Land-use emission leakages from China's dietary shift and afforestation amplify food insecurity and economic losses under the 2 °C target

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Abstract

Shifting to healthy diets and afforestation can reduce greenhouse gas (GHG) emissions and enhance carbon sequestration. However, when implemented unilaterally by China, the world's largest agricultural producer and GHG emitter, these measures may support domestic carbon neutrality while triggering emission leakages abroad through trade-induced land-use changes. Using an integrated environmental-economic model, we find that China's dietary shift with less meat and more dairy leads to an additional 14 Mha of agricultural land use abroad and causes 364 Tg CO_2-eq of emission leakage, more than tripling the domestic mitigation. Similarly, China's afforestation policy expands food production overseas, resulting in 16 Mha of additional agricultural land use and 424 Tg CO_2-eq of emission leakage, largely offsetting the domestic reduction. These emission leakages, primarily driven by land-use change, necessitate more stringent mitigation in non-agricultural sectors to meet the 2 °C target. Without coordinated global actions, these unilateral measures amplify pressures on carbon tax (36%), food price (65%), the population at risk of hunger (145 million people), and economic losses ($74 billion) under the 2 °C target.

Keywords

diet shift; afforestation; food security; land-based mitigation; climate change mitigation.

Main

36 Food systems exert tremendous pressure on planetary boundaries (PB) regarding climate change,

37 ocean acidification, biogeochemical flows, and land-use changes [1]. Within food systems, the

38 agriculture, forestry, and other land use (AFOLU) sector plays a central role, accounting for 20-25%

39 of global greenhouse gas (GHG) emissions [2] and serving as a major source of pollutants that drive

40 acidification and eutrophication [1]. For example, livestock production and fertiliser use emit methane

41 (CH_4) and nitrous oxide (N_2O), which contribute to global warming; ammonia (NH_3), which leads

42 to acidification; and nitrogen (N) and phosphorus (P) losses, which cause eutrophication [1,2]. Given

43 its contribution to both climate change and air pollution, the AFOLU sector is increasingly

44 recognised as a critical component of global mitigation strategies. This is particularly relevant in the

45 context of the Paris Climate Agreement, which seeks to limit global warming to well below 2°C and

46 possibly below 1.5°C above pre-industrial levels [3]. While energy system decarbonisation remains

47 central to this goal, growing evidence suggests that these temperature thresholds are unattainable

48 without concurrent mitigation in the AFOLU sector [4].

49 As the world's largest GHG emitter and agricultural producer and importer, and with just 7% of

50 global arable land supporting 19% of the global population [5], China's actions in the AFOLU domain

51 play a pivotal role for global sustainability. In line with its 2030 carbon peaking and 2060 carbon

52 neutrality goals [6]. China has recently proposed two key demand-side and land-based mitigation

53 measures in the AFOLU domain: (1) a dietary transition toward healthier, less meat-intensive

54 consumption [7], which may reduce domestic agricultural emissions and free up land for afforestation;

55 (2) a large-scale afforestation program provides direct negative emissions through carbon

56 sequestration [8]. However, when implemented unilaterally, such measures may support domestic

57 carbon neutrality while inducing environmental spillovers abroad. For example, Cai, et al. [9] show

58 that the dietary shift toward the Chinese Dietary Guidelines (CDG) nearly triples dairy consumption,

59 potentially triggering emission leakage through trade-induced land-use changes. Likewise, China's

60 afforestation policy [10] may displace food production to other countries, offsetting domestic

61 mitigation gains.

62   Despite growing recognition of these risks, few studies have comprehensively assessed the cross-

63   border impacts of China's demand-side and land-based mitigation measures. Implementing a global

64   uniform carbon tax is widely recognised as the most cost-effective policy instrument for reducing

65   GHG emissions and preventing emission leakage [11,12]. However, the carbon tax is often criticised

66   as regressive, as it places disproportionate burdens on low-income and vulnerable populations,

67   thereby conflicting with the principle of equity and common but differentiated responsibilities [13].

68   Recycling carbon tax revenues to households via lump-sum transfers can help ease these equity

69   concerns and mitigate adverse impacts on food security caused by the carbon tax [13,14]. Nevertheless,

70   how these aforementioned measures interact with the Sustainable Development Goals (SDGs) [15,16],

71   particularly SDG 2 (zero hunger), SDG 13 (climate action), and SDG 15 (life on land), remains

72   underexplored. For instance, China's unilateral actions intended to reduce domestic emissions may

73   inadvertently drive deforestation abroad, thereby amplifying food insecurity and economic losses

74   under the 2 °C target. These outcomes underscore the systemic risks of isolated climate measures,

75   and call for a nexus framework to achieve sustainable development and avoid shifting burdens

76   across borders or sustainability domains.

77   While the nexus concept has long been discussed in sustainability discourse, its application in

78   quantitative assessments has only recently been applied at the global scale [15]. In particular, little

79   attention has been paid to how unilateral climate actions by a major GHG emitter, such as China,

80   interact with globally coordinated measures. Expanding applied general equilibrium (AGE) models

81   or linking them with partial equilibrium (PE) approaches offers a promising pathway to examine

82   such systemic interdependencies at both national and global scales [17-19]. To address these gaps, our

83   study provides a food-land-climate nexus framework that balances food security, land sustainability,

84   and climate goals. Specifically, we develop an integrated environmental-economic model based on

85   an AGE framework to assess the environmental and economic impacts of China's dietary shift and

86   afforestation under the 2 °C target, capturing both domestic impacts in China and cross-border

87   effects on its major food and feed trading partners (MTP, i.e., Brazil, the United States, and Canada).

88   This study makes four key contributions. First, unlike previous studies that analyse individual

89   climate mitigation measures in isolation, it integrates four distinct scenarios (dietary shift,

90   afforestation, carbon tax with revenue recycling, and combined measures) to understand their

4

91 cumulative and interactive effects. Second, while most studies focus on domestic impacts, it

92 examines how China's domestic measures affect MTP, revealing hidden leakage effects, such as

93 increased deforestation abroad. Third, it identifies counterintuitive trade-offs, for example, how

94 China's dietary shifts and afforestation reduce domestic GHG emissions but trigger emission

95 leakages abroad through trade-induced land-use changes. Fourth, it presents an enhanced AGE

96 model that improves the representation of biophysical flows, integrates biophysical data (i.e., land

97 cover, fertiliser use, and emissions), and incorporates a food security module into the economic

98 analysis. This enables a more holistic assessment of climate mitigation measures and supports the

99 design of more coherent and globally aligned sustainability strategies.

100 In this study, we focus on the improvement of one or more components in the food-land-climate

101 nexus. Five scenarios are examined: one baseline (S0) scenario, three scenarios focusing on

102 improving one nexus component, and one combined scenario focusing on improving all nexus

103 components. The baseline (S0) represents the economic and environmental conditions of all sectors

104 (including agriculture, industries, and services) in the entire economies of China and MTP in 2014.

105 The food scenario (S1) aligning with SDG 2 (zero hunger) indicates China's dietary shift towards a

106 less animal-based diet, closing 20% of the gap between current food consumption and the Chinese

107 Dietary Guidelines (CDG) 2022 [20], considering the unaffordability of a complete dietary shift for

108 low-income households. The land scenario (S2) supporting SDG 15 (life on land) represents China's

109 afforestation policy based on China's National Forest Management Plan (2016-2050) [10], which aims

110 to expand forest land in China by 23 Mha (equivalent to 4% of China's agricultural land) by 2030.

111 The climate scenario (S3) in line with SDG13 (climate action) presents a global uniform carbon tax

112 with revenue recycling via lump-sum transfers to achieve the 2 °C climate target set by the Paris

113 Agreement [3], which aims to reduce net total GHG emissions in China and MTP by 25% by 2030.

114 The combined scenario (S4: S1+S2+S3) integrates China's dietary shift (S1) and afforestation

115 policy (S2) with the implementation of a global uniform carbon tax with carbon tax revenue

116 recycling (S3). Key food security (food price, food affordability, food availability, and the

117 population at risk of hunger) and environmental sustainability (cropland and pastureland use,

118 nitrogen and phosphorus fertiliser use, and economy-wide emissions of GHGs, acidification

119 poulltants, and eutrophication pollutants) and economic (sectoral employment and gross domestic

120   product (GDP)) indicators are assessed for China and MTP. Detailed scenario assumptions and are

121   provided in the Supplementary Information (SI).

