A modest mitigation target could address rebound effects of upcycling food waste as feed in China while safeguarding global food security

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Abstract

Feeding livestock with food waste could reduce environmental impacts, but rebound effects, where lower feed costs lead to expanded livestock production, may diminish these benefits. Using an integrated environmental-economic model, we assessed the global impacts of upcycling food waste in China's monogastric livestock production. We found that the upcycling increased monogastric livestock production by 23-36% and raised Chinese economy-wide acidification emissions by 2.5-4.0%. Eutrophication emissions decreased by 0.2% with partial upcycling but increased by 0.2% with all upcycling. Greenhouse gas emissions decreased by 0.5-1.4% due to reduced food waste in landfills and incinerators, along with contractions in non-food production. This upcycling and resource reallocation across food systems enhanced food security in China without compromising its trading partners. An ambitious emission mitigation target (i.e., emission taxes to meet Paris Agreement goals) could counteract rebound effects but risk a 9.4% rise in food prices, threatening global food security. Conversely, a modest emission mitigation target (i.e., emission taxes to maintain baseline levels) provides an opportunity to address rebound effects while safeguarding global food security.

Keywords

circular economy; food waste; food security; environmental sustainability; 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. The surge in demand for ASF, driven by population growth, prosperity, and urbanization, [1,2] is expected to double by 2050, especially in developing countries [3]. This surge in livestock production has exacerbated food-feed competition and exerted tremendous pressure on planetary boundaries (PBs). Currently, 70% of global agricultural land is used for producing animal feed [4], and global livestock production accounts for 13-18% of the total anthropogenic 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]. It has been shown that the global 1.5°C climate target cannot be achieved without mitigating emissions from food systems [8].

Upcycling food waste as animal feed is crucial for reducing environmental impacts and building more circular food systems [9], as global food waste has risen from 1.3 billion tons to 1.6–2.5 billion tons in recent years despite significant reduction efforts [10], with much of it exacerbating GHG emissions and climate change through landfill and incineration [11]. Upcycling food waste as animal feed offers a pathway to mitigate land-related pressures [12], alleviate the food-feed competition [9], and reduce emissions from food systems and improper food waste disposal [13]. This is because low-opportunity-cost feed (LCF), i.e., food waste and food processing by-products, typically compete less for land and natural resources than human-edible feeding crops [9,12,13]. 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) [14].

While many studies acknowledge the environmental benefits of increasing food waste utilisation as feed, significant gaps remain in the existing literature, particularly in three critical areas. First, previous studies [9,12,13] employing linear optimization models to evaluate the environmental impacts of this circular transition may overestimate the environmental benefits by disregarding "rebound effect" (or “Jevons paradox”) [15]. Here we consider the possibility that feeding animals with food waste may lower feed costs and expand livestock production, thus leading to increased emissions—

the "rebound effect". This rebound effect and its knock-on effects on other commodities in the broader economy may further diminish the environmental benefits of feeding animals with food waste. For example, increased demand for feed due to expanded livestock production may intensify the need for cropland and fertilisers to cultivate feeding crops, thereby exacerbating emissions even more. This raises concerns that upcycling food waste as animal feed might enhance food security while potentially compromising environmental sustainability. Second, the "rebound effect" phenomenon has been extensively studied in energy systems [16,17], but its implications in food systems are largely lacking. Although previous studies have explored rebound effect related to a global dietary shift towards plant-based food [18] and halving food loss and waste [19], none have yet explored the rebound effect of upcycling food waste as animal feed. Third, while measures that are not subject to rebound effects, such as implementing economy-wide emissions taxes, could help mitigate livestock expansion resulting from upcycling food waste as feed, the combination of these strategies has not yet been formally explored in scenario analyses. Additionally, while emission taxes may help address rebound effects, they may pose a threat to food security [20]. It remains unclear how to address rebound effects of upcycling food waste as feed while safeguarding food security.

In this study, we fill these gaps and contribute to the existing literature by using an integrated environmental-economic modelling framework based on the applied general equilibrium (AGE) models to assess the environmental and economic consequences of upcycling food waste in China's monogastric livestock production as feed in a global context, and to explore how implementing economy-wide emissions taxes could mitigate rebound effects of this upcycling while safeguarding food security. We focused on China for our study because it is the world's largest animal producer, accounting for 46%, 34%, and 13% of global pork, egg, and poultry meat production in 2018, respectively [21]. Furthermore, 27% of food produced for human consumption are lost or wasted in China [22], implying a substantial opportunity to upcycle food waste as feed. We addressed 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 and its main food and feed trading partners (MTP, including Brazil, the United States, and Canada)? Second, how will these influence global environmental sustainability (i.e., emissions of GHGs, acidification pollutants, and eutrophication pollutants) and food security (i.e., average food price, food affordability, population at risk of hunger, and food availability)? Third, how will implementing economy-wide emissions taxes mitigate rebound effects of this upcycling while safeguarding food security?

The novelty of this study lies in three parts. First, the inclusion of two food waste-related sectors (see Fig. 1 and Methods) within the AGE model 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. Second, the improved framework by bridging monetary AGE models with biophysical (quantity-based) and nutritional (protein and energy-based) constraints allows us to capture the rebound effect of expanded livestock production and its knock-on effects on other commodities, as well as subsequent impacts on global environmental sustainability and food security, in the context of upcycling food waste as feed with and without implementing economy-wide emissions taxes. Third, integrating emissions of GHGs 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 five scenarios: (i) the baseline (S0) scenario representing the economies of China and MTP in 2014; (ii) scenario 1 (S1) upcycling partial food waste as feed (54% of food waste and 100% of food processing by-products) for monogastric livestock production in China; (iii) scenario 2 (S2) upcycling all food waste as feed (100% of food waste and 100% of food processing by-products) for monogastric livestock production in China; (iv) scenario 3 (S3 = S1 + A modest emission mitigation target) implementing economy-wide emission taxes to ensure that emissions of GHGs, acidification pollutants, and eutrophication pollutants in both China and MTP do not exceed their baseline (S0) levels; (v) scenario 4 (S4 = S1 + An ambitious emission mitigation target) implementing economy-wide emission taxes to meet their annual mitigation target of the Intended Nationally Determined Contributions (INDC) under the Paris Agreement [23,24] and China’s “13th Five-Year Plan” [25]. When substituting primary feed (i.e., feeding crops and compound feed) in animal diets with food waste and food processing by-products, we maintained the protein and energy supply for per unit of animal output in all scenarios to prevent imbalances between nutritional (protein and energy) supply and livestock requirements. The scenarios mentioned above were further described in Table 1.

Results

Expanded monogastric livestock production and its knock-on effects on other commodities. China produced about 104 Tg of monogastric livestock (pork: 57 Tg; poultry meat: 18 Tg; egg: 29 Tg) and 53 Tg of ruminant livestock (milk: 42 Tg; beef: 6 Tg; lamb: 4 Tg) products in 2014. We estimated that 226 Tg food waste (54 Tg in dry matter; 7 Tg in crude protein; 690 billion MJ in energy) and 163 Tg food processing by-products (139 Tg in dry matter; 49 Tg in crude protein; 1907 billion MJ in energy) was available in China in 2014, but only 39% of the food waste and 51% of the food processing by-products were recycled as feed, with the remainder disposed in landfills and incinerators (Supplementary Tables 2-3). Unlike previous studies that considered recycling food waste and food processing by-products as feed to be costless [9,12,13], we modelled the rising cost of this recycling process as an increasing percentage of the initial cost of the recycling process itself (Supplementary Table 4), with these costs covered by monogastric livestock producers.

