behaviour of a representative consumer (household to maximise its utility subject to budget constraints) consuming rival goods. The utility function of the consumer in region i is written as:

$$U_i = \prod_s C_{i,s}^{\beta_{i,s}} \tag{2}$$

where consumption goods s refers to cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, sugar crops, other non-food crops, monogastric livestock, ruminant livestock, processed food, and non-food. $C_{i,s}$ is the consumption of the rival good in region i. $\beta_{i,s}$ is the elasticity of utility concerning the consumption of rival good s in region i, i.e., the expenditure share of consumption good s in consumption of rival goods in region i, and $\sum_s \beta_{i,s} = 1$.

Production function

We present the production functions of seventeen producers, namely, cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, sugar crops, other non-food crops, monogastric livestock, ruminant livestock, compound feed, cereal brans, alcoholic pulps, oil cakes, processed food, nitrogen fertiliser, phosphorus fertiliser, and non-food.

The production function of producer j in region i is specified as:

$$Y_{i,j} = A_{i,j}[(KL_{i,j})^{\eta_{1i,j}}(LB_{i,j})^{\eta_{2i,j}}(LD1_{i,j})^{\eta_{3i,j}}(LD2_{i,j})^{\eta_{4i,j}}(NFE_{i,j})^{\eta_{5i,,j}}(PFE_{i,j})^{\eta_{6i,j}}$$

$$(CER_{i,j})^{\eta_{7i,j}}(OSD_{i,j})^{\eta_{8i,j}}(VF_{i,j})^{\eta_{9i,j}}(RT_{i,j})^{\eta_{10i,j}}(SGR_{i,j})^{\eta_{11i,j}}(OTC_{i,j})^{\eta_{12i,j}}$$

$$(COF_{i,j})^{\eta_{13i,j}}(BRAN_{i,j})^{\eta_{14i,j}}(PULP_{i,j})^{\eta_{15i,j}}(CAKE_{i,j})^{\eta_{16i,j}}]^{1-\xi_{i,j}}$$

$$[\left(CERW_{i,j}\right)^{\delta_{1i,j}}\left(OSDW_{i,j}\right)^{\delta_{2i,j}}\left(VFW_{i,j}\right)^{\delta_{3i,j}}\left(RTW_{i,j}\right)^{\delta_{4i,j}}$$

$$\left(BRANW_{i,j}\right)^{\delta_{5i,j}}\left(PULPW_{i,j}\right)^{\delta_{6i,j}}\left(CAKEW_{i,j}\right)^{\delta_{7i,j}}]^{\xi_{i,j}} \tag{3}$$

where $Y_{i,j}$ is the production of sector j in region i. $A_{i,j}$ is the technological parameter of the production of sector j in region i. $KL_{i,j}$, $LB_{i,j}$, $LD1_{i,j}$ and $LD2_{i,j}$ are capital, labour, cropland, and pasture land inputs for production j in region i, respectively. $NFE_{i,j}$, $PFE_{i,j}$, $CER_{i,j}$, $OSD_{i,j}$, $VF_{i,j}$, $RT_{i,j}$, $SGR_{i,j}$, $OTC_{i,j}$, $COF_{i,j}$, $BRAN_{i,j}$, $PULP_{i,j}$, and $CAKE_{i,j}$ are nitrogen fertiliser, phosphorus fertiliser, cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, sugar crops, other non-food crops, compound feed, cereal bran, alcoholic pulp, and oil cake inputs for the production of sector j in region i, respectively. $CERW_{i,j}$, $OSDW_{i,j}$, $VFW_{i,j}$, $RTW_{i,j}$, $BRANW_{i,j}$, $PULPW_{i,j}$, and $CAKEW_{i,j}$ are food waste (i.e., cereal grains waste, oilseeds & pulses waste, vegetables & fruits waste, roots & tubers waste, cereal bran waste, alcoholic pup waste, and oil cake waste) recycling service as feed input for the production of sector j in region i, respectively. $\xi_{i,j}$ ($0<\xi_{i,j}<1$) is the cost share of food waste for the production of sector j in region i. η_f (f=1, 2, 3, …, 16) is the cost share of each factor and intermediate input for production, and $\sum_{f=1}^{16} \eta_f = 1$. δ_f (f=1, 2, 3, …, 7) is the cost share of each food waste input for production, and $\sum_{f=1}^{7} \delta_f = 1$.

We also add several additional constraints on the production of crops (i.e., cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, sugar crops, other non-food crops), livestock (i.e., monogastric livestock, ruminant livestock), and food processing by-products (i.e., cereal brans, alcoholic pulps, oil cakes) based on the information from the social accounting matrices (SAM) (see Appendix Tables 2-3) in the base year of 2014 for China and its trading partners.

Crops can't be produced in a 'factory-like' setting because the chemical processes within plants require specific nutrients that can't be substituted for one another. Different combinations of nutrients, such as nitrogen (N) and phosphorus (P_2O_5), lead to varying crop yields. Thus, we kept the total output of crop as a fixed ratio of nitrogen and phosphorus fertiliser inputs. In other words, the ratio of nitrogen and phosphorus fertiliser inputs for per unit of crop output remained constant across all scenarios. Since livestock productivity is directly tied to the protein and energy levels of feed, the total output of livestock is a fixed ratio of feed inputs. When substituting primary feed (i.e., human-edible feed crops and compound feed) with food waste and food processing by-products, we maintained the protein and energy feed supply for per unit of animal output in all scenarios to prevent imbalances between nutritional (protein and energy) supply and livestock requirements. Since food processing by-products are calculated based on the consumption of food products and specific technical conversion factors, we maintained a constant ratio of by-product output to the consumption of corresponding food products across all scenarios.

When emissions are outputs of the production process, the emissions intensities of greenhouse gases (GHGs) ($\varepsilon_{gg,i,j}$, kg CO_2 equivalent USD^{-1}), acidification pollutants ($\varepsilon_{ga,i,j}$, kg NH_3 equivalent USD^{-1}), and eutrophication pollutants (EP, $\varepsilon_{ge,i,j}$, kg N equivalent USD^{-1}) from producer j in region i are calculated as:

$$\varepsilon_{gg,i,j} = \frac{EM^{+0}_{gg,i,j}}{Y^0_{i,j}} \tag{4}$$

$$\varepsilon_{ga,i,j} = \frac{EM^{+0}_{ga,i,j}}{Y^0_{i,j}} \tag{5}$$

$$\varepsilon_{ge,i,j} = \frac{EM^{+0}_{ge,i,j}}{Y^0_{i,j}} \tag{6}$$

where $EM^{+0}_{gg,i,j}$ is the emissions of GHGs gg (gg=CO_2, CH_4, and N_2O emissions) from producer j in region i in the base run. $EM^{+0}_{ga,i,j}$ is the emissions of acidification pollutants ga (ga=NH_3, NO_x, and SO_2 emissions) from producer j in region i in the base run. $EM^{+0}_{ge,i,j}$ is the emissions of eutrophication pollutants ge (ge= N and P losses) from producer j in region i in the base run. $Y^0_{i,j}$ is the production of producer j in region i in the base run.

Next, the emissions in different scenarios are calculated by multiplying the current production level by corresponding emission intensities. The total emissions of GHGs, acidification and eutrophication pollutants from all producers in region i are calculated as follows:

$$EMG^{+}_{i,j} = \sum_{gg} \varepsilon_{gg,i,j} * Y_{i,j} * Eqv_{gg}$$

for emissions of GHGs gg = CO_2, CH_4, and N_2O emissions

(7)

$$EMA^{+}_{i,j} = \sum_{ga} \varepsilon_{ga,i,j} * Y_{i,j} * Eqv_{ga}$$

for emissions of acidification pollutants ga = NH_3, NO_x, and SO_2 emissions

(8)

$$EME^{+}_{i,j} = \sum_{ge} \varepsilon_{ge,i,j} * Y_{i,j} * Eqv_{ge}$$

for emissions of eutrophication pollutants ge = N and P losses

(9)

where $EMG^{+}_{i,j}$, $EMA^{+}_{i,j}$, and $EME^{+}_{i,j}$ are the total emissions of GHGs, acidification and eutrophication pollutants from producer j in region i, respectively. Eqv_{gg}, Eqv_{ga}, and Eqv_{ge} are the GWP, AP, and EP equivalent factors based on Goedkoop, et al. [3].

Balance equations

In our applied model, we consider factor inputs (i.e., capital, labour, and land) to be mobile between different sectors but immobile between China and MTP. Cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, and other non-food crops are used for direct consumption and intermediate use for monogastric livestock, ruminant livestock, compound feed, food processing by-products (i.e., cereal bran, alcoholic pulp, and oil cake), and processed food production. Food processing by-products (i.e., cereal bran, alcoholic pulp, and oil cake) and compound feed are produced for intermediate use for monogastric livestock and ruminant livestock production. Monogastric livestock, ruminant livestock, processed food, and non-food are used for direct consumption. Nitrogen fertiliser and phosphorus fertiliser are used for cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, and other non-food crops production but not for consumption. We note C for consumption, XNET for net export (exports minus imports), and Y for production. Variables with a bar stand for exogenous ones.

International trade is modelled using the assumption of perfect substitutes between domestic and imported goods, adhering to the Heckscher-Ohlin assumption [4]. With this assumption, production will take place in countries with comparative advantages, meaning goods will be produced in the countries that can produce them most efficiently. To prevent a strong specialisation effect under free international trade, which could reduce some goods' production to zero in a certain region, we set a lower bound of 10% of the original production for each sector in our model.

The balance equations for cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, and other non-food crops in region i are as follows:

$$C_{i,cer} + CER_{i,oap} + CER_{i,ctl} + CER_{i,cof} + CER_{i,bran} + CER_{i,pulp} + CER_{i,otf} + XNET_{i,cer} \leq Y_{i,cer} \qquad (p_{i,cer}) \tag{10}$$

$$C_{i,osd} + OSD_{i,oap} + OSD_{i,ctl} + OSD_{i,cof} + OSD_{i,cake} + OSD_{i,otf} + XNET_{i,osd} \leq Y_{i,osd} \qquad (p_{i,osd}) \tag{11}$$

$$C_{i,vf} + VF_{i,oap} + VF_{i,ctl} + VF_{i,cof} + VF_{i,otf} + XNET_{i,vf} \leq Y_{i,vf} \qquad (p_{i,vf}) \tag{12}$$

$$C_{i,rt} + RT_{i,oap} + RT_{i,ctl} + RT_{i,cof} + RT_{i,otf} + XNET_{i,rt} \leq Y_{i,rt} \qquad (p_{i,rt}) \tag{13}$$

$$C_{i,sgr} + SGR_{i,oap} + SGR_{i,ctl} + SGR_{i,cof} + SGR_{i,otf} + XNET_{i,sgr} \leq Y_{i,sgr} \quad (p_{i,sgr}) \tag{14}$$

$$C_{i,ocr} + OCR_{i,oap} + OCR_{i,ctl} + OCR_{i,cof} + OCR_{i,otf} + XNET_{i,vf} \leq Y_{i,ocr} \quad (p_{i,ocr}) \tag{15}$$

where $CER_{i,oap}$, $CER_{i,ctl}$, $CER_{i,cof}$, $CER_{i,bran}$, $CER_{i,pulp}$, and $CER_{i,otf}$ are cereals used for monogastric livestock, ruminant livestock, compound feed, cereal bran, alcoholic pulp, and processed food production in region i, respectively. $OSD_{i,oap}$, $OSD_{i,ctl}$, $OSD_{i,cof}$, $OSD_{i,bran}$, and $OSD_{i,otf}$ are cereals used for monogastric livestock, ruminant livestock, compound feed, oil cake, and processed food production in region i, respectively. $VF_{i,oap}$, $VF_{i,ctl}$, $VF_{i,cof}$, and $VF_{i,otf}$ are vegetables & fruits used for monogastric livestock, ruminant livestock, compound feed, and processed food production in region i, respectively. $RT_{i,oap}$, $RT_{i,ctl}$, $RT_{i,cof}$, and $RT_{i,otf}$ are roots & tubers used for monogastric livestock, ruminant livestock, compound feed, and processed food production in region i, respectively. $SGR_{i,oap}$, $SGR_{i,ctl}$, $SGR_{i,cof}$, and $SGR_{i,otf}$ are sugar crops used for monogastric livestock, ruminant livestock, compound feed, and processed food production in region i, respectively. $OCR_{i,oap}$, $OCR_{i,ctl}$, $OTC_{i,cof}$, and $OTC_{i,otf}$ are other non-food crops used for monogastric livestock, ruminant livestock, compound feed, and processed food production in region i, respectively. $p_{i,cer}$, $p_{i,osd}$, $p_{i,vf}$, $p_{i,rt}$, $p_{i,sgr}$, and $p_{i,ocr}$ are the shadow prices of cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, and other non-food crops in region i, respectively.

The balance equation for food processing by-products (i.e., cereal bran, alcoholic pulp, and oil cake) in region i is as follows:

$$BRAN_{i,oap} + XNET_{i,bran} \leq Y_{i,bran} \qquad (p_{i,bran}) \tag{16}$$

$$PULP_{i,oap} + XNET_{i,pulp} \leq Y_{i,pulp} \qquad (p_{i,pulp}) \tag{17}$$

$$CAKE_{i,oap} + XNET_{i,cake} \leq Y_{i,cake} \qquad (p_{i,cake}) \tag{18}$$

where $BRAN_{i,oap}$, $PULP_{i,oap}$, and $CAKE_{i,oap}$ are cereal bran, alcoholic pulp, and oil cake used for monogastric livestock production in region i, respectively. $p_{i,bran}$, $p_{i,pulp}$, and $p_{i,cake}$ are the shadow prices of cereal bran, alcoholic pulp, and oil cake in region i.

The balance equation for compound feed in region i is as follows:

$$COF_{i,oap} + COF_{i,ctl} + XNET_{i,cof} \leq Y_{i,cof} \qquad (p_{i,cof}) \tag{19}$$

where $COF_{i,oap}$ and $COF_{i,ctl}$ are compound feed used in monogastric livestock and ruminant livestock production in region i, respectively. $p_{i,cof}$ is the shadow price of compound feed in region i.

The balance equation for monogastric livestock, ruminant livestock, processed food, and non-food in region i is as follows:

$$C_{i,j} + XNET_{i,j} \leq Y_{i,j} \qquad (p_{i,j}) \tag{20}$$

where $p_{i,j}$ is the shadow price of good j in region i.

The balance equation for nitrogen and phosphorus fertiliser in region i is as follows:

$$NFE_{i,cer} + NFE_{i,osd} + NFE_{i,vf} + NFE_{i,rt} + NFE_{i,sgr} + NFE_{i,ocr} + XNET_{i,nfe} \leq Y_{i,nfe} \qquad (p_{i,nfe}) \tag{21}$$

$$PFE_{i,cer} + PFE_{i,osd} + PFE_{i,vf} + PFE_{i,rt} + PFE_{i,sgr} + PFE_{i,ocr} + XNET_{i,pfe} \leq Y_{i,pfe} \qquad (p_{i,pfe}) \tag{22}$$

where $NFE_{i,cer}$, $NFE_{i,osd}$, $NFE_{i,vf}$, $NFE_{i,rt}$, $NFE_{i,sgr}$ and $NFE_{i,ocr}$ are the nitrogen fertiliser used for cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, and other non-food crops production in region i, respectively. $PFE_{i,cer}$, $PFE_{i,osd}$, $PFE_{i,vf}$, $PFE_{i,rt}$, $PFE_{i,sgr}$ and $PFE_{i,ocr}$ are the phosphorus fertiliser used for cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, and other non-food crops production in region i, respectively. $p_{i,nfe}$ and $p_{i,pfe}$ are the shadow prices of nitrogen fertiliser and phosphorus fertiliser in region i, respectively.

