Rebound effects may undermine benefits of upcycling food waste and food processing by-products as animal feed in China

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Abstract

Upcycling food waste and food processing by-products as animal feed could reduce environmental impacts of livestock production, but rebound effects, where lower feed costs lead to livestock production expansion, may diminish these benefits. Using an integrated environmental-economic model, we assess the impacts of upcycling food waste and food processing by-products as feed in China's monogastric livestock production. We find that the upcycling increases monogastric livestock production by 23-36% and raises total emissions of acidification pollutants in China by 2.5-4.0%. In contrast, domestically total greenhouse gas emissions decrease by 0.5-1.4% through less food waste and food processing by-products in landfills and incinerators and contraction of the non-food sector. This upcycling, accompanied by resource reallocation across the entire economy, enhances food security and had significant knock-on effects beyond the agricultural sectors, thereby influencing sectoral employment, gross domestic product, and household welfare. Implementing appropriate emission taxes provides an opportunity to absorb the rebound effects on emissions but may negatively affect food security indicators and shift emission-intensive sectors from China to its trading partners, depending on the height of the taxes. Our study, thus, supports policy design aimed at achieving environmental sustainability and food security.

Keywords

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

Main

Animal-sourced food (ASF), such as meat, milk, and eggs, is the main contributor to the environmental impacts of food systems, including global warming potential (GWP), acidification potential (AP), and eutrophication potential (EP) [1,2]. The global demand for ASF, driven by population growth and increased prosperity and urbanisation, is expected to double by 2050, while growth is more pronounced in low-income countries in Sub-Saharan Africa and South Asia [3-5]. This surge in livestock production has exacerbated food-feed competition and significantly contributed to the exceedance of the planetary boundaries (PBs) for emissions of greenhouse gases (GHGs), acidification pollutants, and eutrophication pollutants [6]. Currently, 70% of global agricultural land is used for producing animal feed [7], and global livestock production accounts for 13-18% of the total anthropogenic greenhouse gas (GHG) emissions [8], 40% of the ammonia (NH_3) and nitrous oxide (N_2O) emissions [9], and around 24% of nitrogen (N) and 55% of phosphorus (P) losses to water bodies [10]. Given that livestock production dominates the environmental impacts of food systems, achieving the global 1.5°C climate target and reducing emissions of acidification and eutrophication pollutants will remain challenging without mitigating emissions from livestock production [11-13].

Globally, the Food and Agriculture Organization (FAO) estimated in 2011 that one-third (1.3 billion tons per year) of food produced for human consumption was lost or wasted [14]. In 2021, the World Wildlife Fund (WWF) estimated this figure at 40% (2.5 billion tons per year) [15], which includes on-farm food losses (not included in earlier studies) and uses updated data on losses in supply chains and waste at retailing and consumption, offering a more comprehensive assessment of food loss and waste. A large proportion of food waste ends up in landfills or incinerators, exacerbating GHG emissions and associated climate change [16]. Upcycling food waste and food processing by-products (also called “low-opportunity-cost feed products (LCFs)”), as animal feed is, thus, crucial for reducing environmental impacts and building more cir cvbbfgydcular food systems, as it offers a pathway to mitigate land-related pressures [17], alleviate the food-feed competition [18], and reduce emissions from food systems and improper food waste disposal [19]. By upcycling food waste and food processing by-products as animal feed, livestock could recycle nutrients that would otherwise be lost in food production [20]. This upcycling prioritises arable land for food rather than feed production, allowing livestock to support the food supply without requiring additional arable land, thereby enhancing food security, reducing the environmental impact of food systems [17-19], and contributing to achieving Sustainable Development Goals (SDGs), including SDG 2 (zero hunger), SDG 6 (clean water and sanitation), SDG 13 (climate action), and SDG 15 (life on land) [21].

While many studies acknowledge the environmental benefits of upcycling food waste and food processing by-products as animal feed, some gaps remain in the existing literature. First, previous studies [17-19] employing linear optimisation models to evaluate the environmental impacts of a transition towards circularity may have overestimated the environmental benefits by disregarding "rebound effects" (also known as "Jevons paradox") [22,23]. The rebound effect, where lower feed costs lead to livestock production expansion, may diminish the environmental benefits of feeding animals with food waste and food processing by-products. The "rebound effect" has been extensively studied in energy systems [24,25], but studies of its implications in food systems are largely lacking. Although previous studies have explored rebound effects related to a global dietary shift towards plant-based food [26] and halving food loss and waste [27], there is still limited understanding of the rebound effect of upcycling food waste and food processing by-products as animal feed. Second, strategies to absorb these negative rebound effects resulting from upcycling food waste and food processing by-products as animal feed have not yet been explored. Implementing synergistic emission taxes that encompass emissions of GHGs and pollutants leading to acidification and eutrophication is considered an effective policy instrument to identify the most economically cost-effective mitigation pathway for achieving given emission mitigation targets [28-31]. Such emission taxes could drive emission reductions by curbing production in emission-intensive sectors (e.g., livestock) while incentivising producers and consumers to transition from emission-intensive activities, commodities, and technologies to cleaner alternatives, ensuring a cost-effective mitigation strategy in the entire economy. Thus, a coordinated strategy that integrates upcycling with emission taxes is essential to help absorb the rebound effects on emissions. For example, several countries, such as the United States, France, Canada, and New Zealand, have implemented various forms of carbon taxes to mitigate GHG emissions [32]. However, unilateral carbon taxes may lead to "carbon leakage", as the production of emission-intensive goods may shift from carbon-constrained regions to those with weaker or no carbon regulations, thereby reducing policy effectiveness [33,34]. As climate change is a global public good, effective GHG mitigation requires coordinated international action, such as the recent commitments to net-zero targets under the Paris Agreement [35,36]. Additionally, an integrated tax plan for taxes on emissions of carbon dioxide (CO_2), nitrogen oxides (NO_x), and sulphur dioxide (SO_2) from energy use in China could reduce socioeconomic and welfare costs by 50% compared to independent plans [31]. This underscores the importance of an integrated tax strategy, combining carbon and other environmental taxes, to achieve a win-win situation for the economy and the environment.

Our study focuses on China because it is the world's largest animal producer, accounting for 46%, 34%, and 13% of global pork, egg, and poultry meat production, respectively, in 2018 [5]. Moreover, around 27% of food produced for human consumption is lost or wasted in China [37], implying a great opportunity to upcycle the discarded food waste as feed. In addition, the Chinese government has proposed to lower the agricultural product processing loss to below 3% by 2035 [38], and to substitute human-edible feed ingredients, such as soybeans and maize, in animal feed with food processing by-products [39]. Furthermore, the Chinese government recently released a national plan to reduce concentrated feedstuffs such as soybean and maize in pig and chicken production sectors through improved feeding strategies, including upcycling food waste and food processing by-products as animal feed [40]. Evidently, there is a great need to better understand potential rebound effects that may influence the expected benefits of upcycling food waste and food processing by-products as animal feed before this action plan is widely implemented in China.

To address these knowledge gaps, this study uses an integrated environmental-economic applied general equilibrium (AGE) modelling approach to assess the impacts of the environmental and economic impacts of upcycling food waste and food processing by-products as feed in China's monogastric livestock production, capturing both domestic effects in China and cross-border impacts on its main food and feed trading partners (MTP, including Brazil, the United States, and Canada) through bilateral trade. We also explore how implementing regional uniform emission taxes across all sectors on economy-wide emissions (i.e., total emissions from all sectors in the entire economy) of GHGs (including CO_2, methane (CH_4), and N_2O), acidification pollutants (including NH_3, NO_x, and SO_2), and eutrophication pollutants (including N and P losses to water bodies) in China and MTP could absorb the rebound effects of this upcycling while safeguarding food security. We address three main research questions. First, how will an increased utilisation of food waste and food processing by-products as feed influence livestock production, food supply, and other non-agricultural sectors in China and MTP? Second, how will an increased utilisation of food waste and food processing by-products influence economy-wide emissions of GHGs, acidification pollutants, and eutrophication pollutants, as well as food security indicators (i.e., average food price, food affordability, population at risk of hunger, and food availability)? Third, how will emission taxes absorb the rebound effects of this upcycling while safeguarding food security?