Our results showed that upcycling 54-100% of food waste and 100% of food processing by-products as feed increased the share of food waste and food processing by-products used as feed within the total feed use by 8-16% in fresh matter, 10-14% in dry matter, 4-6% in protein, and 8-13% in energy (Supplementary Fig. 1). The upcycling, which increased the supply of feed protein by 27-40% (14-21 Tg) and feed energy by 26-39% (883-1318 billion MJ), reduced total feed (i.e., feeding crops, compound feed, food waste, and by-products) cost for per unit of monogastric livestock production by 2.1-3.0%, leading to a 23-36% (24-37 Tg) increase in monogastric livestock production (Fig. 2b). This shift signifies a transition for China from a net importer of monogastric livestock, importing 1% (1.2 Tg) of output in the baseline (S0), to an exporting nation, with 18-25% (24-37 Tg) of output being exported (Fig. 2h). Ruminant livestock production decreased by 3% (2 Tg) as the expansion of monogastric livestock reduced the availability of feeding crops and compound feed to ruminant livestock (Fig. 2b). To meet domestic demand, ruminant livestock imports rose from 1% (0.5 Tg) of output in the baseline (S0) to 4% (2 Tg) (Fig. 2h).

Expanded monogastric livestock production raised the demand for primary feed (i.e., feed crops and compound feed), which outweighed the reduction in primary feed use from substituting it with food waste and food processing by-products. Although total feed demand for ruminant livestock decreased by 0.6% (2 Tg) (Fig. 3f), overall feed demand for both monogastric and ruminant livestock increased by 17-34% (116-236 Tg) due to a 33-67% (118-238 Tg) rise in feed demand for monogastric livestock (Fig. 3e). The upcycling would, thus, change in the feed conversion ratio (FCR, the ratio of fresh feed inputs to live weight gain) and edible feed conversion ratio (eFCR, the amount of human-edible feedstuffs like feeding crops and compound feed used for per unit of live weight gain) for livestock. Despite an increase in FCR for monogastric livestock by 0.22-0.62 kg kg^{-1}, the eFCR decreased by 0.11-0.19 kg kg^{-1}, indicating its reduced reliance on human-edible feedstuffs (Supplementary Fig. 2a). Since feeding crops and compound feed account for only 12% of ruminant feed compared to 88% from grass, the upcycling has a minor impact on ruminant production and feed use. Minute changes were observed in FCR (0.14 kg kg^{-1}) and eFCR (0.01 kg kg^{-1}) for ruminant livestock production (Supplementary Fig. 2b).

The increase in overall feed demand indirectly affected the crop production, crop harvested area, and the use of nitrogen and phosphorus fertilisers, while also prompting crop extensification due to price-driven substitution effects. The expansion of monogastric livestock production, a relatively labour-intensive sector, increased labour demand, leading to a 0.13-0.22% rise in average wages across the Chinese economy (Supplementary Fig. 3a). Consequently, labour became comparatively more expensive than other inputs (i.e., capital, cropland, and fertilisers). As cropland and fertilisers became relatively cheaper, crop producers were incentivised to engage in crop extensification and use more cropland and fertilisers to substitute labour. This led to a 0.8-2.3% (0.3-0.9 Tg) increase in total nitrogen fertiliser use (Fig. 2f & 3a), a 0.8-2.8% (0.1-0.5 Tg) increase in total phosphorus fertiliser use (Fig. 2f & 3b), and a 0.6-13% (1-24 Mha) expansion in the crop cultivated area (Fig. 3c). Crop producers will prioritise reducing the production of relatively labour-intensive crops; for example, roots & tubers and sugar crops decreased by 6-90% (7-108 Tg) and by 15-32% (21-43 Tg) (Fig. 2a). The saved cropland would then be reallocated to increase the production of cereal grains by 0.8-1.5% (4-8 Tg), vegetables and fruits by 1.7-2.7% (7-11 Tg), and other non-food crops by 8-18% (3-6 Tg) (Fig. 2a). Notably, the production of oilseeds & pulses decreased by 1.6% (1 Tg) with partial upcycling but increased by 95% (70 Tg) with all upcycling (Fig. 2a). This variation occurs because oilseeds & pulses are both relatively labor-intensive and cropland-intensive compared to other crops, making their production dependent on the interplay between labour and cropland costs at different levels of upcycling. To meet the a 1.6-2.4% (24-34 Tg) rise in total crop consumption (i.e., used as feeding crops, compound feed, food by-products, processed food, and primary fresh food) (Fig. 2d & 3d), while facing a 1.2-4.4% (15-57 Tg) decline in total crop production (Fig. 2a), crop import reliance rose, with the share of import increasing from 11% (146 Tg) in the baseline (S0) to 15-19% (184-236 Tg) (Fig. 2g).

Adjustments in crop and livestock production also had knock-on effects beyond the agricultural sectors in the broader economy, thus influenced sectoral employment, gross domestic product (GDP), household expenditure, and household welfare (a measure of economic well-being in US dollars). Since our AGE model assumes full employment and free mobility of labour across sectors, following the default setting of standard GTAP [26] model, there is no net loss in employment, and labour is swiftly reallocated from one sector to another. We observed that the 27-43% (11.5-18.4 million people) increase in monogastric livestock employment was largely transferred from a 1.1-1.7% decline in the non-food (i.e., industry and services, detailed in Appendix Table 1) sector, challenging the livelihoods of 11.8-17.5 million people currently employed there (Supplementary Fig. 5a,c). While the non-food sector, which currently accounts for 76.8% of China's total sectoral value-added (Supplementary Fig. 8), experienced a slight relative output decline of 1.0-1.4% (Supplementary Fig. 6a,c), it faced the largest absolute loss of 28-41 billion US dollars (USD, 2014 constant price) (Supplementary Fig. 7a). In contrast, nitrogen and phosphorus fertiliser production surged by 35-36% (13.7-14.0 Tg) and 20-59% (3.5-10.1 Tg) (Fig. 2c), respectively, due to rising demand and decreased production costs, as the shrinking non-food production made key inputs more available to fertiliser production. This notable expansion in fertiliser production highlights China's transition from a net importer, with 3% (1.0 Tg) of nitrogen and 2% (0.3 Tg) of phosphorus fertilisers imported in the baseline (S0), to an exporter, with 22-24% (11.8-12.7 Tg) of nitrogen and 15-34% (3.1-9.3 Tg) of phosphorus fertilisers exported (Fig. 2i). Despite these notable relative increases, the absolute value of fertiliser output (currently representing 0.5% of China's total sectoral value-added, see Supplementary Fig. 8) rose by only 5.4-7.0 billion USD (Supplementary

Fig. 7a), which were considerably smaller than the absolute changes observed in the non-food sector. From an economy-wide perspective, the economic losses in the crop and non-food sectors were largely offset by the expansion of the monogastric livestock and fertiliser sectors (Supplementary Fig. 7a), resulting in a slight overall negative impact on China's economy, with a 0.02-0.07% (0.8-2.6 billion USD) decrease in GDP (Supplementary Fig. 9). Despite the slight negative impact on GDP, slight overall positive impacts were observed on household welfare (0.18-0.32%) and household expenditure (0.15-0.27%) in China (Supplementary Fig. 10) due to a reduction in net exports.

Asymmetric impacts on global environmental sustainability and food security.