For trade balance of all goods:

$$\sum_i XNET_{i,j} = 0 \qquad (p_j) \tag{23}$$

In the applied model, we assume that factor endowments (i.e., capital, labour, cropland, and pasture land) are mobile between different sectors but immobile among the two regions. For the balance equations of production factor inputs:

$$\sum_j KL_{i,j} \leq \overline{KL_i} \qquad (r_i) \tag{24}$$

$$\sum_j LB_{i,j} \leq \overline{LB_i} \qquad (w_i) \tag{25}$$

$$\sum_j LD1_{i,j} \leq \overline{LD1_i} \qquad (k1_i)$$

for sector j = cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, and other non-food crops

(26)

$$\sum_j LD2_{i,j} \leq \overline{LD2_i} \qquad (k2_i)$$

for sector j = ruminant livestock

(27)

where $\overline{KL_i}$, $\overline{LB_i}$, $\overline{LD1_i}$ and $\overline{LD2_i}$ are the factor endowments (i.e., capital, labour, cropland, pasture land) supply in region i, respectively. r_i, w_i, $k1_i$, and $k2_i$ are the shadow prices of capital, labour, cropland, and pasture land in region i, respectively.

If an emission permit system is implemented to control the total emissions of GHGs, acidification and eutrophication pollutants from all producers, then the following relationship holds:

$$\sum_j EMG_{i,j}^{+} \leq \overline{TMG_i^{+}} \qquad (p_{eg,i}) \tag{28}$$

$$\sum_j EMA_{i,j}^{+} \leq \overline{TMA_i^{+}} \qquad (p_{ea,i}) \tag{29}$$

$$\sum_j EME_{i,j}^{+} \leq \overline{TME_i^{+}} \qquad (p_{ee,i}) \tag{30}$$

where TMG_i^{+}, TMA_i^{+}, and TME_i^{+} are the total emissions of GHGs, acidification and eutrophication pollutants from all producers in region i, respectively. $\overline{TMG_i^{+}}$, $\overline{TMA_i^{+}}$, and $\overline{TME_i^{+}}$ are the permitted level of the total emissions of GHGs, acidification and eutrophication pollutants in region i, respectively. Emissions should not be above a certain level for the regeneration of the environment. For benchmarking, the permitted emission level is the total emission level in the base year. For an environmental policy study (scenarios S3-4), the permitted emission level can be an exogenous emission permit determined by the ecological limit. $p_{eg,i}$, $p_{ea,i}$, and $p_{ee,i}$ are the shadow prices of the emissions of GHGs, acidification and eutrophication pollutants in region i, respectively.

Monogastric livestock's total demand for food waste recycling service must be equal to or less than the total supply of food waste recycling service, then the following relationship holds:

$$CERW_{i,oap} \leq \overline{CERW_{i,oap}} \qquad (p_{i,cerw1}) \tag{31}$$

$$OSDW_{i,oap} \leq \overline{OSDW_{i,oap}} \qquad (p_{i,osdw1}) \tag{32}$$

$$VFW_{i,oap} \leq \overline{VFW_{i,oap}} \qquad (p_{i,vfw1}) \tag{33}$$

$$RTW_{i,oap} \leq \overline{RTW_{i,oap}} \qquad (p_{i,rtw1}) \tag{34}$$

$$BRANW_{i,oap} \leq \overline{BRANW_{i,oap}} \qquad (p_{i,branw1}) \tag{35}$$

$$PULPW_{i,oap} \leq \overline{PULPW_{i,oap}} \qquad (p_{i,pulpw1}) \tag{36}$$

$$CAKEW_{i,oap} \leq \overline{CAKEW_{i,oap}} \qquad (p_{i,cakew1}) \tag{37}$$

where $\overline{CERW_{i,oap}}$, $\overline{OSDW_{i,oap}}$, $\overline{VFW_{i,oap}}$, $\overline{RTW_{i,oap}}$, $\overline{BRANW_{i,oap}}$, $\overline{PULPW_{i,oap}}$, and $\overline{CAKEW_{i,oap}}$ are the total supply of food waste (i.e., cereal grains waste, oilseeds & pulses waste, vegetables & fruits waste, roots & tubers waste, cereal bran waste, alcoholic pup waste, and oil cake waste) recycling service. $p_{i,cerw1}$, $p_{i,osdw1}$, $p_{i,vfw1}$, $p_{i,rtw1}$, $p_{i,branw1}$, $p_{i,pulpw1}$, and $p_{i,cakew1}$ are the shadow prices of food waste (i.e., cereal grains waste, oilseeds & pulses waste, vegetables & fruits waste, roots & tubers waste, cereal bran waste, alcoholic pup waste, and oil cake waste) recycling service.

Consumer's total demand for food waste collection service must be equal to or less than the total supply of food waste collection service, then the following relationship holds:

$$C_{i,cerw} \leq \overline{C_{i,cerw}} \qquad (p_{i,cerw2}) \tag{38}$$

$$C_{i,osdw} \le \overline{C_{i,osdw}} \qquad (p_{i,osdw2}) \tag{39}$$
$$C_{i,vfw} \le \overline{C_{i,vfw}} \qquad (p_{i,vfw2}) \tag{40}$$
$$C_{i,rtw} \le \overline{C_{i,rtw}} \qquad (p_{i,rtw2}) \tag{41}$$
$$C_{i,branw} \le \overline{C_{i,branw}} \qquad (p_{i,branw2}) \tag{42}$$
$$C_{i,pulpw} \le \overline{C_{i,pulpw}} \qquad (p_{i,pulpw2}) \tag{43}$$
$$C_{i,cakew} \le \overline{C_{i,cakew}} \qquad (p_{i,cakew2}) \tag{44}$$

where $C_{i,cerw}$, $C_{i,osdw}$, $C_{i,vfw}$, $C_{i,rtw}$, $C_{i,branw}$, $C_{i,pulpw}$, and $C_{i,cakew}$ are the total supply of food waste (i.e., cereal grains waste, oilseeds & pulses waste, vegetables & fruits waste, roots & tubers waste, cereal bran waste, alcoholic pup waste, and oil cake waste) collection service. $p_{i,cerw2}$, $p_{i,osdw2}$, $p_{i,vfw2}$, $p_{i,rtw2}$, $p_{i,branw2}$, $p_{i,pulpw2}$, and $p_{i,cakew2}$ are the shadow prices of food waste (i.e., cereal grains waste, oilseeds & pulses waste, vegetables & fruits waste, roots & tubers waste, cereal bran waste, alcoholic pup waste, and oil cake waste) collection service.

Budget constraint

The budget constraint for a consumer i holds such that the expenditure must be equal to the income:

$$\textstyle\sum_s\left(p_{i,s}C_{i,s}\right) + p_{i,cerw2}C_{i,cerw} + p_{i,osdw2}C_{i,osdw} + p_{i,vfw2}C_{i,vfw} + p_{i,rtw2}C_{i,rtw} + p_{i,branw2}C_{i,branw} + p_{i,pulpw2}C_{i,pulpw} + p_{i,cakew2}C_{i,cakew} = h_i \tag{45}$$

where consumption goods s refers to cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, sugar crops, other non-food crops, monogastric livestock, ruminant livestock, processed food, and non-food. $\sum_s\left(p_{i,s}C_{i,s}\right)$ is the total expenditure on the consumption goods in region i. $p_{i,cerw2}C_{i,cerw}$, $p_{i,osdw2}C_{i,osdw}$, $p_{i,vfw2}C_{i,vfw}$, $p_{i,rtw2}C_{i,rtw}$, $p_{i,branw2}C_{i,branw}$, $p_{i,pulpw2}C_{i,pulpw}$, and $p_{i,cakew2}C_{i,cakew}$ are the payments to the food waste (i.e., cereal grains waste, oilseeds & pulses waste, vegetables & fruits waste, roots & tubers waste, cereal bran waste, alcoholic pup waste, and oil cake waste) collection service in region i. The Negishi weight (α_i) in the welfare function (equation 1) will be chosen such that the budget constraints hold for each representative consumer in region i.

Consumer's income is the sum of the remuneration of initial endowments employed in production and payments to the environmental sector. Given that food waste is either consumed by livestock as feed or consumed by consumers as a cost of collecting food waste from the municipality, we should also include income from food waste treatment. Since goods are tradable, the consumer's income should exclude the export part. Thus, the consumer's income is:

$$h_i = r_i\overline{KL_i} + w_i\overline{LB_i} + k1_i\overline{LD1_i} + k2_i\overline{LD2_i} - \textstyle\sum_j(p_jXNET_{i,j}) + p_{i,cerw1}CERW_{i,oap} + p_{i,osdw1}OSDW_{i,oap} + p_{i,vfw1}VFW_{i,oap} + p_{i,rtw1}RTW_{i,oap} + p_{i,branw1}BRANW_{i,oap} + p_{i,pulpw1}PULPW_{i,oap} + p_{i,cakew1}CAKEW_{i,oap} + p_{i,cerw2}C_{i,cerw} + p_{i,osdw2}C_{i,osdw} + p_{i,vfw2}C_{i,vfw} + p_{i,rtw2}C_{i,rtw} + p_{i,branw2}C_{i,branw} + p_{i,pulpw2}C_{i,pulpw} + p_{i,cakew2}C_{i,cakew} + p_{eg,i}\overline{TMG_i^+} + p_{ea,i}\overline{TMA_i^+} + p_{ee,i}\overline{TME_i^+} \tag{46}$$

where $\sum_j(p_jXNET_{i,j})$ is the income from exports. $p_{i,cerw1}CERW_{i,oap}$, $p_{i,osdw1}OSDW_{i,oap}$, $p_{i,vfw1}VFW_{i,oap}$, $p_{i,rtw1}RTW_{i,oap}$, $p_{i,branw1}BRANW_{i,oap}$, $p_{i,pulpw1}PULPW_{i,oap}$, and $p_{i,cakew1}CAKEW_{i,oap}$ are the income from food waste recycling service in region i. $p_{i,cerw2}C_{i,cerw}$, $p_{i,osdw2}C_{i,osdw}$, $p_{i,vfw2}C_{i,vfw}$, $p_{i,rtw2}C_{i,rtw}$, $p_{i,branw2}C_{i,branw}$, $p_{i,pulpw2}C_{i,pulpw}$, and $p_{i,cakew2}C_{i,cakew}$ are the income from food waste collection service in region i. $p_{eg,i}\overline{TMG_i^+}$, $p_{ea,i}\overline{TMA_i^+}$, and $p_{ee,i}\overline{TME_i^+}$ are the income from selling emission permits of GHGs, acidification and eutrophication pollutants.

The producers' profits are specified as follows:

$$PROF_{i,j} = p_j Y_{i,j} - r_i KL_{i,j} - w_i LB_{i,j} - k1_i LD1_{i,j} - k2_i LD2_{i,j} - p_{cer} CER_{i,j} - p_{osd} OSD_{i,j} - p_{vf} VF_{i,j} - p_{rt} RT_{i,j} - p_{sgr} SGR_{i,j} - p_{ocr} OCR_{i,j} - p_{cof} COF_{i,j} - p_{bran} BRAN_{i,j} - p_{pulp} PULP_{i,j} - p_{cake} CAKE_{i,j} - p_{nfe} NFE_{i,j} - p_{pfe} PFE_{i,j} - p_{i,cerw1} CERW_{i,oap} - p_{i,osdw1} OSDW_{i,oap} - p_{i,vfw1} VFW_{i,oap} - p_{i,rtw1} RTW_{i,oap} - p_{i,branw1} BRANW_{i,oap} - p_{i,pulpw1} PULPW_{i,oap} - p_{i,cakew1} CAKEW_{i,oap} - p_{eg,i} EMG_{i,j}^+ - p_{ea,i} EMA_{i,j}^+ - p_{ee,i} EME_{i,j}^+ \quad (47)$$

Model calibration

As in the literature on AGE models, we followed the Harberger convention [5] to calibrate the model using the base year SAMs. It means that the prices of all goods and factors are set to one, and the quantities of consumption and production goods equal the monetary value of the base year SAMs [6]. We calibrate the parameters in production and utility functions based on the cost shares of inputs in total production output and expenditure shares of consumption goods in total expenditure. In order to calibrate food waste-related parameters and add food waste (i.e., cereal grains waste, oilseeds & pulses waste, vegetables & fruits waste, roots & tubers waste, cereal bran waste, alcoholic pup waste, and oil cake waste) into the SAMs (see Appendix Tables 2-3), our model treats food waste recycling service as feed input for monogastric livestock production (see equation 3), and assumes that consumer buys food waste collection service for consumption (see equation 45).

Definition of scenarios

S0 - Baseline

The baseline (S0) represents the economies of China and MTP in 2014. The total amounts of food waste and food processing by-products and their current use as animal feed and discarded biomass (i.e., landfill and incineration) for China in S0 are presented in Supplementary Tables 4. When substituting primary feed (i.e., feeding crops and compound feed) in animal diets with food waste and food processing by-products, we kept the total protein and total energy supplies for per unit of animal output were kept constant in all scenarios. The cost of increasing the supply of food waste recycling service was modelled as a rising percentage of the initial cost of recycling food waste and food processing by-products as feed (54 dollar ton^{-1}), while the cost of decreasing the supply of food waste collection service was modelled as a declining percentage of the initial cost of collecting food waste and food processing by-products for landfill and incineration (82 dollar ton^{-1}). Physical quantities and prices of food waste recycling service and food waste collection service in China were presented in Supplementary Tables 4-5.

S1 - Partial use of LCFs as feed

Scenario S1 investigated the impacts of upcycling partial LCFs as feed (54% of food waste and 100% of food processing by-products allowed to be used as feed for monogastric livestock). In S1, cross-provincial transportation of food waste was not allowed, which limits the maximum utilisation rate of food waste with high moisture content to 54% in China, according to Fang, et al. [7].

S2 - Full use of LCFs as feed

Scenario S2 analysed the impacts of upcycling full LCFs as feed (100% of food waste and 100% of food processing by-products allowed to be used as feed for monogastric livestock), taking into account economies of scale. In S2, cross-provincial transportation of food waste was allowed in S2. Economies of scale in food waste recycling were considered in S2, where a 1% increase in recycled waste resulted in only a 0.078% rise in recycling costs, indicating that increasing the amount of recycled waste might not necessarily incur additional costs, as reported by Cialani and Mortazavi [8]. This is because, initially, recycling entails high fixed costs, yet as production scales up, marginal costs decrease and then stabilise.