Results

China's dietary shift causes carbon leakage that is more than triple the domestic mitigation.

124   In line with the CDG dietary guidelines, the dietary shift in China (S1) reduces the intake of cereal

125   grains (5%), roots & tubers (10%), and sugar crops (15%) (**Supplementary Table 10**), directly

126   lowering cropland demand. It also decreases the consumption of non-ruminant meat (11%)

127   (**Supplementary Table 10**), which indirectly eases cropland pressure by reducing demand for feed

128   crops. However, the dietary shift leads to only a marginal decline in domestic cropland (0.06%; 0.08

129   Mha) and fertiliser use (0.63%; 0.4 Tg) (**Supplementary Fig. 4a**; **Fig. 1f**) due to increased intake

130   of oilseeds & pulses (10%) and vegetables & fruits (6%) (**Supplementary Table 10**). Similarly,

131   although the reduced intake of ruminant meat (13%) lowers pastureland demand, this effect is

132   largely offset by the increased dairy consumption (52%), resulting in a -0.86% (3 Mha) reduction

133   in domestic pastureland use (**Supplementary Table 10**; **Supplementary Fig. 4e**). The freed

134   agricultural land is converted into a 3.04% (3 Mha) increase in domestic forest land for afforestation

135   (**Supplementary Fig. 4i**). Outside China, pastureland expands by 3.29% (15 Mha), primarily at the

136   expense of forest land (2.95%; 14 Mha) and cropland (0.38%; 0.98 Mha) (**Supplementary Fig.**

137   **4a,e,i**). The largest expansion occurs in the United States, mainly driven by a 10.23% (14.63 Mha)

138   increase in pastureland used for ruminant meat production, while the increase in pastureland for

139   dairy production is relatively small (0.09%; 0.11 Mha) (**Fig. 2a**; **Supplementary Fig. 4e**). This is

140   because China almost offsets rising dairy demand through domestic expansion, but reduces

141   ruminant meat production more than consumption, resulting in a 32% increase in the net import rate

142   of ruminant meat (**Fig. 1b,e,h**).

143   While China's dietary shift delivers domestic co-benefits by reducing agricultural emissions of

144   GHGs, acidification pollutants, and eutrophication pollutants, these reductions are partially or fully

145   offset by cross-sectoral and cross-border emission leakages, with carbon leakage more than tripling

146   the domestic reductions, whereas leakages of other pollutants remain relatively smaller. We find

147 that China's dietary shift reduces domestic GHG emissions from crop and livestock production by

148 1.7% (6 Tg CO_2-eq) and 18.3% (59 Tg CO_2-eq), respectively (**Supplementary Fig. 8a, 9a**).

149 Afforestation of freed-up agricultural land from the dietary contributes an additional 53 Tg CO_2-eq

150 reduction in domestic GHG emissions (**Fig. 2b**). However, these reductions are partially offset by a

151 0.2% (26 Tg CO_2-eq) increase in non-agricultural emissions (**Supplementary Fig. 10a**), as labour

152 saved from the agricultural sectors is reallocated to the expansion of non-agricultural production

153 (**Supplementary Fig. 16a**). As a result, China's economy-wide GHG emissions decline by only 0.8%

154 (93 Tg CO_2-eq) (**Fig. 3a**). Since agricultural activities account for a larger share of economy-wide

155 emissions of acidification and eutrophication pollutants than of GHGs, the dietary shift results in

156 even greater reductions in these pollutants, i.e., 3.47% (1.17 Tg NH_3-eq) and 4.20% (0.42 Tg N-eq),

157 respectively, in China's economy-wide emissions (**Fig. 3e,i**). Meanwhile, China's dietary shift also

158 triggers emissions leakage abroad due to trade-induced land-use changes. Economy-wide emissions

159 leakage amounts to 393% for GHGs, 5% for acidification pollutants, and 18% for eutrophication

160 pollutants, relative to the corresponding domestic reductions in China (**Fig. 3a,e,i**). The substantial

161 carbon leakage is primarily driven by deforestation in the United States, which increases GHG

162 emissions from land-use change by 342 Tg CO_2-eq (**Fig. 2b**).

163 China's dietary shift also raises food security concerns. It leads to a 5.6% (168 kcal capita^{-1} day^{-1})

164 reduction in domestic food availability (**Supplementary Fig. 3a**), resulting in a 55% (77 million

165 people) increase in the population at risk of hunger in China (**Fig. 4b**). Although China consumes

166 less of certain foods under the dietary shift, the global average food price (including primary food

167 products and processed food) increases by 0.12% (**Supplementary Table 17**), primarily due to a

168 0.37-0.83% increase in land rents in both China and MTP (**Supplementary Fig. 11a**). Considering

169 that prices reflect only part of the picture in assessing food affordability, we estimate changes in

170 food affordability by subtracting changes in the average wage across the entire economy from

171 fluctuations in cereal prices, given the role of cereals as a primary calorie source for low-income

172 households. We find that the dietary transition reduces food affordability in both China and MTP

173 by 0.15-0.22% (**Fig. 4a**), driven by a 0.15% increase in global cereal prices and a 0.003-0.070%

174 decline in wages (**Supplementary Fig. 12a, 13a**).

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China's afforestation induces carbon leakage that offsets domestic reduction.

Under China's afforestation policy (S2), domestic forest land expands by 20% (23 Mha), primarily through the conversion of 6% (22 Mha) of pastureland and 0.42% (1 Mha) of cropland (**Supplementary Fig. 4a,e,i**). This reduction in agricultural land decreases crop and livestock production in China by 2% and 4%, respectively, leading to increases in net import rates by 1% for crops and 8% for livestock (**Fig. 1**). The decline in crop production also leads to a 4.60% (2.6 Tg) reduction in domestic fertliser use (**Fig. 1f**). Outside China, pastureland expands by 3.85% (17 Mha), largely at the expense of forest land (3.43%; 16 Mha) and cropland (0.46%; 1.20 Mha) (**Supplementary Fig. 4a,e,i**). The United States accounts for the majority of this expansion, driven primarily by a 7.90% (11.29 Mha) increase in pastureland for ruminant meat production and a 2.53% (2.90 Mha) increase for dairy production (**Fig. 2b**; **Supplementary Fig. 4f**).

Although carbon leakage from China's afforestation is smaller than that from dietary shifts, it fully offsets domestic GHG reductions, whereas larger leakages of acidification and eutrophication pollutants offset nearly half of the domestic domestic mitigations, as land-use change only contributes to GHG emissions rather than these pollutants. We find that China's afforestation reduces domestic GHG emissions from land-use change by 352 Tg CO_2-eq (**Fig. 2d**), alongside additional reductions of 4.9% (18 Tg CO_2-eq) and 9.2% (30 Tg CO_2-eq) from crop and livestock production, respectively (**Supplementary Fig. 8b, 9b**). However, part of the emission reductions is offset by a 0.2% (25 Tg CO_2-eq) increase in non-agricultural emissions (**Supplementary Fig. 10b**)., as labor released from the agricultural sector is redirected to expand non-agricultural activities (**Supplementary Fig. 16e**). Consequently, China's economy-wide GHG emissions decline by 3.4% (374 Tg CO_2-eq) (**Fig. 3b**). This also yields co-benefits by reducing domestic economy-wide emissions of acidification and eutrophication pollutants by 3.51% (1.18 Tg NH_3-eq) and 2.24% (0.22 Tg N-eq), respectively (**Fig. 3f,j**). Furthermore, China's afforestation induces emissions leakage abroad, offsetting 113%, 49%, and 47% of domestic reductions in economy-wide emissions of GHGs, acidification pollutants, and eutrophication pollutants, respectively (**Fig. 3b,f,j**). The carbon leakage is primarily driven by deforestation outside China, which increases GHG emissions from land-use change by 330 Tg CO_2-eq in the United States, 66 Tg CO_2-eq in Brazil, and 10 Tg

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203   CO_2-eq in Canada (**Fig. 2b**). In contrast, the leakages of acidification and eutrophicatiojn pollutants

204   are mainly attributed to increased livestock production in the United States (**Fig. 3f,j**).