Shifts in production, consumption, and trade patterns had asymmetric impacts on global environmental sustainability and food security. In terms of environmental sustainability, our findings revealed trade-offs and synergies among different types of emissions. While emissions from crop, livestock, and fertiliter production in China would rise, other non-agriculture emissions would decline, making the overall impact on economy-wide emissions dependent on which change, the increase or the decrease, was more dominant (Supplementary Fig. 11). We found that expanded monogastric livestock (1.22-1.89 Tg NH_3-eq) production raised Chinese economy-wide emissions of acidification polluants by 2.5-4.0% (0.83-1.36 Tg NH_3-eq) (Fig. 4e). Economy-wide emissions of eutrophication pollutants decreased by 0.2% (0.02 Tg N-eq) with partial upcycling but increased by 0.2% (0.02 Tg N-eq) with all upcycling (Fig. 4f). The 0.5-1.4% (56-163 Tg CO_2-eq) decease in economy-wide GHG emissions was dominated by reduced food waste in landfills and incinerators (119-222 Tg CO_2-eq), along with contractions in non-food (98-145 Tg CO_2-eq) production (Fig. 4d). China's main food and feed trading partners (MTP, including Brazil, the United States, and Canada) experienced a reduction in economy-wide emissions of GHGs by 1.1-1.3% (85-102 Tg CO_2-eq), acidification pollutants by 8-13% (1.13-1.80 Tg NH_3-eq), and eutrophication pollutants by 2.5-4.0% (0.14-0.22 Tg N-eq). These environmental benefits for MTP arise from a reduction in their domestic livestock and fertilizer production, as China shifted from a net importer to an exporter of livestock products and fertilisers (Fig. 2h,i).

For assessing food security, we used four indicators covering two dimensions: two indicators for food availability (dietary calorie availability and the population at risk of hunger) and two indicators for food access (cereals affordability for labour force and the average food price). Our findings suggested that upcycling and resource reallocation across food systems enhanced food security in China without compromising its trading partners. In addition, the reduced cost of collecting food waste for landfill and incineration allowed consumers in China to allocate more of their income to food consumption. The availability dimension of food security showed an increase in dietary calorie availability by 0.16-0.32% (4.3-9.6 kcal capita^{-1} day^{-1}) and a 1.43-2.98% (2.2-4.3 million people) reduction in the population at risk of hunger in China and MTP. More specifically, dietary calorie availability in China increased by 0.16-0.32% (5.2-10.3 kcal capita^{-1} day^{-1}), and the population at risk of hunger, representing 17% of the global population at risk of hunger, decreased by 1.6-3.2% (2.2-4.5 million people) (Fig. 5c,d). In contrast, for MTP, its dietary calorie availability decreased by 0.02-0.03% (0.7-0.9 kcal capita^{-1} day^{-1}), and the population at risk of hunger, accounting for 0.6% of the global population at risk of hunger, rose by 2.3-3.0% (0.1-0.2 million people) (Fig. 5g,h). The access dimension of food security also improved in China and MTP. Globally, the average food price saw a moderate decrease of 0.14-0.23% (Fig. 5a,e). In China, cereals affordability for labour force increased by 0.29-0.47% (Fig. 5b), as a result of a rise in the average wage across the Chinese economy (0.13-0.22%) (Supplementary Fig. 3) and a decrease in cereals price (0.16-0.26%) (Supplementary Fig. 12). In contrast, while cereals affordability for labour force in MTP increased by 0.15-0.28% (Fig. 5f), this increase was smaller compared to the rise in China.

Addressing rebound effects through emission taxes while safeguarding global food security. The above results underscore the asymmetric impacts of upcycling food waste as feed in China on food security and environment sustainability, urging complementary measures and policies to mitigate negative spillovers while safeguarding global food security. To address this, building on the upcycling of partial food waste as feed (S1), we further assessed the impacts of implementing economy-wide emission taxes to achieve two mitigation targets: scenario 3 (S3 = S1 + A modest emission mitigation target), ensuring that emissions of GHGs, acidification pollutants, and eutrophication pollutants in both China and MTP do not exceed their baseline (S0) levels, and scenario 4 (S4 = S1 + An ambitious emission mitigation target) aligned with the annual mitigation target of the Intended Nationally Determined Contributions (INDC) under the Paris Agreement [23,24] and China's "13th Five-Year Plan" [25]. Economy-wide emission taxes would incentivise producers and consumers to shift from emission-intensive commodities to cleaner alternatives, and every good would be primarily produced in regions with relatively lower emission intensities. In this way, this approach allows us to identify the most cost-effective mitigation pathway for achieving a given emission mitigation target.

Our findings demonstrated that a modest mitigation target could address rebound effects of upcycling food waste as feed in China while safeguarding global food security. We found that combining upcycling partial food waste as feed with implementing economy-wide emissions taxes to achieve a modest emission mitigation target (S3) not only reversed the increase in Chinese economy-wide acidification pollutants from 2.5% (0.83 Tg NH_3-eq) to a decrease of 0.1% (0.04 Tg NH3-eq), but also led to an additional reduction of 0.4% (44 Tg CO_2-eq) in GHG emissions and 0.4% (0.04 Tg N-eq) in eutrophication pollutants compared to scenario S1 (Fig. 4d,e,f). In terms of food security, the changes in food security indicators under scenario S3 were nearly identical to those in scenario S1, indicating that achieving a moderate emission mitigation target did not adversely affect global food security (Fig. 5). This is because the modest emission mitigation target involved only a low tax rate on economy-wide emissions of acidification pollutants (3 \$ ton^{-1} NH_3-eq) in China, given that upcycling partial food waste as feed (S1) only increased economy-wide emissions of acidification pollutants in China. The reduction in emissions of all pollutants in China under scenario S3 was mainly attributed to a 2.1% (28 Tg) further decrease in total crop production compared to scenario S1 (Fig. 2a), which led to reductions in GHG emissions by 51 Tg CO_2-eq, emissions of acidification pollutants by 0.82 Tg NH_3-eq, and emissions of eutrophication pollutants by 0.01 Tg N-eq (Fig. 4d,e,f). More specifically, the reduction in emissions resulting from decreased production of crops with relatively high emission intensities (i.e., cereal grains, sugar crops, and other non-food crops) outweighed the emissions increase from the higher production of crops with relatively low emission intensities (i.e., oilseeds & pulses, vegetables & fruits, and roots & tubers) (Supplementary Fig. 11a,b,c), leading to a net reduction in emissions from crop production. In scenario S3, changes in livestock production were similar to those in scenario S1, with a further 0.4% reduction (0.4 Tg) in monogastric livestock and a 0.03% decrease (0.01 Tg) in ruminant livestock production compared to S1 (Fig. 2b). Given that phosphorus fertiliser production was relatively 'cleaner' compared to nitrogen fertiliser production, China further increased phosphorus fertiliser production by 40% (7 Tg) while reducing nitrogen fertiliser production by 6% (2 Tg) compared to scenario S1 (Fig. 2c). As a result, in MTP, economy-wide emissions of GHGs, acidification pollutants and eutrophication pollutants further increased by 0.4% (32 Tg CO_2-eq), 2.3% (0.32 Tg NH_3-eq), and 0.1% (0.01 Tg N-eq), respectively, compared to scenario S1 (Fig. 4d,e,f) due to the shift of emission-intensive production from China to MTP through international trade; nonetheless, emissions of all pollutants in MTP still remained below baseline levels.