S3 - S1 + A modest emission mitigation target

In S3, the equations below showed that the total emissions of GHGs, acidification and eutrophication pollutants from all sectors j in both China and MTP were no more than their baseline (S0) emission levels.

$$\sum_j EMG_{i,j}^{+} \leq \overline{TMG_i^{+}} \qquad (p_{eg,i}) \tag{48}$$

$$\sum_j EMA_{i,j}^{+} \leq \overline{TMA_i^{+}} \qquad (p_{ea,i}) \tag{49}$$

$$\sum_j EME_{i,j}^{+} \leq \overline{TME_i^{+}} \qquad (p_{ee,i}) \tag{50}$$

S4 - S1 + An ambitious emission mitigation target

In S4, the equations below showed that the total emissions of GHGs, acidification and eutrophication pollutants from all sectors j in both China and MTP were no more than the emission thresholds set by China’s and MTP’s annual GHG mitigation targets under the Intended Nationally Determined Contributions (INDC) of the Paris Agreement [9,10], as well as China’s emission reduction goals for acidification and eutrophication pollutants in line with the “14th Five-Year Plan” [11].

$$\sum_j EMG_{CN,j}^{+} \leq 0.974 * \overline{TMG_i^{+}} \qquad (p_{eg,i}) \tag{51}$$

$$\sum_j EMG_{MTP,j}^{+} \leq 0.98 * \overline{TMG_i^{+}} \qquad (p_{eg,i}) \tag{52}$$

$$\sum_j EMA_{CN,j}^{+} \leq 0.975 * \overline{TMA_i^{+}} \qquad (p_{ea,i}) \tag{53}$$

$$\sum_j EMA_{MTP,j}^{+} \leq \overline{TMA_i^{+}} \qquad (p_{ea,i}) \tag{54}$$

$$\sum_j EME_{CN,j}^{+} \leq 0.98 * \overline{TME_i^{+}} \qquad (p_{ee,i}) \tag{55}$$

$$\sum_j EME_{MTP,j}^{+} \leq \overline{TME_i^{+}} \qquad (p_{ee,i}) \tag{56}$$

Supplementary Results

Results related to crop production

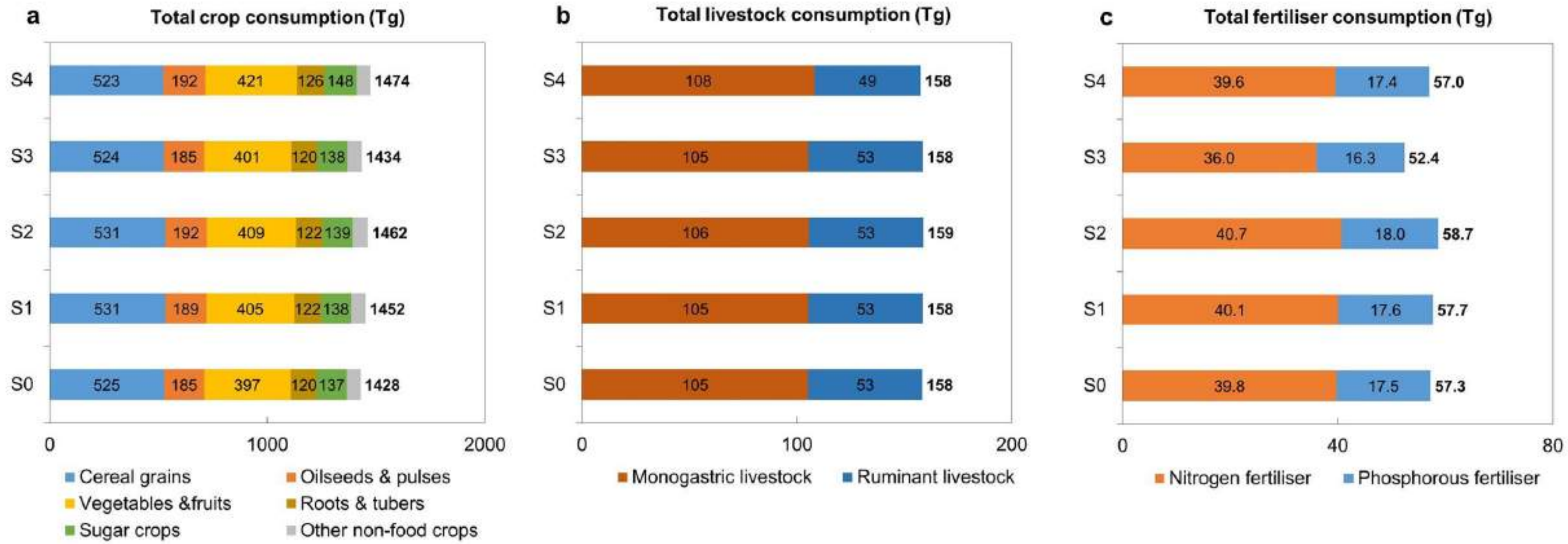
The expansion of monogastric livestock production, a relatively labour-intensive sector, increased labour demand, leading to a 0.13-0.22% rise in average wages across the Chinese economy (Supplementary Fig. 5a). Consequently, labour became comparatively more expensive than other inputs (i.e., capital, cropland, and fertilisers). As cropland and fertilisers became relatively cheaper, crop producers were incentivised to engage in crop extensification and use more cropland and fertilisers to substitute labour. This led to a 0.8-2.3% (0.3-0.9 Tg) increase in total N fertiliser use, a 0.8-2.8% (0.1-0.5 Tg) increase in total P fertiliser use (Supplementary Fig. 4a,b). Crop producers will prioritise reducing the production of relatively labour-intensive crops; for example, roots & tubers and sugar crops decreased by 6-90% (7-108 Tg) and by 15-32% (21-43 Tg) (Supplementary Fig. 6). The saved cropland would then be reallocated to increase the production of cereal grains by 0.8-1.5% (4-8 Tg), vegetables and fruits by 1.7-2.7% (7-11 Tg), and other non-food crops by 8-18% (3-6 Tg) (Supplementary Fig. 6). Notably, the production of oilseeds & pulses decreased by 1.6% (1 Tg) with partial upcycling but increased by 95% (70 Tg) with full upcycling (Supplementary Fig. 6). This variation occurs because oilseeds & pulses are both relatively labour-intensive and cropland-intensive compared to other crops, making their production dependent on the interplay between labour and cropland costs at different levels of upcycling.

Supplementary Discussion

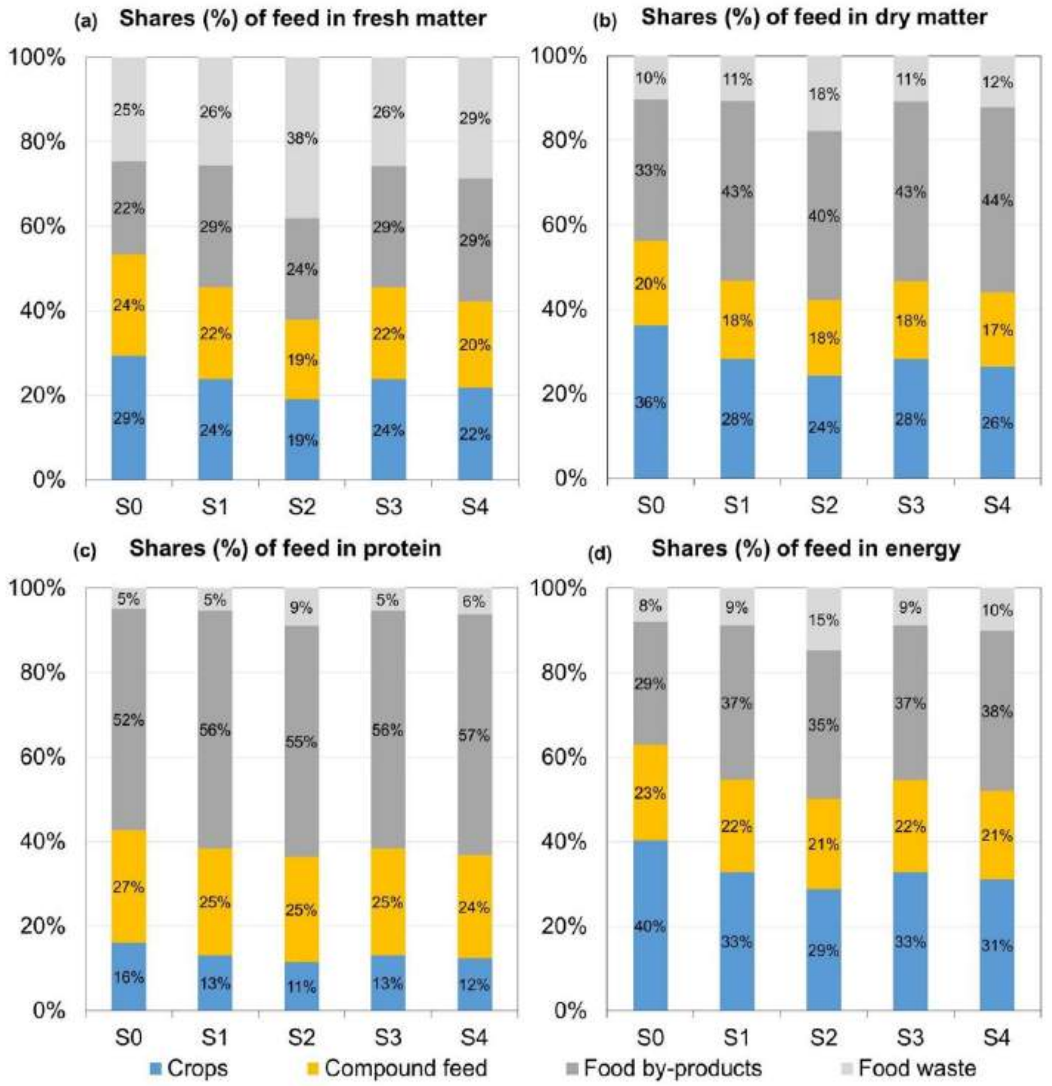
Limitations and future outlook

First, our study assumes free international trade, full mobility of factor endowments (capital, labour, and land) across sectors, and constant income elasticities for all consumption goods. Neglecting trade barriers in our analysis may overestimate the extent of international trade of feed and food. Barriers to the movement of factor endowments across sectors could be included, for example, by introducing separate labour and capital markets for agricultural and non-agricultural sectors or allowing for land shifts within agroecological zones with similar soil, landform, and climatic features, as included in the MAGNET [12] and GTAP-AEZ [13] models. Second, extending our modelling framework to include additional feed types like maize silage, alfalfa hay, and roughage-like by-products would improve the assessment of nutritional balances, particularly in the context of ruminant livestock production. Since these feeds are primarily used for ruminant livestock, which is not our main focus, this falls outside the scope of our study. Third, our analysis concentrates on scenarios outlining technically and physically possible options and does not endeavour to depict policy instruments for achieving the goal of increased utilisation of LCFs as feed, aligning with previous literature on feeding animals with LCFs [7,14-16]. How to design and implement policies that can achieve the goal of increased utilisation of LCFs as feed and implementation of emission taxes should be a pivotal direction for future research. Fourth, in line with SDG 12.3 ("halving food waste") [17], high priority should be placed on reducing food waste. With less food waste available for animal feed, the impacts of upcycling food waste as feed may diminish. However, we consider our estimates of the impacts of upcycling food waste as feed as conservative, as we did not factor in cross-provincial transportation of food waste with high moisture content (except in scenario S2). Last but not least, health impacts resulting from changes in food consumption, such as diet- and weight-related risks [18], could also be considered.

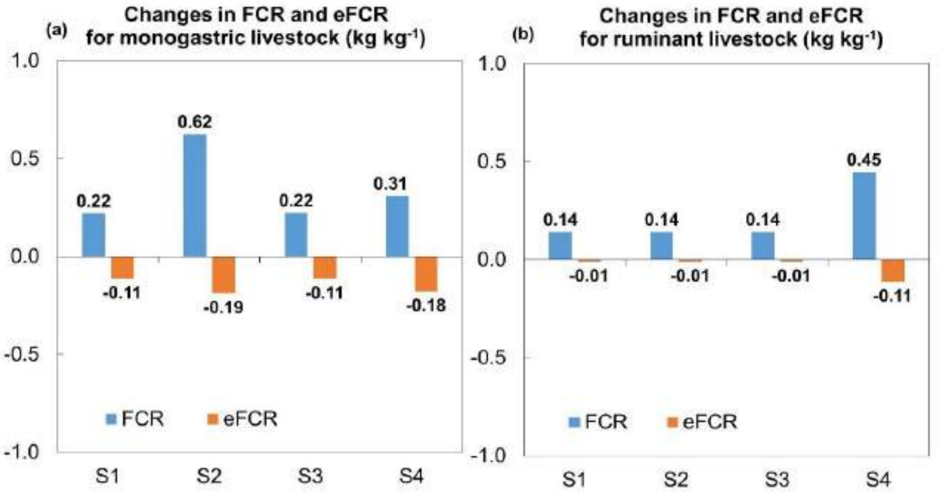
Supplementary Figures



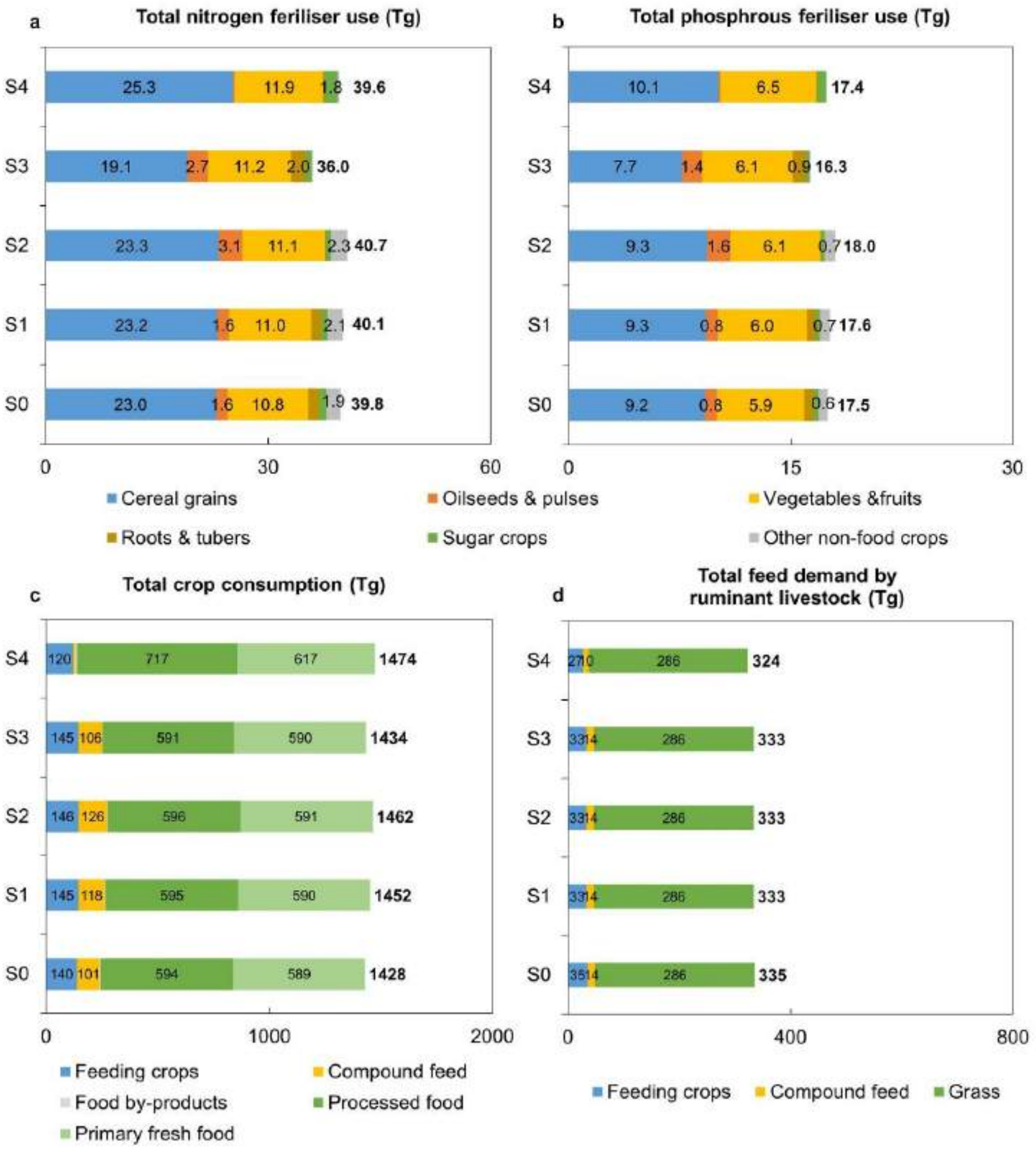
Supplementary Fig. 1 | Total (a) crop, (b) livestock, and (c) fertiliser consumption (Tg) in scenarios. Total crop consumption exclude food waste and food processing by-products used by “food waste recycling service” and “food waste collection service” sectors (see Supplementary Table 4 for detailed data). Total crop consumption includes crop used for intermediate use (i.e, feeding crops, compound feed, food by-products, processed food) and direct consumption (i.e., primary fresh food).