205   China's afforestation has relatively smaller adverse impacts on food security compared to the dietary

206   shift scenario. As it does not alter consumers' dietary patterns, it has no effect on food availability

207   (**Supplementary Fig. 3b**) and the population at risk of hunger (**Fig. 4d**). Although afforestation

208   creates competition between agricultural and forest land in China, and typically pushes up domestic

209   food prices. Given that our model adopts the Heckscher-Ohlin (H-O) assumption, which treats

210   domestic and imported goods as perfect substitutes, it captures only changes in global average prices.

211   The resulting increase in China's domestic prices can be attenuated by expanded agricultural

212   production in MTP, leading to modest global price increases or even slight declines, depending on

213   global supply. We find that the global average food price declines slightly by 0.02%

214   (**Supplementary Table 17**) under China's afforestation, reflecting the combined effect of falling

215   and rising prices across food categories. Specifically, dairy and ruminant meat prices decrease by

216   0.16% as production shifts from China to MTP with relatively higher yields, while crop and non-

217   ruminant meat prices rise by 0.02-0.03% due to their relocation to MTP regions with lower yields

218   compared to China (**Supplementary Fig. 12b**). Food affordability in China and MTP declines by

219   0.02-0.05% (**Fig. 4c**), reflecting the combined effects of rising global cereal prices (0.03%) and

220   wage changes, exacerbated in China by a 0.024% wage decrease and slightly alleviated in MTP by

221   wage increases of 0.001-0.008% (**Supplementary Fig. 12b, 13b**).

222   **A global uniform carbon tax with revenue recycling causes substantial economic losses.**

223   Under the 2 °C climate target (S3), a global uniform carbon tax of $43 t CO_2-eq^{-1} is required to

224   achieve a 25% (4752 Tg CO_2-eq) reduction in net total GHG emissions in China and MTP (**Fig. 3c**).

225   This global uniform carbon tax incentivises the production of each good to concentrate in regions

226   with lower emission intensities and away from GHG-intensive sectors. The target is met through a

227   31% (3408 Tg CO_2-eq) reduction in China and a 17% (1344 Tg CO_2-eq) reduction across MTP (**Fig.**

228   **3c**), indicating that China bears a relatively larger share of the GHG mitigation burden. Among all

229   regions, China achieves the largest absolute GHG emissions reduction, followed by Brazil, the

230   United States, and Canada (**Fig. 3c**). The largest contribution to total GHG emissions reduction

comes from non-agricultural sources in China, especially the non-food sector (3283 Tg CO_2-eq) (**Fig. 3c**; **Supplementary Fig. 10c**). Within the non-agricultural sectors, the fertiliser sectors in China account for the second-largest reduction (314 Tg CO_2-eq) (**Fig. 3c**). Within the AFOLU sectors, the most significant reductions stem from land-use changes associated with forest carbon sequestration in Brazil (760 Tg CO_2-eq) and the United States (188 Tg CO_2-eq) (**Fig. 3c**), owing to their relatively high sequestration intensities and large areas of accessible forest land compared to other regions (**Supplementary Tables 3, 13**). Among livestock products, ruminant meat has the highest GHG intensity and therefore contributes the most to emission reductions (486 Tg CO_2-eq) (**Supplementary Fig. 9c**; **Appendix Table 10**). Ruminant production declines in China (90%), Brazil (90%), and Canada (90%), while it increases in the United States (38%), which benefits from its relatively lower GHG intensity (**Supplementary Figs. 9c**, **15i-l**). In terms of crop sectors, cereal grains exhibit the highest GHG intensity and account for the largest emissions reduction (36 Tg CO_2-eq) among crops (**Supplementary Fig. 8c**). Cereals production expand in Brazil (297%), where GHG intensity is relatively low, while contracting in China (7%), the United States (90%), and Canada (90%) (**Supplementary Figs. 8c**, **15i-l**). The global uniform carbon tax also generates co-benefits by reducing total emissions of acidification and eutrophication pollutants in China and MTP by 12% (5.64 Tg NH_3-eq) and 5% (0.79 Tg N-eq), respectively (**Fig. 3g,k**). Notably, total emissions of eutrophication pollutants in China even increase by 6% (0.61 Tg N-eq) (**Fig. 3k**), reflecting the pollution-swapping effect. These relatively smaller reductions, or even increases in China, compared with GHGs are primarily due to the expansion of non-ruminant meat (98%) and processed food (155%) production in China (**Supplementary Figs. 9g**, **10k**, **15i-l**), which have relatively low GHG intensities but relatively high acidification and eutrophication intensities (**Appendix Tables 10-12**). These shifts in production structure result in GDP losses of 4.9% ($185 billion) for China, 0.4% ($5 billion) for Brazil, 0.1% ($10 billion) for the United States, and 0.4% ($4 billion) for Canada (**Fig. 5e,f**).

Despite the rise in global food prices and worsened affordability under the carbon tax, revenue recycling to households via lump-sum transfers helps mitigate the adverse food security impacts of a global uniform carbon tax aimed at the 2 °C target. The global uniform carbon tax leads to a 162% increase in the global average food price (**Supplementary Table 17**), with particularly sharp

increases in "dirty" agricultural sectors, such as cereal grains (220%) and ruminant meat (212%) (**Supplementary Fig. 12c**). The price of cereals surpasses that of ruminant meat, primarily due to the sharp rise in nitrogen fertiliser prices (474%), as nitrogen fertiliser has the highest GHG intensity among all sectors (**Supplementary Fig. 12c**; **Appendix Table 10**). Food affordability in China and MTP declines by 226-275% (**Fig. 4e**), reflecting the combined effects of soaring cereal prices and falling wages (6-54%) (**Supplementary Figs. 12c, 13c**). The global uniform carbon tax also encourages consumers to shift from "dirty" food products (e.g., cereal grains and ruminant meat) to "clean" food products (e.g., sugar crops and non-ruminant meat) (**Supplementary Fig. 3c**). While such a tax is generally expected to reduce food availability due to higher production costs and food prices, we observe a notable increase in China (7%; 221 kcal capita^{-1} day^{-1}), in contrast to declines in Brazil (18%; 617 kcal capita^{-1} day^{-1}), the United States (23%; 746 kcal capita^{-1} day^{-1}), and Canada (4%; 127 kcal capita^{-1} day^{-1}) (**Supplementary Fig. 3c**). This counterintuitive outcome in China is largely due to the concurrent revenue recycling via lump-sum transfers. In China, the recycled carbon tax revenue amounts to 0.21% of GDP, higher than in Brazil (0.06%), the United States (0.05%), and Canada (0.07%) (**Supplementary Fig. 19**), thereby mitigating the adverse effects on food availability. As a result, the population at risk of hunger decreases by 102 million people in China but increases by 37 million people in Brazil (**Fig. 4f**). Following the FAO approach, risk of hunger is not applied to high-income countries; thus, no changes are reported for the United States and Canada.

Combined measures amplify food insecurity and economic losses under the 2 °C target.

Combined measures (S4) undermine the feasibility of the 2 °C climate target (S3) due to emission leakages, primarily driven by land-use change, from China's dietary shift (S1) and afforestation (S2), which raise the global uniform carbon tax by 36%, from \$43 t CO₂-eq^{-1} in S3 to \$59 t CO₂-eq^{-1} in S4. This is because emission leakages, primarily driven by land-use change, increase pressure on CO_2 emissions in the energy system to achieve the same 2 °C climate target, thereby necessitating more stringent mitigation in non-agricultural sectors (an additional reduction of 296 Tg CO₂-eq relative to S3) (**Fig. 3c,d**). This heightened constraint on the energy system results in a 12-384% increase in non-agricultural sector prices, particularly for nitrogen fertiliser (384%) and non-food

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goods (99%) (**Supplementary Fig. 12c,d**). While still meeting the same 2 °C climate target, the combined measures (S4) result in additional reductions in total emissions in China compared to S3: 7% (764 Tg CO_2-eq) for GHGs, 8% (2.75 Tg NH_3-eq) for acidification pollutants, and 4% (0.41 Tg N-eq) for eutrophication pollutants. In contrast, MTP sees increases in total emissions relative to S3: 10% (764 Tg CO_2-eq) for GHGs, 0.2% (0.03 Tg NH_3-eq) for acidification pollutants, and 1% (0.07 Tg N-eq) for eutrophication pollutants (**Fig. 3c,d,g,h,k,i**). Despite these regional disparities, the combined measures (S4) deliver co-benefits by reducing total emissions of acidification and eutrophication pollutants in China and MTP by an additional 6% (2.72 Tg NH_3-eq) and 2% (0.33 Tg N-eq), respectively, compared to S3 (**Fig. 3g,h,k,i**). These additional reductions largely stem from the 127% (215 kcal capita^{-1} day^{-1}) lower China's non-ruminant meat consumption in S4 compared to S3 (**Supplementary Fig. 3c,d**), resulting from the dietary shift (S1). In contrast to S3, where the carbon tax alone promotes consumption substitution from ruminant to non-ruminant meat, S4 avoids the resulting increase in emissions from non-ruminant meat, which has relatively high acidification and eutrophication intensities.