In contrast, we observed that an ambitious emission mitigation target could counteract rebound effects and achieve further emission reduction, but it posed a risk to global food security. Our analysis revealed that combining upcycling partial food waste as feed with implementing economy-wide emissions taxes to achieve an ambitious emission mitigation target (S4) raised the average global food price by 9.4% (Fig. 5a,e) and reduced cereals affordability for labour force by 20% in China (Fig. 5b) and 15% in MTP (Fig. 5f). On the one hand, the negative impact on the access dimension of food security in China and MTP was due to the high tax rates on economy-wide emissions in both regions required to achieve the ambitious emission target: 5 \$ ton^{-1} CO_2-eq , 788 \$ ton^{-1} NH_3-eq, and 6969 \$ ton^{-1} N-eq in China, and 2.5 \$ ton^{-1} CO_2-eq in MTP. To be more specific, prices in relatively 'dirtier' agricultural sectors increased significantly, for example, cereal grains by 12.4%, monogastric livestock by 9.6%, and ruminant livestock by 20.7% (Supplementary Fig. 12). On the other hand, the impact on the the availability dimension of food security was negative for MTP but positive for China in scenario S4. High emission tax rates led to increased food prices and a 3.27% (108.2 kcal $capita^{-1}$ day^{-1}) reduction in dietary calorie availability in MTP, mostly from cereal grains (47.2 kcal $capita^{-1}$ day^{-1}), monogastric livestock (9.6 kcal $capita^{-1}$ day^{-1}), and ruminant livestock (73.6 kcal $capita^{-1}$ day^{-1}) (Fig. 5h). Consequently, the population at risk of hunger in MTP increased by 346% (18.3 million people) (Fig. 5g). Conversely, the 3.64% (116.2 kcal $capita^{-1}$ day^{-1}) increase in dietary calorie availability in China resulted from higher calorie availability from crops (111.1 kcal $capita^{-1}$ day^{-1}) and monogastric livestock (11.0 kcal $capita^{-1}$ day^{-1}), which outweighed the decrease from ruminant livestock (5.9 kcal $capita^{-1}$ day^{-1}) (Fig. 5d). As a result, the population at risk of hunger in China declined by 35.85% (50.4 million people) (Fig. 5g). Globally, we observed a 0.12% (8.0 kcal capita^{-1} day^{-1}) increase in dietary calorie availability and a 22% (32.1 million people) reduction in the population at risk of hunger in China and MTP. In China, the 2.6% reduction in economy-wide GHG emissions (305 Tg CO_2-eq) and the 2.5% decrease in emissions of acidification pollutants (0.88 Tg NH_3-eq) were both largely driven by a 2.3% further reduction in non-food production compared to scenario S1. This reduction contributed to a further decrease of 240 Tg CO_2-eq in GHG emissions and 0.42 Tg NH_3-eq in emissions of acidification pollutants compared to scenario S1 (Fig. 4d,e). Additionally, a 2.0% reduction in economy-wide emissions of eutrophication pollutants (0.21 Tg N-eq) in China was mainly achieved by shifting livestock production from ruminant livestock to monogastric livestock. Although monogastric livestock production was slightly higher than in the baseline (S0), it remained below levels in scenario S1, leading to a reduction of 0.09 Tg N-eq in emissions compared to scenario S1 (Fig. 4f; Supplementary Fig. 11). Furthermore, reduced ruminant livestock production led to an additional decrease of 0.12 Tg N-eq in emissions compared to scenario S1 (Fig. 4f; Supplementary Fig. 11). For MTP, the 2.0% reduction in economy-wide GHG emissions (162 Tg CO_2-eq) was largely attributed to reductions in total crop and livestock production, which accounted for a further decrease of 37 Tg CO_2-eq and 55 Tg CO_2-eq in GHG emissions, respectively, compared to scenario S1 (Fig. 4d). Meanwhile, this reduction in economy-wide GHG emissions also led to a 4.6% (0.63 Tg NH_3-eq) decrease in emissions of acidification pollutants and a 4.6% (0.26 Tg N-eq) decrease in emissions of eutrophication pollutants in MTP (Fig. 4e,f).

Discussion

Our integrated environmental-economic framework complements previous linear optimisation studies [9,12,13], which overlooked market-mediated responses via the price system by considering both direct and indirect (price-induced) effects of upcycling food waste as feed. In contrast to previous linear optimisation studies that assume livestock production remains unchanged as long as feed protein and energy are maintained, our modelling framework enables us to capture the indirect "rebound effect" of 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 environmental impacts when transitioning to more circular food systems. 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. Our results, thus, enhance the understanding of synergies and trade-offs between economic impacts and multiple environmental stresses associated with upcycling food waste as feed.

Policy implications.

Policymakers focused on reducing the environmental impact of food systems and enhancing food security may find our findings particularly informative, as we unveiled the asymmetric impacts of upcycling food waste as feed on food security and environment sustainability. The reduction in GHG emissions, coupled with the enhancements in food security, underscores the rationale for policymakers to promote the adoption of feeding food waste strategies. This also aligns with China's recent emphasis on carbon neutrality and food security as leading priorities [27,28]. Despite these benefits of upcycling food waste as feed, policymakers should remain vigilant regarding indirect effects and spillovers, particularly the unintended increases in emissions of acidification and eutrophication pollutants. To avoid unintended negative environmental impacts and achieve the dual dividend of environmental sustainability and food security, it is essential to carefully design and implement tailored, complementary policies and measures rather than relying on a single, one-size-fits-all solution. Therefore, our findings hold following policy implications.

First, our study highlights the need to integrate both food security and environmental sustainability into policy decisions to leverage potential win-win opportunities. This is crucial, as our findings reveal that upcycling food waste as feed in China has asymmetric impacts on food security and environment sustainability, largely due to rebound effects of expanded monogastric livestock production. Since the food system plays a crucial role in driving numerous global and regional environmental challenges [29], any alterations in food systems are likely to have significant environmental effects[30,31]. Thus, there is a strong interconnection between food security and environmental sustainability. Given that food security and environmental sustainability represent major challenges for humanity, efforts to address both issues are anticipated to increase, as demonstrated by initiatives such as the United Nations Sustainable Development Goals [14]. Consequently, policymakers should pay closer attention to the interconnection between food security and environmental sustainability to better leverage potential synergies and minimize trade-offs [32]. In China, the responsibility for food security and environmental sustainability often falls to different government agencies, highlighting the pressing need for improved coordination and consistency within the government to effectively tackle these intertwined issues [33]. Our study provides an example of how combining upcycling food waste as feed with implementing economy-wide emissions taxes can address rebound effects while safeguarding global food security. We find that an ambitious emission mitigation target (i.e., emission taxes to meet Paris Agreement goals) could counteract rebound effects but has negative impacts on global food security. Conversely, a modest emission mitigation target (i.e., emission taxes to maintain baseline levels) provides an opportunity to address rebound effects while safeguarding global food security. In this way, our analysis demonstrates how a carefully designed policy combination can achieve the dual dividend of environmental sustainability and food security.

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 [9,12,13,34]. 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. While upcycling food waste as feed has been shown not to affect livestock productivity [10], to gain acceptance and adoption among livestock producers, food waste protein production must demonstrate its economic competitiveness against conventional feed proteins such as cereals and oilseeds. Upcycling all food waste as feed necessitates various investments and policies to support the construction of municipal food waste collection plants to efficiently collet, sanitize, and package food waste for sale to livestock producers as feed [12]. Achieving near-full use of food waste as feed appears feasible in China in the future due to several reasons. The food waste treatment industry (i.e., food waste collection service and food waste recycling service) has seen significant development and expansion in recent years [35]. Reinforced policies on municipal solid waste separation and collection guarantee a stable feed supply for monogastric livestock production [36]. For example, the Chinese government recently launched an action plan to reduce reliance on soybean imports, which includes a key initiative to trial feed production from food waste in 20 cities by 2025 [37]. Additionally, the geographic proximity of industrial livestock farms to municipal food waste collection plants further facilitates the success of upcycling food waste as feed for monogastric livestock production [35].

Third, our study assumes that individuals employed in non-agricultural sectors can shift to agricultural-related sectors under a constant total labour supply within the economy, following the default settings of standard GTAP [26] model. 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 specialised 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 structure in China [38] 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.

Future outlooks.

