The asymmetric impacts of feeding China's monogastric livestock with food waste on food security and environment sustainability

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Abstract

Around 1.3 billion tonnes of food waste are produced in the world, which are mainly disposed in landfills and incinerators, and are a significant source of greenhouse gas (GHG) emissions. While feeding animals with food waste may decrease such emissions, potential "rebound effect" remain unexplored. We used an integrated environmental-economic modelling framework to assess the impacts of upcycling food waste in China's monogastric livestock production in a global context. We found that upcycling 54-100% of food waste as feed increased monogastric livestock production (25-37%) and average wage across the Chinese economy (0.18-0.22%), with negative indirect effects such as increased total agricultural land use (0.5-0.6%) and economy-wide emissions of acidification (3-6%) and eutrophication (0.5-0.8%) pollutants in China. Synergy effects from less food waste in landfills and incinerators, along with the contraction in non-food production, decreased Chinese economy-wide GHG emissions (0.5-0.9%). While feeding food waste strategies enhanced food availability (6-12 kcal capita^{-1} day^{-1}) and affordability (0.38-0.49%) in China, it slightly reduced food availability (0.5-1.0 kcal capita^{-1} day^{-1}) and increased affordability (0.18-0.22%) in its trading partners. Our results highlight the asymmetric impacts of feeding China's monogastric livestock with food waste on food security and environment sustainability, urging complementary measures and policies to mitigate negative spillovers when promoting more circular food systems.

Keywords

circular economy; food waste; food security; environmental sustainability; environmental-economic modelling; rebound effects.

Main

The surge in demand for animal-sourced food (ASF) such as meat, milk, and eggs is driven by population growth, prosperity, and urbanisation [1,2]. The global demand for ASF is projected to double by 2050; the increase will occur, particularly in developing countries [3]. Livestock production expansion has driven global demand for animal feed as well as land used for feed crops, intensifying the food-feed competition and causing serious environmental concerns. Currently, 70% of global agricultural land is used for producing animal feed [4], and global livestock production account for 13-18% of the total anthropogenic greenhouse gas (GHG) emissions [5], 40% of the ammonia (NH_3) and nitrous oxide (N_2O) emissions [6], and around 24% of nitrogen (N) and 55% of phosphorus (P) losses to water bodies [7].

Globally, approximately 1.3 billion tons of food (roughly one-third of the total amount of food produced for human consumption) are lost or wasted each year, a considerable portion of which is disposed in landfills or incinerators, further exacerbating GHG emissions and climate change [8]. Upcycling food waste to substitute human-edible feed crops in animal diets may decrease GHG emissions associated with landfill and incineration and is crucial for building circular food systems [9]. Further, low-opportunity-cost feed (LCF), i.e., food waste and food processing by-products, typically compete less for land and natural resources than cereals and oilseeds, which are the main compounds of concentrated feed for monogastric livestock [9-11]. Feeding animals with food waste offers a pathway to mitigate land-related pressures [10], alleviate the food-feed competition [9], and reduce emissions from improper food waste disposal [11]. Increased utilisation of food waste 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) [12].

Building more circular food systems through increased utilisation of food waste as feed may also result in indirect effects and spillovers, which have not yet been investigated. First, feeding animals with food waste may lower feed costs and boost farm profits, which may drive livestock production expansion and lead to increased emissions—a phenomenon known as the "rebound effect" or “Jevons paradox” [13]. Second, increased utilisation of food waste as feed will not only impact consumers and producers of livestock but also have knock-on effects on other commodities in the broader economy. For instance, heightened demand for feed due to expanded monogastric livestock production may drive up crop production, leading to increased demand for land, fertilisers, and associated emissions. In addition, less food waste in landfills and incinerators may contribute to lower GHG emissions. Reducing cropland areas and GHG emissions are seen as the two key environmental benefits of feeding animals with food waste [9-11]. However, the possible rebound effect of expanded livestock production and its knock-on effects on other commodities could alter the expected outcome in terms of reducing agricultural land use and emissions. In essence, while previous studies [9-11] acknowledge the environmental benefits of increasing food waste utilisation as feed, their employment of linear optimisation models may overestimate the environmental benefits by disregarding market-mediated responses via the price system (i.e., holding costs and prices constant). Third, the food price may change, which could influence the availability and access dimensions of food security [14]. For example, the increased food production will enhance food availability, leading to lower food prices, but the expanded livestock production will stimulate labour demand, thus raising the economy-wide average wage. Food affordability is determined by fluctuations in the prices of a food consumption basket relative to changes in consumer income [15]. However, solely focusing on food price fluctuations without considering income changes resulting from increased food waste utilisation as feed may lead to biased conclusions on changes in food affordability.

Applied general equilibrium (AGE) models based on microeconomic theory are useful tools for analysing the economy-wide effects (i.e., production, consumption, and trade) of a transition to a circular economy [16,17]. AGE models can depict sectoral interactions, international trade, and consumer responses to changing prices and incomes, making them valuable tools for assessing the consequences of the transition towards more circular food systems. However, this requires that monetary AGE models do fully account for biophysical (quantity-based) and nutritional (protein and energy-based) livestock feeding constraints, which are crucial for analysing the environmental and economic impacts of feeding animals with food waste. Although previous studies [18-21] have endeavoured to integrate biophysical and nutritional livestock feeding constraints into AGE models, none have yet explored the potential impacts of upcycling discarded food waste as animal feed. Moreover, AGE models such as GTAP-E [22], GTAP-AEZ [23], GTAP-BIO [24], and MAGNET [25] primarily focus on GHG emissions and overlook other pollutants. It is crucial to encompass not only GHG emissions but also pollutants leading to acidification (i.e., NH_3 emissions to air) and eutrophication (i.e., N and P losses to water bodies) from livestock production within the AGE framework, given that livestock contributes more to these pollutants than to GHG [26-29]. Yet, no studies have done that so far.

In this study, we analysed the possible environmental and economic consequences of upcycling food waste in China's monogastric livestock production in a global context. China is the world's largest animal producer, and accounted for 46%, 34%, and 13% of the global pork, egg, and poultry meat production in 2018, respectively [30], making it a focal point of our study. We address three main research questions, emphasising indirect effects and spillovers not directly covered in previous studies. First, how will an increased utilisation of food waste as feed influence livestock production, food supply, and other sectors in China? Second, how will these influence GHG emissions and the pollutants emissions leading to acidification and eutrophication? Third, how will an increased utilisation of food waste as feed influence food availability and food affordability, which are crucial indicators of food security, if we account for changes in food prices and wages that provide the main source of consumer income? The novelty of this study lies in the improvement of an integrated environmental-economic framework by bridging monetary AGE models with biophysical (quantity-based) and nutritional (protein and energy-based) constraints. This improved framework may capture the rebound effect of expanded livestock production, its knock-on effects on other commodities, and the changes in food prices and consumer income when promoting circular food systems through increased utilisation of food waste as feed. Furthermore, integrating emissions of GHG and pollutants that lead to acidification and eutrophication into the AGE framework simultaneously allows us to discern the trade-offs and synergies associated with each type of emission.

We examined two scenarios with changed animal diets and compared these scenarios to a baseline (S0) scenario for the year 2014 without changing animal diets. Scenario S1 investigated the environmental and economic impacts of allowing partial use of food waste as feed (54% of food waste and 100% of food processing by-product waste) for monogastric livestock. Scenario S2 analysed the environmental and economic impacts of allowing full use of food waste as feed, taking into account economies of scale. 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. [10], whereas it was allowed in S2. 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, as reported by Cialani and Mortazavi [31]. The inclusion of two food waste-related sectors (see Fig. 1 and Methods) in the enhanced framework makes it capable of exploring the potential reuse of discarded food waste as animal feed. These sectors include the food waste recycling service sector for recycling food waste as animal feed and the food waste collection service sector for collecting food waste for landfill or incineration. The consumer price of food includes both the market price of food and the cost of collecting food waste by the municipality. In terms of recycling food waste as feed, monogastric livestock production bears the associated cost. When substituting primary feed (i.e., crops and compound feed) in animal diets with food waste, we maintain the protein and energy supply per unit of animal output in all scenarios to prevent imbalances between nutritional (protein and energy) supply and livestock requirements. The scenarios mentioned above are further described in Table 1.

Results

Impacts on livestock production, food supply, and other sectors.

China produced about 103 Tg of monogastric livestock products (pork: 57 Tg; poultry: 18 Tg; egg: 29 Tg) in 2014. The food recycling service sector recycled only 39% of food waste and 51% of by-product waste as feed (see Table 1). Expanding this sector to accomplish the goal of upcycling 54-100% of food waste as feed provided 18-28% more feed protein and 22-69% more feed energy for monogastric livestock production compared to current feed sources. This led to a 3.4-4.1% reduction in feed costs for per animal output, boosting profits for monogastric livestock producers and driving a 25-37% expansion in production (Fig. 2a). This shift also signals a transition for China from a net importer of monogastric livestock (with 1.1% of output imported in our baseline scenario S0) to an exporting nation of monogastric livestock (with 24-35% of output exported) (Fig. 3e). Increased shares of food waste use (9-14% in dry matter, 4-6% in protein, and 8-12% in energy; see Supplementary Fig. 1) within total feed use led to an equivalent decrease in demand for primary feed (i.e., crops and compound feed) for per unit of monogastric livestock production.

To quantify the contribution of human–edible feedstuffs to the animal-based food supply, we defined the eFCR (edible Feed Conversion Ratio) [32] as the quantity of human–edible feedstuffs included in the total feed to produce one unit of live weight gain of livestock production. Increased utilisation of food waste as feed alters FCR (feed conversion ratio, a ratio between the fresh matter of feed inputs and the live weight gain of livestock production) and eFCR. Despite a moderate increase in FCR (0.16-0.56 $kg\cdot kg^{-1}$) for monogastric livestock, the decreased eFCR (0.14-0.23 $kg\cdot kg^{-1}$) demonstrates reduced utilisation of human-edible feed crops for per unit of monogastric livestock production (Fig. 2b). However, the total demand of human-edible feed crops in monogastric livestock production increased by 9.5-9.9% (see Supplementary Fig. 2) due to expanded monogastric livestock production, intensifying demand for cropland by 0.4-0.6% (Fig. 2c). Negligible changes (less than 0.001 $kg\cdot kg^{-1}$) were observed in FCR and eFCR in ruminant livestock production due to minute changes in the production and feed use of ruminant livestock.

Feeding food waste strategies increased demand for feed crops and compound feed, driven by expanded monogastric livestock production, leading to a 0.18-0.22% rise in the average wage across the Chinese economy (see Supplementary Fig. 3), given that the crop and livestock sectors comprise 19% of the total labor supply. Consequently, labour became relatively more expensive compared to other factor inputs such as capital, cropland, and pasture land (see Supplementary Fig. 3). Consequently, producers will substitute labour with these relatively cheaper factor inputs. Ruminant livestock production remained nearly static, with the rise in labor costs offset by a corresponding increase in pasture land usage, driving a 0.5-0.7% increase in demand for pasture land (Fig. 2c). Crop producers will prioritise reducing the production of relatively labour-intensive crops; for example, roots & tubers are expected to decrease by 7-90% and sugar crops by 17-27% (Fig. 2c,d). The cropland saved from the reduced production of relatively labour-intensive crops will be reallocated to increase the production of crops that require relative more cropland or capital, such as cereal grains (1-3%), vegetables & fruits (2-3%), and other non-food crops (34-105%) (Fig. 2c,d). The larger percentage changes in other non-food crop production, compared to cereal grains and vegetables & fruits, can be attributed to initially low share acreage in total cropland occupation, accounting for less than 0.5% (see Supplementary Fig. 4). Notably, the production of oilseeds & pulses decreased by 8% when partial use of food waste as feed was allowed but increased by 71% when full use was allowed (Fig. 2c,d). This phenomenon arises because oilseeds & pulses are not only relatively cropland-intensive but also labour-intensive crops compared with other crops so the changes in their production depend on the interplay between labour and cropland costs under different scenarios.

Changes in crop production will alter their self-sufficiency ratios (SSRs, a ratio between domestic production and domestic utilisation). We found that the SSRs of roots & tubers and sugar crops decreased by 8-90% and 17-27%, respectively (Fig. 3e). The SSR of oilseeds & pulses increased by 26% when full use of food waste as feed was allowed, but decreased by 4% when allowing a partial use of food waste as feed (Fig. 3e). When full use of food waste as feed was allowed, the imports of cereal grains and other non-food crops decreased by 1.5 and 1.2 times of the initial levels, which led to complete self-sufficiency for these crops (see Supplementary Fig. 5).