We examine five scenarios: (i) the baseline (S0) scenario represents the economic and environmental conditions of all sectors (including agriculture, industries, and services) in the entire economies of China and MTP in 2014; (ii) scenario S1 involves partially upcycling food waste and food processing by-products (54% of food waste and 100% of food processing by-products) as feed for monogastric livestock production in China; (iii) scenario S2 involves fully upcycling food waste and food processing by-products (100% of food waste and 100% of food processing by-products) as feed for monogastric livestock production in China; (iv) scenario S3 adds a modest emission mitigation target to S1 which entails implementing regional uniform emission taxes across all sectors to ensure that economy-wide emissions of GHGs, acidification pollutants, and eutrophication pollutants in China and MTP do not exceed their baseline (S0) levels; (v) scenario 4 adds an ambitious emission mitigation target to S1 which entails implementing regional uniform emission taxes across all sectors to meet China's and the MTP's annual economy-wide GHG mitigation targets under the Intended Nationally Determined Contributions (INDC) of the Paris Agreement [35,36], as well as China's emission reduction goals for economy-wide emissions of acidification and eutrophication pollutants in line with the "14th Five-Year Plan" [41]. In S1, cross-provincial transportation of food waste with high moisture content is not allowed, which limits the maximum utilisation rate of food waste to 54% in China, according to Fang, et al. [17], whereas it is allowed in S2. We consider food waste (cereal grains waste, vegetables & fruits waste, roots & tubers waste, and oilseeds & pulses waste) during distribution, retailing, and consumption (both households and out-of-home), as well as food processing by-products (cereal bran, alcoholic pulp, and oil cakes). When substituting primary feed (i.e., feed crops and compound feed) in animal diets with food waste and food processing by-products, the total protein and total energy supplies per unit of animal output are kept constant in all scenarios. Detailed information on the scenarios and sensitivity analysis is provided in Supplementary Information (SI).

Results

Overview of current utilisation of food waste and food processing by-products.

According to the FAO [5], China produced about 104 Tg (1 Tg = 10^6 tons) of monogastric livestock products (pork: 57 Tg; poultry meat: 18 Tg; egg: 29 Tg) and 53 Tg of ruminant livestock products (milk: 42 Tg; beef: 6 Tg; lamb: 4 Tg) in 2014. We estimate that 226 Tg food waste (equivalent to 54 Tg in dry matter; 7 Tg in crude protein; 690 billion MJ in energy) and 155 Tg food processing by-products (equivalent to 139 Tg in dry matter; 49 Tg in crude protein; 1907 billion MJ in energy) was available in China in 2014, but only 39% of the food waste and 51% of the food processing by-products were recycled as feed for monogastric livestock production, with the remainder disposed in landfills and incinerators (Supplementary Tables 3-4). The limited use of food waste for feed production in China is primarily due to the early stage of industrialization of recycling food waste as feed, which currently has a low processing capacity [42], and the reliance of industrialized livestock production on concentrated feed [3]. In addition, despite being protein-rich, food processing by-products, such as unprocessed oil cakes, contain anti-nutritional factors that may hinder protein absorption by animals. Although fermentation can effectively eliminate these anti-nutritional factors and enhance digestion and growth performance [43], its limited adoption in China leads to a large amount of these by-products being discarded in landfills or incinerators.

Rebound effects of livestock production expansion.

Unlike previous studies that considered recycling food waste and food processing by-products as feed to be costless [17-19], we assume that increasing costs of more recycled food waste and food processing by-products as feed are born by monogastric livestock producers, and consumers benefit from decreasing costs associated with less waste ending up in landfills and incinerators. We find that upcycling 54-100% of food waste and 100% of food processing by-products as feed in scenarios

S1 and S2 increases the share of food waste and food processing by-products used as feed within the total feed use in dry matter from 43% in S0 to 53-58% in S1 and S2 (Supplementary Fig. 2b). Upcycling increases the supply of feed protein by 27-40% and feed energy by 26-39%, and reduces total feed cost (including feed crops, compound feed, food waste, and food processing by-products) per unit of monogastric livestock production by 2.1-3.0%. Consequently, the upcycling expands monogastric livestock production by 23-36% in S1 and S2 (Fig. 2b). This expansion improves China's comparative advantage in monogastric livestock trade in the global market, transforming it from a net importer (importing 1% of output in S0) to a net exporter (exporting 18-25% of output in S1 and S2) (Fig. 2e) while displacing production in its trading partners, which declines by 41-63% (Supplementary Fig. 8b,d). As a result, total monogastric livestock production across China and its trading partners increases only slightly (0.08-0.18%), leading to a minute decline (0.11-0.19%) in the global monogastric livestock price (Supplementary Fig. 15). Ruminant livestock production decreases by 3% as the expansion of monogastric livestock reduced the availability of feed crops and compound feed to ruminant livestock (Fig. 2b). To meet domestic demand, ruminant livestock imports rises from 1% of output in the baseline (S0) to 4% (Fig. 2e).

Expanded monogastric livestock production raises the demand for primary feed (i.e., feed crops and compound feed), which surprisingly outweigh the reduction in primary feed use by substituting it with food waste and food processing by-products. The overall feed demand for both monogastric and ruminant livestock increases by 17-34% due to a 33-67% rise in feed demand in fresh form for monogastric livestock (Fig. 3b). The upcycling increases the feed conversion ratio (FCR, the ratio of fresh feed inputs to live weight gain) for monogastric livestock by 0.22-0.62 kg kg^{-1}, but decreases the edible feed conversion ratio (eFCR, the amount of human-edible feedstuffs, i.e., feed crops and compound feed, used for per unit of live weight gain) by 0.11-0.19 kg kg^{-1}, indicating its reduced reliance on human-edible feedstuffs (Supplementary Fig. 3a). Since feed crops and compound feed account for only 12% of ruminant feed (compared to 88% from grass, see Supplementary Fig. 4d), upcycling has a minor impact on ruminant production and its FCR and eFCR (Supplementary Fig. 3b). The growing demand for crops used as animal feed increases reliance on crop imports, with the import share rising from 11% in the baseline (S0) to 15-19% (Fig. 2d), considering that the total crop production declines by 1.2-4.4% (Fig. 2a). Despite the decline in crop production, the cultivated crop area expands by 0.6-13% (Fig. 3a), driven by higher labor costs (Supplementary Fig. 5) and reduced labor availability (Supplementary Fig. 7), which incentivise crop producers to substitute labour with increased cropland use. Detailed impacts on crop production structure, as well as the use of nitrogen and phosphorus fertilisers, are presented in Supplementary Results (see Supplementary Figs. 4 and 6).

Adjustments in crop and livestock production also have knock-on effects beyond the agricultural sectors in the broader economy, thereby influencing sectoral employment, gross domestic product (GDP), and household welfare (a measure of economic well-being in US dollars). We observe that the increase of 11.5-18.4 million people employed in monogastric livestock production is largely a transfer from the non-food sector (i.e., industries and services; detailed in Appendix Table 1) (Supplementary Fig. 7a,c). Output in the non-food sector declines slightly by 1.0-1.4% (Supplementary Fig. 8a,c) with an absolute loss of 28-41 billion US dollars (USD, 2014 constant price) (Supplementary Fig. 9a). In contrast, nitrogen and phosphorus fertiliser production surges by 35-36% and 20-59% (Fig. 2c), respectively, due to rising demand and decreased production costs, as the shrinking non-food sector improves the availability of inputs to fertiliser production. As a consequence, China becomes an exporter of nitrogen and phosphorus fertiliser (Fig. 2f). The absolute value of fertiliser output rises by 5.4-7.0 billion USD (Supplementary Fig. 9a), which compensates for less than one-fifth of the total output decrease of the non-food sector. The economic losses in the crop and non-food sectors are largely offset by the expansion of the monogastric livestock and fertiliser sectors (Supplementary Fig. 9a). The overall impact on China's economy is a 0.02-0.07% (0.8-2.6 billion USD) decrease in GDP (Supplementary Fig. 11) and a slight positive impact on household welfare (0.18-0.32%) (Supplementary Fig. 12).

Asymmetric impacts of upcycling food waste and food processing by-products on food security and environment sustainability.

We find that the 23-36% expansion in monogastric livestock production under scenarios S1 and S2, along with its knock-on effects beyond the agricultural sectors, increase Chinese economy-wide emissions of acidification pollutants by 2.5-4.0% (Fig. 4b) and eutrophication pollutants by ±0.2% (Fig. 4c). In contrast, the 0.5-1.4% decrease in economy-wide GHG emissions in China is caused by less food waste and food processing by-products in landfills and incinerators and contraction of the non-food sector (Fig. 4a). Economy-wide emissions in MTP are reduced by 1.1-1.3% for GHGs, by 8-13% for acidification pollutants, and by 2.5-4.0% for eutrophication pollutants. These environmental benefits for MTP arise from a reduction in their domestic livestock and fertiliser production, as China shifts from a net importer to an exporter of livestock products and fertilisers (Fig. 2e,f).