Despite the integrated and holistic approach, our 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 [39] and GTAP-AEZ [40] 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 [12,13,34], 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 endeavour 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 [9,12,13,34]. 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") [14], high priority should be placed on reducing food waste. With less food waste available for animal feed, the impacts of upcycling food waste as feed may diminish. However, we consider our estimates of the impacts of upcycling food waste as feed as conservative, as we did not factor in cross-provincial transportation of food waste with high moisture content (except in scenario S2). Fourth, the integrated environmental-economic framework we presented here could be expanded to evaluate health impacts resulting from changes in food consumption, such as diet- and weight-related risks [41]. A framework that integrates these three aspects would enhance policy design aimed at achieving the triple benefits of environmental sustainability, food security, and public health. 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. While the static model has limitations in short-term policy analysis, it minimises assumptions and uncertainties about future economic conditions by not considering technological and resource changes over time, allowing us to isolate the impact of feeding China's monogastric livestock with food waste. Despite the need for further research, our study provides a starting point by offering an integrated environmental-economic framework that addresses synergies and trade-offs within the food-land-water-climate nexus and supports policy design aimed at achieving the dual dividend of environmental sustainability and food security. Moreover, 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 [42-46]. For this study, we developed a global comparative static AGE model, a modified version of an integrated environmental-economic model, [47-49] 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 [50].

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. [51], 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 [26] was used to calibrate our AGE model and provide dollar-based quantities. We designed a sectoral aggregation scheme comprising 16 sectors (see Appendix Table 1) from the original GTAP database to produce social accounting matrices (SAM) (see Appendix Tables 2-3) in our study. Data on physical quantities (see Supplementary Table 1) of crop and livestock production was obtained from FAO [21]. Feed production was extracted from "Feed" in the FAO food balance sheet. Grass from natural grassland was derived from Miao and Zhang [52]. We only included grass from natural grassland where ruminant livestock is grazing for feed, and grass from remaining grassland was excluded. Data on the trade shares matrix was calculated from the data from the UN Comtrade Database [53]. 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, the ratio of fresh feed inputs to live weight gain), and edible feed conversion ratio (eFCR, the amount of human-edible feedstuffs like feeding crops and compound feed used for per unit of live weight gain) [54] 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 the 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 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).

Modelling assessment of food waste.

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 [55]. Food waste was quantified separately for each type of food product using data on food consumption and China-specific food loss and waste fractions [22] following the FAO methodology [56]. 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 [57]. The total amounts of food waste and food processing by-products and their current use as animal feed and discarded biomass (i.e., landfill and incineration) for China in S0 were presented in Supplementary Tables 3.

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 [58]. 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 their income to both the consumption of goods and food waste collection services, but they derive utility solely 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.

Environmental impact assessment.

In this study, 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. [59]. Data on CO_2, CH_4, and N_2O emissions were obtained from the Climate Analysis Indicators Tool (CAIT) [60]. All GHG emissions calculations in our model follow the IPCC Tier 2 approach [61]. We derived NH_3, NO_x, and SO_2 emissions from Liu, et al. [62], Huang, et al. [63], and Dahiya, et al. [64], respectively. We considered NO_x emissions from energy use only, as agriculture's contribution to NO_x emissions is generally small ($\leqslant$2%). We used the global eutrophication database of food and non-food provided by Hamilton, et al. [7] to obtain data on N and P losses to water bodies. 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). More detailed information about emissions sources of greenhouse gases, acidification pollutants, and eutrophication pollutants across various sectors of the model was provided in Appendix Table 4. The sector-level emissions of GHGs (Tg CO_2 equivalents), acidification pollutants (Tg NH_3 equivalents), and eutrophication pollutants (Tg N equivalents), as well as the US dollar-based emission intensities of GHGs (t CO_2 equivalents million USD^{-1}), acidification pollutants (t NH_3 equivalents million USD^{-1}), and eutrophication pollutants (t N equivalents million USD^{-1}), were presented in Appendix Tables 5-7 and Appendix Tables 8-10, 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).

We focused on two types of agricultural land, i.e., cropland and pastureland. We updated the GTAP data on crop harvested areas using the FAO [21] database. In our model, pastureland was defined as areas where ruminant grazing occurs, which explains the difference between pastureland and grassland statistics. The remaining grassland in was exclued due to their primary ecological functions rather than agricultural use. Additionally, we derived data on nitrogen and phosphorous fertiliser use by crop types and countries from Ludemann, et al. [65].

Food security indicators.

The FAO [66] 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). In this study, we focused on indicators related to the first two dimensions. The availability dimension is illustrated using two indicators. First, food availability is defined as the 'calories per capita per day available for consumption,' as calculated by our model. Second, '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 [67]. The approach has been widely used in agricultural economic models to evaluate the risk of food insecurity [20,68,69]. In essence, the population at risk of hunger is determined by multiplying the prevalence of undernourishment (PoU) by the total population and is based on dietary energy availability calculated by our model. According to the FAO approach, it is assumed that there is no risk of hunger for high-income countries in Europe, North America, and Oceania. Consequently, the population at risk of hunger is not applied to the United States and Canada (refer to reference [20,68,69] for additional details). The access dimension is tied to people's purchasing power, which depends on food prices, dietary habits, and income trends [70]. First, our model could calculate the average food (including primary agricultural products and processed food) price, which does not account for income changes. Second, given that cereals (including paddy rice, wheat, and other cereals) are the primary diet component for the low-income population, we could calculate changes in cereals affordability for labour force by subtracting changes in the average wage across the whole economy from fluctuations in cereal prices.

Definition of scenarios.

We examined five scenarios: one baseline (S0) scenario representing the economies of China and MTP in 2014, two scenarios involving changes in animal diets without mitigation targets and two scenarios with both changes in animal diets and mitigation targets. These scenarios were compared to a 2014 baseline (S0) scenario without changing animal diets. When substituting primary feed (i.e., human-edible feed crops and compound feed) with food waste and food processing by-products, we maintained the protein and energy feed supply for per unit of animal output in all scenarios to prevent imbalances between nutritional (protein and energy) supply and livestock requirements. The scenarios mentioned above were further described in Table 1 and SI.

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

S2 - Full use of food waste as feed. Scenario S2 analysed the environmental and economic impacts of upcycling all food waste as feed (100% of food waste and 100% of food processing by-products allowed to be used as feed for monogastric livestock), taking into account economies of scale. In S2, cross-provincial transportation of food waste was allowed in S2. Economies of scale in food waste recycling were considered in S2, where a 1% increase in recycled waste resulted in only a 0.078% rise in recycling costs, indicating that increasing the amount of recycled waste might not necessarily incur additional costs, as reported by Cialani and Mortazavi [71]. This is because, initially, recycling entails high fixed costs, yet as production scales up, marginal costs decrease and stabilise.

S3 - S1 + A modest emission mitigation target. In scenario S3, economy-wide uniform emission taxes were applied across all sectors (crop, livestock, and non-food) at the regional level to achieve a modest emission mitigation target, ensuring that emissions of GHGs, acidification pollutants, and eutrophication pollutants in both China and MTP do not exceed their baseline (S0) levels. For a given emission mitigation target for each type of pollutant, the AGE model can endogenously determine the emission taxes for various pollutants (expressed in \$ per ton of CO_2 equivalents, \$ per ton of NH_3 equivalents, and \$ per ton of N equivalents). This approach is the most commonly used in the literature [20,69,72,73] and allowsus to identify the most cost-effective mitigation pathway for achieving a given emission mitigation target.

S4 - S1 + An ambitious emission mitigation target. In scenario S4, economy-wide uniform emission taxes were implemented across all sectors (crop, livestock, and non-food) at the regional level to achieve an ambitious emission mitigation target. This ensures that emissions of GHGs, acidification pollutants, and eutrophication pollutants in both China and MTP remain within the emission thresholds set by their annual mitigation target of the Intended Nationally Determined Contributions (INDC) under the Paris Agreement [23,24] and China's "13th Five-Year Plan" [25].

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).