Without coordinated global actions, combined measures (S4) amplify food insecurity and economic losses under the 2 °C target (S3). The global average food price increases by an additional 65% compared to S3 (**Supplementary Table 17**), with the most pronounced increases observed in cereal grains (73%) and ruminant meat (166%) (**Supplementary Fig. 12c,d**). Food affordability in China and MTP declines by 73-86% compared to S3 (**Fig. 4e,g**), driven by surging cereal prices and wage reductions (0-14%) (**Supplementary Figs. 12c-d, 13c-d**). Food availability in China falls by 13% (389 kcal capita^{-1} day^{-1}), while it increases inBrazil (17%; 584 kcal capita^{-1} day^{-1}), the United States (21%; 680 kcal capita^{-1} day^{-1}), and Canada (10%; kcal capita^{-1} day^{-1}) relative to S3 (**Supplementary Fig. 3c,d**). The decline in China is mainly due to the dietary shift (S1), while the increases in MTP reflect a 0.01-0.02% higher share of recycled carbon tax revenue relative to GDP in S4 (**Supplementary Fig. 19**), due to the higher carbon tax compared to S3. Consequently, the population at risk of hunger rises in China (179 million people) but falls in Brazil (35 million people), relative to S3 (**Fig. 4f,h**). Combined measures (S4) also lead to additional GDP losses relative to S3: 1.5% ($57 billion) in China, 0.2% ($3 billion) in Brazil, 0.1% ($10 billion) in the United States, and 0.5% ($4 billion) in Canada (**Fig. 5e,f,g,h**).

Discussion

Emission leakages through trade-induced land-use changes

In this study, we provide a food-land-climate nexus framework that balances food security, land sustainability, and climate goals. Specifically, we analyse the synergies and trade-offs arising from individually implemented mitigation measures (dietary shift, afforestation, and carbon tax with revenue recycling) as well as their combined application. Synergies reflect mutually complementary effects, while trade-offs indicate competing outcomes that constrain the effectiveness of the measures. Our comprehensive analysis underscores the importance of accounting for land-use emission leakages from China's unilateral mitigation measures to facilitate coordinated global actions in reducing GHG emissions, particularly in the context of global climate challenges. This is because emission leakages through trade-induced land-use changes undermine the feasibility of the 2 °C climate target, thereby amplifying food insecurity and economic losses.

Our findings should not be interpreted as a dismissal of mitigation measures, nor as a suggestion that they may do more harm than good. Rather, they serve as a caution against relying on a single, one-size-fits-all solution and, thus, highlight the need to deliberately and carefully combine mitigation strategies to minimise unintended consequences. Among the measures analysed, China's dietary shift and afforestation complement the global uniform carbon tax in reducing domestic GHG emissions. While these benefits are consistent with previous findings [9,11,21], our study further reveals competing mechanisms that trigger land-use emission leakages abroad, which are overlooked in earlier studies. We find that China's dietary shift with less meat and more dairy leads to an additional 14 Mha of agricultural land use abroad and causes 364 Tg CO_2-eq of emission leakage, more than tripling the domestic mitigation. Similarly, China's afforestation policy expands food production overseas, resulting in 16 Mha of additional agricultural land use and 424 Tg CO_2-eq of emission leakage, largely offsetting the domestic reduction. These emission leakages due to unilateral climate actions echo the findings of Zhong, et al. [22], who showed that meeting the 2030 agricultural and forestry targets of European Green Deal (EGD) could raise demand for agricultural land outside the EU by 24 Mha and result in 759 Mt CO_2-eq of land-use emissions, more than doubling the EGD's carbon removal target for land, land-use-change and forestry sectors. We demonstrate that a global

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345 uniform carbon tax to meet the 2 °C target increases the food price by 162% and reduces food

346 affordability in China and MTP by 226-275%. This aligns with Hasegawa, et al. [11], who found that

347 stringent climate mitigation policy would worsen food insecurity more than climate change itself.

348 Additionally, we find that recycling carbon tax revenues to households via lump-sum transfers leads

349 to heterogeneous food security outcomes: China sees improved food availability (7%; 221 kcal

350 capita^{-1} day^{-1}) and reduced population at risk of hunger (102 million people), whereas Brazil faces

351 declined food availability (18%; 617 kcal capita^{-1} day^{-1}) and increased population at risk of hunger

352 (37 million people). These disparities are driven by differences in recycled carbon tax revenue as a

353 share of GDP, with China at 0.21% and Brazil at 0.06%. Given that land-use emission leakages

354 require more stringent mitigation in non-agricultural sectors to meet the 2 °C target, we reveal that,

355 without coordinated global actions, these unilateral measures amplify pressures on carbon tax (36%),

356 food price (65%), the population at risk of hunger (145 million people), and economic losses ($74

357 billion) under the 2 °C target. These unintended trade-offs highlight the need for more integrated

358 and holistic measures to reduce emissions without amplifying food insecurity and economic losses.

359 **Possible measures to mitigate emission leakages, food insecurity, and economic losses**

360 To mitigate emission leakages due to China's unilateral measures, additional or alternative measures

361 are needed. First, one promising approach is to align global dietary patterns with the EAT-Lancet

362 recommendations, rather than the CDG, to further reduce demand for animal-based food. In the food

363 (S1) scenario, land-use emission leakage is primarily driven by China's attempt to close 20% of the

364 gap between current consumption and the CDG target, requiring a 52% increase in dairy intake. The

365 CDG recommendation of dairy intake (300-500 g day^{-1}) may be unrealistic, given the high

366 prevalence of lactose intolerance and bone fractures in China compared to Western populations [7].

367 By contrast, the EAT-Lancet diet recommends a considerably lower dairy intake (250 g day^{-1}). Guo,

368 et al. [23] showed that adopting the EAT-Lancet diet in China would reduce life-cycle GHG emissions

369 and land-use carbon opportunity costs by 24% and 47%, respectively, compared to the CDG diet,

370 due to the EAT diet's smaller increases in dairy intake and greater reductions in red meat

371 consumption. Furthermore, Sun, et al. [24] demonstrated that shifting to EAT-Lancet diet in high-

372 income countries could reduce agricultural emissions from their diets by 61% and sequester 98.3 Gt

373   CO_2-eq, about 14 years of global agricultural emissions. A recent study by Humpenöder, et al. [25]

374   found that a global shift to the EAT-Lancet diet could increase the 1.5 °C-compatible peak carbon

375   budget from 500 to 625 Gt CO_2-eq, thereby enhancing the feasibility of achieving 1.5 °C pathways

376   under the Paris Agreement. Nevertheless, shifting consumer behaviour toward healthier diets

377   remains challenging. In upper-middle- to high-income countries, healthy diets are estimated to be

378   22-34% cheaper than current diets, whereas in low- to lower-middle-income countries, they are 18-

379   29% more expensive [26]. Targeted educational programs, behavioural nudges, and informative

380   labeling, such as carbon and health labels, can raise consumer awareness of sustainable food choices

381   and promote sustainable dietary transitions [27,28]. Financial support, such as transfer payments or

382   revised food distribution schemes, is also crucial in providing economic incentives for the adoption

383   of healthy diets [25]. Second, agricultural intensification and afforestation on abandoned cropland

384   could help mitigate land-use emission leakages. In the land (S2) scenario, such leakages are

385   primarily driven by competition between agricultural and forest land within China, which reduces

386   domestic food production and increases reliance on imports, thereby triggering agricultural

387   expansion abroad. On the one hand, agricultural intensification could reduce agricultural land use,

388   thereby alleviating the pressure from large-scale afforestation. This is particularly relevant given

389   that direct global forest protection has been shown to be more effective than the EU's deforestation-

390   linked import restrictions, which are undermined by emissions leakage and economic losses [29].