Despite the 1-4% decrease in total crop production (Fig. 3a), the total fertiliser demand increased by 2-6% (Fig. 3c,d) because of changes in fertiliser demand by the crop type pattern (see Supplementary Fig. 4). Since fertiliser sectors are relatively energy-intensive, fertiliser producers could obtain profits by substituting labour with comparatively cheaper energy (mainly coal). This shift resulted in a 38-40% increase in nitrogen fertiliser production and a 24-64% increase in phosphorus fertiliser production. Consequently, China shifts from a net importer of nitrogen (with 3% of output imported in S0) and phosphorus (with 2% of output imported in S0) fertilisers to an exporting nation of nitrogen (with 27-31% of output exported) and phosphorus (with 20-52% of output exported) fertilisers (Fig. 3f). The significant changes in fertiliser production can be attributed to its initially low share of value-added in gross domestic product (GDP), accounting for less than 0.5% (see Supplementary Fig. 6). From the whole-economy perspective, upcycling food waste in monogastric livestock production as feed prompts a shift of workers from non-agricultural sectors to agricultural-related sectors, leading to an expansion in agricultural production and a contraction in non-agricultural production except for fertiliser sectors (Fig. A6).

Impacts on emissions.

Changes in production structure will lead to alterations in emissions of GHG (measured by CO_2-eq), acidification (measured by NH_3-eq), and eutrophication pollutants (measured by N-eq). Our findings revealed trade-offs between reductions in GHG emissions and an increase in emissions of acidification and eutrophication pollutants in China. Upcycling 54-100% of food waste as feed increased economy-wide emissions of acidification (3-6%) and eutrophication (0.5-0.8%) pollutants (Fig. 4b,c) in China, primarily due to the expansion of monogastric livestock production with relatively high emission intensities of these pollutants. The economy-wide GHG emissions decreased by 0.5-0.9% in China (Fig. 4a), despite the rise in GHG emissions from expanded livestock and fertiliser production, indicating synergy effects from less food waste in landfills and incinerators, alongside the contraction in non-food production.

Increased utilisation of food waste as feed will reduce China's reliance on imports of livestock products and fertilisers, resulting in its transition from a net importer to an exporting nation of these commodities (Fig. 3e,f). Consequently, China's main food and feed trading partners (MTP, including Brazil, the United States, and Canada) will experience environmental benefits, including reduced emissions of GHG (1.2-1.5%), acidification (9-14%), and eutrophication pollutants (3-4%). These environmental benefits for MTP stem from saving their domestic production of livestock and fertiliser because China transitions from a net importer of these commodities to an exporting nation of these commodities.

Impacts on food security and household welfare.

Subsequently, changes in production and prices may also influence not only food supply but also household welfare. We evaluated the availability and access dimensions of food security using food availability (daily per capita dietary calorie availability) and food access (per capita affordability and the average price of the current diet) as indicators. The composition of the current diet was outlined in Supplementary Fig. 7. Since prices offer only partial insight into food affordability, we used changes in the average price of a food consumption basket (current diet) in relation to the economy-wide average wage that provides the main source of consumer income (see Supplementary Fig. 8), as a proxy for food affordability.

Our findings indicated that upcycling 54-100% of food waste as feed slightly increased food availability (0.19-0.37%) and food affordability (0.38-0.49%) in China, which was related to lower food prices (0.20-0.27%) and higher average wage across the Chinese economy (0.18-0.22%) (Fig. 5a,b; Fig. A9). The increased food availability (0.19-0.37%, 6-12 kcal capita^{-1} day^{-1}) in China could sustain an additional 2.6-5.2 million people (Table A6). Concomitantly, there was a marginal decrease in food availability (0.02-0.03%, 0.5-1.0 kcal capita^{-1} day^{-1}) in MTP (Table A6). Overall, this initiative could potentially feed 2.5-5.0 million more people in China and MTP together. The increased food affordability in China aligned with a drop in the average price of the current diet (0.20-0.27%) and an increased average wage (0.18-0.22%) (Fig. A9). While food affordability rose for MTP (0.19-0.21%), the increase was smaller than for China (0.38-0.49%) (see Supplementary Fig. 9). Further, household welfare (a measure of economic well-being in million $) increased by 0.19-0.38% in China but decreased by 0.01-0.03% in MTP (see Supplementary Fig. 9). More detailed results on changes in prices by sectors are provided in Supplementary Fig. 10.

Discussion

This study uses an integrated environmental-economic framework to evaluate the possible environmental and economic consequences of upcycling food waste in China's monogastric livestock production in a global context. The novelty of this study lies in incorporating biophysical (quantity-based) and nutritional (protein and energy-based) constraints into monetary AGE models, thereby addressing a key limitation of current AGE models [19,21]. Feeding monogastric livestock with food waste will induce price changes and have knock-on effects on other commodities in the broader economy, potentially impacting changes in wage, land rent, and rental price of capital. Our approach complements previous linear optimisation studies [9-11], which overlooked market-mediated responses via the price system by considering both direct and indirect (price-induced) effects of increased utilisation of food waste as feed. Our results, thus, enhance the understanding of synergies and trade-offs between economic impacts and multiple environmental stresses associated with the increased utilisation of food waste as animal feed while respecting biophysical and nutritional constraints on livestock production.

Feeding monogastric livestock with food waste contributes significantly to the transition from linear to more circular food systems and alleviates food-feed competition. We found that upcycling 54-100% of food waste in monogastric livestock production significantly increased the shares of food waste use (9-14% in dry matter, 4-6% in protein, and 8-12% in energy) within total feed use for per unit of monogastric livestock production in China, which is crucial for the transition towards circular food systems. Despite a moderate increase in FCR (0.16-0.56 $kg \cdot kg^{-1}$) for monogastric livestock, the decreased eFCR (0.14-0.23 $kg \cdot kg^{-1}$) indicates reduced utilisation of human-edible feed crops for per unit of monogastric livestock production. These findings of changes in FCR and eFCR align with findings from Fang, et al. [10] and Gatto, et al. [19].

Feeding waste strategies can also address China's dependence on imported feed. While the 95% SSR redlines were maintained for main staple crops (wheat, rice, and maize), China became increasingly reliant on the imports of soybean, with 66% of the global soy trade purchased by China in 2017 to meet 90% of domestic demand [33]. This reliance on external sources presents food security risks [34], which are becoming an increasingly pressing global concern. We found that allowing the full utilisation of food waste as feed reduced cereal grain imports to 1.5 times their initial levels, achieving complete self-sufficiency, while oilseeds & pulses imports decreased by 26%, consistent with expectations outlined by Fang, et al. [10]. The decrease in imports of oilseeds & pulses can also reduce the environmental pressure associated with deforestation in Brazil, as 59% of Brazil's soybean exports associated with deforestation are attributed to China [35]. Feeding food waste strategies additionally reduced the economy-wide GHG emissions decreased by 0.5-0.9% in China due to less food waste in landfills and incinerators as well as the contraction in non-food production. This supports China's commitment to achieving carbon neutrality by 2060 [36].

While our study confirms the benefits of feeding food waste strategies observed in other studies, we also uncover some indirect and spillover effects associated with increased food waste utilisation as feed, aspects overlooked in prior linear optimisation studies [9-11,37]. 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 expanded livestock production induced by lower feed costs. The rebound effect of increased livestock production and its knock-on effects on other commodities cannot be overlooked, as these potential trade-offs and negative spillovers may alter the expected outcome in terms of reducing agricultural land use and emissions when transitioning to more circular food systems.

The first possible economic spillover effect is a 25-37% expansion of monogastric livestock production in China. This surge is attributed to the provision of 18-28% more feed protein and 22-69% more feed energy for monogastric livestock production through upcycling 54-100% of food waste as feed. Consequently, reduced feed costs and amplified profits for livestock producers incentivise livestock expansion. The expanded livestock production has been confirmed by Tong, et al. [38], who argue that allowing full use of food waste as feed could increase pork production by 14-29% even when holding costs and prices constant. This shift also signifies China's transition from a net importer of monogastric livestock (with 1.1% of output imported in our baseline scenario S0) to an exporting nation of monogastric livestock (with 24-35% of output exported). It is in line with the target of the " 95% SSR target for pork" proposed in 2020 [39] to restore the domestic supply capacity under the outbreak of African swine fever [40,41]. The expansion of monogastric livestock production, coupled with increased demand for feed crops and compound feed, drove up labour demand, generating a second positive spillover in the average wage across the Chinese economy (0.18-0.22%). Consequently, there was a shift toward substituting labour with other relatively cheaper factor inputs, such as capital, cropland, and pasture land, to choose the cheapest combination of inputs. This generates a third negative spillover effect of expanded monogastric livestock production: heightened agricultural land (cropland and pasture land) demand. In spite of reduced reliance on human-edible feed crops for per unit of monogastric livestock production, our model results indicate that the total demand for human-edible feed crops in livestock production will increase by 9.5-9.9%, intensifying demand for cropland by 0.4-0.6%. Meanwhile, the rise in labor costs also stimulate the use of pasture land for ruminant livestock production, driving a 0.5-0.7% increase in demand for pasture land. Crop producers will prioritise reducing the production of relatively labour-intensive crops (i.e., roots & tubers: 7-90%; sugar crops: 17-27%) and increasing the production of relatively cropland-intensive or capital-intensive crops (cereal grains: 1-3%; vegetables & fruits: 2-3%; other non-food crops: 34-105%). The production of oilseeds & pulses exhibits intriguing dynamics: its production decreased by 8% when partial use of food waste as feed was allowed but increased by 71% when full use was allowed. This phenomenon arises because oilseeds & pulses are not only relatively cropland-intensive but also labour-intensive crops. When partial use of food waste as feed is allowed, the increased cost of labour outweighs the decreased cost of cropland, resulting in reduced production. Conversely, when full use of food waste as feed is allowed, the futher reduced cost of cropland outweighs the increased cost of labour, leading to increased production. Labour, however, can also be substituted by comparatively cheaper energy (mainly coal) for fertiliser production, attributed to the energy-intensive nature of fertiliser sectors. This shift led to a 38-40% increase in nitrogen fertiliser production and a 24-64% increase in phosphorus fertiliser production. This also generates another negative environmental spillover effect by increasing GHG emissions related to fertiliser production. Our results are confirmed by Gatto, et al. [19] who have assessed the impact of subsidising the upcycling of agricultural residues and by-products as feed, revealing increases in agricultural wage, livestock production, and agricultural land use.

Economic spillovers into monogastric livestock sector also unexpectedly reverses the expected outcome in terms of reducing emissions. Our results indicated that feeding food waste strategies increased economy-wide emissions of pollutants associated with acidification (3-6%) and eutrophication (0.5-0.8%) in China, primarily driven by the expansion of monogastric livestock production. In spite of increased GHG emissions from expanded livestock and fertiliser production, China's economy-wide GHG emissions declined by 0.5-0.9% due to less food waste in landfills and incinerators as well as the contraction in non-food production. The positive contribution to lower GHG emissions through interactions with non-agricultural sectors also illustrates the relevance of using an general equilibrium model rather than an agricultural partial equilibrium model. The GHG-related environmental benefits of the increased food waste as animal feed are acknowledged by prior linear optimisation studies [9-11,37]; however, in our economy-wide perspective, the primary reduction in GHG emissions stems from less food waste in landfills and incinerators. Due to differing scenario setups and objectives, the results of the linear optimisation studies, as argued by Gatto, et al. [19], are largely incomparable to those in our economy-wide models. Linear optimisation studies often explore extreme scenarios by holding costs and prices constant, contrast sharply with our economy-wide models, which accounts for market-mediated responses via the price system and rational economic behavior of agents to closely mirror real-world conditions. This disparity presents challenges in replicating such scenarios within our economy-wide models, as the monetary constraints and rational economic behaviors modeled in our analysis diverge from the extreme scenarios exclusively detectable in linear optimisation models. Yet, these two modelling approaches could complement each other and support researchers and decision-makers by offering diverse perspectives on the same issue. Prior linear optimization studies could benefit from insights into the potential rebound effects uncovered by our economy-wide models, which potentially diminish the anticipated environmental benefits of feeding food waste strategies. Conversely, economy-wide models could gain valuable insights into envisioning a sustainable future by examining scenarios that disregard market-mediated responses via the price system.