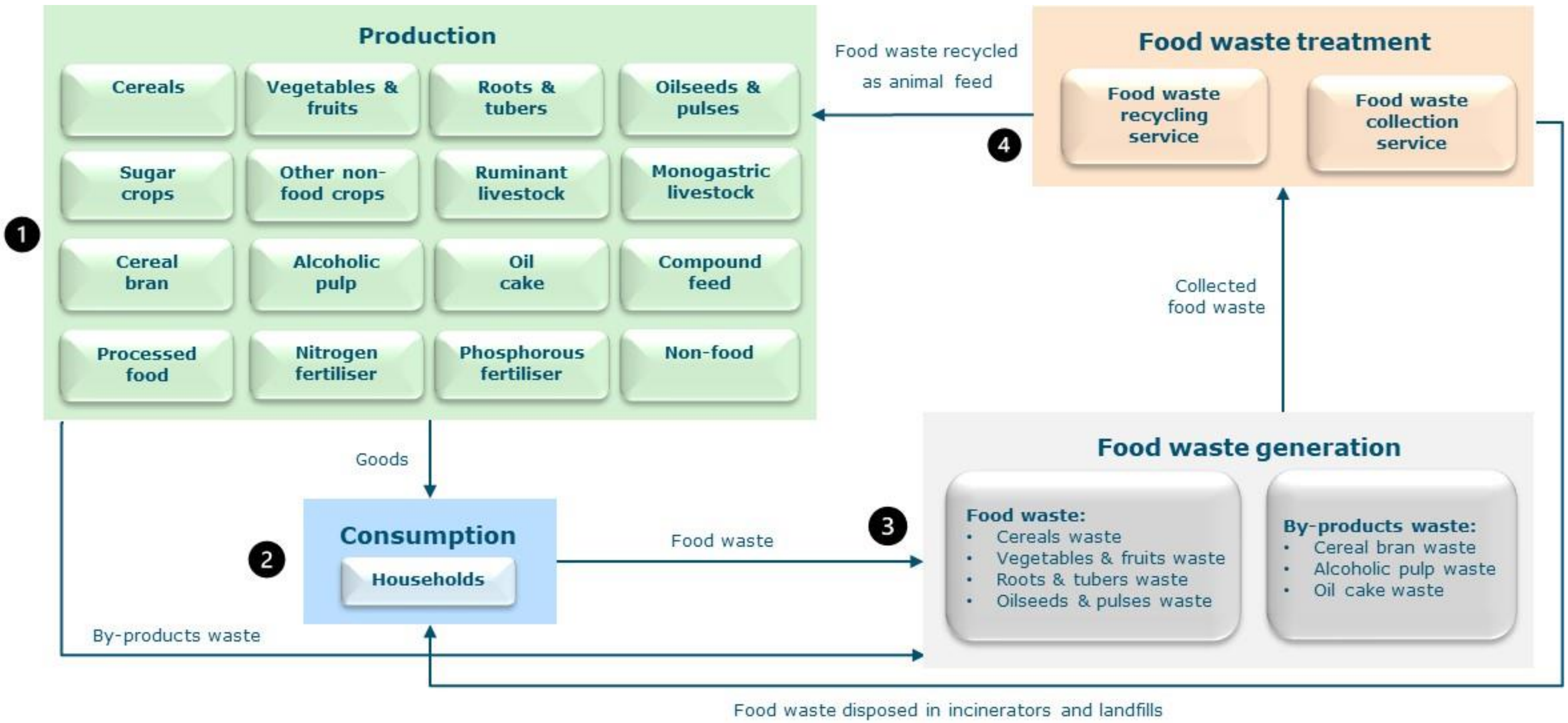

Fig. 1 | Representation of the economy in China in an applied general equilibrium (AGE) framework with the module of food waste treatment. The framework includes four parts: (1) Production; (2) Consumption; (3) Food waste generation; (4) Food waste treatment. The generated food waste is sent either to the 'food waste recycling service' sector or the 'food waste collection service' sector. The food waste recycling service sector 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.

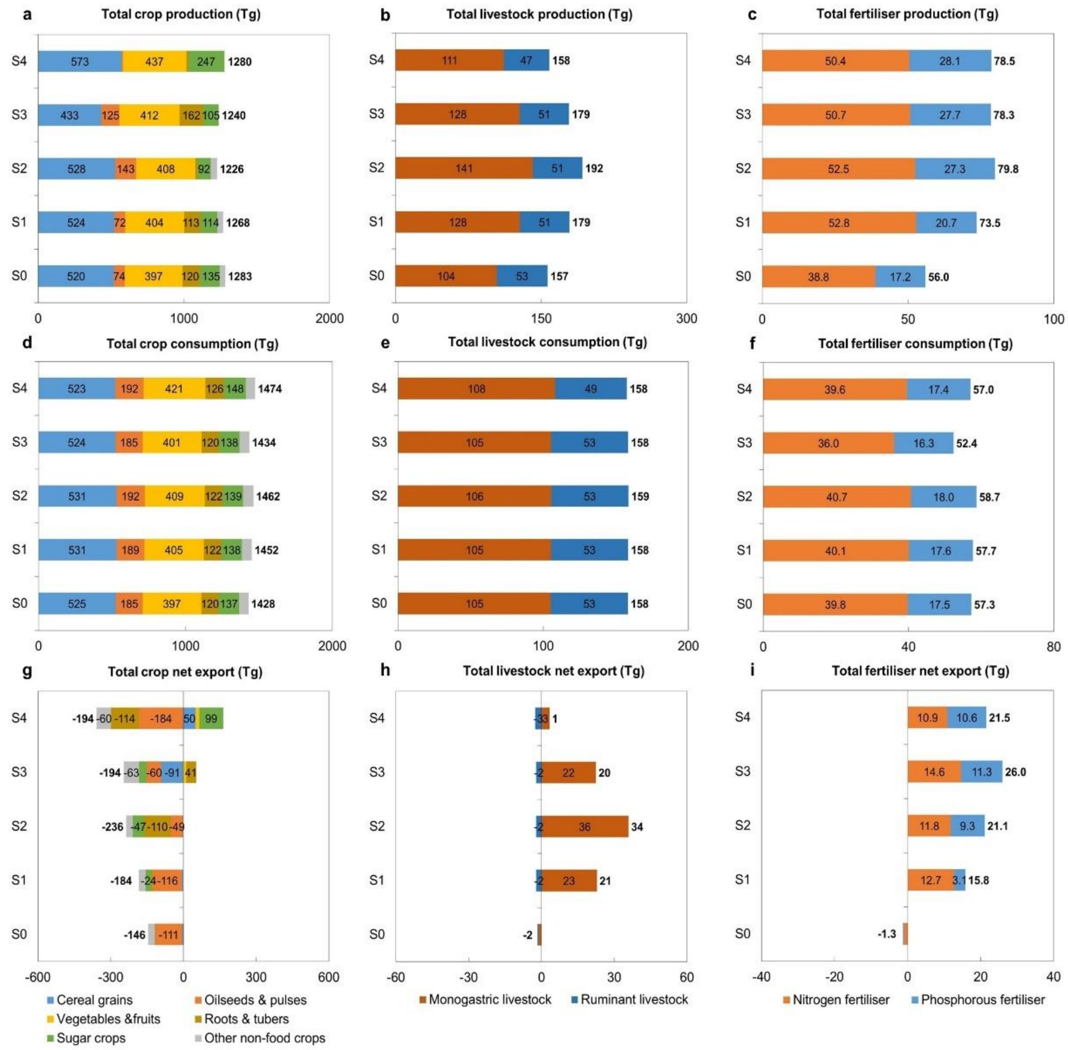

Fig. 2 | Impacts of upcycling food waste in China's monogastric livestock as feed on domestic production, consumption, and trade of total crop, livestock, and fertiliser. (**a, d, g**) Total crop production (Tg), consumption (Tg), and net export (Tg) in scenarios. (**b, e, h**) Total livestock production (Tg), consumption (Tg), and net export (Tg) in scenarios. (**c, f, i**) Total fertiliser production (Tg), consumption (Tg), and net export (Tg) in scenarios. Total crop production and consumption exclude food waste and food processing by-products used by "food waste recycling service" and "food waste collection service" sectors (see Supplementary Table 3 for detailed data). Total crop consumption includes crop used for intermediate use (i.e, feeding crops, compound feed, food by-products, processed food) and direct consumption (i.e., primary fresh food). Definitions of scenarios (S1 - 'Partial use of food waste as feed'; S2 - 'Full use of food waste as feed'; S3 - 'S1 + A modest emission mitigation target'; S4 - 'S1 + An ambitious emission mitigation target') are described in Table 1.