391   However, large-scale global afforestation may exacerbate competition between agricultural and

392   forest land, raising concerns over potential trade-offs between climate goals and food security [30].

393   Doelman, et al. [31] found that globally large-scale land-based mitigation (~600 Mha by 2050) could

394   raise food prices by 11%, reduce food availability by 230 kcal capita^{-1} day^{-1}, and put 230 million

395   more people at risk of hunger, but these negative impacts on food security could be offset by a 9%

396   increase in crop yields and 3% intensification in ruminant production. Nevertheless, the feasibility

397   of future crop yield increases remains uncertain due to the negative impacts of climate change on

398   crop yields [31]. On the other hand, afforestation on abandoned cropland could also mimise the

399   competition for land. Wu, et al. [32] found that China had 31 Mha of abandoned cropland, 83% of

400   which (26 Mha) was suitable for recultivation, exceeding the 23 Mha afforestation target by 2030

401   in S2. Therefore, afforestation on abandoned cropland in China could fully meet, and even surpass,

402   China's 2030 afforestation goal. However, abandoned cropland can serve diverse purposes beyond

403   afforestation, such as recultivation, bioenergy with carbon capture and storage, and natural

404   vegetation regrowth, offering various benefits for climate mitigation, ecosystem restoration, and

405   biodiversity conservation [32,33]. Thus, a more pragmatic strategy is to balance these objectives rather

406   than dedicating all abandoned land solely to afforestation [34].

407   Combining climate policies with mixed carbon tax revenue recycling schemes and technology

408   transfer initiatives may support climate mitigation goals while avoiding increased food insecurity

409   and economic losses. In the climate (S3) scenario, we find that recycling carbon tax revenues to

410   households via lump-sum transfers helps mitigate the adverse food security impacts of a global

411   uniform carbon tax aimed at the 2 °C target, but this still comes at the cost of substantial economic

412   losses. Critics argue that a global uniform carbon tax may be unrealistic and inequitable, conflicting

413   with the principle of equity and common but differentiated responsibilities [35]. Frank, et al. [36] found

414   that targeting carbon taxes at land-rich, low-population countries could achieve mitigation with

415   minimal food security impacts. However, Hasegawa, et al. [37] warned that exempting certain regions

416   or sectors from the carbon tax could shift a greater and often more costly mitigation burden onto the

417   rest of the system, potentially jeopardizing the achievement of ambitious climate targets. Therefore,

418   complementary measures are needed to mitigate these negative impacts. For example, de Bruin and

419   Yakut [14] found that combining carbon taxation with allocating carbon tax revenues between sales

420   tax reductions and household transfers can achieve a triple dividend by reducing emissions,

421   increasing GDP, and enhancing income equality. Additionally, including the transfer of land- and

422   emission-saving production technologies from developed to developing countries in climate

423   agreements could support global mitigation efforts while fostering economic development [38].

424   Despite its integrated approach, this study has some limitations that necessitate some follow-up.

425   First, model simplifications, such as fixed budget shares for consumers, fixed cost shares for

426   producers, and the absence of trade barriers, may exaggerate trends but are appropriate for

427   illustrating potential synergies or trade-offs within the food-land-climate nexus. Second, our model

428   includes only China and MTP (i.e., Brazil, the United States, and Canada), which may underestimate

429   land-use emission leakages from China's dietary shift and afforestation, thereby further amplifying

430   food insecurity and economic losses under the 2 °C target. Although MTP accounts for over 75%

431   of China's total food and feed trade value, expanding the model to include the rest of the world,

432   especially China's main importers of dairy products and ruminant meat, such as Australia and New

433   Zealand, could provide a more comprehensive assessment. Nonetheless, the omission is expected

434   to result in limited underestimation, given Oceania's small contribution (0.3%) to the global

435   accessible forest land area [39]. Third, our analysis does not account for the adverse effects of climate

436   change on crop yields, which may lead to an underestimation of food insecurity and economic losses

437   under the 2 °C target, though this would not alter our main conclusions. While this falls beyond the

438   scope of our study, future research could include it in the food-land-climate nexus, as explored in

439   previous research [11,40]. Fourth, our static modelling framework reflects current economic conditions

440   and does not capture long-term dynamics (e.g., population growth, economic development, evolving

441   trade policies) or external shocks (e.g., African swine fever, the US-China trade war, COVID-19)

442   that may reshape agri-food systems. Future work could address these gaps through dynamic

443   modelling and additional scenario analyses. Further details on these limitations are provided in the

444   Supplementary Discussion.

445   **Methods**

446   **The integrated environmental-economic model and database.**

447   We developed a global comparative static applied general equilibrium (AGE) model, a modified
448   version of an integrated environmental-economic model, [41] and improved the representation of
449   agriculture, forestry, and other land use (AFOLU) (6 crop types, 3 livestock categories, and 1
450   forestry sector) and non-agricultural (compound feed, food processing by-products, processed food,
451   fertilisers, and non-food) sectors. While the static nature of the model limits its applicability to short-
452   term policy analysis, previous studies have demonstrated that it reduces reliance on assumptions
453   and uncertainties regarding future population and economic growth [40]. This allows us to isolate the
454   impacts of China's dietary shift and afforestation under the 2 °C target from other complex and
455   uncertain factors, by grounding the analysis in current economic conditions. Although such
456   interactions are important, they are beyond the scope of this study. Moreover, dynamic models
457   require extensive assumptions on these factors, which can complicate the interpretation of climate-
458   induced impacts and increase uncertainty [42].

459   AGE models, grounded in microeconomic theory, represent the entire economy by integrating
460   utility-maximising consumers, profit-maximising producers, and market-clearing conditions, where
461   relative prices adjust to equate total supply (domestic production and imports) and total demand
462   (intermediate use, household consumption, and exports), under perfect competition. For
463   international trade, our AGE model adopts the Heckscher-Ohlin (H-O) assumption, treating
464   domestic and imported goods as perfect substitutes, leading to specialised production in countries
465   with comparative advantages where goods can be produced most efficiently. A comprehensive
466   description of our AGE model specifications is provided in the Supplementary Information (SI).

467   In contrast to the previous version, which combined Brazil, the United States, and Canada into a
468   single region representing China's main food and feed trading partners (MTP), the current version

469 of our model disaggregates them into separate regions. This four-region framework, including China,
470 Brazil, the United States, and Canada, allows for a more detailed representation of bilateral trade
471 flows and enhances the model's capacity to assess context-specific impacts of land-use change. We
472 select 2014 as the reference year, as it is the most recent year in the Global Trade Analysis Project
473 (GTAP) database [43] at the time of our study. Our model is solved using the general algebraic
474 modelling system (GAMS) software package [44]. The rest of the world (RoW) is excluded from our
475 analysis, as GTAP [43] trade flow data indicates that, in 2014, over 75% of China's total food and feed
476 trade value was concentrated with MTP, whereas trade with RoW accounted for only 25%. This
477 highlights the central role of China's domestic food production in shaping its trade balance with
478 MTP, and our model can effectively capture the most significant trade flows influencing China's
479 food system.