Social spillover effects on food availability and affordability varies across China and its main food and feed trading partners. Some studies [42,43] evaluated food affordability primarily by considering changes in prices without accounting for income fluctuations, which may alter conclusions on changing food affordability. Since prices offer only partial insight into food affordability, we use changes in the average price of a food consumption basket (current diet) in relation to the average wage as a proxy for food affordability. We found increased food affordability in China (0.38-0.49%) aligned with a drop in the average price of the current diet (0.20-0.27%) and an increased average wage (0.18-0.22%), with a smaller increase in food affordability observed for MTP (0.19-0.21%) compared to China. Increased food availability in China could sustain 2.6-5.2 million more people, while a slight decrease in availability among trading partners risks hunger for 0.1-0.2 million people. Nonetheless, global food availability is improved, as China's increase exceeds the decline in its trading partners. This suggests that increased feeding of food waste to pigs in China has impacts that extend beyond borders, a type of telecoupled impact. [44,45]

Our findings unveiled the asymmetric impacts of feeding China's monogastric livestock with food waste on food security and environment sustainability. The concurrent reduction in GHG emissions, coupled with the enhancements in food availability and affordability, underscores the rationale for policymakers to promote the adoption of feeding food waste strategies. This aligns with China's recent emphasis on carbon neutrality and food security as leading priorities [46,47]. Despite these benefits of increased utilisation of food waste as feed, policymakers should remain vigilant regarding indirect effects and spillovers, particularly the unintended increases in agricultural land use and emissions of acidification and eutrophication pollutants, and be prepared to implement complementary measures and policies to mitigate these negative effects. Therefore, our findings hold following policy implications.

First, on the one hand, implementing economy-wide taxes on emissions of acidification and eutrophication pollutants alongside feeding food waste strategies could help mitigate the rebound effect of expanded monogastric livestock production, thus alleviating pressures on agricultural land use and reducing these emissions. This approach aligns with the recommendation of Gatto, et al. [20], who proposed using economy-wide GHG taxes to address the rebound effect of non-food sectors with increased GHG emissions during the global EAT-Lancet diet transition. The Chinese government has enacted several environmental policies aimed at reducing emissions of pollutants linked to acidification and eutrophication from agriculture and improving water quality. These policies include (i) Improvement of manure recycling [48], and (ii) Prevention and Treatment of Water Pollution ("Ten-Point Water Plan") [49]. On the other hand, adopting nitrogen mitigation measures for livestock manure could also alleviate the rebound effect of expanded production of monogastric livestock, given that poorly managed livestock manure is identified as the primary source of pollutants associated with acidification and eutrophication in China [50]. The estimated rate of manure nitrogen recycling to the field in China, accounting for 32% of total nitrogen excretion [50], significantly lags behind figures reported in the United States (75%) [51] and European Union (EU) countries (80%) [52]. Covering slurry stores and implementing low-NH_3 emission manure applications have been embraced by over 90% of farmers in the Netherlands and Denmark [53]. However, surveys conducted in China indicate that less than 20% of pig farms have adopted these measures. Policy instruments such as tax incentives and financial grants could accelerate the adoption of these technologies in China to mitigate the unintended increases in emissions of acidification and eutrophication pollutants. Despite the decrease in Chinese economy-wide GHG emissions, it is worth noting that the GHG environmental benefits do not originate from feed crop production but rather from the less food waste in landfills and incinerators. Therefore, China could achieve greater GHG environmental benefits through intensive crop production [54] and the adoption of improved fertilizer production technologies [55]. These measures are also consistent with the implementation of the "zero fertilizer growth" policy [56] in 2015 to reduce fertiliser use.

Second, we dodge the question of the policy instruments used to achieve the goal of increased utilisation of food waste as feed by exogenously raising the cost of recycling food waste as feed and lowering the cost of collecting food waste for landfill and incineration. This exogenous shift is similar to key publications on feeding food waste strategies [9-11,37]. We assume that the “food waste recycling service” sector exogenously expands its production to achieve the goal of increased utilisation of food waste as feed, leading to an equivalent decrease in the production of the “food waste collection service” sector. This implies that the capital and labour markets for food waste are not included in our analysis. This seems acceptable as the shares of value-added related to food waste in China’s total GDP amount to less than 0.5% (see Supplementary Fig. 6). Achieving close to the full use of food waste as feed seems possible in China because the food waste treatment industry (i.e., food waste collection service and food waste recycling service) is well developed and expanding recently [57]. The current reinforced policies on municipal solid waste separation and collection [58] in China guarantee a stable feed supply for monogastric livestock production. Additionally, the geographic proximity of industrial livestock farms to municipal food waste collection plants further facilitates the success of upcyling food waste as feed for monogastric livestock production [57]. However, allowing 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 [10]. In addition, 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. Our results demonstrated that upcycling 54-100% of food waste as feed increased feed protein supply by 18-28% and feed energy supply by 22-69% for monogastric livestock production, leading to a 3.4-4.1% reduction in feed costs for per animal output.

Third, our study assumes that individuals employed in non-agricultural sectors can shift to agricultural-related sectors under a constant total labor supply within the economy, following the default settings of standard GTAP [59] and USAGE [60] models. However, constraints on labour mobility, especially in the short term, may exist. On one hand, policies should facilitate the transition of workers towards agricultural sectors by lowering barriers to agricultural jobs through specialized training and educational programs, which could provide workers with enhanced opportunities to consider alternative employment paths. On the other hand, the current agricultural and non-agricultural production in China [61] implies that such shifts may require individuals employed in non-agricultural sectors to relocate from major non-agricultural production regions (i.e., southern China) to regions specialising in agricultural production (i.e., northern China). These relocations could incur tangible costs, which are likely to impact disadvantaged individuals and communities disproportionately.

Despite the integrated and holistic approach, this study has some limitations that necessitate some follow-up. 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 [25] and GTAP-AEZ [23] models. Second, expanding 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. While the estimated FCRs for the monogastric livestock sector closely align with reference estimates observed in literature [10,11,37], our estimates for ruminant livestock are somewhat lower compared to the literature. However, as 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 endeavor to depict policy instruments for achieving the goal of increased utilisation of food waste as feed, aligning with key literature on feeding food waste strategies [9-11,37]. Crucial questions remain how to design and implement policies that can achieve the goal of increased utilisation of food waste as feed, which falls outside the scope of this study but should be a pivotal direction for future research. Fourth, in line with SDG 12.3 ("halving food waste") [12], high priority should be placed on reducing food waste. With less food waste available for animal feed, the impacts of increased utilisation of food waste as feed may diminish. However, we consider our estimates of the impacts of increased utilisation of 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, we stress that the model simplifies the real world and draws conclusions from a static model with aggregated goods under current economic conditions. The outbreak of African swine fever in China is not considered in our model, which may overestimate the capacity to feed more food waste to pigs and expand the pig sector. This gives a direction for further study on developing a dynamic AGE model to include such events. Despite its limitations in short-term policy analysis, the static model, without considering technological and resource changes over time, allows us to minimise assumptions and uncertainties about future economic conditions while also isolating the impact of feeding China's monogastric livestock with food waste.

This study serves as a step towards bridging monetary AGE models with biophysical (quantity-based) and nutritional (protein and energy-based) constraints and explores the possible environmental and economic consequences of upcycling food waste in China's monogastric livestock production. While feeding food waste strategies offers benefits, such as reducing GHG emissions and improving food availability and affordability, policymakers should implement complementary measures and policies from an economy-wide perspective to address unintended increases in agricultural land use and emissions of acidification and eutrophication pollutants when promoting more circular food systems. 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.

Methods

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 [62-66]. For this study, we developed a global comparative static AGE model, a modified version of an integrated environmental-economic model, [67-69] 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. Our model is solved using the general algebraic modelling system (GAMS) software package [70].

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. [19], we converted dollar-based quantities to physical quantities (Tg) to allow the tracing of biophysical flows through the global economy. Global Trade Analysis Project (GTAP) version 10 database [59] was used to calibrate our AGE model and provide dollar-based quantities. Data on physical quantities (see Table A1) for crop and livestock production was obtained from FAO [30], FAO [71], and Miao and Zhang [72]. Feed production was extracted from “Feed” in the FAO food balance sheet. For illustrative purposes, 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 GTAP database. Our model aggregated livestock sectors in GTAP 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 feed in line with the dietary constraints of each livestock type in our model allows us to calculate the nutritional balance (crude protein and gross energy), feed conversion ratios (FCR, a ratio between the fresh matter of feed inputs and the live weight gain of livestock production), and edible feed conversion ratio (eFCR, the quantity of human–edible feedstuffs included in the total feed to produce one unit of live weight gain of livestock production) [32] for each livestock sector. First, we obtained the physical quantities (Tg) of livestock sectors and defined the feed supply in terms of physical quantities, energy, and protein required to produce this output of livestock. Then, the composition of total feed supplied to each livestock sector is specified, indicating the physical quantities, energy, and protein of feed products. The protein and energy supply for per kg animal feed remains preserved in all scenarios to avoid cases where livestock productivity is greatly affected when primary feed (i.e., crops and compound feed) is substituted with food waste. As we do not fully represent livestock diets by omitting hay, crop residues, and roughage-like by-products, FCRs for livestock, especially ruminant livestock, are slightly different from FCRs in the literature. Further model details, nutritional balance, and detailed composition of animals’ diets are available in the Supplementary Information (SI).

Food waste and food processing by-products available in China in 2014 were included in our study. 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 potential risks of pathogen transfer, including foot-and-mouth and classical swine fever [73]. Food waste was quantified separately for each type of food product using data on food consumption and China-specific food loss and waste fractions [74] following the FAO methodology [75]. 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 [76]. The total amounts of food waste and food processing by-products and their current use as animal feed in S0 for China are presented in Supplementary Table 2.

Our model incorporated a detailed module of food waste treatment by introducing two food waste-related sectors, i.e., food waste collection service and food waste recycling service. The representation of the economy in China in an AGE framework with the module of food waste treatment is shown in Figure 1. The food waste recycling service sector produces food waste recycling services to recycle food waste as feed for monogastric livestock production. The food waste collection service sector produces food waste collection services to collect 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. In our study, food waste generation was added as a margin commodity, similar to how GTAP treated transport costs following Peterson [77]. This means that the consumer price of food includes both the market price of food and the cost of collecting food waste from the municipality. In this way, the new food commodity can be seen as a composite bundle of the original food commodity and the food waste collection service required to collect food waste associated with the consumption of that food commodity. Consumers allocate income to the consumption of goods and food waste collection services, deriving utility only from the consumption of goods. In this way, decreased expenditure on food waste collection services does not alter consumers' utility function. 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. Since the value of food waste generation needs to be taken from the 'wtr' demand of consumers and monogastric livestock 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. Physical quantities and prices of food waste recycling service and food waste collection service in China were presented in Supplementary Tables 3-4.

We included three main environmental impacts of food systems, i.e., global warming potential (GWP, caused by 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. [78]. Data on CO_2, CH_4, and N_2O emissions were obtained from the Climate Analysis Indicators Tool (CAIT) [79]. We derived NH_3, NO_x, and SO_2 emissions from Liu, et al. [80], Huang, et al. [81], and Dahiya, et al. [82], 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 emissions to water bodies. We first obtained the total GHG emissions and pollutants leading to acidification and eutrophication for the food and non-food sectors in the base year. 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. Emissions per sector were calculated based on the emission database mentioned above and additional literature provided in SI by multiplying the physical quantity of an activity undertaken (in tons) and the corresponding emissions coefficient (tons of CO_2, NH_3, or N equivalents per unit of activity undertaken). The sector-level emissions of GHG (Tg CO_2 equivalents), acidification pollutants (Tg NH_3 equivalents), and eutrophication pollutants (Tg N equivalents) are presented in see Supplementary Tables 12-14, respectively. Furthermore, since food processing by-products are joint products with potential economic value to producers, 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 5).

Definition of scenarios. We examined two scenarios with changed animal diets and compared these scenarios to a baseline (S0) scenario in 2014 without changing animal diets. Scenario S1 investigated the environmental and economic impacts of allowing partial use of food waste as feed (54% of food waste and 100% of food processing by-product waste allowed to be used as feed for monogastric livestock). Scenario S2 analysed the environmental and economic impacts of allowing full use of food waste as feed, taking into account economies of scale. 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. [10], whereas it 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 [31]. This is because, initially, recycling entails high fixed costs, yet as production scales up, marginal costs decrease and stabilise. When substituting primary feed (i.e., human-edible feed crops and compound feed) with food waste, we maintain the protein and energy supply per unit of animal output in all scenarios to prevent imbalances between nutritional (protein and energy) supply and livestock requirements. The scenarios mentioned above are further described in Table 1.