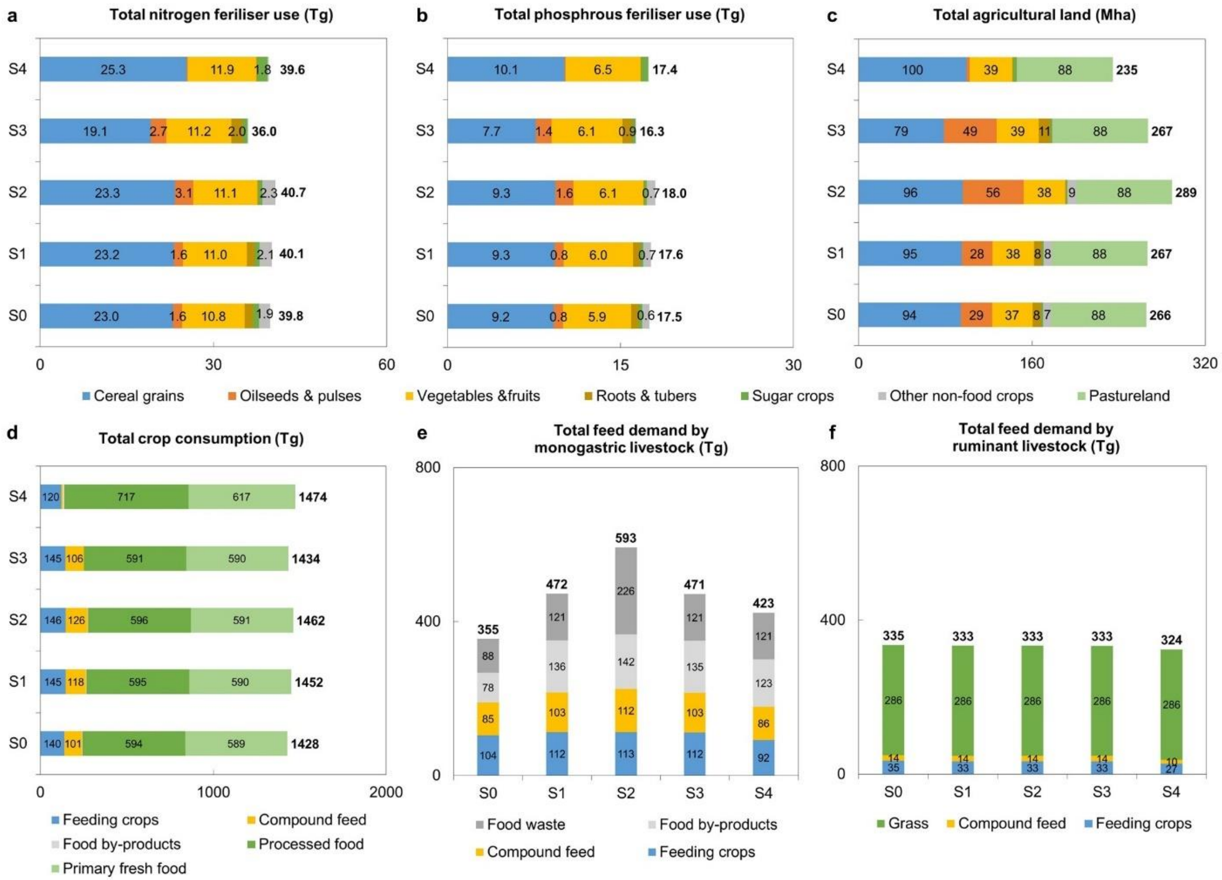

Fig. 3 | Impacts of upcycling food waste in China's monogastric livestock as feed on domestic total fertiliser use, harvested area, crop consumption, and feed demand. (**a**) Total nitrogen fertiliser use (Tg), (**b**) phosphorous fertiliser use (Tg), (**c**) agricultural land (crop harvested area and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

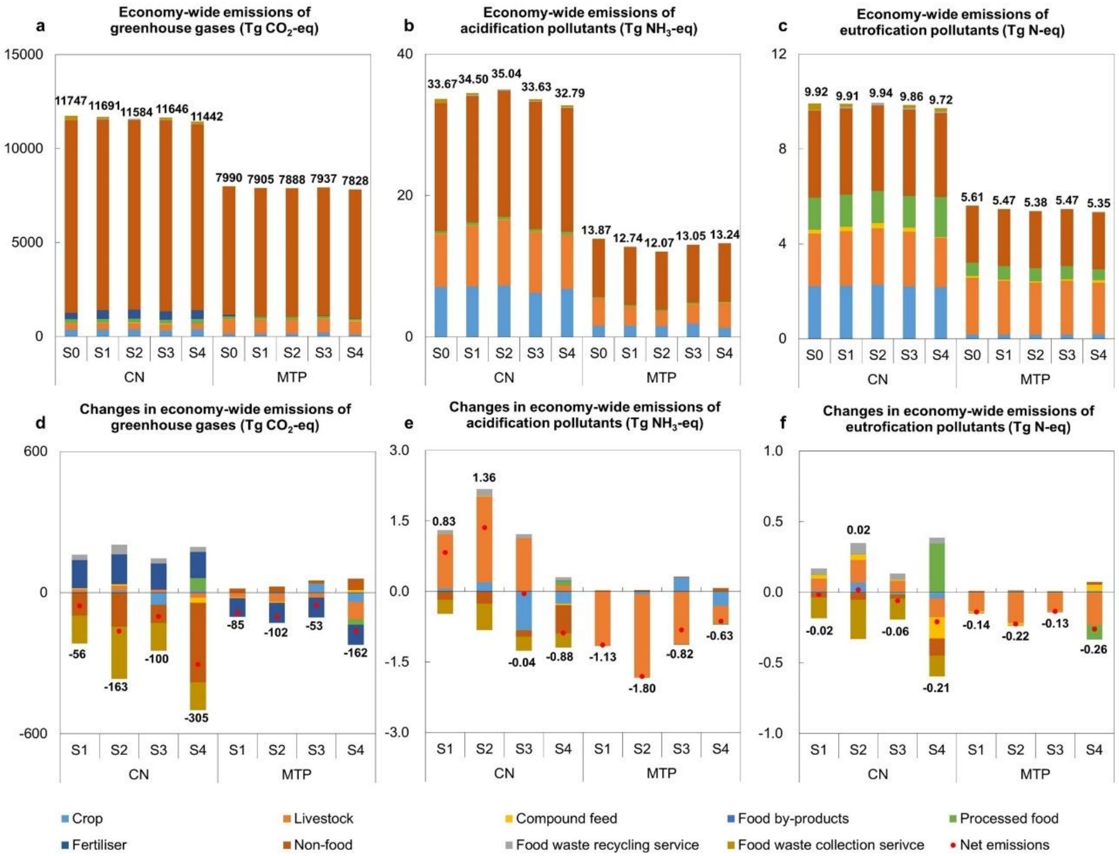

Fig. 4 | Impacts of upcycling food waste in China's monogastric livestock as feed on economy-wide emissions in China (CN) and China's main food and feed trading partners (MTP). (**a**) Economy-wide emissions of greenhouse gases (Tg CO_2-eq), (**b**) acidification pollutants (Tg NH_3-eq), and (**c**)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