480 To enable the simulation of one of our key scenarios, China's dietary shift towards the Chinese
481 Dietary Guidelines 2022 [20], we enhance the model's representation of biophysical flows, thereby
482 improving its ability to track changes in food demand. Following Gatto, et al. [45] and Chepeliev [46],
483 we convert dollar-based quantities (million USD) to physical quantities (Tg; $1 \text{ Tg} = 10^6$ tons) to
484 trace biophysical flows throughout the global economy. GTAP version 10 database [43] is used to
485 calibrate our AGE model and provide dollar-based quantities. We designed a sectoral aggregation
486 scheme comprising 18 sectors (see **Appendix Table 1**) based on the original GTAP database to
487 produce social accounting matrices (SAM) (see **Appendix Tables 2-5**) in our study. In the SAMs
488 from the GTAP database, dollar-based material balances for the reference year ensure that dollar-
489 based production quantities for each commodity across countries equal the sum of dollar-based
490 intermediate demand across sectors, dollar-based final demand from the representative consumer,
491 and dollar-based net exports. Dollar-based bilateral trade quantities in the GTAP database are
492 constructed based on the reconciled UN Comtrade Database [47], while physical bilateral trade
493 quantities are obtained from [5] trade data. To construct bilateral trade flows in physical quantities,
494 we prioritise FAO-reported imports over exports, assuming that import data is more reliable since
495 importers have a stronger incentive to provide accurate trade records for taxation purposes. We then
496 apply the RAS approach (also referred to bi-proportional balancing) [48] to balance physical bilateral
497 trade quantities, ensuring consistency between FAO-reported totals for exporting and importing
498 countries. Physical production quantities of crops, livestock, and fertilisers (see **Supplementary
499 Table 2**) are obtained from [5]. In addition to primary food products, we also account for food
500 processing by-products, which include cereal bran, alcoholic pulp (including distiller's grains from
501 maize ethanol production, brewer's grains from barley beer production, and distiller's grains from
502 liquor production), and oil cakes (including soybean cake and other oil cakes). Physical production
503 quantities of food processing by-products (see **Supplementary Table 2**) are estimated by
504 multiplying the production quantities of primary food products by FAO technical conversion factors
505 for various by-products [49]. Physical feed production quantities are extracted from the "Feed"
506 category in the FAO Food Balance Sheet (FBS). Since the FAO does not provide feed data by
507 livestock type, following Gatto, et al. [50], we allocated the physical production quantities of feed
508 across livestock sectors based on dollar-based feed demand shares across livestock sectors in the
509 SAMs from the GTAP database. To establish the link between dollar-based and physical quantities,
510 we define material intensity coefficients (kg USD^{-1}) for each commodity at the regional level as the
511 ratios of physical production quantities from FAO to dollar-based production quantities from GTAP,
512 estimating these coefficients using reference year data. This approach allows us to compute physical
513 material balances once dollar-based material balances are determined after each model run.
514 However, physical material balances may not hold for all commodities and countries. To address
515 this, we adjust physical production quantities to ensure that the supply of each commodity aligns
516 exactly with FAO-FBS data for further comparisons. This adjustment ensures consistent tracing of
517 material flows in both dollar-based and physical quantities for each commodity across countries.
518 For simulations using the static AGE model, physical material flows change proportionally to the
519 corresponding dollar-based material flows in response to an exogenous shock.

520 **Modelling land use change and forest carbon supply.**
521 In our model, the allocation of land is determined through a constant elasticity of transformation
522 (CET) function, which is widely used in the previous literature [40,42,51,52]. The rent-maximising
523 landowner initially determines the allocation of land among three land cover types, i.e., cropland,
524 pastureland, and forest land, based on relative returns to land. Subsequently, the landowner allocates

525 cropland among various crops and pastureland between dairy products and ruminant meat. A three-
526 tier structure of land supply is provided in **Supplementary Fig. 1**. Physical area of cropland,
527 pastureland, and forest land is obtained from [5]. Following the GTAP land use and land cover
528 database [39,53,54], we align the land cover data in our AGE model with FAO land cover data (see
529 **Supplementary Table 3**). We classify forest land into accessible and inaccessible categories based
530 on the country-specific shares of accessible forest area, using data provided by Sohngen, et al. [55]
531 and Baldos and Corong [54]. Accessible forests include both managed (timber-producing) and
532 unmanaged forests. Following Golub, et al. [42], inaccessible forest land is excluded from our
533 modelling. We assume that deforestation only occurs on the accessible forest land, which is less
534 costly to access than the remaining inaccessible forest land. Crop sectors account for 100% of total
535 cropland, while livestock sectors use 100% of total pastureland. Dairy and ruminant meat production
536 are assumed to use pastureland, whereas non-ruminant livestock production does not use land. The
537 forestry sector uses 100% of the accessible portion of total forest land cover. While we acknowledge
538 that other land types (SavnGrasslnd, Shrubland and inaccessible forests) are also potentially
539 cultivable, Golub, et al. [42] have demonstrated that this modelling approach captures the majority of
540 land classified by the FAO as potential arable land.

541 We link our AGE model with the Global Timber Model (GTM) [56,57], a partial equilibrium, dynamic
542 optimisation model representing the global forestry sector, by calibrating the forest carbon
543 sequestration component of the AGE model. The calibration procedure aimsto ensure that both
544 models produce consistent forest carbon sequestration responses under the same carbon tax rate.
545 The calibration is performed by adjusting the incremental annual forestry carbon sequestration
546 intensities to mimic the GTM's assumptions. In line with GTM, we assume that one hectare of new
547 forest sequesters the same amount of carbon, regardless of whether it is converted from cropland or
548 pastureland. Following Hertel, et al. [51] and Golub, et al. [58], forest carbon stocks can be increased by
549 increasing the biomass on existing forest acreage (the intensive margin) or by expanding forest land
550 (the extensive margin). Annual forestry carbon sequestration intensities (see **Supplementary Table
551 13**), derived from Nguyen, et al. [59] and China's National Forest Management Plan (2016-2050) [10],
552 are distributed evenly over a depreciation period of 20 years, as suggested by [60] and BSI [61].
553 Additional details are provided in Supplementary Methods.

554 **Environmental impact assessment.**

555 While the previous version of our model accounted only for production-related emissions across all
556 sectors in the entire economies of China and MTP, the current version expands the scope to include
557 greenhouse gas emissions from land-use change, specifically emissions associated with net forest
558 conversion (i.e., changes in forest carbon stocks). Detailed information about emission sources
559 across sectors is provided in **Appendix Table 6**.

560 Three main environmental impacts of food systems are distinguished, i.e., global warming potential
561 (GWP, caused by greenhouse gas (GHG) emissions, including carbon dioxide(CO_2), methane (CH_4),
562 and nitrous oxide (N_2O) emissions; converted to CO_2 equivalents), acidification potential (AP,
563 caused by pollutants leading to acidification, including ammonia (NH_3), nitrogen oxides (NO_x), and
564 sulphur dioxide (SO_2) emissions; converted to NH_3 equivalents), and eutrophication potential (EP,
565 caused by pollutants leading to eutrophication, including nitrogen (N) and phosphorus (P) losses;
566 converted to N equivalents). The conversion factors for GWP, AP, and EP are derived from
567 Goedkoop, et al. [62]. Data on CO_2, CH_4, and N_2O emissions are obtained from the Climate Analysis
568 Indicators Tool (CAIT) [63]. GHG emissions calculations in our model follow the IPCC National
569 GHG Emission Guidelines [64]. We derive NH_3, NO_x, and SO_2 emissions from Liu, et al. [65], Huang,
570 et al. [66], and Dahiya, et al. [67], respectively. We consider NO_x emissions from energy use only, as
571 agriculture's contribution to NO_x emissions is generally small (≤2%) [68]. We use the global
572 eutrophication database of food and non-food provided by Hamilton, et al. [69] to obtain data on N
573 and P losses to water bodies. Emissions of food processing by-products (i.e., cereal bran, alcoholic
574 pulp, oil cake) are derived from Mackenzie, et al. [70]. We attribute the environmental impacts
575 between the main (e.g., cereal flour) and joint products (e.g., cereal bran) according to their relative
576 economic values (see **Supplementary Table 8**). We derive nitrogen and phosphorous fertiliser use
577 by crop types and countries from Ludemann, et al. [71]. The total emissions of GHGs, acidification
578 pollutants, and eutrophication pollutants for the food and non-food sectors in the base year are
579 calculated first. Then, we allocate the total emissions to specific sectors according to the shares of
580 emissions per sector in total emissions to unify the emission data from different years. The sectoral-

581 level emissions as well as the US dollar-based emission intensities of GHGs (t CO_2 equivalents
582 million USD^{-1}), acidification pollutants (t NH_3 equivalents million USD^{-1}), and eutrophication
583 pollutants (t N equivalents million USD^{-1}) are presented in **Appendix Tables 7-12**.

Food security indicators.