Data availability

The data and parameters that support the economic model in this study are available from the GTAP version 10 database (https://www.gtap.agecon.purdue.edu/databases/v10/), which was used under license for the current study. Data are available with permission from the GTAP Centre. The other data that support splitting 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 from the original database GTAP 10 are publicly available at FAOSTAT (http://www.fao.org/faostat/en/#data) and the UN Comtrade Database (https://comtrade.un.org/data). The authors declare that all other data supporting the findings of this study are available within the article and its Supplementary Information files, or are available from the corresponding author upon reasonable request.

Code availability

The authors declare that the GAMS codes for producing the results of this study are available from the corresponding author upon reasonable request.

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Author contributions

W.L., X.Z., H.P.W., O.O., and Y.H. designed the research; W.L. and X.Z. developed the model; W.L., X.Z., H.P.W., O.O., and Y.H. analysed data; W.L., X.Z., H.P.W., O.O., and Y.H. wrote the paper. All authors contributed to the analysis of the results. All authors read and commented on various drafts of the paper.

Competing interests

The authors declare no competing interests.

Additional information

Details about the data, methods, and framework are presented in Supplementary Information (SI).

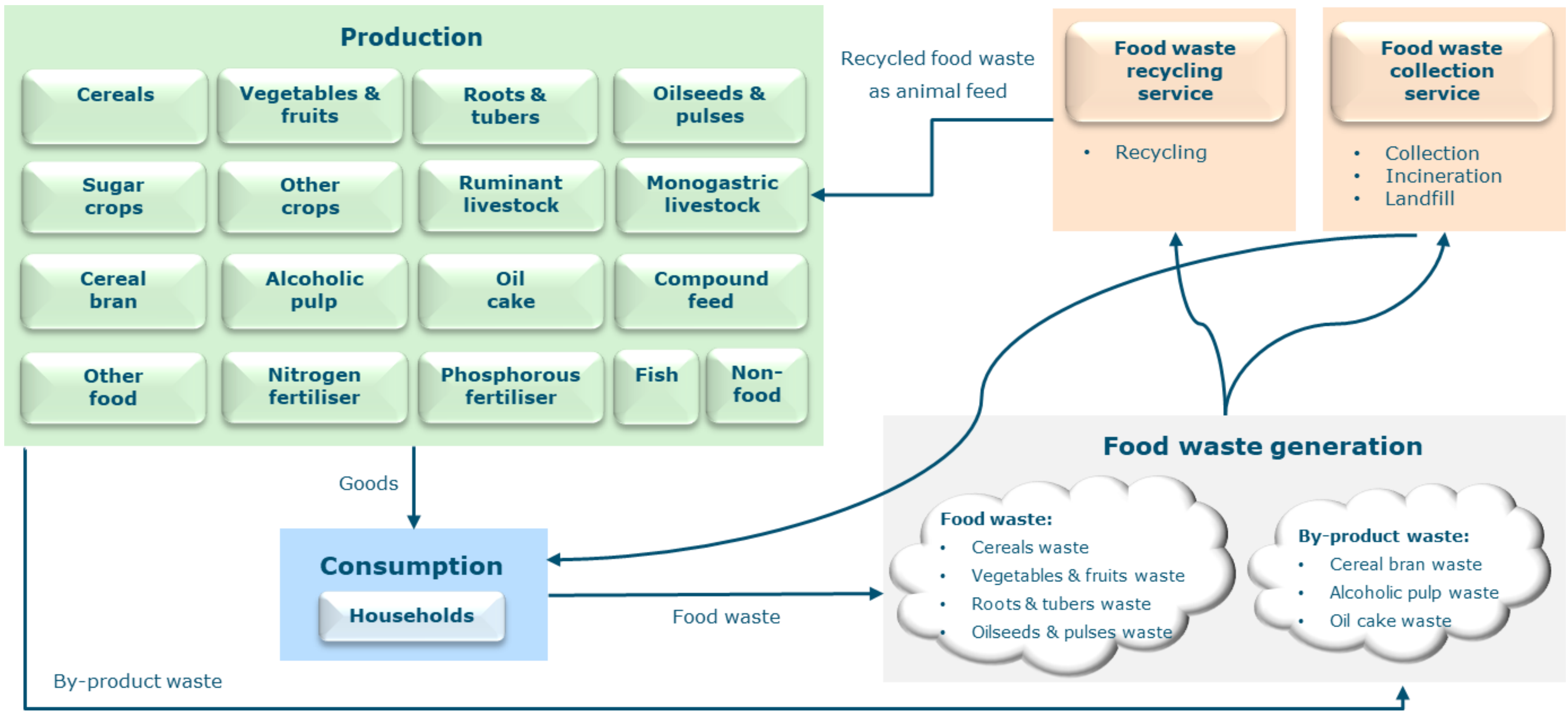

Fig. 1 | Representation of the economy in China in an AGE framework with the module of food waste treatment. The generated food waste is sent either to the ‘food waste recycling service’ sector or the ‘food waste collection service’ sector. The food waste recycling service sector produces food waste recycling services to recycle food waste as feed for monogastric livestock production. The food waste collection service sector produces food waste collection services to collect 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 by the municipality. In terms of recycling food waste as feed, monogastric livestock production bears the associated cost. Detailed information is presented in Methods and Supplementary Information.

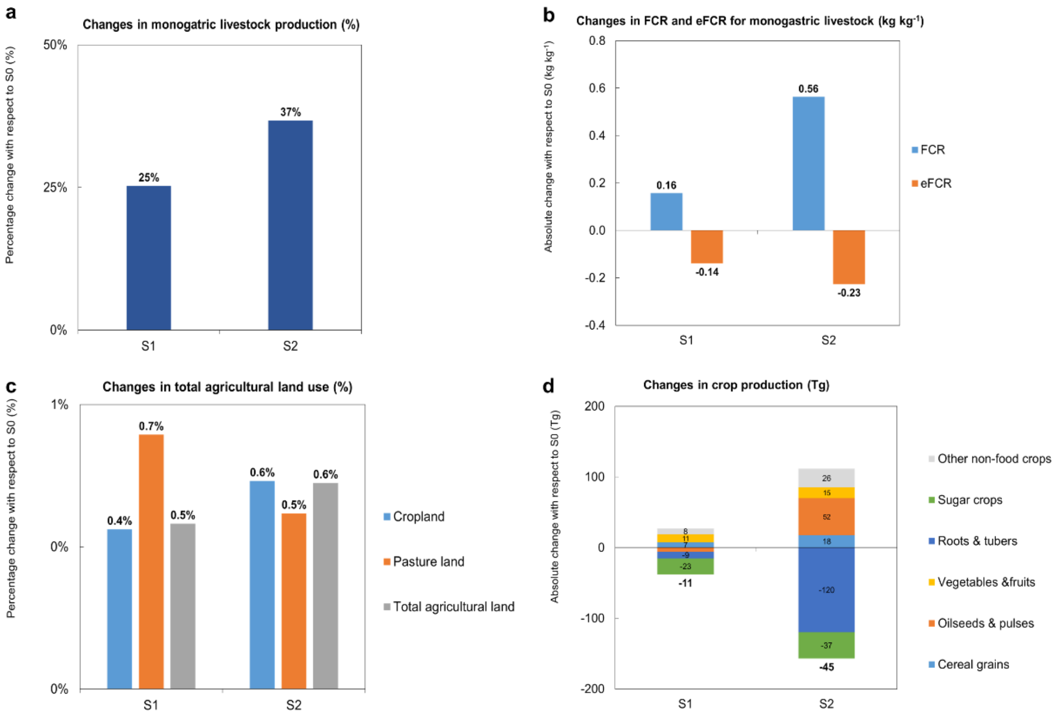

Fig. 2 | Impacts of upcycling food waste in monogastric livestock as feed on domestic livestock and crop production in China. (**a**) Percentage changes (%) in monogastric livestock production in scenarios with respect to S0. (**b**) Absolute changes (kg kg^{-1}) in feed conversion ratio (FCR) and edible feed conversion ratio (eFCR) for monogastric livestock in scenarios with respect to S0. (**c**) Percentage shares (%) for cropland and pasture land occupation with respect to S0. (**d**) Absolute changes (Tg) in crop production in scenarios with respect to S0. Definitions of scenarios (S1- 'Allowing partial use of food waste as feed'; S2- 'Allowing full use of food waste as feed with economies of scale') are described in Table 1.

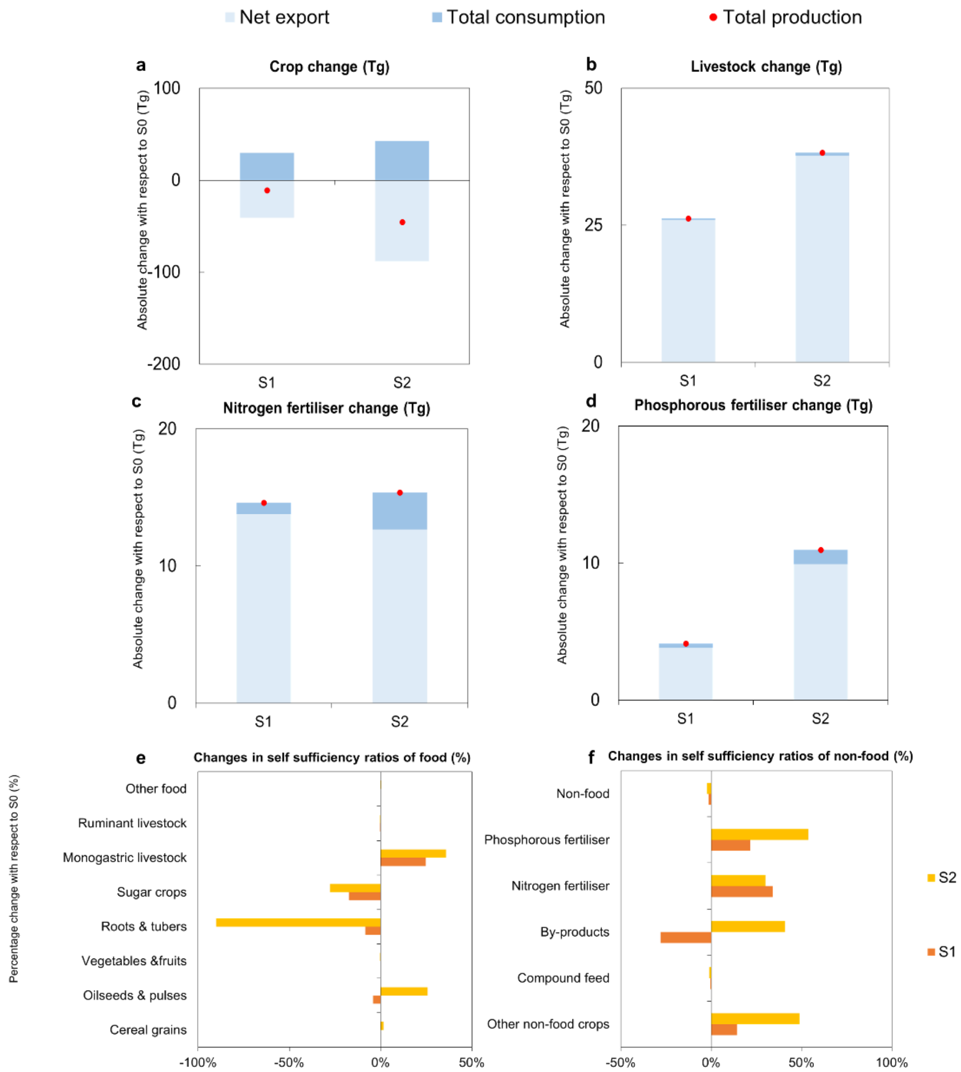

Fig. 3 | Impacts of upcycling food waste in monogastric livestock as feed on domestic production, consumption, and trade of food and non-food in China. a–d, absolute changes (Tg) in China's (**a**) crop consumption, production, and net exports, (**b**) livestock consumption, production, and net exports, (**c**) nitrogen fertiliser consumption, production, and net exports, and (**d**) phosphorous fertiliser consumption, production, and net exports in scenarios with respect to S0 in China. **d–e,** percentage changes (%) in self-sufficiency ratios (SSRs) of (**d**) food and (**e**) non-food. Definitions of scenarios (S1- 'Allowing partial use of food waste as feed'; S2- 'Allowing full use of food waste as feed with economies of scale') are described in Table 1.