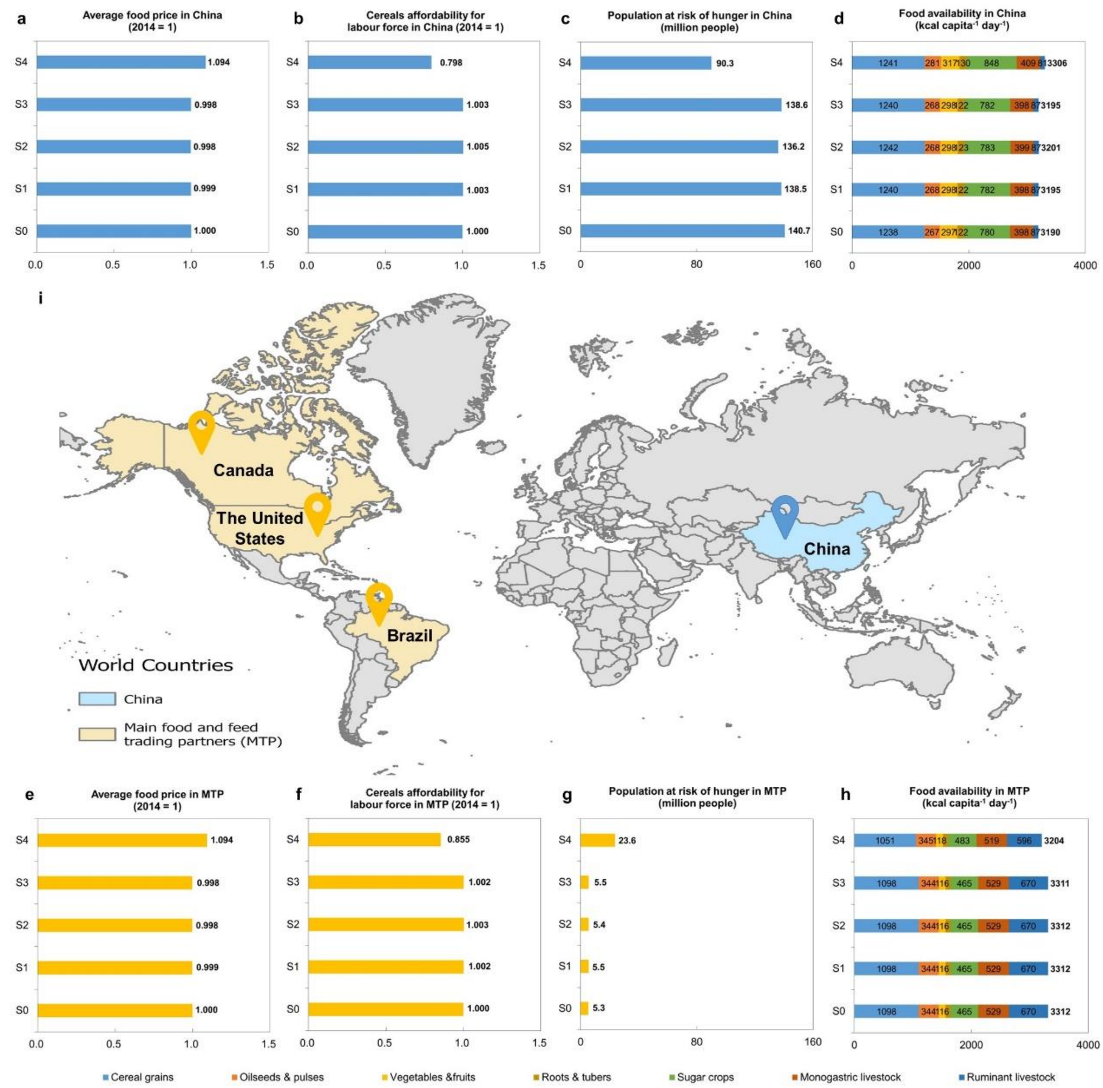

Fig. 5 | Impacts of upcycling food waste in monogastric livestock as feed on food security indicators in China (CN) and China's main food and feed trading partners (MTP). (**a**) Average food (including primary agricultural products and processed food) price, (**b**) cereals affordability for labour force, (**c**) population at risk of hunger (million people), and (**d**) food availability (kcal capita^{-1} day^{-1}) in scenarios in China. (**e**) Average food (including primary agricultural products and processed food) price, (**f**) cereals affordability for labour force, (**g**) population at risk of hunger (million people), and (**d**) food availability (kcal capita^{-1} day^{-1}) in scenarios in MTP. (**i**) Geographic location of China and MTP. 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 in Europe, North America, and Oceania. Consequently, the population at risk of hunger is not applied to the United States and Canada (detailed in reference [20,68,69]). Definitions of scenarios (S1 - 'Partial use of food waste as feed'; S2 - 'Full use of food waste as feed'; S3 - 'S1 + A modest emission mitigation target'; S4 - 'S1 + An ambitious emission mitigation target') are described in Table 1. Credit: World Countries base map, Esri (https://hub.arcgis.com/datasets/esri::world-countries/about).

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

Scenarios [a]	**Food waste used as animal feed in its total supply** [b]	**Emission mitigation target**
S0: Baseline	Food waste: 39% By-products: 51%	No
S1: Partial use of food waste as feed [c]	Food waste: 54% By-products: 100%	No
S2: Full use of food waste as feed [c]	Food waste: 100% By-products: 100%	No
S3: S1 + A modest emission mitigation target [d]	Food waste: 54% By-products: 100%	Implementing economy-wide emission taxes to control emissions of greenhouse gases, acidification pollutants, and eutrophication pollutants in both China and its main food and feed trading partners (MTP, including Brazil, the United States, and Canada) no more than their baseline (S0) levels.
S4: S1 + An ambitious emission mitigation target [d]	Food waste: 54% By-products: 100%	Implementing economy-wide emission taxes to reduce emissions of greenhouse gases by 2.6% in China and 2.0% in MTP in line with their annual mitigation target of Intended Nationally Determined Contributions (INDC) under the Paris Agreement [23,24]. Implementing economy-wide emission taxes to reduce emissions of acidification and eutrophication pollutants in China by 2.5% and 2.0%, respectively, according to the annual mitigation target set by the "13th Five-Year Plan" [25]. Implementing economy-wide emission taxes to control emissions of acidification and eutrophication pollutants in MTP no more than the baseline (S0) level.

[a] When substituting primary feed (i.e., crops and compound feed) in animal diets with food waste and food processing by-products, we maintained the protein and energy supply for 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. [12], whereas it was allowed in S2.

[c] The cost of increasing the supply of food waste recycling service is modelled as a rising percentage of the initial cost of recycling food waste as feed (54 dollar ton^{-1}), while the cost of decreasing the supply of food waste collection service is modelled as a declining percentage of the initial cost of collecting food waste for landfill and incineration (82 dollar ton^{-1}). Economies of scale in food waste recycling were considered in S2, where a 1% increase in recycled waste resulted in only a 0.078% rise in recycling costs, indicating that increasing the amount of recycled waste might not necessarily incur additional costs, as reported by Cialani and Mortazavi [71]. This is because, initially, recycling entails high fixed costs, yet as production scales up, marginal costs decrease and stabilise. The total amounts of food waste and food processing by-products and their current use as animal feed and discarded biomass (i.e., landfill and incineration) for China in S0 were presented in Supplementary Tables 3. Physical quantities and prices of food waste recycling service and food waste collection service in China were presented in Supplementary Tables 3-4.

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