For assessing food security, we use four indicators covering two dimensions: two indicators for food availability, i.e., dietary energy availability and the population at risk of hunger; two indicators for food access, i.e., cereals affordability for labour force and the average food price (including primary food products and processed food). Population at risk of hunger is estimated by multiplying the prevalence of undernourishment (PoU) by the total population, with PoU determined based on dietary energy availability from our model and other parameters following the FAO-based approach [44,45]. Cereals affordability for labour force is estimated by subtracting changes in the average wage across the entire economy from fluctuations in cereal prices. Our findings suggest that upcycling, accompanied by resource reallocation across the entire economy, enhances food security in China without compromising that of its trading partners. In addition, the reduced cost of collecting food waste and food processing by-products for landfill and incineration enables consumers in China to allocate more of their income to food consumption. Since the cost of food waste collection for landfill and incineration was quite small in the baseline (S0), the impact of reduced collection costs only has a modest positive effect on most food security indicators. Globally, the average food price declines by 0.1-0.2% (Fig. 5a,e). In China, dietary energy availability increases by 0.2-0.3%, and the population at risk of hunger, representing 17% of the global population at risk of hunger, decreases by 1.6-3.2% (Fig. 5c,d). Cereals affordability for labour force increases by 0.3-0.5% (Fig. 5b), as a result of a rise in the average wage across the Chinese economy (0.13-0.22%) (Supplementary Fig. 5) and a decrease in cereals price (0.16-0.26%) (Supplementary Fig. 15).

Absorbing rebound effects through emission taxes.

The modest mitigation target of S3 absorbs the rebound effects estimated for S1 of upcycling food waste and food processing by-products as feed in China (Fig. 4) and safeguards global food security.

Changes in food security indicators under S3 are nearly identical to those in S1 (Fig. 5). This is due to the implementation of a low tax rate on emissions of acidification pollutants (3 \$ ton^{-1} NH_3-eq) in China. The reduction in emissions of all pollutants in S3 is mainly attributed to a decrease in total crop production compared to S1 (Fig. 2a; Fig 4; Supplementary Fig. 14a,b,c). Monogastric and ruminant livestock production decreases slightly by 0.40% and 0.03%, respectively, in scenario S3 compared to S1 (Fig. 2b). The reduction in total feed cost per unit of monogastric livestock production in S3 remains virtually unchanged from S1. Phosphorus fertiliser production increases by 40% while nitrogen fertiliser production decreases by 6% compared to S1 (Fig. 2c). As a result, emissions increases in MTP compared to S1 (Fig. 4) due to a shift of emission-intensive production from China to MTP. Nonetheless, emissions of all pollutants in MTP still remain below baseline (S0) levels.

The ambitious emission mitigation target of S4 counteracts the rebound effects further and achieves a further emission reduction but poses a risk to food security, as the average global food price increases by 9.4% (Fig. 5a,e) and cereals affordability for labour force decreases by 20.2% in China (Fig. 5b) and by 14.5% in MTP (Fig. 5f). The negative impact on food security in China and MTP is a result of the higher tax rates on emissions in both regions (5 \$ ton^{-1} CO_2-eq , 788 \$ ton^{-1} NH_3-eq, and 6969 \$ ton^{-1} N-eq in China; 2.5 \$ ton^{-1} CO_2-eq in MTP). Emission taxes on acidification and eutrophication pollutants are significantly higher than those on GHGs because their lower emission levels compared to GHGs (see Appendix Tables 5-7) required higher tax rates to achieve the same mitigation target. Food availability in MTP decreases by 3.3%, while it increases by 3.6% in China (Fig. 5d,h). The rise in food availability in China is primarily due to the reallocation of crops from animal feed to human food consumption (Supplementary Fig. 4c) following the reduction in livestock production in S4 compared to S1 (Fig. 2b). Additionally, this increase is further supported by a dietary shift in China from animal-based food to more affordable, energy-dense plant-based food (Supplementary Table 8), which accounts for 84% of food availability in S0 (Supplementary Fig. 16a). Consequently, the population at risk of hunger in MTP increases by 346% but declines in China by 36% (Fig. 5 c,g). The 2.6% and 2.5% reduction in economy-wide emissions of GHGs and acidification pollutants in China in S4 are largely driven by the non-food production contraction compared to S1 (Fig. 4a,b). The 2.0% reduction in economy-wide emissions of eutrophication pollutants (Fig. 4c) in China is primarily driven by 16% less monogastric livestock production and a 7% decline in ruminant livestock production in S4 compared to S1 (Fig. 2b; Supplementary Fig. 14f). The total feed cost per unit of monogastric livestock production in S4 decreases by an additional 2.3% compared to S1, driven by a shift in feed composition from human-edible feedstuffs (i.e., feed crops and compound feed) to less expensive food processing by-products and food waste. This transition is reflected in a further 0.07 kg kg^{-1} reduction in eFCR for monogastric livestock (Supplementary Fig. 3a). For MTP, the 2.0% reduction in economy-wide GHG emissions can largely be attributed to reductions in total crop and livestock production (Fig. 4a). Meanwhile, economy-wide emissions of acidification and eutrophication pollutants decrease both by 5% in MTP (Fig. 4b,c).

Discussion

We explore the possible environmental and economic consequences of upcycling food waste and food processing by-products in China's monogastric livestock production, and provide possible solutions to absorb the rebound effects in China and safeguard global food security. Our study serves as a step towards bridging monetary AGE models with biophysical and nutritional (i.e., protein and energy) constraints. Our integrated environmental-economic framework complements previous linear optimisation studies [17-19], which overlooked market-mediated effects via the price system. Our modelling framework captures the indirect "rebound effect" of livestock production expansion induced by lower feed costs and its knock-on effects beyond the agricultural sectors, which may undermine the expected environmental benefits in the transition to more circular food systems. Further, we show that changes in China's food systems has significant cross-border impacts on its trading partners through bilateral trade.

Upcycling food waste and food processing by-products as animal feed.

The primary challenges in upcycling food waste and processing by-products as animal feed are concerns over food and feed safety and potential animal health risks. For example, European Union (EU) legislation prohibits food waste in animal feed due to disease transmission concerns [46]. In contrast, feeding animals with food waste is more prevalent in Asian countries such as China, South Korea, and Japan, driven by growing demand for animal-sourced food, resource constraints that prioritise food production over feed, and the preference for low-cost alternative feeds among small-scale farms [16]. Extensive field-based evidence has demonstrated that feeding animals with properly treated food waste is safe for animals with minimal health risks [47]. Thermal treatment methods, including heating, drying, and dehydration, are the most commonly used approaches to effectively reduce pathogen transmission risks and ensure food and feed safety [16]. While upcycling food waste as feed has been shown not to affect livestock productivity [16], to gain acceptance and adoption among livestock producers, livestock production from food waste must demonstrate its economic competitiveness against conventional feed [47]. Upcycling food waste and food processing by-products as feed necessitates various investments and policies to support the construction of municipal food waste collection plants to efficiently collect, sanitize, and package discarded food waste and food processing by-products for sale to livestock producers as feed [17]. Achieving near-full use of food waste and food processing by-products as feed appears feasible in China in the future due to several reasons. First, the food waste treatment industries (i.e., food waste collection service and food waste recycling service) have seen significant development and expansion in recent years [48]. Second, reinforced policies on municipal solid waste separation and collection guarantee a stable feed supply for monogastric livestock production [49,50]. For example, the Chinese government recently launched an action plan to reduce reliance on soybean imports, which includes a key initiative to give a trial to feed production from food waste in 20 cities by 2025 [51]. Additionally, the geographic proximity of industrial livestock farms to municipal food waste processors in China further facilitates the feasibility of upcycling [48].

Rebound effects of upcycling food waste and food processing by-products as animal feed.

Our findings are particularly informative for policymakers focusing on reducing the environmental impact of food systems and enhancing food security, as we unveil the asymmetric impacts of upcycling food waste and food processing by-products as feed on food security and environment sustainability. A decreased eFCR for monogastric livestock reflects reduced reliance on human-edible feedstuffs per unit of production. While these benefits align with prior findings, our study additionally identifies rebound effects of expanded livestock production, its knock-on effects beyond the agricultural sectors, and cross-border impacts on other countries, which were missing in previous linear optimisation studies [17-19]. We find that partially or fully upcycling food waste and by-products as feed, intended to reduce livestock demand for human-edible feedstuffs and lower emissions, can backfire, as a 2.1-3.0% reduction in feed costs drives a 23-36% expansion in monogastric livestock production, ultimately increasing emissions. This livestock expansion is consistent with findings by Tong, et al. [52], who argued that upcycling food waste as feed could increase pork production in China by 14-29%, even when costs and prices remain constant. Additionally, this expansion, along with its knock-on effects beyond the agricultural sectors, increase economy-wide emissions of acidification and eutrophication pollutants in China by 2.5-4.0% and by ±0.2%, respectively, in scenarios S1 and S2. In contrast, the 0.5-1.4% decrease in economy-wide GHG emissions in China is caused by less food waste and food processing by-products in landfills and incinerators and contraction of the non-food sectors. China's trading partners obtain environmental benefits through reducing their domestic livestock and fertiliser production, as China shifts from a net importer to an exporter of livestock products and fertilisers. This upcycling, accompanied by resource reallocation across the entire economy, enhances food security in China without compromising that of its trading partners. Our estimation of the rebound effect aligns with Wang, et al. [53], who, using an agro-economic model, found that accelerated investments in technology and infrastructure, which drive crop yield increases in China, would not only increase GHG emissions from agriculture, forestry, and other land-use sectors due to expanded crop production for export but also improve domestic food security by lowering food prices. Our results also echo the findings of Hegwood, et al. [27], who argued that rebound effects could offset more than half of avoided food loss and waste, thereby reducing environmental benefits while enhancing food security. Our analysis, thus, enhances the understanding of synergies and trade-offs between economic impacts and multiple environmental stresses associated with upcycling food waste and food processing by-products as animal feed.