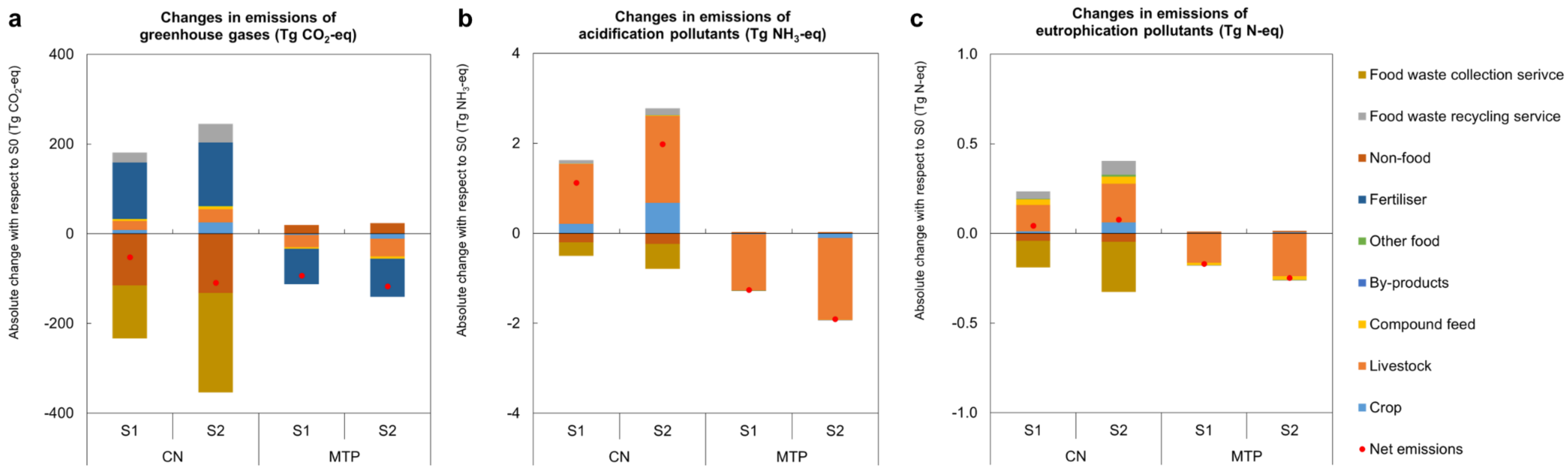

Fig. 4 | Impacts of upcycling food waste in monogastric livestock as feed on emissions in China (CN) and China's main food and feed trading partners (MTP). Absolute changes in (**a**) emissions of greenhouse gases (Tg CO_2-eq), (**b**) acidification pollutants (Tg NH_3-eq), and (**c**) eutrophication pollutants (Tg N-eq) in scenarios with respect to S0. Here, MTP includes Brazil, the United States, and Canada. Definitions of scenarios (S1- 'Allowing partial use of food waste as feed'; S2- 'Allowing full use of food waste as feed with economies of scale') are described in Table 1.

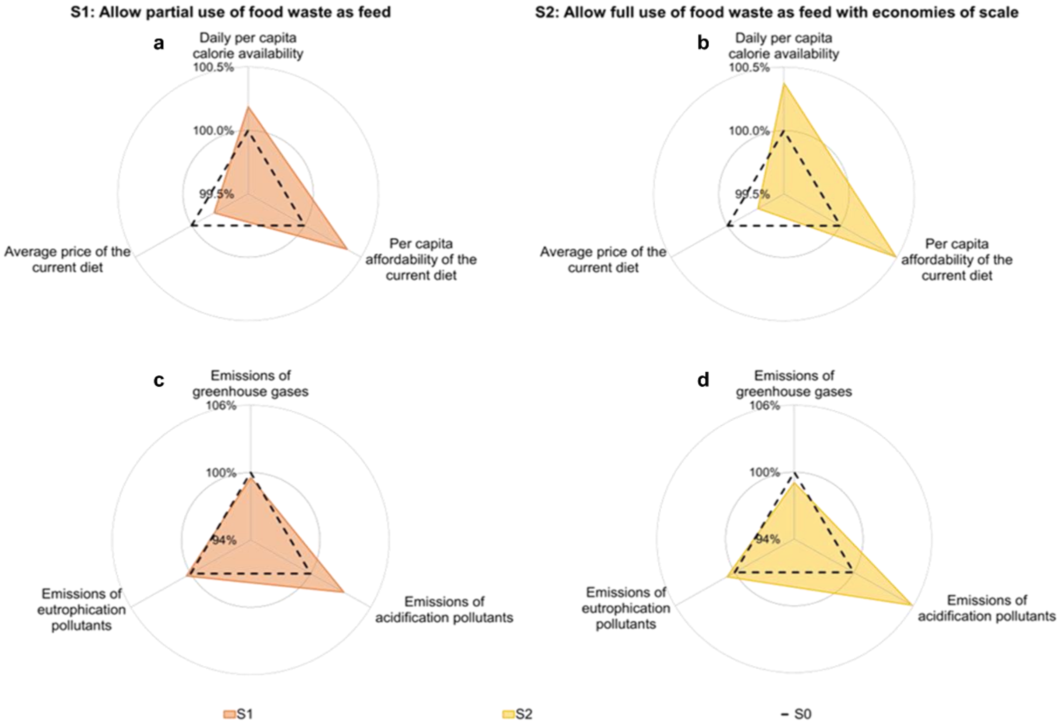

Fig. 5 | Impacts of upcycling food waste in monogastric livestock as feed on domestic sustainability in China. Percentage changes (%) of food security-related (i.e., daily per capita calorie availability, per capita affordability, and average price of the current diet) and environment sustainability-related (emissions of greenhouse gases, acidification pollutants, and eutrophication pollutants) indicators in (**a, c**) scenario S1 and (**b, d**) scenario S2 with respect to S0. Definitions of scenarios (S1- 'Allowing partial use of food waste as feed'; S2- 'Allowing full use of food waste as feed with economies of scale') are described in Table 1.

Table 1 | Summary of key assumptions used in the quantification of feed use in scenarios S0, S1, and S2 in China.

Scenarios [a]	Food waste as animal feed in its total supply [b]	Detailed explanation [c]
S0: Baseline	Food waste: 39% By-products: 51%	
S1: Allowing partial use of food waste as feed	Food waste: 54% By-products: 100%	Increasing the supply of food waste recycling service and decreasing the supply of food waste collection service to achieve 54% of food waste and 100% by-product waste being recycled as feed for monogastric livestock production.
S2: Allowing full use of food waste as feed with economies of scale	Food waste: 100% By-products: 100%	Increasing the supply of food waste recycling service and decreasing the supply of food waste collection service to achieve 100% of food waste and 100% by-product waste being recycled as feed for monogastric livestock production.

[a] When substituting primary feed (i.e., crops and compound feed) in animal diets with food waste, we maintain the protein and energy supply per unit of animal output in all scenarios to prevent imbalances between nutritional (protein and energy) supply and livestock requirements.

[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. [10], whereas it was allowed in S2.

[c] We increase the supply of food waste recycling service by exogenously raising the cost of recycling food waste as feed (54 dollar ton^{-1}) and decrease the supply of food waste recycling service by exogenously lowering the cost of collecting food waste for landfill and incineration (82 dollar ton^{-1}). Detailed information regarding the cost calculation is provided in Supplementary Table A4. 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 [31]. This is because, initially, recycling entails high fixed costs, yet as production scales up, marginal costs decrease and stabilise.

SUPPLEMENTARY INFORMATION

The asymmetric impacts of feeding China's monogastric livestock with food waste on food security and environment sustainability

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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, other food, fish, 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, other food, nitrogen fertiliser, phosphorus fertiliser, fish, and non-food.

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

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

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

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

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

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

(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 $\sum_{f=1}^{7} \delta_f = 1$.

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:

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

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

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

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

$$\tag{8}$$

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

for emissions of eutrophication pollutants ge = N and P losses

$$\tag{9}$$

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, by-products (i.e., cereal bran, alcoholic pulp, and oil cake), and other food production. 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, fish, other 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.

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

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

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

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

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

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

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

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 other 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 other 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 other 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 other 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 other 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 other 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 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, fish, other 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:

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

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

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, 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:

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

$$C_{i,osdw} \leq \overline{C_{i,osdw}} \qquad (p_{i,osdw2}) \tag{39}$$

$$C_{i,vfw} \leq \overline{C_{i,vfw}} \qquad (p_{i,vfw2}) \tag{40}$$

$$C_{i,rtw} \leq \overline{C_{i,rtw}} \qquad (p_{i,rtw2}) \tag{41}$$

$$C_{i,branw} \leq \overline{C_{i,branw}} \qquad (p_{i,branw2}) \tag{42}$$

$$C_{i,pulpw} \leq \overline{C_{i,pulpw}} \qquad (p_{i,pulpw2}) \tag{43}$$

$$C_{i,cakew} \leq \overline{C_{i,cakew}} \qquad (p_{i,cakew2}) \tag{44}$$

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:

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

where consumption goods s refers to cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, sugar crops, other non-food crops, monogastric livestock, ruminant livestock, other food, fish, and non-food. $\sum_s (p_{i,s}C_{i,s})$ 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 food waste collection service sector. Since goods are tradable, the consumer's income should exclude the export part. Thus, the consumer's income is:

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

where $\sum_j (p_j XNET_{i,j})$ 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.

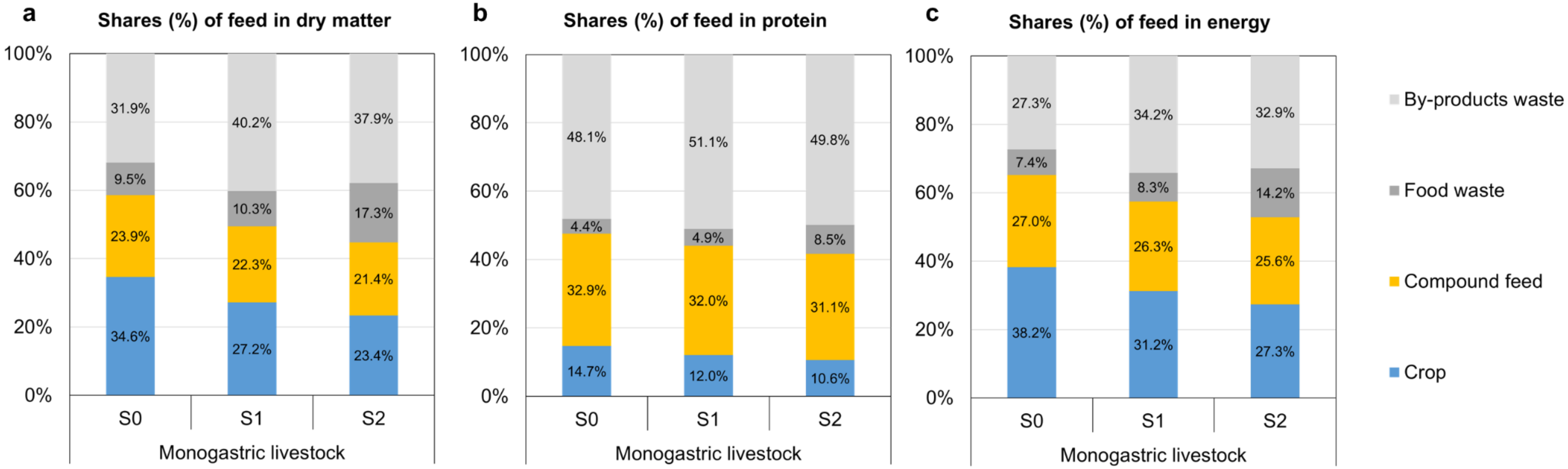
The producers' profits are specified as follows:

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

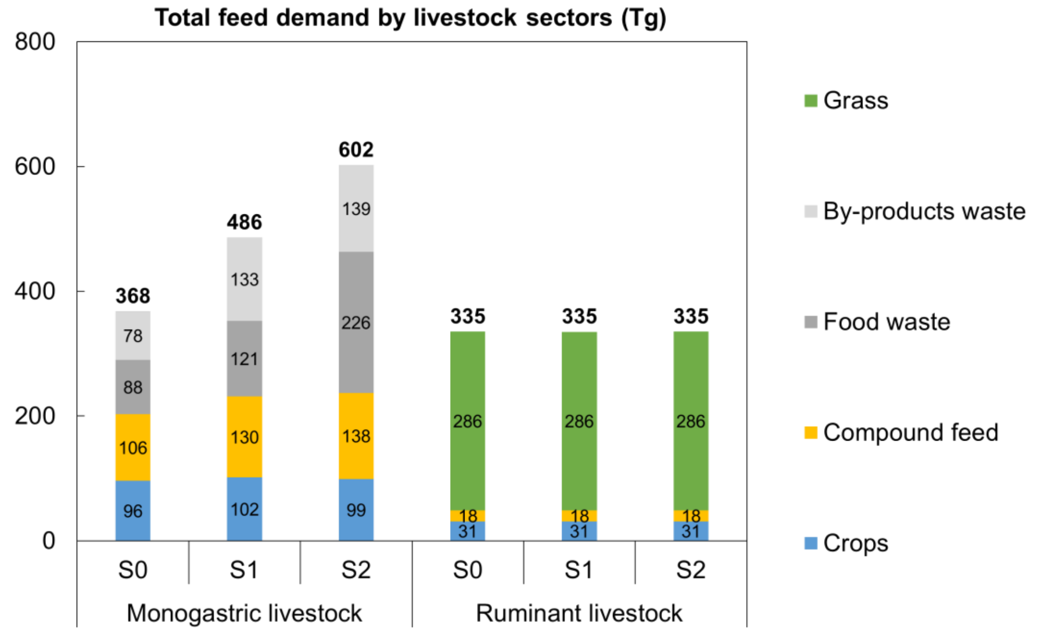
Model calibration

As in the literature on AGE models, we followed the Harberger convention [4] 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 [5]. 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 Supplementary Tables 10-11), 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)).