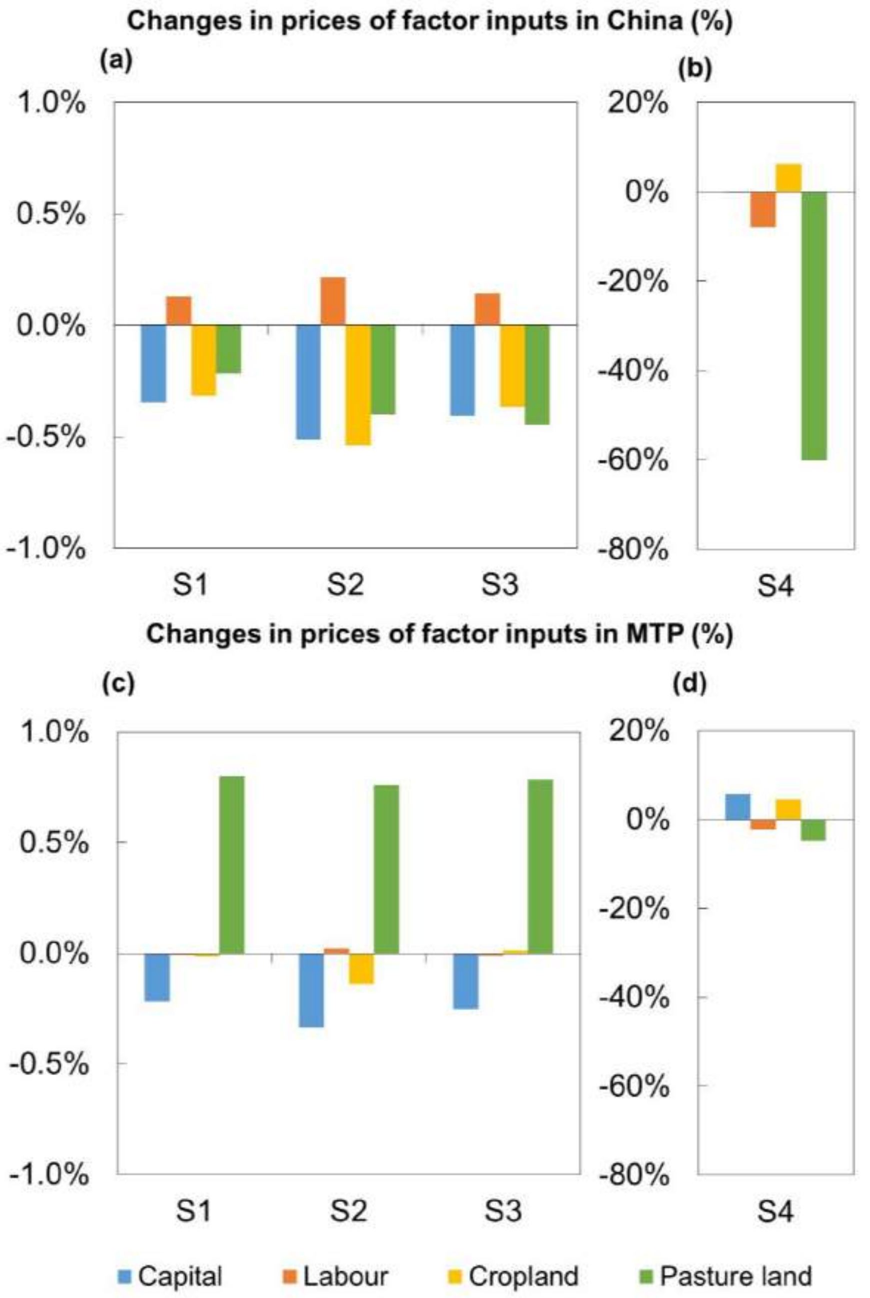
Supplementary Fig. 2 | Shares (%) of each type of feed within the total feed use for monogastric livestock production, categorized by (a) fresh matter, (b) dry matter, (c) protein, and (d) energy in China in scenarios.



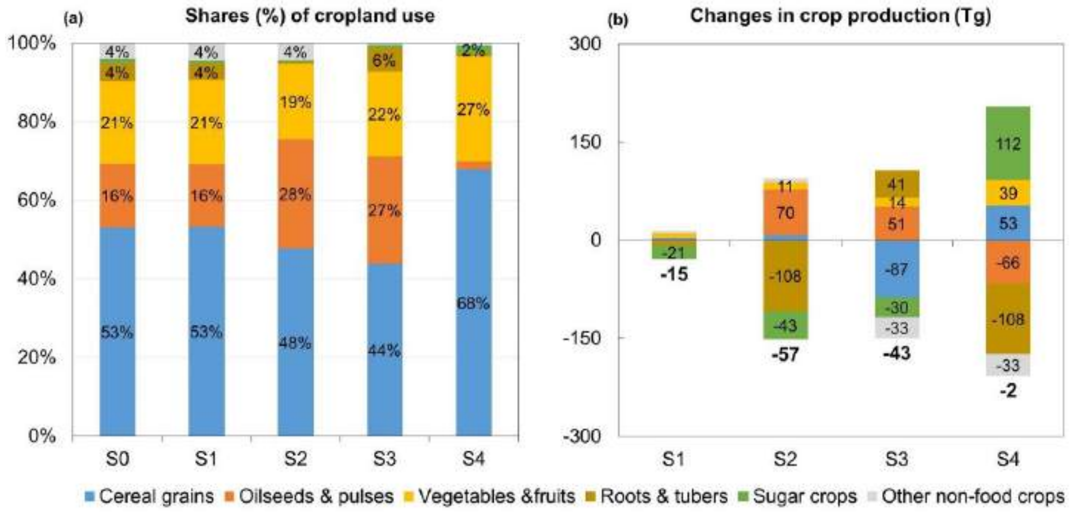
Supplementary Fig. 3 | Changes in FCR (kg kg^{-1}) and eFCR (kg kg^{-1}) for (a) monogastric livestock and (b) ruminant livestock production in China in scenarios with respect to the baseline (S0).



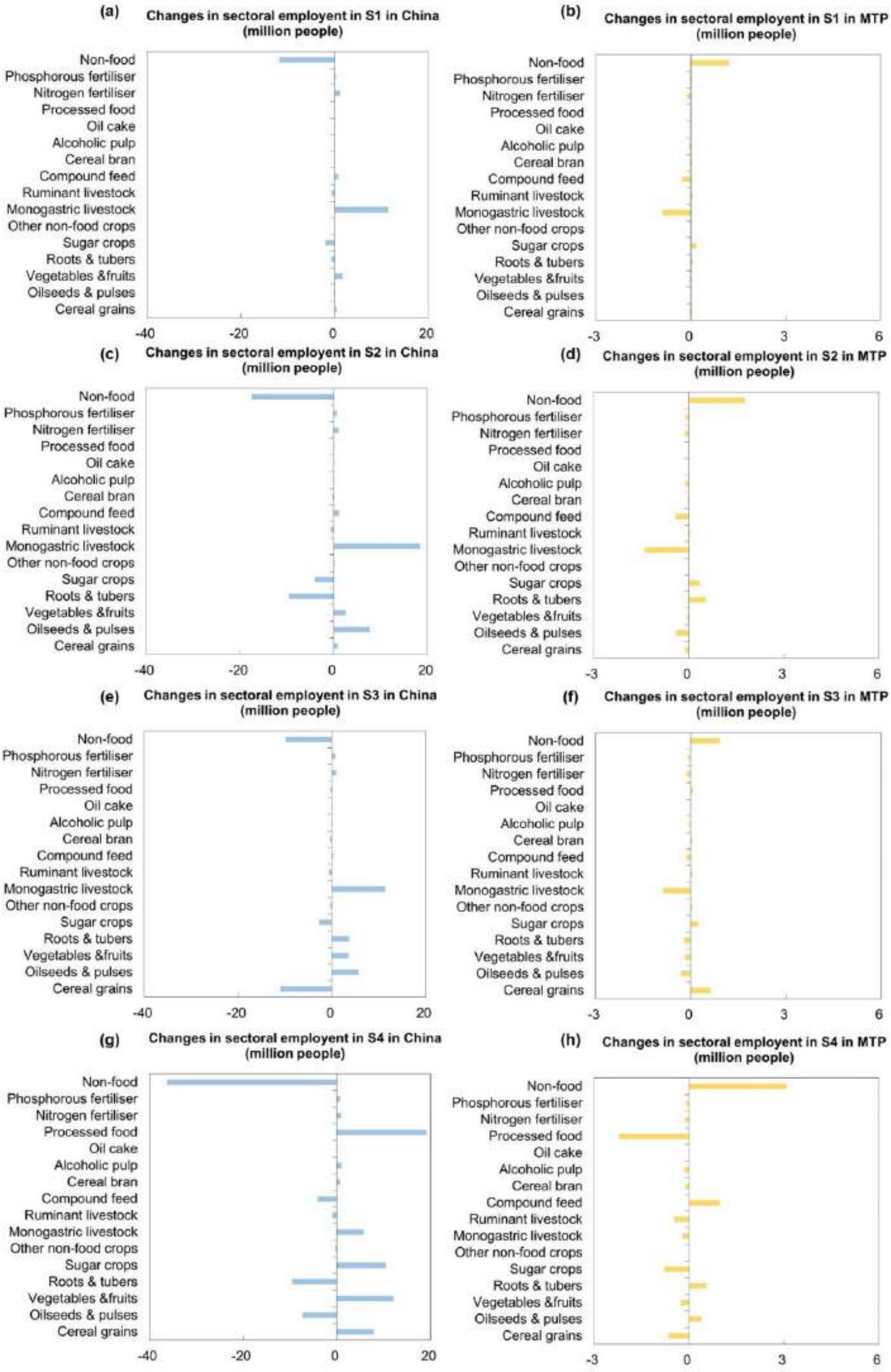
Supplementary Fig. 4 | (a) Total nitrogen fertiliser use (Tg), (b) phosphorous fertiliser use (Tg), (c) crop consumption (Tg), and (d) feed demand by ruminant livestock (Tg) in scenarios..



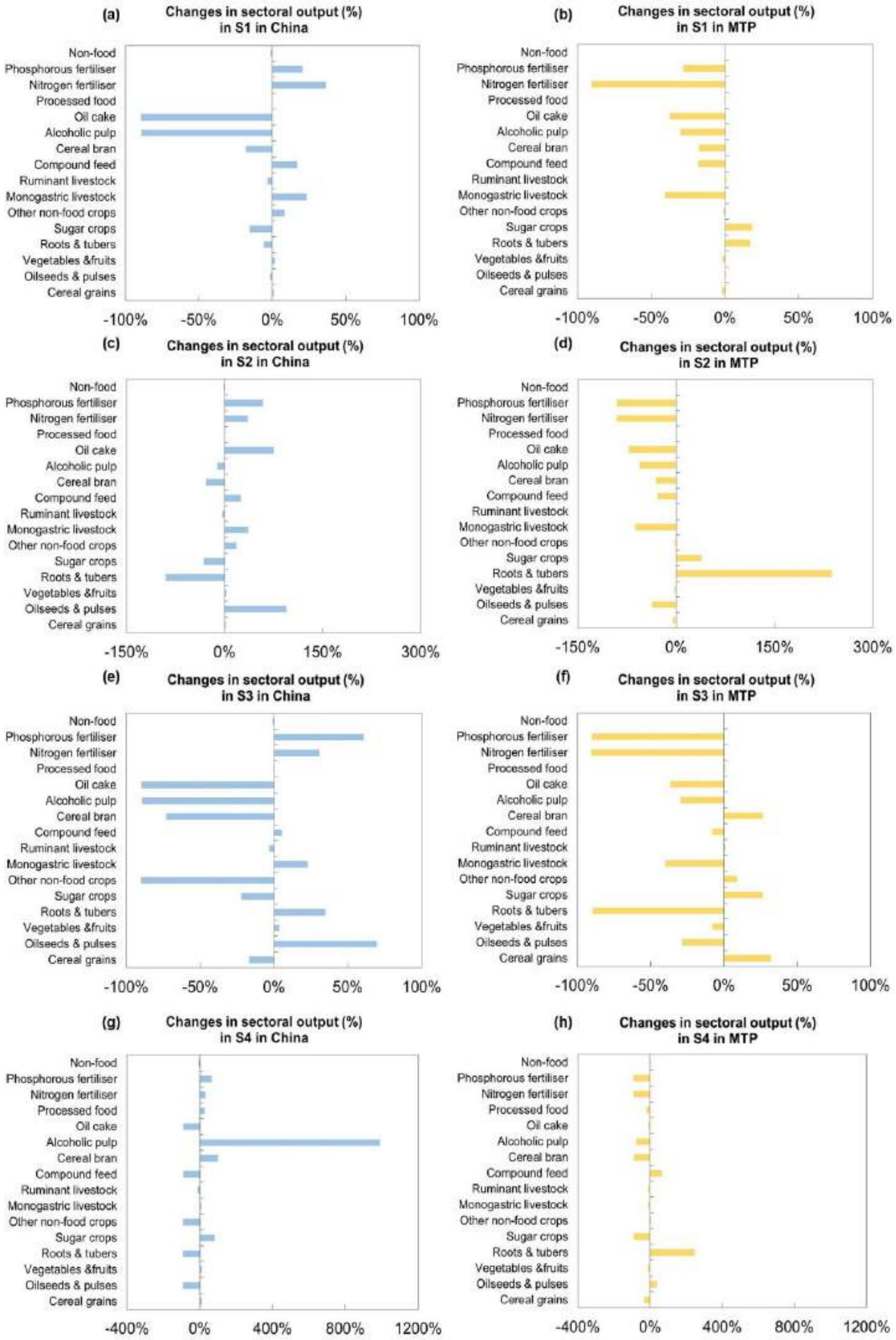
Supplementary Fig. 5 | Changes (%) in prices of factor inputs in China in scenarios (a) S1-3 and (b) S4 with respect to the baseline (S0). Changes (%) in prices of factor inputs in MTP in scenarios (c) S1-3 and (d) S4 with respect to the baseline (S0).