Interconnection between food security and environmental sustainability.

Our study highlights the need to integrate both food security and environmental sustainability into policy decisions to leverage potential win-win opportunities, especially under the current challenges such as climate change and resource constraints. In essence, policymakers should pay closer attention to the interconnection between food security and environmental sustainability to better leverage potential synergies and minimise trade-offs [54]. The reduction in GHG emissions, coupled with the enhancements in food security, underscores the rationale for policymakers to promote upcycling food waste and food processing by-products as feed. This also aligns with China's recent emphasis on carbon neutrality and food security as leading priorities [55,56]. However, policymakers should remain vigilant regarding indirect effects and spillovers, particularly the unintended increases in emissions of acidification and eutrophication pollutants. We implement two emission taxes to absorb the rebound effects of upcycling food waste and food processing by-products as feed in China. Our findings show that ambitious emission taxes counteract rebound effects but lead to a 9.4% rise in food prices, thereby threatening global food security. This aligns with findings of Hasegawa, et al. [29], who revealed the risk of increased food insecurity under stringent global climate change mitigation policy. Conversely, modest emission taxes provide an opportunity to absorb the rebound effects in China and safeguard global food security. Therefore, to avoid unintended negative environmental impacts and achieve the dual dividend of environmental sustainability and food security, it is essential to carefully design and implement tailored, complementary policies rather than relying on a single, one-size-fits-all solution. In China, the responsibility for food security and environmental sustainability falls on different government agencies, highlighting the pressing need for improved coordination and consistency within the government to effectively tackle these intertwined issues [57]. In addition, a globally coordinated mitigation policy is imperative for reducing the exceedance of the planetary boundaries for emissions of GHGs, acidification pollutants, and eutrophication pollutants, as unilateral environmental policies can lead to "carbon leakage" by outsourcing the production of emission-intensive goods to countries which lack environmental regulations [33].

Despite its integrated approach, this study has some limitations that necessitate some follow-up. First, while some simplifications, such as assumptions of fixed budget shares for consumers, fixed cost shares for producers, and the absence of trade barriers, in our model may exaggerate trends, they are appropriate for illustrating rebound effects. Second, our model overlooks sub-national heterogeneity, and future research could address this by improving spatial resolution to provide more precise region-specific policy insights. Third, we use dollar-based shares to allocate physical material flows without accounting for variations in product quality along the global supply chain. This may introduce uncertainties in converting dollar-based quantities to physical quantities, yet it remains a common approach [26,58]. In the absence of a universally accepted method to resolve this issue, future research could further address this limitation. Fourth, our estimates of impacts are performed in the static modelling framework based on the current economic conditions. It does not account for long-term dynamics (e.g., population growth, economic development, evolving trade policies) or external shocks (e.g., African swine fever, the US-China trade war, COVID-19), which could reshape crop and livestock production portfolios and impact global food security and environmental sustainability. Future research could incorporate dynamic modelling and extra scenario analyses to better capture these uncertainties. Since we recognise the importance of accounting for uncertainties in model results, we conducted a sensitivity analysis and decomposed uncertainties into five major sources: (1) feasibility of upcycling food waste and food processing by-products as feed; (2) conversion of dollar-based quantities to physical quantities; (3) substitution of cropland with other inputs for crop production; (4) cereal self-sufficiency target; (5) cleaner crop and livestock production technology. While potential data variations may moderately influence the magnitude of our results, they do not alter the overall trends of food security indicators and environmental impacts, and our main conclusions remain plausible. Further details on these limitations and uncertainties are detailed in the Supplementary Discussion. In essence, our integrated environmental-economic framework supports policy design aimed at achieving the dual dividend of environmental sustainability and food security. Our analysis holds significant policy implications not only for China, a key global market for food and feed, but also serves as a blueprint for other populous emerging economies striving to achieve a better balance between food security and environmental sustainability.

Methods

The integrated environmental-economic model and database.

The integrated environmental-economic model based on an applied general equilibrium (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 [59-63]. For this study, we developed a global comparative static AGE model, a modified version of an integrated environmental-economic model, [33,64-68] and enhanced sectoral representation for agricultural (6 crop types and 2 livestock categories) and non-agricultural (compound feed, food processing by-products, processed food, fertilisers, food waste treatment, and non-food) sectors (see Fig. 1). While the static model limits its applicability to short-term policy analysis, prior studies have shown that it minimises assumptions and uncertainties about future conditions on population and economic growth [30,69]. This allows us to isolate the impact of upcycling food waste and food processing by-products as animal feed in China under current economic conditions.

AGE models grounded in microeconomic theory represent the entire economy by integrating consumer demand, producer decisions, and market clearing into a unified framework. Consumers maximise utility by allocating income across goods and services within budget constraints, given prices and initial endowments. Producers maximise profits by selecting optimal input combinations based on production technology and given prices under perfect competition, following a zero-profit condition. This condition means that output values match input costs, preventing excess profits in constant returns to scale firms, as new firms increase supply, lower prices, and drive profits to zero, while firms incurring losses will exit the market, maintaining market equilibrium. The market clearance condition states that a market is in equilibrium when total supply equals total demand. In line with this principle, the economy reaches equilibrium when total supply matches total demand across all markets, with relative prices adjusting until consumers and producers can meet their effective demand and supply. Total supply consists of domestic production and imports, while total demand includes intermediate use by firms, household consumption, and exports. The resulting equilibrium prices ensure that all markets are cleared. For international trade, our AGE model adopted the Heckscher-Ohlin (H-O) trade assumption, treating domestic and imported goods as perfect substitutes. Under this assumption, production occurs in countries with comparative advantages, meaning goods are produced where they can be most efficiently produced. Detailed specifications of our AGE model can be found in the Supplementary Information (SI).

Our model distinguishes two regions: China and its main food and feed trading partners (MTP, including Brazil, the United States, and Canada). We select 2014 as the reference year, as it is the latest available year in the Global Trade Analysis Project (GTAP) database [70] at the time of our research. Our model is solved using the general algebraic modelling system (GAMS) software package [71]. We exclude the rest of the world (RoW) because, according to GTAP [70] trade flow data, MTP accounts for over 75% of China's total food and feed trade value in 2014, while China's trade share with RoW is smaller at 25%. Detailed information on China's domestic use and trade shares of food and feed products with MTP and RoW is provided in Supplementary Table 9. We observe that China maintains nearly 99% self-sufficiency in monogastric livestock production, with imports accounting for only 1% (0.8% from MTP and 0.2% from RoW; see Supplementary Table 9). Furthermore, monogastric livestock production in China and MTP together represents approximately 50% of global production (Supplementary Table 10).Thus, China's domestic food production plays a primary role in shaping its trade balance with MTP. Our two-region framework effectively captures the most significant trade flows influencing China's food system, while simplifying the model calculations.