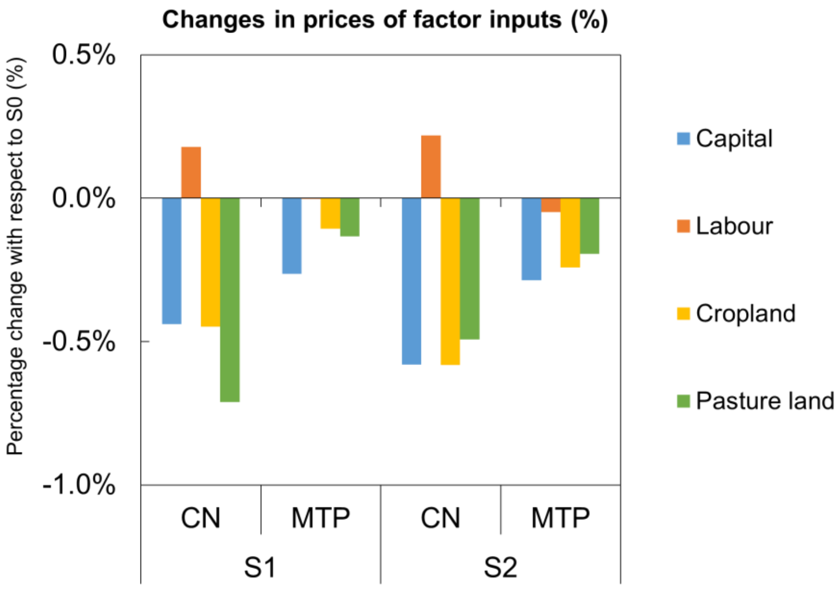
Supplementary Figures



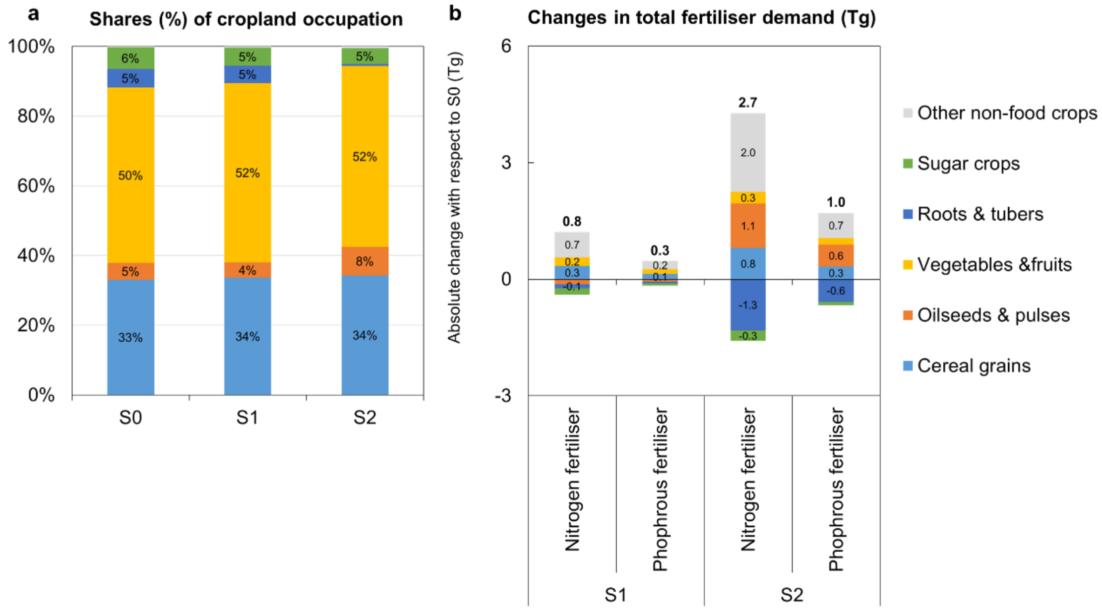
Supplementary Fig. 1 | Percentage shares (%) for each feed type of changes in feed in (a) dry matter, (b) protein, and (c) energy within total feed use for per kg of monogastric livestock production in scenarios.



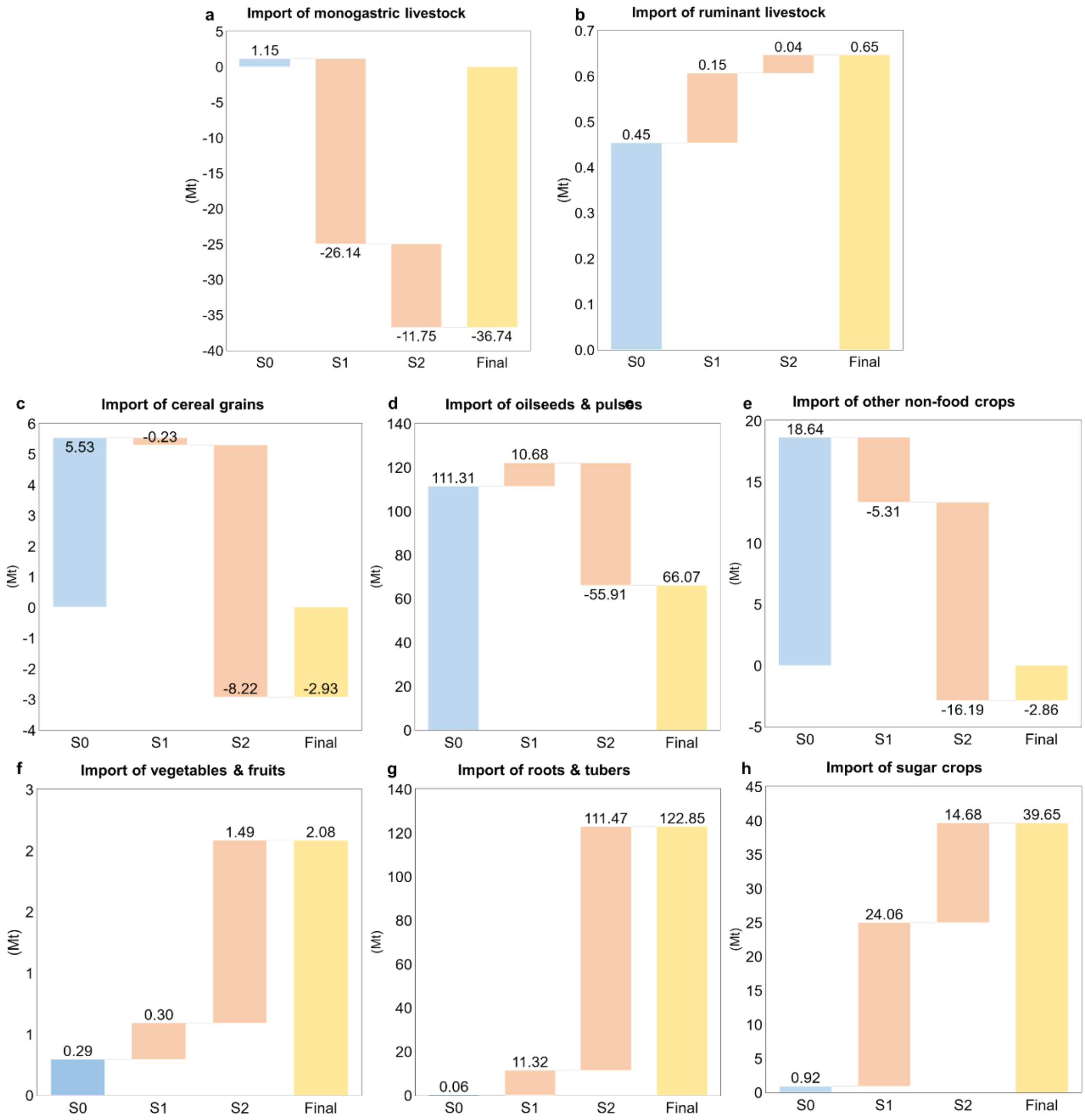
Supplementary Fig. 2 | Total feed demand (Tg) by livestock sectors in China in scenarios.



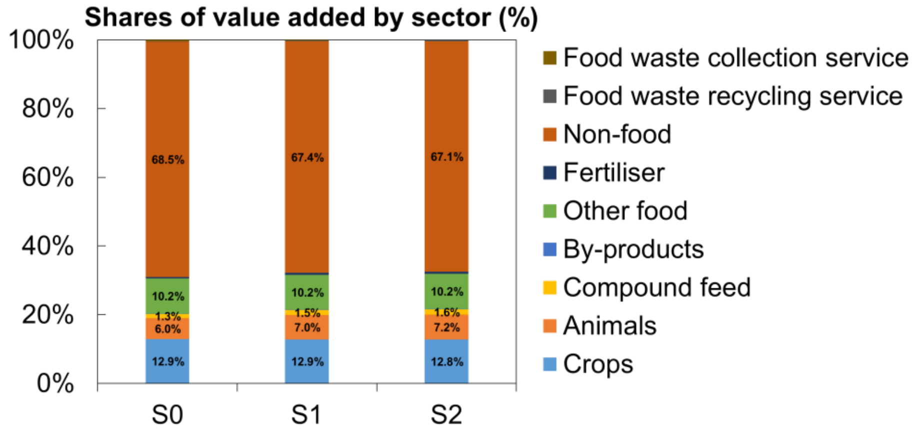
Supplementary Fig. 3 | Percentage changes (%) in prices of factor inputs in China (CN) and China’s main food and feed trading partners (MTP) in scenarios with respect to S0.



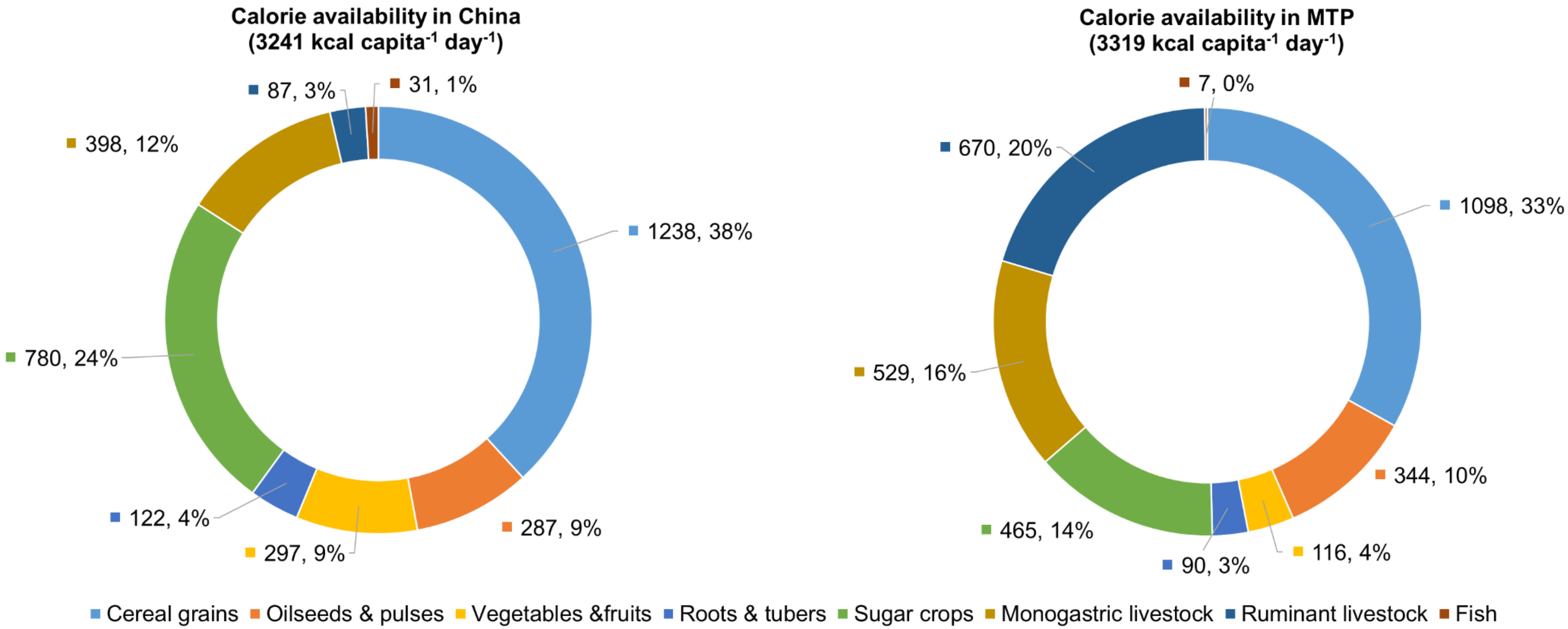
Supplementary Fig. 4 | (a) Percentage shares (%) for each crop of changes in total cropland occupation in scenarios. (b) Absolute changes (Tg) in total fertiliser demand by crops in China in scenarios with respect to S0.