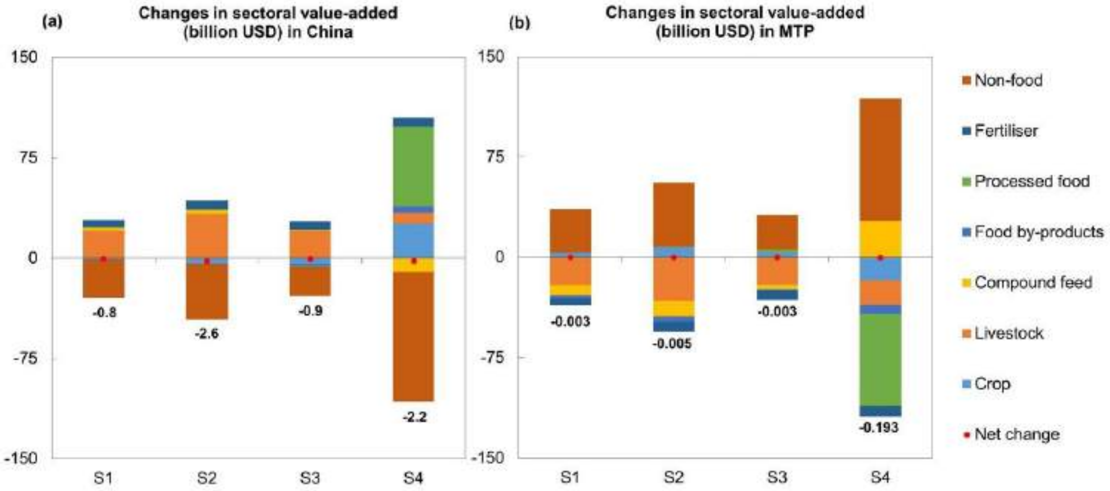
Supplementary Fig. 6 | (a) Shares (%) of each type of crop within the total cropland use in China in scenarios. (b) Changes (Tg) in crop production in China in scenarios with respect to the baseline (S0).



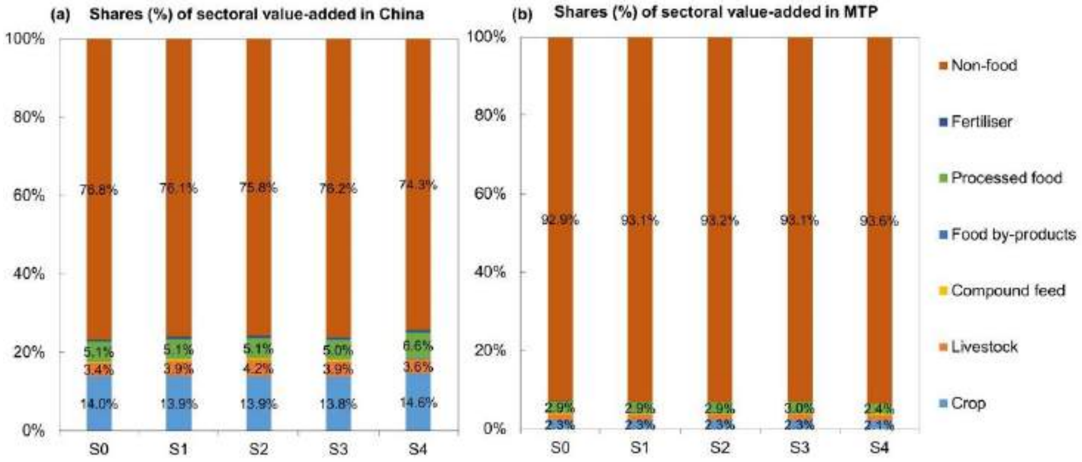
Supplementary Fig. 7 | Changes (million people) in sectoral employment in China in scenarios (a) S1, (c) S2, (e) S3, and (g) S4 with respect to the baseline (S0). Changes (million people) in sectoral employment in MTP in scenarios (b) S1, (d) S2, (f) S3, and (h) S4 with respect to the baseline (S0).



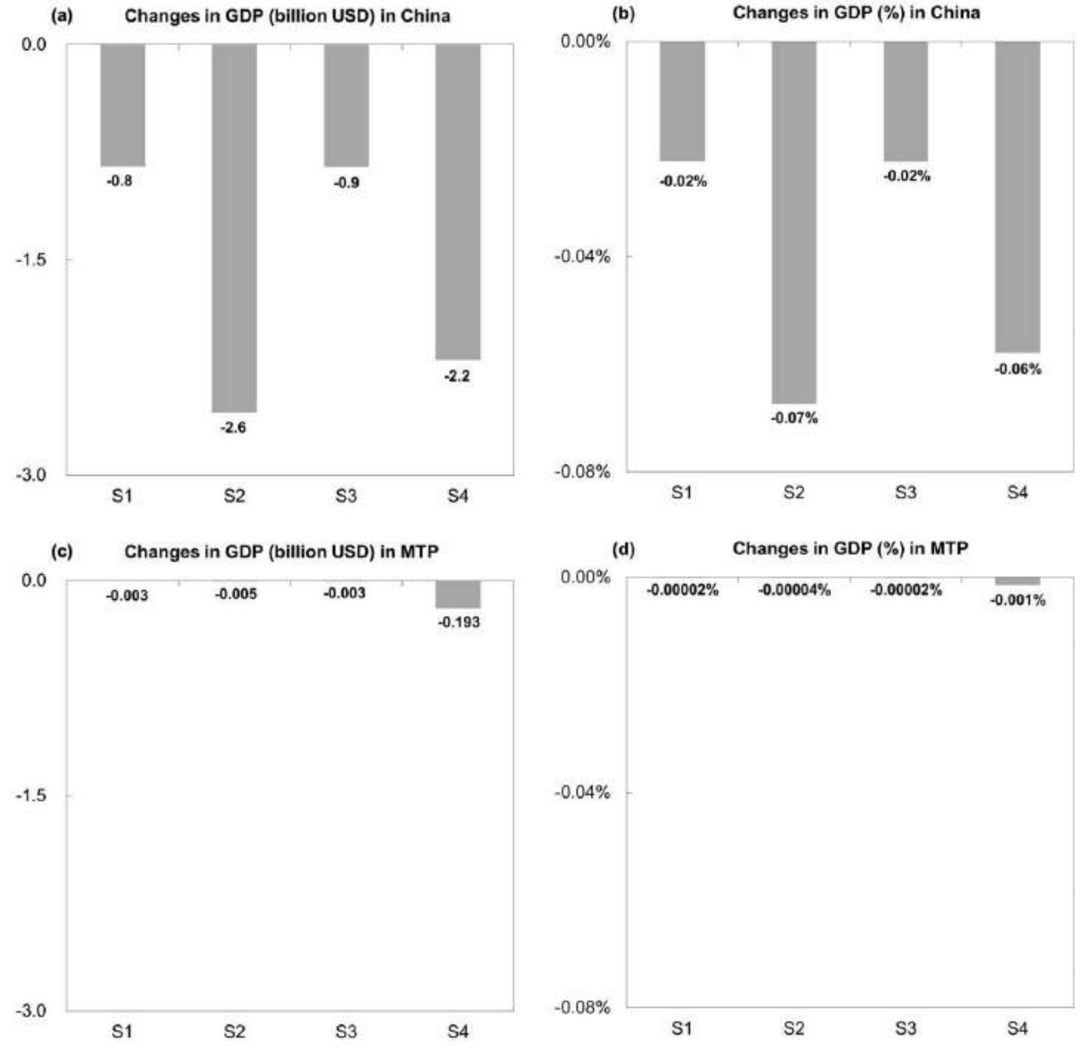
Supplementary Fig. 8 | Changes (%) in sectoral output (i.e., the value of production) in China in scenarios (a) S1, (c) S2, (e) S3, and (g) S4 with respect to the baseline (S0). Changes (%) in sectoral output (i.e., the value of production) in MTP in scenarios (b) S1, (d) S2, (f) S3, and (h) S4 with respect to the baseline (S0).



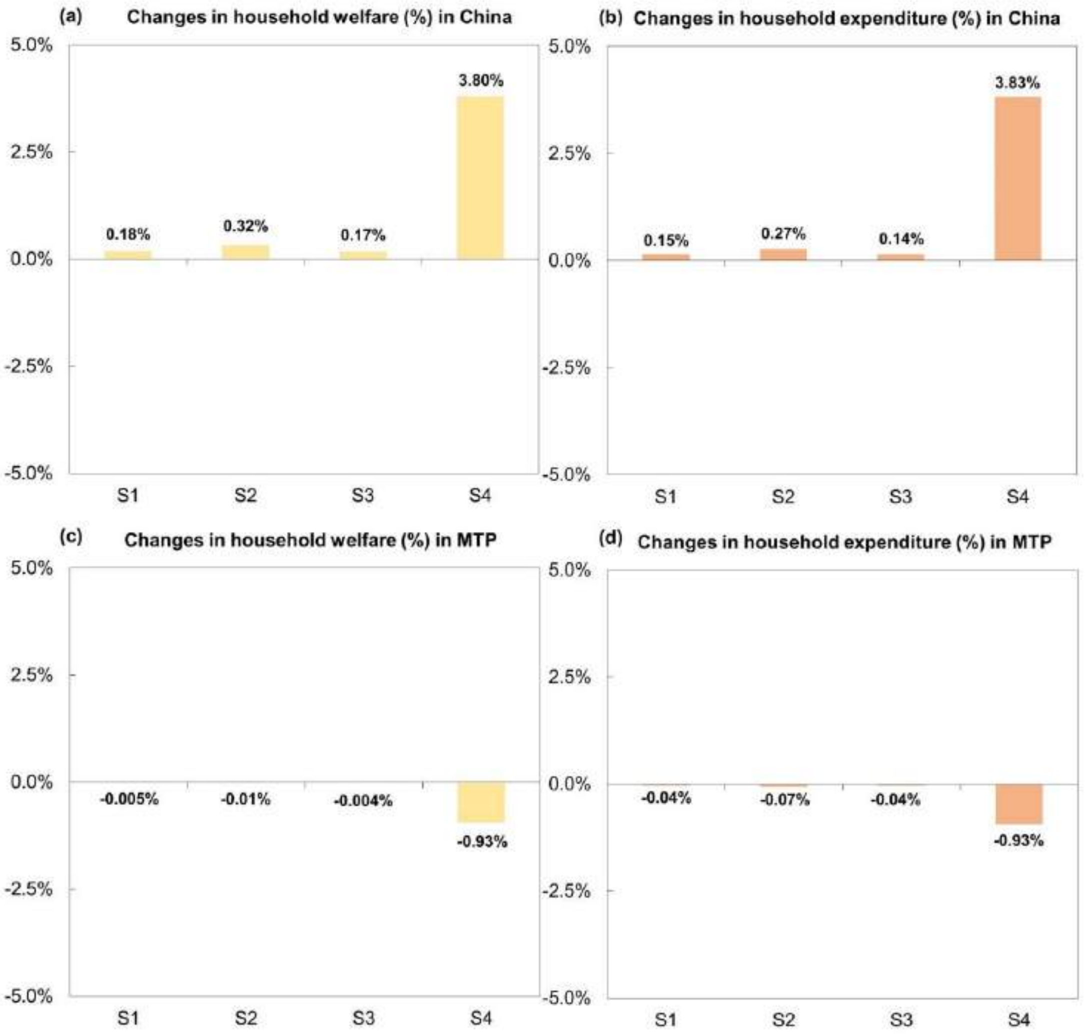
Supplementary Fig. 9 | Changes (billion USD) in sectoral value-added (a) in China and (b) MTP in scenarios with respect to the baseline (S0).



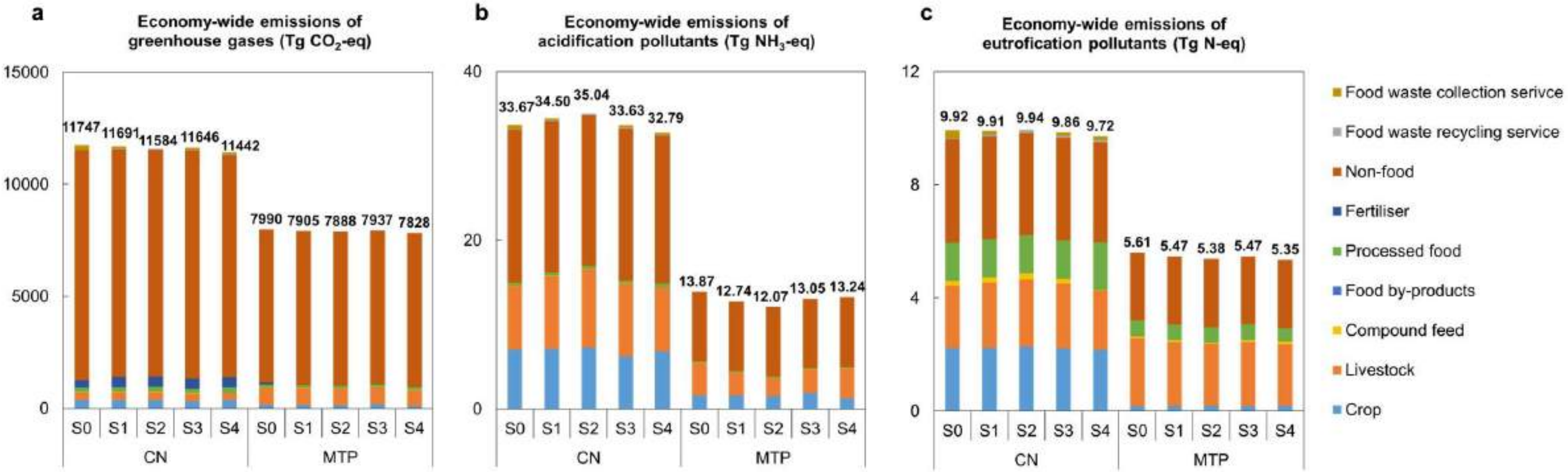
Supplementary Fig. 10 | Shares (%) of sectoral value-added in (a) China and (b) MTP in scenarios.