Modelling circularity in livestock production requires a detailed representation of biophysical flows to consider nutritional balances and livestock feeding requirements due to increased utilisation of food waste and food processing by-products as feed for monogastric livestock production. Following Gatto, et al. [26] and Chepeliev [58], we convert dollar-based quantities (million USD) to physical quantities (Tg; 1 Tg = 10^6 tons) to allow the tracing of biophysical flows through the global economy. GTAP version 10 database [70] is used to calibrate our AGE model and provided dollar-based quantities. We designed a sectoral aggregation scheme comprising 16 sectors (see Appendix Table 1) based on the original GTAP database to produce social accounting matrices (SAM) (see Appendix Tables 2-3) in our study. In the SAMs from the GTAP database, dollar-based material balances for the reference year ensure that dollar-based production quantities for each commodity across countries equal the sum of dollar-based intermediate demand across sectors, dollar-based final demand from the representative consumer, and dollar-based net exports. Dollar-based bilateral trade quantities in the GTAP database are constructed based on the reconciled UN Comtrade Database [72], while physical bilateral trade quantities are obtained from FAO [5] trade data. To construct bilateral trade flows in physical quantities, we prioritise FAO-reported imports over exports, assuming that import data is more reliable since importers have a stronger incentive to provide accurate trade records for taxation purposes. We then apply the RAS approach (also referred to bi-proportional balancing) [73] to balance physical bilateral trade quantities, ensuring consistency between FAO-reported totals for exporting and importing countries. Physical production quantities of crops, livestock, and fertilisers (see Supplementary Table 2) are obtained from FAO [5]. Physical feed production quantities are extracted from the "Feed" category in the FAO Food Balance Sheet (FBS). Since the FAO does not provide feed data by livestock type, following Gatto, et al. [74], we allocated the physical production quantities of feed across livestock sectors based on dollar-based feed demand shares across livestock sectors in the SAMs from the GTAP database. The physical production quantity of grass from natural grassland is derived from Miao and Zhang [75]. We only include grass from natural grassland where ruminant livestock is grazing for feed, and grass from remaining grassland is excluded. The dollar-based production quantity of grass is estimated as the value flow from the pastureland to ruminant livestock in the SAMs from the GTAP database. To establish the link between dollar-based and physical quantities, we define material intensity coefficients (kg USD^{-1}) for each commodity at the regional level as the ratios of physical production quantities from FAO to dollar-based production quantities from GTAP, estimating these coefficients using reference year data. This approach allows us to compute physical material balances once dollar-based material balances are determined after each model run. However, physical material balances may not hold for all commodities and countries. To address this, we adjust physical production quantities to ensure that the supply of each commodity aligns exactly with FAO-FBS data for further comparisons. This adjustment ensures consistent tracing of material flows in both dollar-based and physical quantities for each commodity across countries. For simulations using the static AGE model, physical material flows change proportionally to the corresponding dollar-based material flows in response to an exogenous shock.

Livestock categories are aggregated into two sectors, i.e., monogastric livestock (including pigs, broilers, and laying hens) and ruminant livestock (including dairy cattle, other cattle, and sheep & goats). Furthermore, the inclusion of animal-specific dietary constraints in our model allows us to calculate the nutritional balance (crude protein and digestible energy), feed conversion ratios (FCR, the ratio of fresh feed inputs to live weight gain), and edible feed conversion ratio (eFCR, the amount of human-edible feedstuffs, i.e., feed crops and compound feed, used for per unit of live weight gain) [76] for each livestock sector. First, we estimate the physical quantities of feed protein (Tg) and energy (billion MJ) required to produce the physical output of each livestock sector (Tg) in the reference year based on the FAO-FBS data and nutritional (i.e., protein and energy) contents of feed sub-groups (see Supplementary Table 7). Then, we obtain the initial composition of total feed (including feed crops, compound feed, food waste, food processing by-products, and grass) supplied to each livestock sector in the reference year. When substituting primary feed (i.e., feed crops and compound feed) in animal diets with food waste and food processing by-products, the total protein and total energy supplies per unit of animal output are kept constant in all scenarios. Our FCRs for ruminant livestock are slightly different from FCRs in the literature, as we do not fully account for maize silage, alfalfa hay, and roughage-like by-products, but this bias does not affect the impacts of upcycling food waste and food processing by-products for monogastric livestock production. Further model details, nutritional balance, and detailed composition of animals' diets are available in the SI.

Modelling amounts and impacts of food waste and food processing by-products.

In this study, we consider food waste and food processing by-products. Food waste is considered a local resource within China, while food processing by-products can be traded between China and MTP. We focus on food intended for human consumption that is wasted during distribution, retailing, and consumption (both households and out-of-home), as it has a high potential for upcycling as animal feed. In contrast, food loss, which occurs earlier in the supply chain, is often driven by poor infrastructure and is not easily prevented or repurposed for feed use [17]; therefore, it is excluded from our analysis. Additionally, we only consider plant-sourced food waste because animal-sourced food waste may pose a risk of pathogen transfer, including foot-and-mouth and classical swine fever [77]. Food waste is quantified separately for each type of food product by multiplying primary food products after processing by China-specific food waste fractions [37] following the FAO methodology [78]. Four types of food waste are 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 are estimated by multiplying the production quantities of primary food products by FAO technical conversion factors for various by-products [79]. The total amounts of food waste and food processing by-products and their current use as animal feed and discarded biomass (i.e., landfill and incineration) for China in S0 are presented in Supplementary Table 4.

Our model incorporate two food waste treatment sectors, i.e., "food waste collection service" and "food waste recycling service" (Figure 1). The food waste recycling service sector recycles food waste and food processing by-products as feed for monogastric livestock production. The food waste collection service sector collects food waste and food processing by-prodcuts for landfill and incineration. Waste collection, treatment and disposal activities were included in the "Waste and water (wtr)" sector in the GTAP database. Food waste generation is added as a margin commodity, similar to how GTAP treated transport costs following Peterson [80]. Thus, the consumer price of food includes both the market price of food and the cost of collecting food waste and food processing by-products. Consumers spend their income on both consumption of goods and food waste collection service, but they derive utility solely from the consumption of goods. In terms of recycling food waste and food processing by-products as feed, monogastric livestock producer bears the associated cost. By multiplying the quantities of food waste with the unit costs of food waste treatment, we can calculate the economic value of food waste generation. Physical quantities and prices of food waste recycling and collection services in China are presented in Supplementary Tables 4-5.

Environmental impact assessment.

Economy-wide emissions considered in our study are limited to the production-related stages from all sectors in the entire economies of China and MTP, excluding land use change and household consumption. Specifically, emissions from both agricultural (6 crop types and 2 livestock categories) and non-agricultural (compound feed, food processing by-products, processed food, fertilisers, food waste treatment, and non-food) production are quantified. In line with other studies [81], land use is considered to be constant here, allowing to focus on changes in total emissions from all sectors in the entire economy without addressing the impacts of context-specific land use change. Detailed information about emission sources across sectors is provided in Appendix Table 4.

Three main environmental impacts are distinguished, i.e., global warming potential (GWP, caused by greenhouse gas (GHG) emissions, including carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) emissions; converted to CO_2 equivalents), acidification potential (AP, caused by pollutants leading to acidification, including ammonia (NH_3), nitrogen oxides (NO_x), and sulphur dioxide (SO_2) emissions; converted to NH_3 equivalents), and eutrophication potential (EP, caused by pollutants leading to eutrophication, including nitrogen (N) and phosphorus (P) losses to water bodies; converted to N equivalents). The conversion factors for GWP, AP, and EP are derived from Goedkoop, et al. [82].

Data on CO_2, CH_4, and N_2O emissions are obtained from the Climate Analysis Indicators Tool (CAIT) [83]. GHG emissions calculations in our model follow the IPCC National GHG Emission Guidelines [84]. We derive NH_3, NO_x, and SO_2 emissions from Liu, et al. [85], Huang, et al. [86], and Dahiya, et al. [87], respectively. We consider NO_x emissions from energy use only, as agriculture's contribution to NO_x emissions is generally small ($\leqslant 2\%$) [88]. We use the global eutrophication database of food and non-food provided by Hamilton, et al. [10] to obtain data on N and P losses to water bodies. Emissions of food processing by-products (i.e., cereal bran, alcoholic pulp, oil cake) are derived from Mackenzie, et al. [89]. We attribute the environmental impacts between the main (e.g., cereal flour) and joint products (e.g., cereal bran) according to their relative economic values (see Supplementary Table 6). Emissions of food waste recycling and collection services are obtained from Alsaleh and Aleisa [90], Hong, et al. [91], and Hong, et al. [92].

The total emissions of GHGs, acidification pollutants, and eutrophication pollutants from all sectors in the entire economy in the base year are calculated first. Then, we allocate 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. The sectoral-level emissions, as well as the dollar-based emission intensities of GHGs (ton CO_2 equivalents million USD^{-1}), acidification pollutants (ton NH_3 equivalents million USD^{-1}), and eutrophication pollutants (ton N equivalents million USD^{-1}) are presented in Appendix Tables 5-10.

Two types of land use, i.e., cropland and pastureland, are distinguished. We update the GTAP data on crop harvested areas using the FAO [5] database. Pastureland is defined as areas where ruminant grazing occurs. We derive nitrogen and phosphorous fertiliser use by crop types and countries from Ludemann, et al. [93].

Food security indicators.

The FAO [94] defines food security as encompassing four key dimensions: availability (adequate food supply), access (sufficient resources to obtain food), utilisation (nutritious and safe diets), and stability (consistent access to food over time). We focuse on the first two dimensions. First, food availability is defined as "calories per capita per day available for consumption". "Population at risk of hunger" refers to the portion of people experiencing dietary energy (calorie) deprivation lasting more than a year following the FAO-based approach [44,45]. This approach has been widely used in agricultural economic models to evaluate the risk of food insecurity [29,95,96]. In essence, the population at risk of hunger is determined by multiplying the prevalence of undernourishment (PoU) by the total population. According to the FAO, the PoU is based on dietary energy availability calculated by our model, the mean minimum dietary energy requirement (MDER), and the coefficient of variation (CV) of the domestic distribution of dietary energy consumption in a country. It is assumed that there is no risk of hunger for high-income countries; consequently, the population at risk of hunger is not applied to the United States and Canada [29,95,96]. Second, the access dimension is tied to people's purchasing power, which depends on food prices, dietary habits, and income trends [97]. We calculate the average food price (including primary food products and processed food) and estimated changes in food affordability by subtracting changes in the average wage across the entire economy from fluctuations in cereal prices.