585 The FAO [72] defines food security as encompassing four key dimensions: availability (adequate food
586 supply), access (sufficient resources to obtain food), utilisation (nutritious and safe diets), and
587 stability (consistent access to food over time). We focus on the first two dimensions. First, food
588 availability is defined as "calories per capita per day available for consumption". "Population at risk
589 of hunger" refers to the portion of people experiencing dietary energy (calorie) deprivation lasting
590 more than a year following the FAO-based approach [73]. In essence, the population at risk of hunger
591 is determined by multiplying the prevalence of undernourishment (PoU) by the total population.
592 According to the FAO, the PoU is based on dietary energy availability calculated by our model, the
593 mean minimum dietary energy requirement (MDER), and the coefficient of variation (CV) of the
594 domestic distribution of dietary energy consumption in a country. It is assumed that there is no risk
595 of hunger in high-income countries; consequently, the population at risk of hunger is not applied to
596 the United States and Canada. Second, the access dimension is tied to people's purchasing power,
597 which depends on food prices, dietary habits, and income trends. We calculate the average food
598 price (including primary food products and processed food) and estimate changes in food
599 affordability by subtracting changes in the average wage across the entire economy from
600 fluctuations in cereal prices.

Definition of scenarios.

602 We examined five scenarios: one baseline (S0) scenario and four scenarios of improvements in the
603 food-land-climate nexus components. The latter four scenarios were compared to the 2014 baseline
604 (S0) scenario. The scenarios are further described below and in **Supplementary Table 1**.

605 **S0 - Baseline.** The baseline (S0) represents the economic and environmental conditions of all sectors
606 (including agriculture, industries, and services) in the entire economies of China and MTP in 2014.
607 It corresponds to the Representative Concentration Pathway (RCP) 8.5 [74], the worst-case climate
608 change scenario in which emissions continue to increase with no mitigation. Note that the potential
609 negative impacts of climate change on crop yields are not considered in this study; therefore,
610 uncertainties associated with crop yield responses under alternative climate futures (RCPs) assessed
611 in previous research [11,40] fall beyond the scope of our analysis.

612 **S1 - Food scenario: A dietary shift in China.** Shifting towards less animal-based and more plant-
613 based diets is the most effective demand-side option to mitigate climate change [75]. Meat
614 consumption in China has exceeded the recommended consumption levels reported by the EAT-
615 Lancet diet [76] and the Chinese Dietary Guidelines (CDG) 2022 [20]. While the EAT-Lancet diet offers
616 a widely recognised global benchmark for healthy and sustainable eating, its limited consideration
617 of cultural, age-, and gender-specific dietary needs presents challenges for national-level
618 implementation [76]. In scenario S1, we focus on the CDG diet [20], which offers culturally tailored
619 dietary recommendations and promotes healthy Eastern dietary patterns that are more compatible
620 with Chinese preferences than global guidelines. Specifically, we simulate an exogenous dietary
621 shift in China toward the CDG diet recommendations. We first estimate the gap in food consumption
622 between current levels in China and the recommended targets in the CDG diet. Subsequently, we
623 adjust China's food consumption patterns to close 20% of this gap, considering the unaffordability
624 of a complete dietary shift for low-income households. Detailed information about the dietary shift
625 in China is provided in **Supplementary Table 10**.

626 **S2 - Land scenario: An afforestation policy in China.** Afforestation, with its potential for negative
627 GHG emissions, is widely recognised as a cost-effective and readily available climate change
628 mitigation option [21]. In line with its commitment to achieving carbon neutrality by 2060, the Chinese
629 government has proposed an ambitious afforestation target to support this goal. In scenario S2, we
630 simulate an afforestation policy in China based on the National Forest Management Plan (2016-
631 2050) [10]. This plan outlines an ambitious tree-planting program to expand forest land in China by
632 23 Mha (equivalent to 4% of China's agricultural land) by 2030.

S3 - Climate scenario: A global uniform carbon tax with carbon tax revenue recycling. Implementing a global uniform carbon tax on economy-wide GHG emissions (i.e., total emissions from all sectors in the entire economy) is considered an effective policy instrument to identify the most cost-effective mitigation pathway to achieve the climate targets outlined in the Paris Agreement [11,12]. In scenario S3, we implement a global uniform carbon tax to achieve a 25% reduction in net total GHG emissions in China and MTP by 2030. This aligns with the 2°C climate stabilisation target [77] outlined in the Paris Agreement [3], which corresponds to the RCP2.6 and requires global GHG emissions to peak by 2025 and decline by 25% by 2030. We select the 2°C target instead of the 1.5°C target because Matthews and Wynes [78] have demonstrated that while current global efforts are insufficient to limit warming to 1.5°C, they provide a greater than 95% chance of staying below 2°C. However, the carbon tax entails economic trade-offs, as it raises production costs, reduces output and welfare, and imposes greater distortions with deeper emission reductions [14]. Moreover, the carbon tax is often criticised as regressive, as it places disproportionate burdens on low-income and vulnerable populations, thereby conflicting with the principle of equity and common but differentiated responsibilities [13]. Recycling carbon tax revenues to households via lump-sum transfers can help ease these equity concerns and mitigate the adverse impacts on food security caused by the carbon tax [14]. Therefore, we combine a global uniform carbon tax with revenue recycling via lump-sum transfers in S3, which could help advance social equity and support a just climate transition.

S4 - Combined scenarios: S1+S2+S3. In the combined scenario S4, all measures are combined to examine their potential synergies or trade-offs in the food-land-climate nexus. This scenario integrates China's dietary shift (S1) and afforestation policy (S2) with the implementation of a global uniform carbon tax with carbon tax revenue recycling (S3).

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 agriculture, forestry, and other land use (AFOLU) (6 crop types, 3 livestock categories, and 1 forestry sector) and non-agricultural (compound feed, food processing by-products, processed food, fertilisers, 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

870 W.L., X.Z., H.P.W., Y.H., and L.M.P. designed the research. W.L., X.Z., and L.M.P. developed the
871 model. W.L. ran the model and performed the analysis. W.L. collected and analysed data. W.L.
872 wrote the paper with contributions from X.Z., H.P.W., O.O., Y.H., L.M.P., L.G., and K.B.. All
873 authors contributed to the interpretation of the results and commented on the manuscript.

Competing interests

875 The authors declare no competing interests.

Additional information

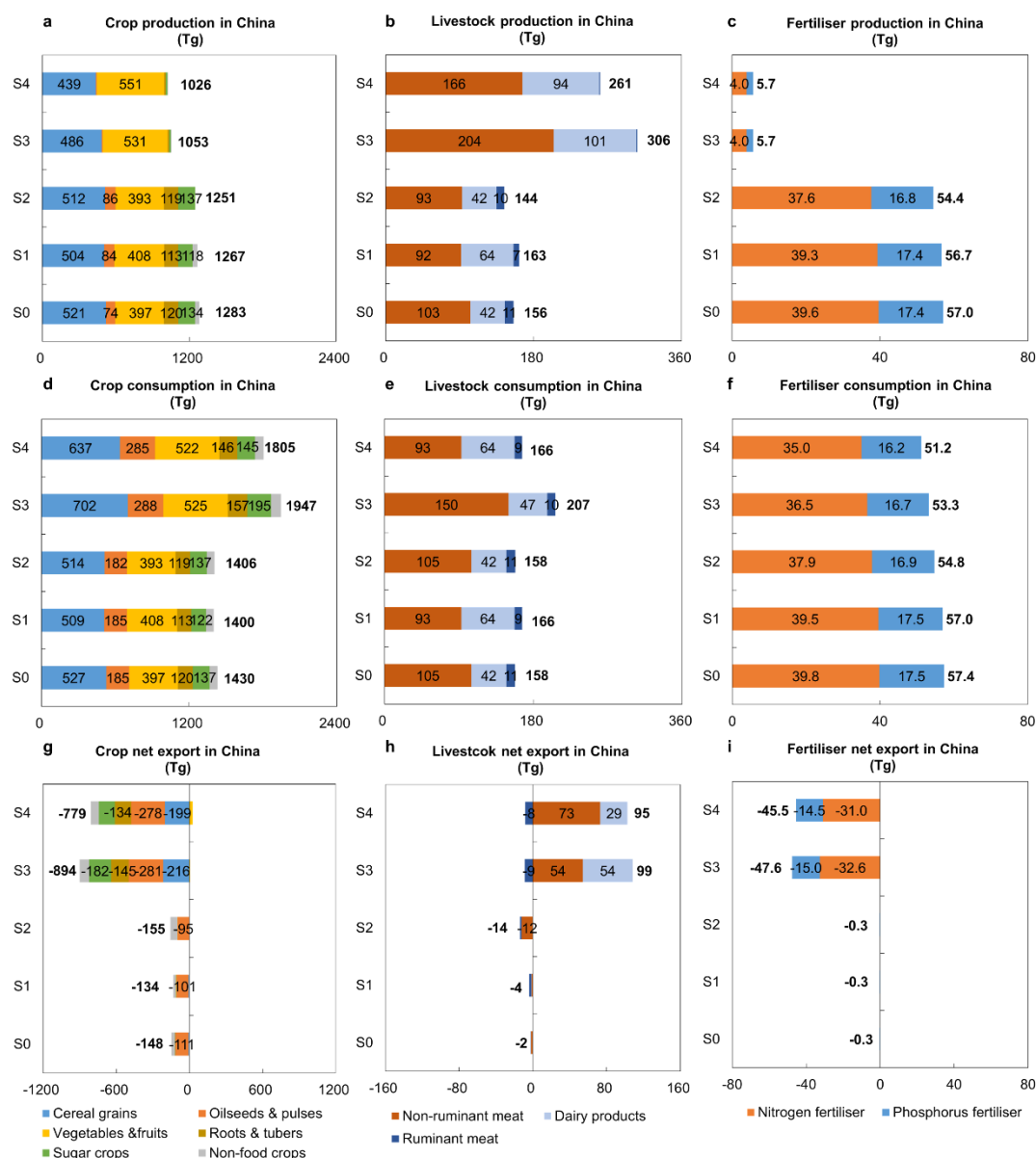877 Details about the data, methods, and framework are presented in the Supplementary Information
878 (SI).