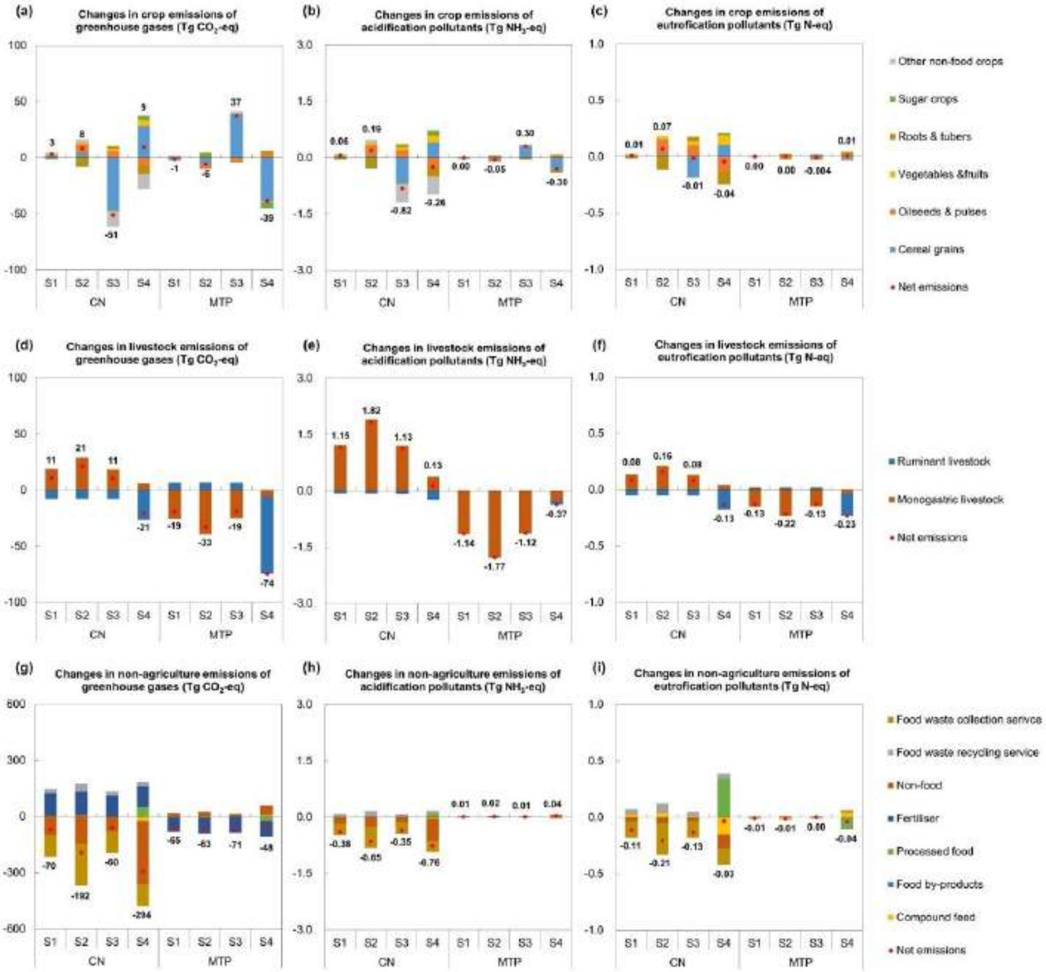
Supplementary Fig. 11 | (a) Absolute changes (billion USD) and (b) relative changes (%) in GDP in China in scenarios with respect to the baseline (S0). (c) Absolute changes (billion USD) and (d) relative changes (%) in GDP in MTP in scenarios with respect to the baseline (S0).



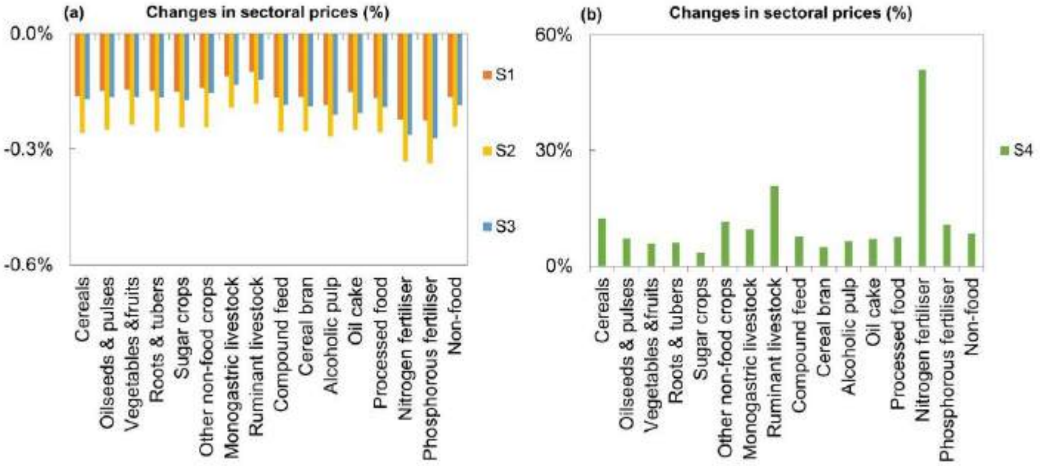
Supplementary Fig. 12 | Changes (%) in (a) household welfare and (b) household expenditure in China in scenarios with respect to the baseline (S0). Changes (%) in (c) household welfare and (d) household expenditure in MTP in scenarios with respect to the baseline (S0).



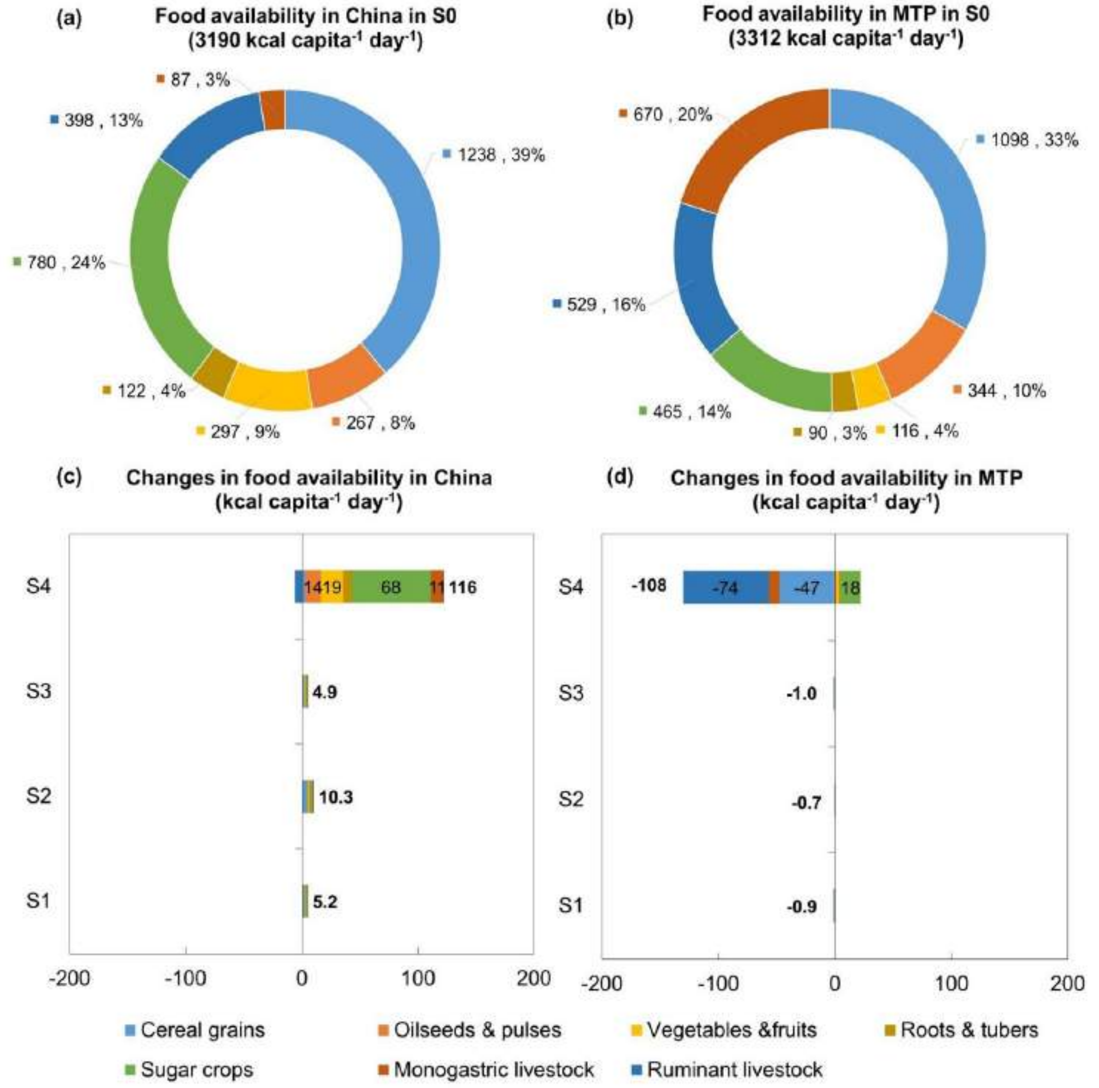
Supplementary Fig. 13 | (a) Economy-wide emissions of greenhouse gases (Tg CO2-eq), (b) acidification pollutants (Tg NH3-eq), and (c) eutrophication pollutants (Tg N-eq) in China and MTP in scenarios.



Supplementary Fig. 14 | Changes in crop emissions of (a) greenhouse gases (Tg CO_2-eq), (b) acidification pollutants (Tg NH_3-eq), and (c) eutrophication pollutants (Tg N-eq) in China and MTP in scenarios with respect to the baseline (S0). Changes in livestock emissions of (d) greenhouse gases (Tg CO_2-eq), (e) acidification pollutants (Tg NH_3-eq), and (f) eutrophication pollutants (Tg N-eq) in China and MTP in scenarios with respect to the baseline (S0). Changes in non-agriculture emissions of (g) greenhouse gases (Tg CO_2-eq), (h) acidification pollutants (Tg NH_3-eq), and (i) eutrophication pollutants (Tg N-eq) in China and MTP in scenarios with respect to the baseline (S0).



Supplementary Fig. 15 | Changes (%) in sectoral prices in scenarios (a) S1-S3 and (b) S4 with respect to the baseline (S0).



Supplementary Fig. 16 | Composition of food availability (%; kcal capita^{-1} day^{-1}) in (a) China and (b) MTP in the baseline (S0). Changes in food availability (kcal capita^{-1} day^{-1}) in (c) China and (d) MTP in scenarios with respect to the baseline (S0).

Supplementary Tables

Supplementary Table 1 | Summary of key assumptions used in scenario narratives and compensatory measures in China.

Scenarios [a]	Food waste used as animal feed in its total supply [b]	Emission mitigation target
S0: Baseline	Food waste: 39% By-products: 51%	No
S1: Partial use of food waste as feed [c]	Food waste: 54% By-products: 100%	No
S2: Full use of food waste as feed [c]	Food waste: 100% By-products: 100%	No
S3: S1 + A modest emission mitigation target [d]	Food waste: 54% By-products: 100%	Implementing economy-wide emission taxes to control emissions of greenhouse gases, acidification pollutants, and eutrophication pollutants in both China and its main food and feed trading partners (MTP, including Brazil, the United States, and Canada) no more than their baseline (S0) levels.
S4: S1 + An ambitious emission mitigation target [d]	Food waste: 54% By-products: 100%	Implementing economy-wide emission taxes to reduce emissions of greenhouse gases by 2.6% in China and 2.0% in MTP in line with their annual mitigation target of Intended Nationally Determined Contributions (INDC) under the Paris Agreement [9,10]. Implementing economy-wide emission taxes to reduce emissions of acidification and eutrophication pollutants in China by 2.5% and 2.0%, respectively, according to the annual mitigation target set by China's "14th Five-Year Plan" [11]. Implementing economy-wide emission taxes to control emissions of acidification and eutrophication pollutants in MTP no more than the baseline (S0) level.

[a] When substituting primary feed (i.e., feeding crops and compound feed) in animal diets with food waste and food processing by-products, we kept the total protein and total energy supplies for per unit of animal output were kept constant in all scenarios.

[b] In S1, cross-provincial transportation of food waste with high moisture content was not allowed, which limits the maximum utilisation rate of food waste to 54% in China, according to Fang, et al. [7], whereas it was allowed in S2.

[c] The cost of increasing the supply of food waste recycling service is modelled as a rising percentage of the initial cost of recycling food waste and food processing by-products as feed (54 dollar ton^{-1}), while the cost of decreasing the supply of food waste collection service is modelled as a declining percentage of the initial cost of collecting food waste and food processing by-products for landfill and incineration (82 dollar ton^{-1}). Economies of scale in food waste recycling were considered in S2, where a 1% increase in recycled waste resulted in only a 0.078% rise in recycling costs, indicating that increasing the amount of recycled waste might not necessarily incur additional costs, as reported by Cialani and Mortazavi [8]. This is because, initially, recycling entails high fixed costs, yet as production scales up, marginal costs decrease and then stabilise. The total amounts of food waste and food processing by-products and their current use as animal feed and discarded biomass (i.e., landfill and incineration) for China in S0 were presented in Supplementary Tables 3. Physical quantities and prices of food waste recycling service and food waste collection service in China were presented in Supplementary Tables 4-5.

[d] The main environmental problem associated with food systems depends on emissions from economic activities. Therefore, the introduction of economy-wide emission taxes could subsequently influence the way food is produced, inducing a shift away from emission-intensive production to cleaner alternatives. These policies aim to reduce emissions by pricing environmental emissions. Shadow prices of emissions, derived from the marginal value of the emission balance equations, ensure that total emissions by all producers remain below a specified emission threshold. For a given emission mitigation target for each type of pollutant, the AGE model can endogenously calculate the shadow prices of emissions of various pollutants.

Supplementary Table 2 | Physical quantities (Tg) in fresh form for each product in China (CN) and its main food and feed trading partners (MTP) in S0.

	CN	MTP
Cereal grains [a]	521.33	595.93
Oilseeds & pulses [a]	74.04	255.65
Vegetables & fruits [a]	397.23	116.39
Roots & tubers [a]	119.82	54.76
Sugar crops [a]	133.61	792.67
Other non-food crops [a]	36.48	23.24
Monogastric livestock [a]	103.15	18.65
Ruminant livestock [a]	52.53	46.28
Compound feed [b]	102.60	103.00
Cereal bran [c]	31.05	12.01
Alcoholic pulp [c]	45.60	76.09
Oil cake [c]	86.42	84.02
Processed food [d]	593.20	580.80
Nitrogen fertiliser	39.60	13.65
Phosphorous fertiliser	17.43	3.13
Grass [e]	286.22	0.00

[a] Physical quantities of cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, sugar crops, other non-food crops, monogastric livestock, ruminant livestock, nitrogen fertiliser, and phosphorous fertiliser were obtained from FAO [19]. Here, physical quantities of cereal grains waste, oilseeds & pulses waste, vegetables & fruits waste, and roots & tubers waste were excluded and presented in Supplementary Table 3.

[b] Compound feed production data was calculated according to the weighted averages of crops included in the compound feed at the national level.

[c] Physical quantities of cereal bran, alcoholic pulp, and oil cake were estimated from the consumption of corresponding food products and specific technical conversion factors [20].