Definition of scenarios.

To estimate the impacts of increased utilisation of food waste and food processing by-products as animal feed on food security and the environment, we examined five scenarios, including one baseline (S0) scenario representing the economic and environmental conditions of all sectors (including agriculture, industries, and services) in the entire economies of China and MTP in 2014, two scenarios involving increased utilisation of food waste and food processing by-products as animal feed, and two scenarios involving increased utilisation of food waste and food processing by-products as animal feed combined with emission mitigation measures. We implement regional uniform emission taxes across all sectors on economy-wide emissions of GHGs, acidification pollutants, and eutrophication pollutants in China and MTP under the partial use of food waste and food processing by-products as animal feed (scenario S1), considering the perishability and collection challenges of food waste, as well as the reduced availability of food waste for feed according to SDG 12.3 ("halving food waste") [21]. The latter four scenarios are compared to the 2014 baseline (S0) scenario. The scenarios are further described below and in Supplementary Table 1.

To ensure the feasibility of upcycling food waste and food processing by-products as feed, scenarios S1-S4 incorporate four key assumptions related to food waste source separation, collection, transportation, pre-treatment technologies, and consumer acceptance. First, policies on food waste source separation and collection, currently implemented mainly in major cities such as Beijing and Shanghai [49,50], are assumed to gradually expand nationwide, accompanied by increasing awareness and participation in food waste separation among households and restaurants in China. Second, food waste collection and transportation logistics are designed to improve alongside policy developments and infrastructure expansion. With increased financial support from the Chinese government, including investments in infrastructure and technological innovations, pilot food waste collection systems already operational in over 33 cities [98] are expected to scale up nationwide, enhancing waste disposal infrastructure and ensuring sufficient capacity for efficient collection and transportation of food waste. Third, pre-treatment technologies, including sorting, shredding, thermal treatment of drying and dehydration, deodorizing, fermentation, hydrolysis, and extrusion of food waste into feed pellets [90], are considered to remove excess moisture, reduce perishability, and extend shelf life, thereby enhancing the feasibility of cross-provincial transportation of food waste. Fourth, consumer acceptance of livestock products raised on food waste-based feed may be improved over time. Extensive field-based evidence has demonstrated that properly treated food waste is safe for animals with minimal health risks [47], and targeted education programs and community outreach can help address consumer concerns about product safety and enhance acceptance of food waste-based animal feed.

S1 - Partial use of food waste and food processing by-products as feed. Scenario S1 analyses the impacts of partially upcycling food waste and food processing by-products (54% of food waste and 100% of food processing by-products) as feed for monogastric livestock production in China. Cross-provincial transportation of food waste is not allowed in S1, which limits the maximum utilisation rate of food waste with high moisture content to 54% in China, according to Fang, et al. [17].

S2 - Full use of food waste and food processing by-products as feed. Scenario S2 analyses the impacts of fully upcycling food waste and food processing by-products (100% of food waste and 100% of food processing by-products) as feed for monogastric livestock production in China. Cross-provincial transportation of food waste is allowed in S2 because we consider that new technology would become available for processing food waste with high moisture content. Economies of scale in food waste recycling are considered in S2; a 1% increase in recycled waste results in only a 0.078% rise in recycling costs [99]. Thus, as production scales up, marginal costs decrease and then stabilise.

S3 - S1 + A modest emission mitigation target. We implement regional uniform emission taxes across all sectors to achieve a modest emission mitigation target, assuming that economy-wide emissions of GHGs, acidification pollutants, and eutrophication pollutants in China and MTP do not exceed their baseline (S0) levels. For a given emission mitigation target for each type of pollutant, the AGE model can endogenously determine the emission taxes for various pollutants (expressed in \$ per ton of CO_2 equivalents, \$ per ton of NH_3 equivalents, and \$ per ton of N equivalents). This approach is commonly used in the literature [29,31,96,100] and allows to identify the most economically cost-effective mitigation pathway for achieving given emission mitigation targets.

S4 - S1 + An ambitious emission mitigation target. We implement regional uniform emission taxes across all sectors to achieve an ambitious emission mitigation target, assuming that economy-wide emissions of GHGs, acidification pollutants, and eutrophication pollutants in China and MTP remain within the emission thresholds set by China's and the MTP's annual GHG mitigation targets under the Intended Nationally Determined Contributions (INDC) of the Paris Agreement [35,36], as well as China's emission reduction goals for acidification and eutrophication pollutants in line with the "14th Five-Year Plan" [41].

Estimation of feed cost and cost savings under various scenarios.

The total feed cost per unit of monogastric livestock production is calculated by dividing the total feed cost (including feed crops, compound feed, food waste, and food processing by-products) by the economic output of monogastric livestock in China. In S0, the costs of feed crops (including cereal grains, oilseeds & pulses, vegetables & fruits, roots & tubers, and sugar crops), compound feed, and select food processing by-products (including distiller's grains from liquor production, brewer's grains from barley beer production, and oil cake), along with the economic output of monogastric livestock, are derived from their market prices based on the SAMs in the GTAP database. Specifically, value flows from feed crops, compound feed, and these food processing by-products to monogastric livestock, as indicated in the SAMs, represent the corresponding feed costs. The costs of additional food processing by-products (including cereal bran and distiller's grains from maize ethanol production) are determined by multiplying their physical production quantities (tons) from FAO by their corresponding prices (USD ton^{-1}), which are calculated using data from UN Comtrade Database [72]. Detailed information on the costs of feed crops, compound feed, and food processing by-products is provided in the notes under Appendix Table 1 in SI. The cost of food waste (cereal grains waste, vegetables & fruits waste, roots & tubers waste, and oilseeds & pulses waste) is estimated based on the price of food waste recycling service, which includes the cost of sorting, shredding, thermal treatment of drying and dehydration, deodorizing, fermentation, hydrolysis, and extrusion of food waste into feed pellets (see Supplementary Table 5). In S1-S4, the cost associated with the increased utilisation of food waste and food processing by-products as feed is also estimated using the price of food waste recycling service. The cost savings from increased utilisation of food waste and food processing by-products as feed are then determined by comparing the total feed cost per unit of monogastric livestock production across scenarios S1-S4 with S0.

Sensitivity analysis.

To evaluate the robustness of our results and assess the relative importance of key input parameters, we conducted a sensitivity analysis and decomposed uncertainties into five major sources: (1) feasibility of upcycling food waste and food processing by-products as feed; (2) conversion of dollar-based quantities to physical quantities; (3) substitution of cropland with other inputs for crop production; (4) cereal self-sufficiency target; (5) cleaner crop and livestock production technology. We employed the one-at-a-time method to assess the sensitivity of food security indicators and environmental impacts to variations in these uncertainty sources. This approach, widely used in marginal impact analysis, isolates the effect of a single input variable while keeping all others constant [101]. The larger the ratio of relative output change to relative input change, the greater the sensitivity of the results to that parameter. Further details on the sensitivity analysis are provided in Supplementary Discussion.

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/). The other data that support splitting agricultural (6 crop types and 2 livestock categories) and non-agricultural (compound feed, food processing by-products, processed food, fertilisers, 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 authors upon reasonable request.

Code availability

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

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Author contributions

W.L., X.Z., H.P.W., 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).