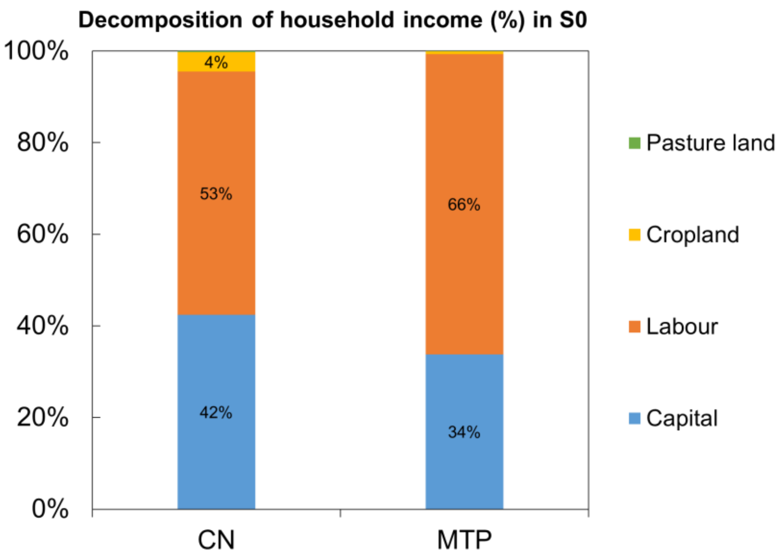
Supplementary Fig. 5 | Absolute changes (Tg) in China's imports of (a) monogastric livestock, (b) ruminant livestock, (c) cereal grains, (d) oilseeds &pulses, (e) other non-food crops, (f) vegetables & fruits, (g) roots & tubers, and (h) sugar crops. The lengths of orange bars indicate the absolute change in each scenario compared with the previous scenario. The length of the final bar is the value for S2.



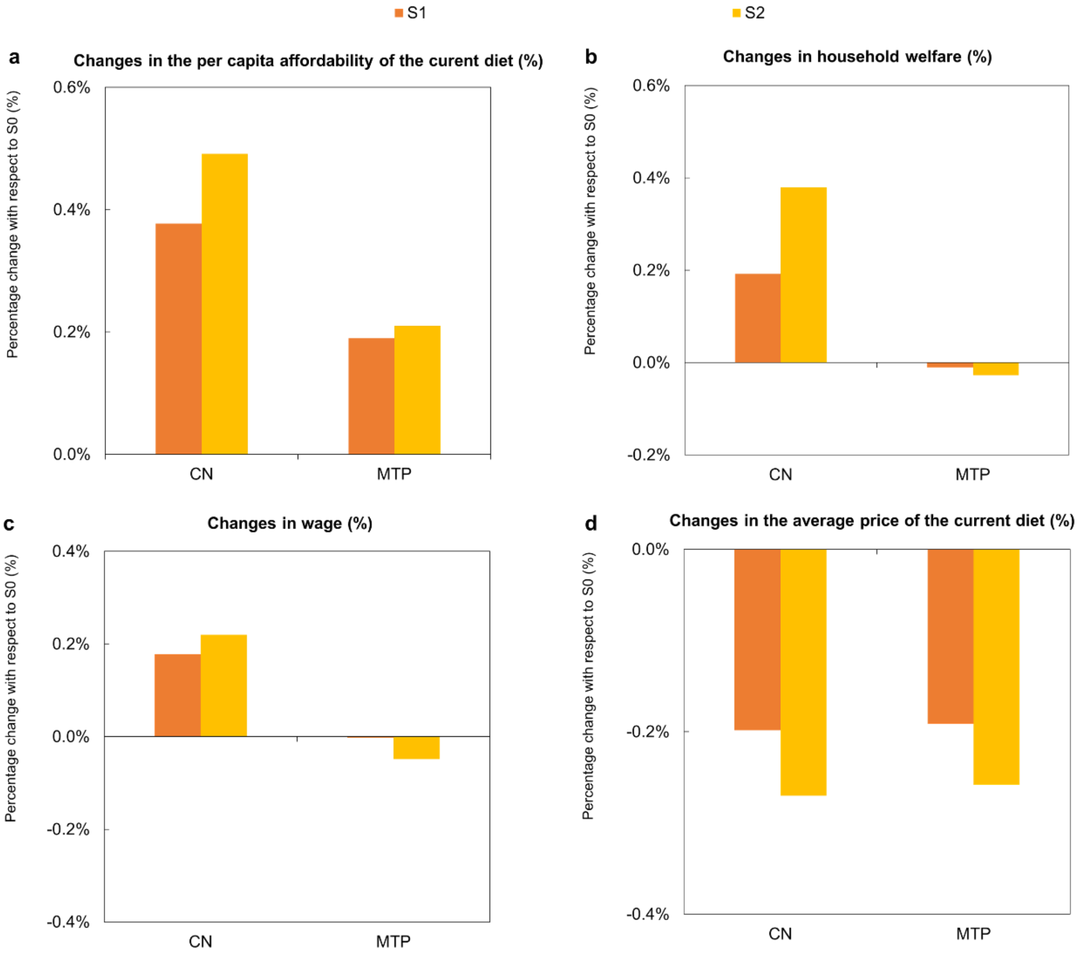
Supplementary Fig. 6 | Shares (%) of value-added by sector in Chinese GDP in scenarios.



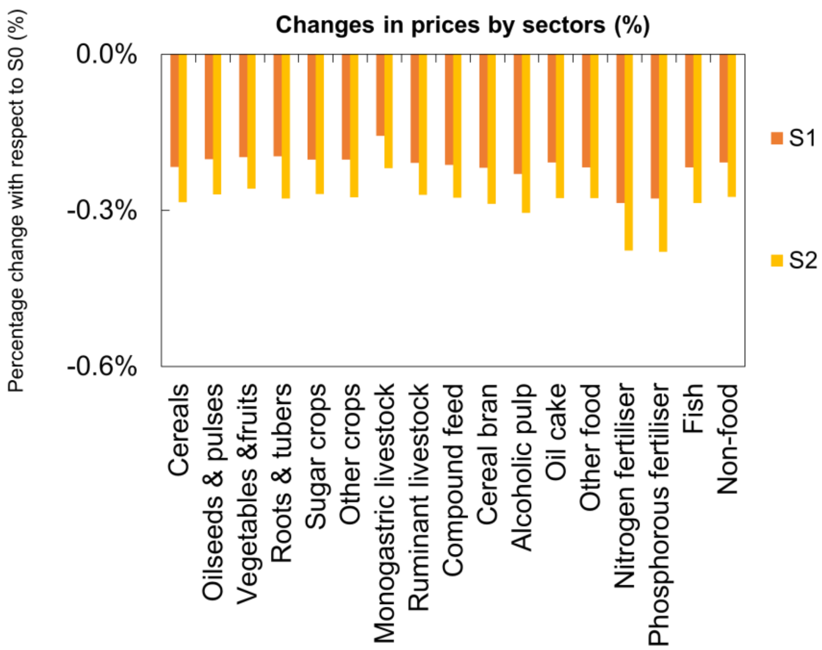
Supplementary Fig. 7 | Calorie availability per capita per day by food types in China and China's main food and feed trading partners (MTP) in S0.



Supplementary Fig. 8 | Decomposition of household income in China and China's main food and feed trading partners (MTP) in S0.



Supplementary Fig. 9 | Percentage changes (%) in (a) per capita affordability of the current diet, (b) household welfare, (c) wage, and (d) the average price of the current diet in China (CN) and China's main food and feed trading partners (MTP) in scenarios with respect to S0.



Supplementary Fig. 10 | Percentage changes (%) in prices by sectors in scenarios with respect to S0.

Supplementary Tables

Supplementary Table 1 | Physical quantities (Tg) for each product or service 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]	572.24	116.39
Roots &tubers [a]	133.14	54.76
Sugar crops [a]	133.61	792.67
Other non-food crops [a]	24.98	19.27
Monogastric livestock [a]	103.15	18.65
Ruminant livestock [a]	52.53	46.28
Fish [b]	12.51	0.66
Compound feed [c]	128.00	103.10
Cereal bran [d]	11.37	12.01
Alcoholic pulp [d]	3.41	76.09
Oil cake [d]	58.06	84.02
Grass [e]	286.22	0.00
Nitrogen fertiliser [a]	39.60	13.65
Phosphorous fertiliser [a]	17.43	3.13

[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 [6]. Here physical quantities of cereal grains waste, oilseeds &pulses waste, vegetables &fruits waste, and roots &tubers waste were excluded and presented in Table A3.
[b] Fish production data was derived from FAO [7].
[c] Compound feed production data was calculated according to the weighted averages of feeding crops included in the compound feed at the national level.
[d] Physical quantities of cereal bran, alcoholic pulp, and oil cake were estimated from the consumption of corresponding food products and specific technical conversion factors [8]. Here, physical quantities of cereal bran, alcoholic pulp, and oil cake only include quantities recycled as feed for monogastric livestock, and quantities collected as waste for landfill and incineration are excluded and presented in Table A3.
[e] Grass from natural grassland was derived from Miao and Zhang [9]. Here, grass refers to grass from natural grassland where ruminant livestock is grazing for feed, and grass from remaining grassland is not included. We do not present grass production data in MTP due to data unavailability.

Supplementary Table 2 | Physical quantities (Tg) of food waste and food processing by-products and their utilisation in the baseline (S0) for China.

	Total (Tg)	Used as feed (%)	Unused biomass (%) [c]
Cereals waste	36.09	39% [a]	Landfill (40%) & incineration (21%)
Vegetables & fruits waste	175.01	39% [a]	Landfill (40%) & incineration (21%)
Roots & tubers waste	13.32	39% [a]	Landfill (40%) & incineration (21%)
Oil seeds & pulses waste	1.27	39% [a]	Landfill (40%) & incineration (21%)
Cereal bran	31.34	36% [b]	Landfill (42%) & incineration (22%)
Alcoholic pulp	42.34	16% [b]	Landfill (55%) & incineration (29%)
Oil cake	84.66	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. [10]. Thus, we assumed that a similar proportion (39%, the mean of values in Japan and South Korea [11]) of food waste was being used as feed in China in 2014 in S0.

[b] The utilisation rates of by-products recycled as feed in China in 2014 in S0 were based on Fang, et al. [10].

[c] The current whereabouts of unused biomass were based on Kaza, et al. [12].

Supplementary Table 3 | Physical quantities (Tg) of food waste and by-product waste to food waste recycling service and food waste collection service in China in S0.

		Physical quantity (Tg)	
	Total (Tg)	Food waste recycling service [a]	Food waste collection service [a]
Cereal grains waste [b]	36.09	14.08	22.02
Vegetables & fruits waste [b]	175.01	67.76	107.25
Roots & tubers waste [b]	13.32	5.20	8.13
Oilseeds & pulses waste [b]	1.27	0.50	0.78
Cereal bran waste [c]	19.97	0.00	19.97
Alcoholic pulp waste [c]	38.94	0.00	38.94
Oil cake waste [c]	26.59	0.00	26.59
Total	311.19	87.53	223.66

[a] Physical quantities of food waste recycling service and food waste collection service refer to how much food waste is recycled as feed for monogastric livestock production and how much food waste is collected for landfill and incineration.