[d] Processed food was calculated according to the weighted averages of crops included in the processed food at the national level.

[e] Grass from natural grassland was derived from Miao and Zhang [21]. Here, grass refers to grass from natural grassland where ruminant livestock is grazing for feed, and grass from remaining grassland is excluded. We do not present grass production data in MTP due to data unavailability.

Supplementary Table 3 | Utilisation rates (%) of food waste and food processing by-products in the baseline (S0) for China.

	Used as feed (%)	Discarded biomass (%) [c]
Cereals waste	39% [a]	Landfill (40%) & incineration (21%)
Vegetables & fruits waste	39% [a]	Landfill (40%) & incineration (21%)
Roots & tubers waste	39% [a]	Landfill (40%) & incineration (21%)
Oil seeds & pulses waste	39% [a]	Landfill (40%) & incineration (21%)
Cereal bran	36% [b]	Landfill (42%) & incineration (22%)
Alcoholic pulp	16% [b]	Landfill (55%) & incineration (29%)
Oil cake	72% [b]	Landfill (18%) & incineration (10%)

[a] In China, quantitative empirical data on food waste recycled as feed for monogastric livestock was not available. We infer that the practices of feeding food waste to monogastric livestock in Japan and South Korea are rather similar to those in China, following Fang, et al. [7]. Thus, we assumed that a similar proportion (39%, the mean of values in Japan and South Korea [22]) of food waste was being used as feed in China in 2014 in S0.

[b] The utilisation rates of food processing by-products recycled as feed in China in 2014 in S0 were based on Fang, et al. [7].

[c] Excluding the portion of food waste and food processing by-products recycled as feed, 66% of the remaining amount in China in 2014 was sent to landfills, while 34% was incinerated, according to Kaza, et al. [23] and Bhada-Tata and Hoornweg [24].

Supplementary Table 4 | Physical quantities (Tg) of food waste and food processing by-products and their utilisation in China in S0.

	Total in fresh form (Tg)	Total in dry matter (Tg)	Total in crude protein (Tg)	Total in energy (billion MJ)	Physical quantity in fresh form (Tg)	
					Used as feed [a]	Discarded biomass [b]
Total food waste	**226**	**54**	**7**	**690**	**88**	**138**
1) Cereal grains waste [b]	36.09	31.40	3.14	447	14.08	22.02
2) Vegetables & fruits waste [b]	175.01	17.50	2.98	183	67.76	107.25
3) Roots & tubers waste [b]	13.32	3.46	0.28	42	5.20	8.13
4) Oilseeds & pulses waste [b]	1.28	1.19	0.18	18	0.50	0.78
Total food processing by-products	**163**	**139**	**49**	**1907**	**78**	**85**
1) Cereal bran [c]	31.05	27.63	4.42	338	11.08	19.97
2) Alcoholic pulp [c]	45.60	34.20	9.23	439	6.66	38.94
3) Oil cake [c]	86.42	76.91	35.38	1130	59.80	26.59
Total	**389**	**192**	**56**	**2597**	**166**	**223**

[a] The amount of food waste used as feed corresponds to the quantity directed to the "food waste recycling service" sector. The amount of food processing by-products used as feed are not directed to the "food waste recycling service" sector; instead, these by-products with economically values are purchased directly by livestock producers in the feed market. When upcycling the discarded biomass of food waste and food processing by-products, these biomass are directed to the "food waste recycling service" sector.

[b] Discarded biomass of food waste and food processing by-products refers to the quantity collected for landfill and incineration, meaning the amount directed to the "food waste collection service" sector.

Supplementary Table 5 | Prices of food waste recycling service and food waste collection service in China. [a]

	Food waste treatment	Price [b] (dollar ton^{-1})	Weighted price [c] (dollar ton^{-1})
Food waste recycling service	Recycling waste as feed	54	54
Food waste collection service	Collection	40	82
	Landfill	31	
	Incineration	64	

[a] Food waste recycling service refers to recycling food waste as feed for monogastric livestock production, and food waste collection service means collecting food waste for landfill and incineration.

[b] The process of recycling food waste and food processing by-products as animal feed involves sorting, shredding, thermal treatment, fermentation, hydrolysis, and extrusion to create animal feed, as outlined by Alsaleh and Aleisa [25]. Collection includes pick up, transfer, and transport to final disposal site for food waste. By multiplying the quantity of food waste with the price of food waste treatment, we can calculate the value of food waste generation. The prices of food waste recycling service and food waste collection service are obtained from Alsaleh and Aleisa [25], Kaza, et al. [23] and Bhada-Tata and Hoornweg [24]. Since the value of food waste generation needs to be taken from the "wtr" demand of consumers and monogastric producers, we further checked whether or not the value of food waste generation is more than 80% of the initial demand of "wtr". If it is higher than 80% of the "wtr" demand, the value of food waste generation is scaled down.

[c] The weighted price of food waste collection service = collection price (40 \$/t) + 66%*landfill price (31\$/t)+34%*incineration price (64\$/t)=82\$/t.

Supplementary Table 6 | The economic and mass allocation of food processing main and by-products. [a]

	Main and by-products	By-product group	Economic share (%)	Mass share (%)
Cereal flour production [a]	Cereal flour	-	93%	86%
	Cereal bran	Cereal bran	7%	14%
Maize ethanol production [b]	Maize ethanol	-	83%	49%
	Distillers' grain from maize ethanol	Alcoholic pulp	17%	51%
Barley beer production [b]	Barley beer	-	98%	82%
	Brewers' grain from barley beer	Alcoholic pulp	2%	18%
Liquor production [b]	Liquor	-	97%	25%
	Distillers' grain from liquor	Alcoholic pulp	3%	75%
Vegetable oil production [c]	Soybean oil	-	44%	23%
	Soybean oil cake	Oil cake	56%	77%
	Other oil	-	66%	43%
	Other oil cake	Oil cake	34%	57%

[a] Data source: Haque, et al. [26], Mackenzie, et al. [27], Nyhan, et al. [28], and Pourmehdi and Kheiralipour [29]

Supplementary Table 7 | Estimated mean dry matter (DM, %), crude protein (CP, %), and energy (MJ kg DM^{-})contents of feed sub-groups in China (CN) and its main food and feed trading partners (MTP). [a]

	Dry matter (DM, %)		Crude protein (CP, %)		Energy (MJ kg DM^{-1})	
	CN	MTP	CN	MTP	CN	MTP
Cereal grains	89	89	11	10	18.25	18.82
Oilseeds &pulses	74	86	22	32	19.72	19.78
Vegetables &fruits	10	10	19	19	13.80	13.80
Roots &tubers	29	29	5	5	21.54	21.54
Sugar crops	69	69	16	16	19.68	19.68
Compound feed	48	70	34	23	18.61	19.36
Cereal bran	89	89	16	16	12.24	12.24
Alcoholic pulp	75	75	27	27	12.84	12.84
Oil cake	89	89	46	47	14.69	14.94
Cereal grains waste	87	-	10	-	14.25	-
Vegetables & fruits waste	10	-	17	-	10.45	-
Roots & tubers waste	26	-	8	-	12.15	-
Oilseeds & pulses waste	94	-	15	-	14.70	-
Cereal bran waste	89	-	16	-	12.24	-
Alcoholic pulp waste	75	-	27	-	12.84	-
Oil cake waste	89	-	46	-	14.69	-
Grass	27	27	12	12	11.20	11.20

[a] The values were weighted averages of feed types included in the groups at the national level. Data were sourced from the NUFER database [30], MITERRA-EUROPE database [31], NRC [32], NRC [33], NRC [34], NRC [35], and China Feed–database Information Network Centre ((http://www.chinafeeddata.org.cn/).

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Appendix Tables

Appendix Table 1 | Sectoral aggregation scheme.

Aggregated sectors	GTAP original sectors
Cereal grains	“Paddy rice (pdr)”, “Processed rice (pcr)”, “Wheat (wht)”, and “Cereals grains nec (gro)” sectors
Oilseeds & pulses	“Oil seeds (osd)” sector, and pulses split from the original “Vegetables& fruits (v_f)” sector
Vegetables & fruits	“Vegetables, fruits, nuts (v_f)” sector after splitting out pulses, and roots & tubers
Roots &tubers	Split from the original “Vegetables& fruits (v_f)” sector
Sugar crops	“Sugar cane & Sugar beet (c_b)” and Sugar (sgr)” sectors
Other non-food crops	“Plant-based fibers (pfb)”, and “Crops nec (ocr)” sectors
Monogastric livestock	“Animal products nec (oap)” and “Meat products nec (omt)” sectors
Ruminant livestock	“Cattle, sheep, goats, horses (ctl)”, “Meat: cattle, sheep, goats, horses (cmt)”, “Raw milk (rmk)”, “Wool, silk-worm cocoons (wol)”, and “Dairy products (mil)” sectors
Compound feed [a]	Split from the original “Food products nec (ofd)” sector
Cereal bran [a]	Split from the original “Food products nec (ofd)” sector
Alcoholic pulp [a]	Distiller’s grains from maize ethanol production split from the original “Food products nec (ofd)” sector; Distiller’s grains from liquor production and brewer’s grains from barley beer production split from the original “Beverages and Tobacco products (b_t)” sector
Oil cake [a]	Split from the original “Vegetable oils and fats (vol)” sector
Processed food [a]	“Food products nec (ofd)” sector after splitting out splitting out compound feed, cereal bran, and distiller's grains from maize ethanol production; “Beverages and Tobacco products (b_t)” sector after splitting out distiller’s grains from liquor production and brewer’s grains from barley beer production; Vegetable oils and fats (vol)” sector after splitting out oil cake
Nitrogen fertiliser [b]	Split from the original “Manufacture of chemicals and chemical products (chm)” sector
Phosphorous fertiliser [b]	Split from the original “Manufacture of chemicals and chemical products (chm)” sector
Food waste recycling service [c]	Split from the original “Waste and water (wtr)” sector

Aggregated sectors	GTAP original sectors
Food waste collection service [c]	Split from the original “Waste and water (wtr)” sector
Non-food	“Manufacture of chemicals and chemical products (chm)” sector after splitting out nitrogen fertiliser and phosphorous fertiliser; “Waste and water (wtr)” sector after splitting out food waste recycling service and food waste collection service; “Forestry (frs)”, “Fishing (fsh)”, “Coal (coa)”, “Oil (oil)”, “Gas (gas)”, “Minerals nec (oxt)”, “Petroleum, coal products (p_c)”, “Electricity (ely)”, “Gas manufacture, distribution (gdt)”, “Textiles （tex)”, “Wearing apparel (wap)”, “Leather products (lea)”, “Wood products (lum)”, “Paper products, publishing (ppp)”, “Manufacture of pharmaceuticals, medicinal chemical and botanical products (bph)”, “Manufacture of rubber and plastics products (rpp)”, “Mineral products nec (nmm)”, “Ferrous metal (i_s)”, “Metal nec (nfm)”, “Metal products (fmp)”, Electronic equipment (ele)”, “Manufacture of electrical equipment (eeq)”, “Manufacture of machinery and equipment n.e.c. (ome)”, “Motor vehicles and parts (mvh)”, “Transport equipment nec (otn)”, “Manufactures nec (omf)”, “Construction (cns)”, “Wholesale and retail trade; repair of motor vehicles and motorcycles (trd)”, “Accommodation, Food and service activities (afs)”, “Land transport and transport via pipelines (otp)”, “Warehousing and support activities (whs)”, “Sea transport (wtp)”, “Air transport （atp)”, “Communication (cmn)”, “Financial services nec (ofi)”, “Insurance (ins)”, “Real estate activities (rsa)”, “Other Business Services nec (obs)”, “Recreation & other services (ros)”, “Other Services (Government) (osg)”, “Education (edu)”, “Human health and social work (hht)”, “Dwellings: ownership of dwellings (imputed rents of houses occupied by owners) (dwe)” sectors

[a] Compound feed was split from the “Food products nec (ofd)” sector in the original GTAP database. The substance flow from “Food products nec (ofd)” to monogastric livestock and ruminant livestock was compound feed. Cereal bran and distiller’s grains from maize ethanol production were taken from the newly-splitted sector of compound feed according to the shares of economic values of cereal bran and distiller’s grains from maize ethanol production in the total economic value of compound feed. Economic values of cereal bran and distiller’s grains from maize ethanol production were calculated by multiplying the physical quantity (in tons) and the corresponding price (dollar per ton). Distiller’s grains from liquor production and brewer’s grains from barley beer production were split from the “Beverages and Tobacco products (b_t)” sector in the original GTAP database. The substance flow from “Beverages and Tobacco products (b_t)” to monogastric livestock were distillers' grains from liquor production and brewers' grains from barley beer production. Oil cake was split from the “Vegetable oils and fats (vol)” sector in the original GTAP database. The substance flow from the “Vegetable oils and fats (vol)” sector to monogastric livestock was oil cake.

[b] The nitrogen and phosphorus fertilisers were taken from the original 'Manufacture of chemicals and chemical products' sector following the method of Sturm [36] and Bartelings, et al. [37].

[c] Food waste recycling service and food waste collection service were split from the “Waste and water (“wtr”) sector in the original GTAP database according to the shares of economic values of food waste recycling service and food waste collection service in the total economic value of “Waste and water (“wtr”) sector. The economic values of food waste recycling service and food waste collection service were calculated by multiplying the physical quantity (in tons) and the corresponding

price (dollar per ton). Since the value of food waste generation needs to be taken from the 'wtr' demand of consumers and monogastric producers, we further checked whether or not the value of food waste generation is more than 80% of the initial demand of "wtr". If it is higher than 80% of the 'wtr' demand, the value of food waste generation are scaled down.