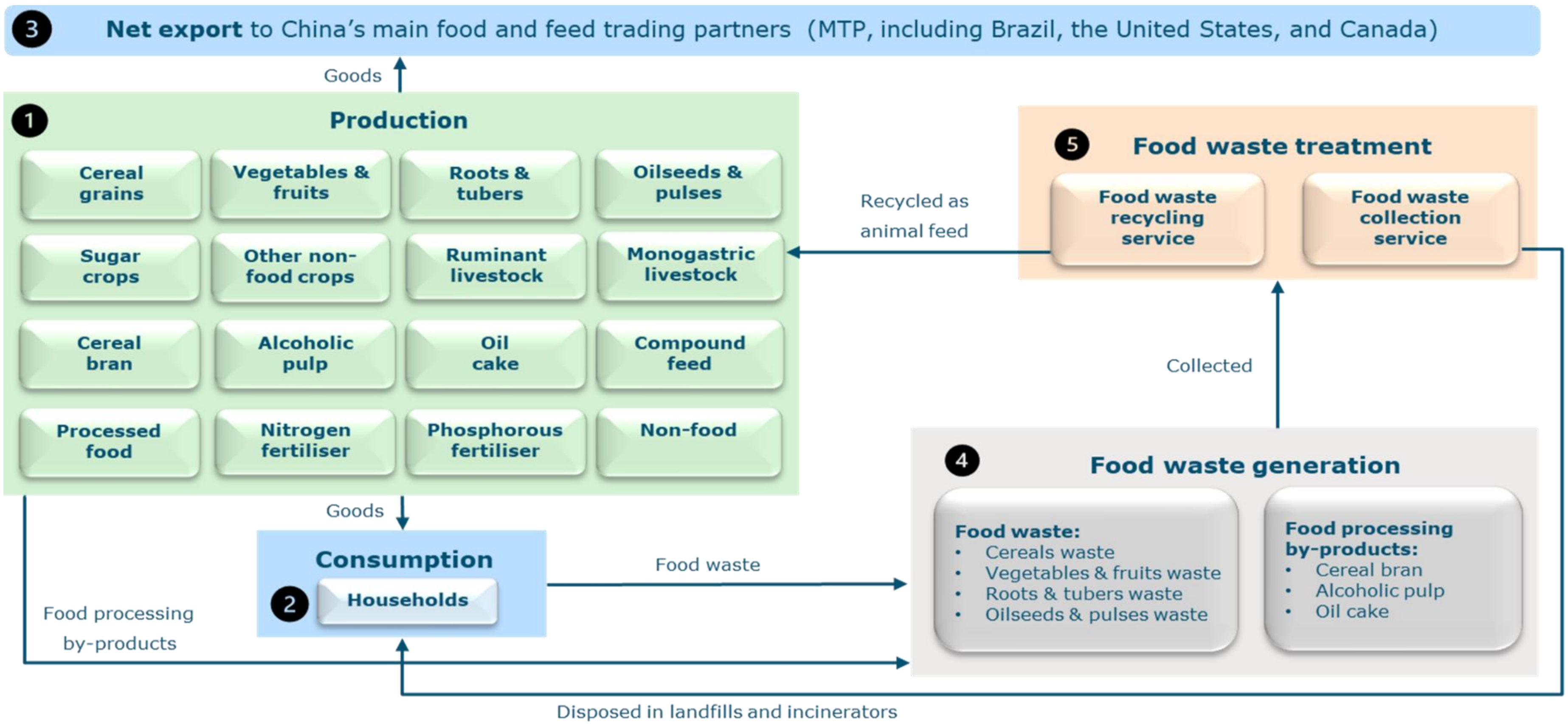

Fig. 1 | Representation of the economy in China in the applied general equilibrium (AGE) framework with food waste and food processing by-products. The framework includes four parts: (1) Production; (2) Consumption; (3) Net export; (4) Food waste generation; (5) Food waste treatment. The generated food waste and food processing by-products are sent either to the "food waste recycling service" sector or the "food waste collection service" sector. The food waste recycling service sector recycles food waste and food processing by-products as feed for monogastric livestock production. The food waste collection service sector collects food waste and food processing by-products for landfill and incineration. The consumer price of food includes both the market price of food and the cost of collecting food waste and food processing by-products. The monogastric livestock producer bears the cost of recycling food waste and food processing by-prodcuts as feed. Detailed information is presented in Methods and Supplementary Information.

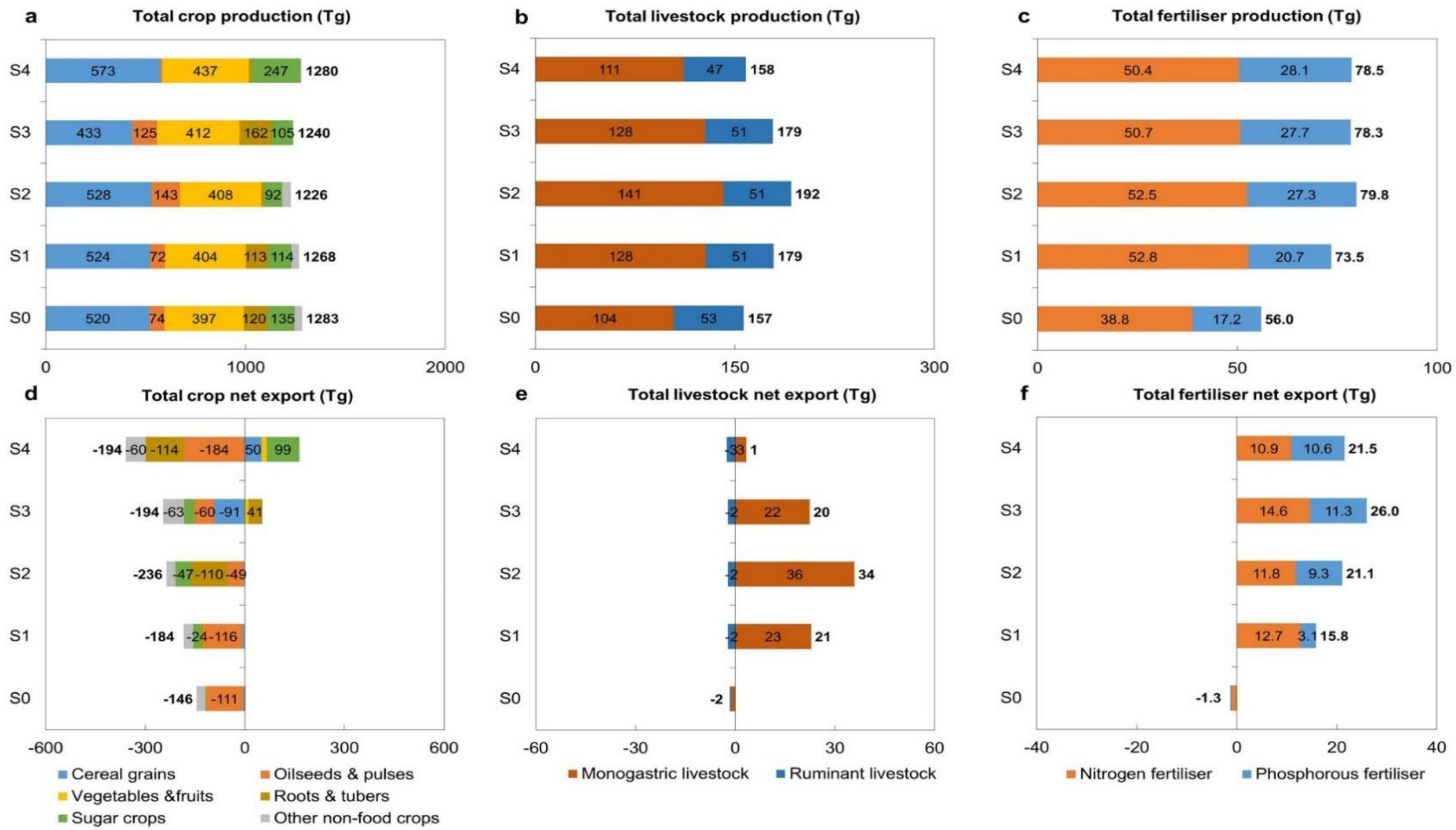

Fig. 2 | Impacts of upcycling food waste and food processing by-products as feed in China's monogastric livestock sector on domestic production and net export of total crop, livestock, and fertiliser. Total (**a**) crop, (**b**) livestock, and (**c**) fertiliser production (Tg) in scenarios. Total (**d**)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

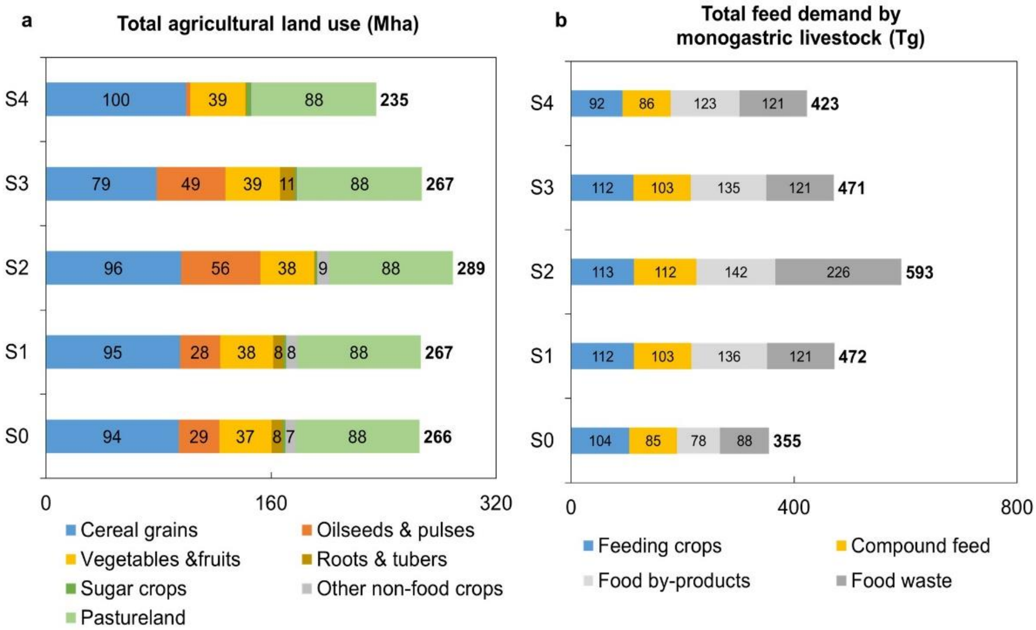

Fig. 3 | Impacts of upcycling food waste and food processing by-products as feed in China's monogastric livestock sector on domestic total agricultural land use and feed demand. (**a**) Total agricultural land use (crop harvested area and pastureland) (Mha) and (**b**) feed demand by monogastric livestock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