[b] Physical quantities of food waste (i.e., cereal grains waste, vegetables & fruits waste, roots & tubers waste, and oilseeds & pulses waste) were quantified separately for each type of food product using data on food consumption and China-specific food loss and waste fractions [13] following the FAO methodology [14]. In China, quantitative empirical data on food waste used 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. [10]. Thus, we assumed that a similar proportion (39%, the mean of values in Japan and South Korea [11]) of food waste was being used as feed in China in 2014 in S0, and the remaining food waste was collected for landfill and incineration.

[c] Physical quantities of by-product waste (i.e., cereal bran waste, alcoholic pulp waste, and oil cake waste) collected for landfill and incineration were estimated by detracting physical quantities of by-products recycled as feed for monogastric livestock (36%, 16%, 72% of total physical quantities of by-products according to Fang, et al. [10]) from total physical quantities of by-products.

Supplementary Table 4 | 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 involves sorting, shredding, thermal treatment, fermentation, hydrolysis, and extrusion to create animal feed, as outlined by Alsaleh and Aleisa [15]. Excluding the food waste recycled as feed, 66% of the remaining food waste in China in 2014 was collected for landfill, while 34% was incinerated, according to Kaza, et al. [12] and Bhada-Tata and Hoornweg [16]. 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 [15], Kaza, et al. [12] and Bhada-Tata and Hoornweg [16]. 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 5 | The economic and mass allocation of 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. [17], Mackenzie, et al. [18], Nyhan, et al. [19], and Pourmehdi and Kheiralipour [20]

Supplementary Table 6 | Food availability (kcal capita^{-1} day^{-1}) and the additional number of population (million people) to be fed as the current diet in China (CN) and China's main food and feed trading partners (MTP) in scenarios.

		Food availability (kcal capita^{-1} day^{-1})	Additional number of people to be fed as the current diet (million people)
S0	CN	3241.0	0
	MTP	3319.3	0
S1	CN	3247.1	2.6
	MTP	3318.8	-0.1
S2	CN	3253.1	5.2
	MTP	3318.4	-0.2

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
Grass	27	27	12	12	11.20	11.20
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	-

[a] The values were weighted averages of feed types included in the groups at the national level. Data were sourced from the NUFER database [21], MITERRA-EUROPE database [22], NRC [23], NRC [24], NRC [25], NRC [26], and China Feed–database Information Network Centre((http://www.chinafeeddata.org.cn/).

473 Supplementary Table 8 | Physical quantities of feed demand (Tg) by livestock sectors in China in scenarios.

Feed demand	Monogastric livestock (Tg)			Ruminant livestock (Tg)		
	S0	S1	S2	S0	S1	S2
Cereal grains	77.66	79.18	76.05	24.51	24.47	24.50
Oilseeds & pulses	3.15	3.31	3.24	0.74	0.73	0.74
Vegetables &fruits	11.84	15.33	17.12	3.17	3.17	3.17
Roots & tubers	2.75	2.97	2.93	0.74	0.74	0.74
Sugar crops	0.78	0.77	0.72	2.13	2.13	2.13
Compound feed	106.45	129.83	139.33	17.63	17.59	17.62
Cereal bran	11.08	12.36	12.52	-	-	-
Alcoholic pulp	6.67	7.05	7.01	-	-	-
Oil cake	59.83	28.51	34.79	-	-	-
Cereal grains waste	14.08	19.49	36.09	-	-	-
Vegetables & fruits waste	67.76	93.82	175.01	-	-	-
Roots & tubers waste	5.20	7.19	13.32	-	-	-
Oilseeds and pulses waste	0.50	0.69	1.27	-	-	-
Cereal bran waste	-	19.97	19.97	-	-	-
Alcoholic pulp waste	-	38.94	38.94	-	-	-
Oil cake waste	-	26.59	26.59	-	-	-
Grass	-	-	-	286.22	286.22	286.22
Total (Tg)	368	486	605	335	335	335

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Supplementary Table 9 | 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
Other 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
Food waste collection service [c]	Split from the original “Waste and water (wtr)” sector
Fish	“Fishing (Fsh)” sector

Aggregated sectors	GTAP original sectors
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 [27] and Bartelings, et al. [28].

[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 mongastric 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.

Supplementary Table 10 | 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	fsh	nf	CONS	XNET	TOT
cer	0	0	0	0	0	0	29229	9055	11363	1372	67	0	81831	0	0	0	0	61825	-2016	192727
osd	0	0	0	0	0	0	1002	230	8312	0	0	182	42993	0	0	0	0	5092	-34661	23150
vf	0	0	0	0	0	0	5685	1495	18959	0	0	0	98059	0	0	0	0	145756	-139	269815
rt	0	0	0	0	0	0	595	157	1986	0	0	0	10270	0	0	0	0	15265	-15	28259
sgr	0	0	0	0	0	0	192	515	1280	0	0	0	6619	0	0	0	0	24553	-903	32256
ocr	0	0	0	0	0	0	664	262	197	0	0	0	1021	0	0	0	0	1282	-1465	1963
oap	0	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	0	63546	-484	63062
cof	0	0	0	0	0	0	45882	7458	0	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	0	27	3398
pulp	0	0	0	0	0	0	800	0	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	0	-10	205
otf	0	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	0	-78	12721
pfe	2412	211	1542	169	83	163	0	0	0	0	0	0	0	0	0	0	0	0	-28	4551
fsh	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	15571	2154	17725
nf	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2547713	352518	2900231
LAD1	53323	7694	80962	8445	9849	396	0	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	0	-10240	0	0
LAB	94995	11819	148120	15450	17556	631	62255	24592	6707	959	155	8	89845	4413	1579	9208	1531587	-2019880	0	0
CAP	34602	2905	35711	3725	4455	151	23777	9057	5390	1067	180	15	102185	8308	2972	8517	1368643	-1611662	0	0
TRA	0	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	17725	2900231			

	cer	osd	vf	rt	sgr	ocr	oap	ctl	cof	bran	pulp	cake	otf	nfe	pfe	fsh	nf	CONS	XNET	TOT
cerw	0	0	0	0	0	0	754	0	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	0	8806		
rtw	0	0	0	0	0	0	278	0	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	0	64		
branw	0	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	0	3197		
cakew	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2184		

[a] Data source: GTAP [29]. 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=other food. nfe=nitrogen fertiliser. pfe=phosphorous fertiliser. fsh=fish. 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.

Supplementary Table 11 | 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	fsh	nf	CONS	XNET	TOT
cer	0	0	0	0	0	0	3794	34288	4450	1023	414	0	32927	0	0	0	0	16597	2016	95511
osd	0	0	0	0	0	0	69	301	3307	0	0	2009	17059	0	0	0	0	1938	34661	59344
vf	0	0	0	0	0	0	354	1110	8351	0	0	0	43966	0	0	0	0	50755	139	104675
rt	0	0	0	0	0	0	37	116	875	0	0	0	4605	0	0	0	0	5316	15	10963
sgr	0	0	0	0	0	0	58	1037	1598	0	0	0	7759	0	0	0	0	16038	903	27392
ocr	0	0	0	0	0	0	130	413	943	0	0	0	4929	0	0	0	0	13124	1465	21003
oap	0	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	0	214439	484	214923
cof	0	0	0	0	0	0	30067	32726	0	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	0	-27	4203
pulp	0	0	0	0	0	0	4967	0	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	0	10	2393
otf	0	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	0	78	4655
pfe	1547	1164	87	47	92	231	0	0	0	0	0	0	0	0	0	0	0	0	28	3195
fsh	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6983	-2154	4828
nf	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	13043344	-352518	12690826
LAD1	22886	13940	25013	2605	2260	5474	0	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	0	-15132	0	0
LAB	31115	17269	34446	3585	14182	5957	35369	71060	23869	1730	2795	231	203920	2038	1461	1581	8508850	-8959458	0	0
CAP	37435	26030	44998	4688	10603	8655	19600	58739	18547	1450	2155	153	198941	2618	1734	3247	4181976	-4621570	0	0
TRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	-312868	312868	
TOT	95511	59344	104675	10963	27392	21003	101056	214923	61939	4203	5365	2393	514107	4655	3195	4828	12690826			

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

[a] Data source: GTAP [29]. 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=other food. nfe=nitrogen fertiliser. pfe=phosphorous fertiliser. fsh=fish. 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.

Supplementary Table 12 | Total emissions of greenhouse gases (Tg CO_2 equivalents) in China (CN) and its main food and feed trading partners (MTP).[a]

	CN	MTP
Cereal grains	276.61	118.98
Oilseeds &pulses	8.33	9.88
Vegetables &fruits	54.88	3.34
Roots &tubers	7.46	0.82
Sugar crops	4.58	6.33
Other crops	15.55	20.73
Monogastric livestock	79.37	63.77
Ruminant livestock	245.04	700.30
Compound feed	25.39	16.03
Cereal bran	0.00752	0.00288
Alcoholic pulp	0.0001148	0.0000318
Oil cake	0.01580	0.01422
Other food	204.54	130.82
Nitrogen fertiliser	324.09	80.29
Phosphrous fertiliser	24.53	9.06
Fish	0.00	0.00
Non-food	10238.21	6825.11
Food waste recycling service	16.37	0.00
Food waste collection service	221.98	0.00
Total	11746.93	7985.49

[a] Data source: Climate Analysis Indicators Tool (CAIT) [30]. Emissions related to N fertiliser production were allocated to the N fertiliser sector, while emissions related to N fertiliser application were distributed to the crop sectors. The data on N and P fertiliser use by crop types and countries were derived from Ludemann, et al. [31]. Emissions of by-products (i.e., cereal bran, alcoholic pulp, oil cake) were derived from Mackenzie, et al. [18]. Emissions of food waste recycling service and food waste collection service were obtained from Alsaleh and Aleisa [15], Hong, et al. [32], and Hong, et al. [33]

Supplementary Table 13 | Total emissions of acidification pollutants (Tg NH_3 equivalents) in China (CN) and its main food and feed trading partners (MTP).[a]

	CN	MTP
Cereal grains	3.94	0.94
Oilseeds &pulses	0.29	0.15
Vegetables &fruits	1.89	0.05
Roots &tubers	0.26	0.01
Sugar crops	0.16	0.09
Other crops	0.54	0.34
Monogastric livestock	5.22	2.88
Ruminant livestock	2.21	1.05
Compound feed	0.04	0.02
Cereal bran	0.000328	0.000126
Alcoholic pulp	0.00000067	0.00000019
Oil cake	0.00080	0.00073
Other food	0.35	0.16
Nitrogen fertiliser	0.0009	0.0035
Phosphrous fertiliser	0.0007	0.0029
Fish	0.00	0.00
Non-food	18.10	8.21
Food waste recycling service	0.06	0.00
Food waste collection service	0.56	0.00
Total	33.61	13.92

[a] Data source: Liu, et al. [34], Huang, et al. [35], and Dahiya, et al. [36]. Emissions of by-products (i.e., cereal bran, alcoholic pulp, oil cake) were derived from Mackenzie, et al. [18]. Emissions of food waste recycling service and food waste collection service were obtained from Alsaleh and Aleisa [15], Hong, et al. [32], and Hong, et al. [33]

Supplementary Table 14 | Total emissions of eutrophication pollutants (Tg N equivalents) in China (CN) and its main food and feed trading partners (MTP).[a]

	CN	MTP
Cereal grains	1.04	0.06
Oilseeds &pulses	0.15	0.05
Vegetables &fruits	0.88	0.04
Roots &tubers	0.12	0.01
Sugar crops	0.02	0.01
Other crops	0.01	0.01
Monogastric livestock	0.58	0.38
Ruminant livestock	1.63	2.02
Compound feed	0.17	0.07
Cereal bran	0.0000147	0.0000056
Alcoholic pulp	0.00000029	0.00000008
Oil cake	0.000037	0.000034
Other food	1.35	0.56
Nitrogen fertiliser	0.0002	0.0007
Phosphrous fertiliser	0.0002	0.0009
Fish	0.00	0.00
Non-food	3.66	2.40
Food waste recycling service	0.0303	0.0000
Food waste collection service	0.2790	0.0000
Total	9.92	5.61

[a] Data source: Hamilton, et al. [37]. Emissions of by-products (i.e., cereal bran, alcoholic pulp, oil cake) were derived from Mackenzie, et al. [18]. Emissions of food waste recycling service and food waste collection service were obtained from Alsaleh and Aleisa [15], Hong, et al. [32], and Hong, et al. [33]

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