Appendix Table 2 | The social accounting matrix in the base year of 2014 for China (million $).[a]

	cer	osd	vf	rt	sgr	ocr	oap	ctl	cof	bran	pulp	cake	otf	nfe	pfe	nf	CONS	XNET	TOT
cer	0	0	0	0	0	0	29229	9055	11363	1372	67	0	81831	0	0	0	61825	-2016	192727
osd	0	0	0	0	0	0	1002	230	8312	0	0	182	42993	0	0	0	5092	-34661	23150
vf	0	0	0	0	0	0	5685	1495	18959	0	0	0	98059	0	0	0	145756	-139	269815
rt	0	0	0	0	0	0	595	157	1986	0	0	0	10270	0	0	0	15265	-15	28259
sgr	0	0	0	0	0	0	192	515	1280	0	0	0	6619	0	0	0	24553	-903	32256
ocr	0	0	0	0	0	0	664	262	197	0	0	0	1021	0	0	0	1282	-1465	1963
oap	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	176874	-3205	173669
ctl	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	63546	-484	63062
cof	0	0	0	0	0	0	45882	7458	0	0	0	0	0	0	0	0	0	854	54194
bran	0	0	0	0	0	0	3371	0	0	0	0	0	0	0	0	0	0	27	3398
pulp	0	0	0	0	0	0	800	0	0	0	0	0	0	0	0	0	0	-398	402
cake	0	0	0	0	0	0	215	0	0	0	0	0	0	0	0	0	0	-10	205
otf	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	432109	714	432823
nfe	7396	521	3479	471	313	621	0	0	0	0	0	0	0	0	0	0	0	-78	12721
pfe	2412	211	1542	169	83	163	0	0	0	0	0	0	0	0	0	0	0	-28	4551
nf	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2563284	354672	2917956
LAD1	53323	7694	80962	8445	9849	396	0	0	0	0	0	0	0	0	0	0	-160670	0	0
LAD2	0	0	0	0	0	0	0	10240	0	0	0	0	0	0	0	0	-10240	0	0
LAB	94995	11819	148120	15450	17556	631	62255	24592	6707	959	155	8	89845	4413	1579	1542959	-2022044	0	0
CAP	34602	2905	35711	3725	4455	151	23777	9057	5390	1067	180	15	102185	8308	2972	1374997	-1609499	0	0
TRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	312868	-312868	
TOT	192727	23150	269815	28259	32256	1963	173669	63062	54194	3398	402	205	432823	12721	4551	2917956			
cerw	0	0	0	0	0	0	754	0	0	0	0	0	0	0	0	0	1808		
vfw	0	0	0	0	0	0	3631	0	0	0	0	0	0	0	0	0	8806		

	cer	osd	vf	rt	sgr	ocr	oap	ctl	cof	bran	pulp	cake	otf	nfe	pfe	nf	CONS	XNET	TOT
rtw	0	0	0	0	0	0	278	0	0	0	0	0	0	0	0	0	667		
osdw	0	0	0	0	0	0	27	0	0	0	0	0	0	0	0	0	64		
branw	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1639		
pulpw	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3197		
cakew	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2184		

[a] Data source: GTAP [38]. cer=cereal grains. osd=oilseeds & pulses. vf=vegetables & fruits. rt= roots & tubers. sgr=sugar crops. ocr=other non-food crops. oap=monogastric livestock. ctl=ruminant livestock. cof=compound feed. bran=cereal bran. pulp=alcoholic pulp. cake=oil cake. otf=processed food. nfe=nitrogen fertiliser. pfe=phosphorous fertiliser. nf=non-food. CONS=consumption. XNET=net export. TOT=total. LAD1=cropland. LAD2=pasture land. LAB=labour. CAP=capital. TRA=trade. cerw=cereal grains waste. osdw= oilseeds & pulses waste. vfw=vegetables & fruits waste. rtw= roots & tubers waste. branw=cereal bran waste. pulpw=alcoholic pulp waste. cakew=oil cake waste.

Appendix Table 3 | The social accounting matrix in the base year of 2014 for China's main food and feed trading partners (MTP) (million $).[a]

	cer	osd	vf	rt	sgr	ocr	oap	ctl	cof	bran	pulp	cake	otf	nfe	pfe	nf	CONS	XNET	TOT
cer	0	0	0	0	0	0	3794	34288	4450	1023	414	0	32927	0	0	0	16597	2016	95511
osd	0	0	0	0	0	0	69	301	3307	0	0	2009	17059	0	0	0	1938	34661	59344
vf	0	0	0	0	0	0	354	1110	8351	0	0	0	43966	0	0	0	50755	139	104675
rt	0	0	0	0	0	0	37	116	875	0	0	0	4605	0	0	0	5316	15	10963
sgr	0	0	0	0	0	0	58	1037	1598	0	0	0	7759	0	0	0	16038	903	27392
ocr	0	0	0	0	0	0	130	413	943	0	0	0	4929	0	0	0	13124	1465	21003
oap	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	97851	3205	101056
ctl	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	214439	484	214923
cof	0	0	0	0	0	0	30067	32726	0	0	0	0	0	0	0	0	0	-854	61939
bran	0	0	0	0	0	0	4229	0	0	0	0	0	0	0	0	0	0	-27	4203
pulp	0	0	0	0	0	0	4967	0	0	0	0	0	0	0	0	0	0	398	5365
cake	0	0	0	0	0	0	2383	0	0	0	0	0	0	0	0	0	0	10	2393
otf	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	514821	-714	514107
nfe	2528	940	131	38	255	685	0	0	0	0	0	0	0	0	0	0	0	78	4655
pfe	1547	1164	87	47	92	231	0	0	0	0	0	0	0	0	0	0	0	28	3195
nf	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	13050326	-354672	12695654
LAD1	22886	13940	25013	2605	2260	5474	0	0	0	0	0	0	0	0	0	0	-72178	0	0
LAD2	0	0	0	0	0	0	0	15132	0	0	0	0	0	0	0	0	-15132	0	0
LAB	31115	17269	34446	3585	14182	5957	35369	71060	23869	1730	2795	231	203920	2038	1461	8550058	-8999086	0	0
CAP	37435	26030	44998	4688	10603	8655	19600	58739	18547	1450	2155	153	198941	2618	1734	4145596	-4581943	0	0
TRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	-312868	312868	
TOT	0	0	0	0	0	0	3794	34288	4450	1023	414	0	32927	0	0	0	16597	2016	95511
cerw	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
vfw	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		

	cer	osd	vf	rt	sgr	ocr	oap	ctl	cof	bran	pulp	cake	otf	nfe	pfe	nf	CONS	XNET	TOT
rtw	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
osdw	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
branw	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
pulpw	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
cakew	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		

[a] Data source: GTAP [38]. cer=cereal grains. osd=oilseeds & pulses. vf=vegetables & fruits. rt= roots & tubers. sgr=sugar crops. ocr=other non-food crops. oap=monogastric livestock. ctl=ruminant livestock. cof=compound feed. bran=cereal bran. pulp=alcoholic pulp. cake=oil cake. otf=processed food. nfe=nitrogen fertiliser. pfe=phosphorous fertiliser. nf=non-food. CONS=consumption. XNET=net export. TOT=total. LAD1=cropland. LAD2=pasture land. LAB=labour. CAP=capital. TRA=trade. cerw=cereal grains waste. osdw= oilseeds & pulses waste. vfw=vegetables & fruits waste. rtw= roots & tubers waste. branw=cereal bran waste. pulpw=alcoholic pulp waste. cakew=oil cake waste.

Appendix Table 4 | Emissions sources of greenhouse gases, acidification pollutants, and eutrophication pollutants across various sectors of the model. [a]

Sectors	Emissions of greenhouse gases (Tg CO_2 equivalents)	Emissions of acidification pollutants (Tg NH_3 equivalents)	Eutrophication pollutants (Tg N equivalents)
Crop	• Rice methane (CH_4) • Synthetic fertiliser and manure application (N_2O)	• Synthetic fertiliser and manure application (NH_3)	• Synthetic fertiliser and manure application (N and P losses)
Livestock	• Enteric fermentation (CH_4) • Manure management (CH_4 and N_2O) • Manure grassland (N_2O)	• Manure management (NH_3) • Manure grassland (NH_3)	• Manure management (N and P losses) • Manure grassland (N and P losses)
Non-agriculture	• Energy use (CO_2, CH_4, and N_2O)	• Energy use (NH_3, NO_x and SO_2)	• Energy use (N and P losses)

[a] Emissions from the production of N and P fertilisers were attributed to the respective fertiliser sector, while emissions from the application of these fertilisers were assigned to the crop sectors to prevent double counting. Data on N and P fertiliser use by crop types and countries were derived from Ludemann, et al. [39]. Manure data by animals were derived from FAO [19]. Allocation of manure for each crop was assumed to be consistent with the allocation of N fertiliser for each crop.

Appendix Table 5 | Total emissions of greenhouse gases (Tg CO_2 equivalents) in China (CN) and its main food and feed trading partners (MTP).[a]

	CN		MTP	
	Total	Total (%)	Total	Total (%)
Cereal grains	276.61	2.35	118.98	1.49
Oilseeds & pulses	8.33	0.07	9.88	0.12
Vegetables &fruits	54.88	0.04	3.34	0.08
Roots &tubers	7.46	0.47	0.82	0.04
Sugar crops	4.58	0.06	6.33	0.01
Other non-food crops	15.55	0.13	20.73	0.26
Monogastric livestock	79.37	0.68	63.77	0.80
Ruminant livestock	245.04	2.09	700.30	8.77
Compound feed	25.39	0.22	16.03	0.20
Cereal bran	0.00752	0.00006	0.00288	0.00004
Alcoholic pulp	0.0001148	0.0000010	0.0000318	0.0000004
Oil cake	0.01580	0.00013	0.01422	0.00018
Processed food	204.54	1.74	130.82	1.64
Nitrogen fertiliser	324.09	2.76	80.29	1.01
Phosphorus fertiliser	24.53	0.21	9.06	0.11
Non-food	10238.21	87.16	6825.11	85.47
Food waste recycling service	16.37	0.14	0.00	0.00
Food waste collection service	221.98	1.89	0.00	0.00
Total	11747	100.00	7985	100.00

[a] Data source: Climate Analysis Indicators Tool (CAIT) [40]. Emissions of food processing by-products (i.e., cereal bran, alcoholic pulp, oil cake) were derived from Mackenzie, et al. [27]. Emissions of food waste recycling service and food waste collection service were obtained from Alsaleh and Aleisa [25], Hong, et al. [41], and Hong, et al. [42]

Appendix Table 6 | Total emissions of acidification pollutants (Tg NH_3 equivalents) in China (CN) and its main food and feed trading partners (MTP).[a]

	CN		MTP	
	Total	Total (%)	Total	Total (%)
Cereal grains	3.94	11.71	0.94	6.77
Oilseeds & pulses	0.29	0.86	0.15	1.08
Vegetables & fruits	1.89	0.47	0.05	0.62
Roots & tubers	0.26	5.63	0.01	0.38
Sugar crops	0.16	0.77	0.09	0.10
Other non-food crops	0.54	1.60	0.34	2.47
Monogastric livestock	5.22	15.53	2.88	20.70
Ruminant livestock	2.21	6.58	1.05	7.56
Compound feed	0.04	0.13	0.02	0.13
Cereal bran	0.000328	0.0010	0.000126	0.0009
Alcoholic pulp	0.00000067	0.0000020	0.00000019	0.0000013
Oil cake	0.00080	0.0024	0.00073	0.0052
Processed food	0.35	1.05	0.16	1.11
Nitrogen fertiliser	0.0009	0.003	0.0035	0.025
Phosphorus fertiliser	0.0007	0.002	0.0029	0.021
Non-food	18.10	53.83	8.21	59.03
Food waste recycling service	0.06	0.18	0.00	0.00
Food waste collection service	0.56	1.66	0.00	0.00
Total	33.61	100.00	13.92	100.00

[a] Data source: Liu, et al. [43], Huang, et al. [44], and Dahiya, et al. [45]. Emissions of food processing by-products (i.e., cereal bran, alcoholic pulp, oil cake) were derived from Mackenzie, et al. [27]. Emissions of food waste recycling service and food waste collection service were obtained from Alsaleh and Aleisa [25], Hong, et al. [41], and Hong, et al. [42]

Appendix Table 7 | Total emissions of eutrophication pollutants (Tg N equivalents) in China (CN) and its main food and feed trading partners (MTP).[a]

	CN		MTP	
	Total	Total (%)	Total	Total (%)
Cereal grains	1.04	10.47	0.06	1.15
Oilseeds & pulses	0.15	1.48	0.05	0.93
Vegetables & fruits	0.88	0.20	0.04	0.12
Roots & tubers	0.12	8.84	0.01	0.69
Sugar crops	0.02	1.20	0.01	0.21
Other non-food crops	0.01	0.11	0.01	0.24
Monogastric livestock	0.58	5.89	0.38	6.79
Ruminant livestock	1.63	16.46	2.02	35.96
Compound feed	0.17	1.70	0.07	1.21
Cereal bran	0.0000147	0.0001	0.0000056	0.0001
Alcoholic pulp	0.00000029	0.0000030	0.00000008	0.0000015
Oil cake	0.000037	0.0004	0.000034	0.0006
Processed food	1.35	13.66	0.56	9.95
Nitrogen fertiliser	0.0002	0.002	0.0007	0.012
Phosphorus fertiliser	0.0002	0.002	0.0009	0.015
Non-food	3.66	36.88	2.40	42.71
Food waste recycling service	0.0303	0.31	0.0000	0.00
Food waste collection service	0.2790	2.81	0.0000	0.00
Total	9.92	100.00	5.61	100.00

[a] Data source: Hamilton, et al. [46]. Emissions of food processing by-products (i.e., cereal bran, alcoholic pulp, oil cake) were derived from Mackenzie, et al. [27]. Emissions of food waste recycling service and food waste collection service were obtained from Alsaleh and Aleisa [25], Hong, et al. [41], and Hong, et al. [42]

Appendix Table 8 | Emission intensities of greenhouse gases (t CO_2 equivalents million USD^{-1}) in China (CN) and its main food and feed trading partners (MTP).[a]

	CN	MTP
Cereal grains	1435	1246
Oilseeds & pulses	360	166
Vegetables &fruits	203	32
Roots &tubers	264	75
Sugar crops	142	231
Other non-food crops	7922	987
Monogastric livestock	457	631
Ruminant livestock	3886	3258
Compound feed	469	259
Cereal bran	2.2	0.7
Alcoholic pulp	0.3	0.01
Oil cake	77	6
Processed food	473	254
Nitrogen fertiliser	25477	17248
Phosphorus fertiliser	5390	2836
Non-food	3509	538
Food waste recycling service	3490	0
Food waste collection service	12087	0

[a] Data source: Calculated by our study.

Appendix Table 9 | Emission intensities of acidification pollutants (t NH_3 equivalents million USD^{-1}) in China (CN) and its main food and feed trading partners (MTP).[a]

	CN	MTP
Cereal grains	20.44	9.84
Oilseeds & pulses	12.53	2.53
Vegetables & fruits	7.00	0.48
Roots & tubers	9.20	0.91
Sugar crops	4.96	3.29
Other non-food crops	275.09	16.19
Monogastric livestock	30.06	28.50
Ruminant livestock	35.04	4.89
Compound feed	0.74	0.32
Cereal bran	0.10	0.03
Alcoholic pulp	0.002	0.00004
Oil cake	3.90	0.31
Processed food	0.81	0.31
Nitrogen fertiliser	0.07	0.75
Phosphorus fertiliser	0.15	0.91
Non-food	6.20	0.65
Food waste recycling service	12.79	0.00
Food waste collection service	30.49	0.00

[a] Data source: Calculated by our study.

Appendix Table 10 | Emission intensities of eutrophication pollutants (t N equivalents million USD^{-1}) in China (CN) and its main food and feed trading partners (MTP).[a]

	CN	MTP
Cereal grains	5.40	0.63
Oilseeds & pulses	6.48	0.84
Vegetables & fruits	3.26	0.38
Roots & tubers	4.25	0.91
Sugar crops	0.62	0.37
Other non-food crops	5.09	0.48
Monogastric livestock	3.34	3.76
Ruminant livestock	25.85	9.40
Compound feed	3.14	1.13
Cereal bran	0.004	0.001
Alcoholic pulp	0.001	0.00001
Oil cake	0.18	0.01
Processed food	3.12	1.09
Nitrogen fertiliser	0.02	0.15
Phosphorus fertiliser	0.04	0.28
Non-food	1.25	0.19
Food waste recycling service	6.46	0.00
Food waste collection service	15.19	0.00

[a] Data source: Calculated by our study.