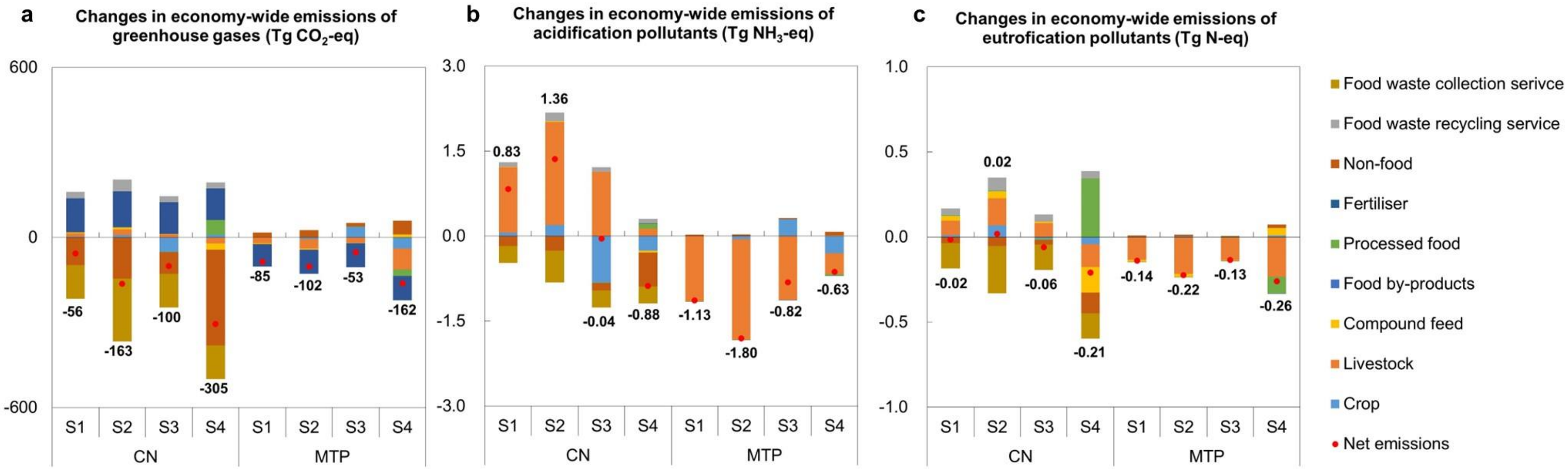

Fig. 4 | Impacts of upcycling food waste and food processing by-products as feed in China's monogastric livestock sector on economy-wide emissions in China (CN) and China's main food and feed trading partners (MTP). Changes in (**a**) economy-wide emissions of greenhouse gases (GHGs) (Tg CO_2-eq), (**b**)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

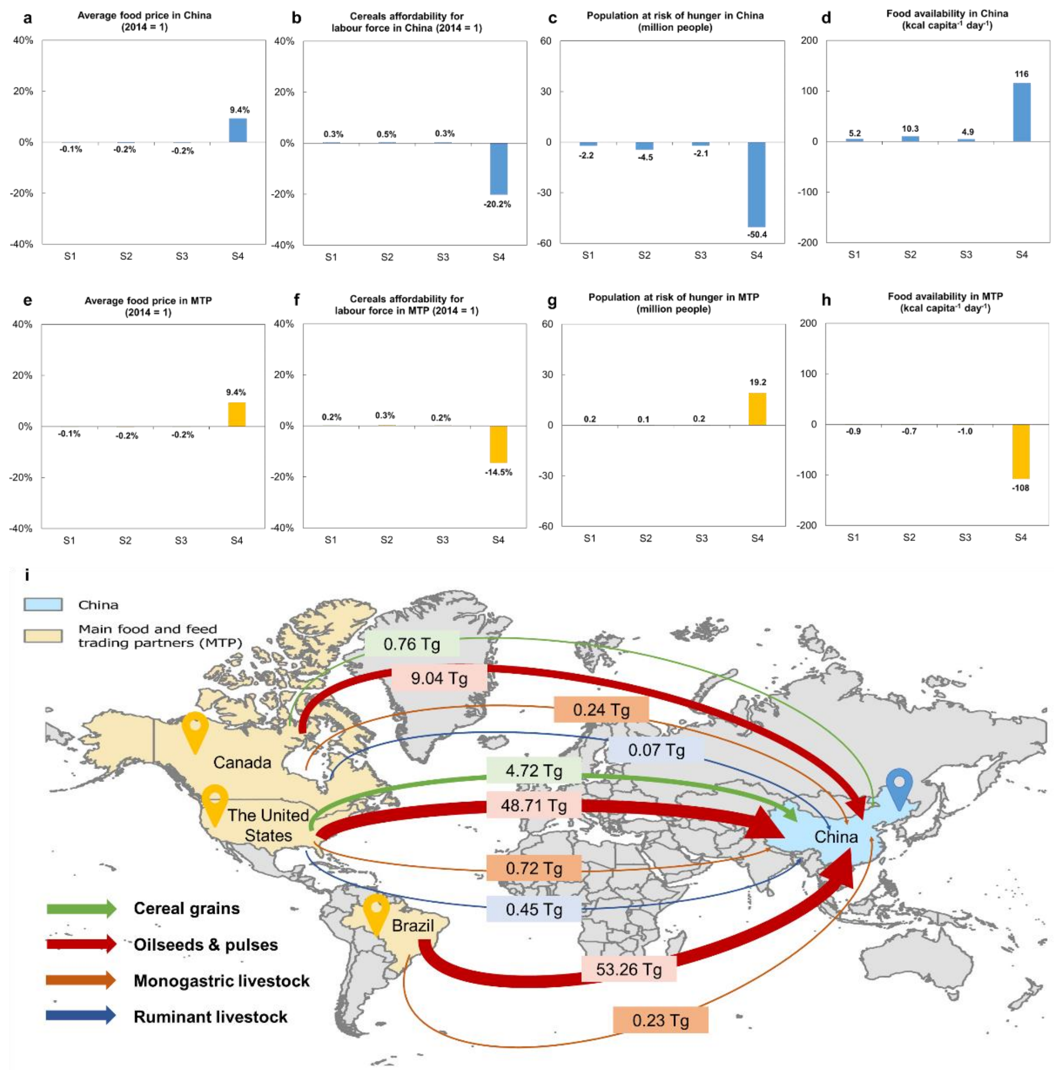

Fig. 5 | Impacts of upcycling food waste and food processing by-products as feed in monogastric livestock sector on food security indicators in China (CN) and China's main food and feed trading partners (MTP). Changes in (**a**) average food price (including primary food products and processed food), (**b**) cereals affordability for labour force, (**c**) population at risk of hunger (million people; S0 = 140.7 million people), and (**d**) food availability (kcal capita^{-1} day^{-1}) in China in scenarios with respect to the baseline (S0). Changes in (**e**) average food price (including primary food products and processed food), (**f**) cereals affordability for labour force, (**g**) population at risk of hunger (million people; S0 = 5.3 million people), and (**d**) food availability (kcal capita^{-1} day^{-1}) in MTP in scenarios with respect to the baseline (S0). (**i**) Net imports (Tg) of main food and feed products from MTP to China in the baseline (S0). MTP includes Brazil, the United States, and Canada. According to the FAO approach, it is assumed that there is no risk of hunger for high-income countries; consequently, the population at risk of hunger is not applied to the United States and Canada [29,95,96]. Definitions of scenarios (S1 - 'Partial use of food waste and food processing by-products as feed'; S2 - 'Full use of food waste and food processing by-products as feed'; S3 - 'S1 + A modest emission mitigation target'; S4 - 'S1 + An ambitious emission mitigation target') are described in Supplementary Table 1. Credit: World Countries base map, Esri (https://hub.arcgis.com/datasets/esri::world-countries/about).