a Crop production in China (Tg)

S4	439	551			**1026**
S3	486	531			**1053**
S2	512	86	393	119 137	**1251**
S1	504	84	408	113 118	**1267**
S0	521	74	397	120 134	**1283**

b Livestock production in China (Tg)

S4	166	94	**261**
S3	204	101	**306**
S2	93	42 10	**144**
S1	92	64 7	**163**
S0	103	42 11	**156**

c Fertiliser production in China (Tg)

S4	4.0		**5.7**
S3	4.0		**5.7**
S2	37.6	16.8	**54.4**
S1	39.3	17.4	**56.7**
S0	39.6	17.4	**57.0**

d Crop consumption in China (Tg)

S4	637	285	522	146 145		**1805**
S3	702	288	525	157 195		**1947**
S2	514	182	393	119 137		**1406**
S1	509	185	408	113 122		**1400**
S0	527	185	397	120 137		**1430**

e Livestock consumption in China (Tg)

S4	93	64 9	**166**
S3	150	47 10	**207**
S2	105	42 11	**158**
S1	93	64 9	**166**
S0	105	42 11	**158**

f Fertiliser consumption in China (Tg)

S4	35.0	16.2	**51.2**
S3	36.5	16.7	**53.3**
S2	37.9	16.9	**54.8**
S1	39.5	17.5	**57.0**
S0	39.8	17.5	**57.4**

g Crop net export in China (Tg)

S4	**-779**	-134 -278 -199
S3	**-894**	-182 -145 -281 -216
S2	**-155**	-95
S1	**-134**	-101
S0	**-148**	-111

h Livestcok net export in China (Tg)

S4	-8 73 29	**95**
S3	-9 54 54	**99**
S2	-14 -12	
S1	-4	
S0	-2	

i Fertiliser net export in China (Tg)

S4	**-45.5** -14.5 -31.0
S3	**-47.6** -15.0 -32.6
S2	**-0.3**
S1	**-0.3**
S0	**-0.3**

Legend: Cereal grains, Oilseeds & pulses, Vegetables & fruits, Roots & tubers, Sugar crops, Non-food crops; Non-ruminant meat, Dairy products, Ruminant meat; Nitrogen fertiliser, Phosphorus fertiliser

Fig. 1 | Impacts of mitigation measures on domestic production, consumption, 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 consumption (Tg) in scenarios. Total (**g**) crop, (**h**) livestock, and (**i**) fertiliser net export (Tg) in scenarios. Total crop production excludes discarded food waste and food processing by-products (see **Supplementary Table 2** 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) (see **Supplementary Fig.5** for detailed data). Definitions of scenarios (S1 - Food scenario: A dietary shift in China; S2 - Land scenario: An afforestation policy in China; S3 - Climate scenario: A global uniform carbon tax with carbon tax revenue recycling; S4 - Combined scenarios: S1+S2+S3) are described in **Supplementary Table 1**.
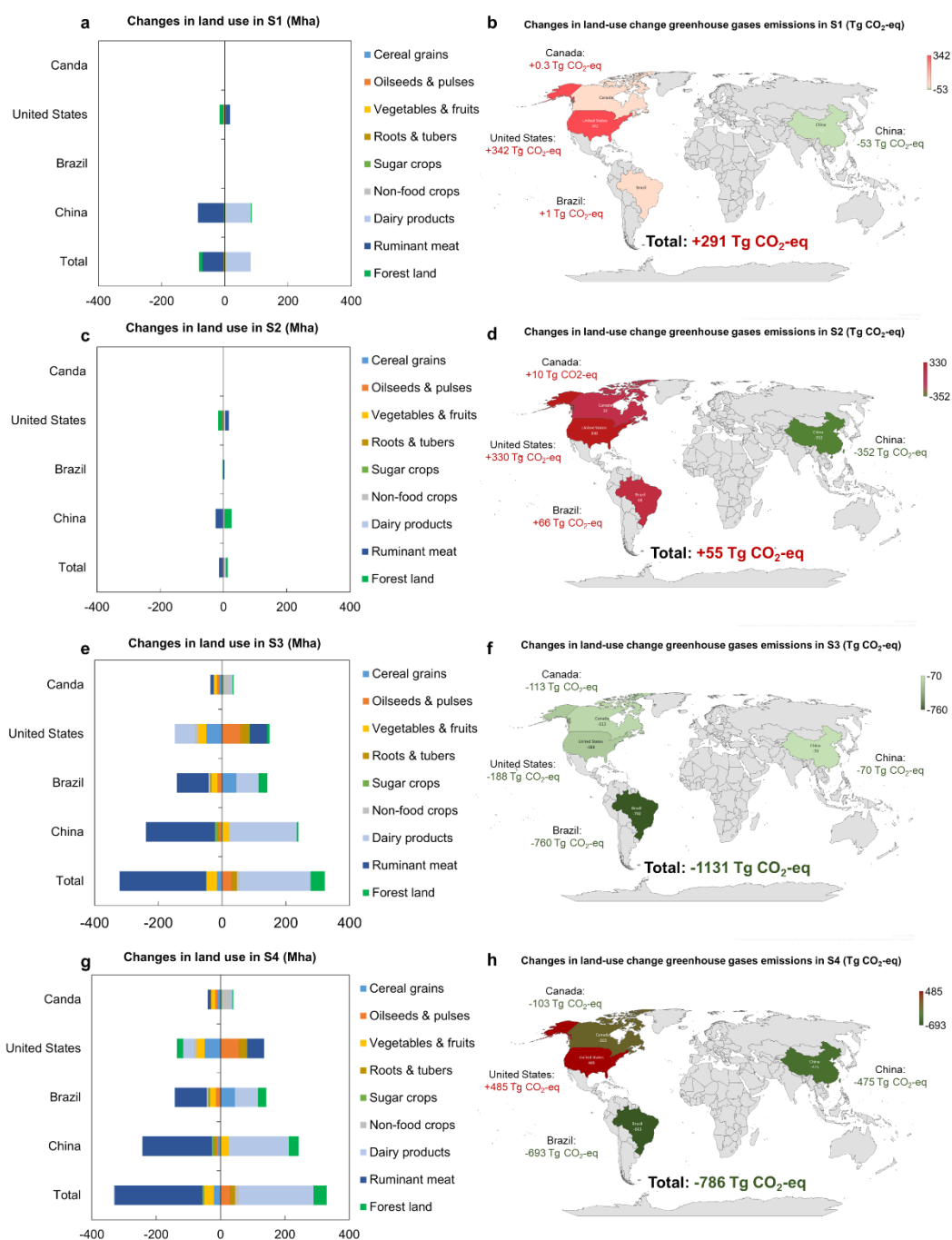
891

Fig. 2 | Impacts of mitigation measures on land-use changes and land-use change greenhouse gases emissions in China and its main food and feed trading partners (MTP, including Brazil, the United States, and Canada). Changes in land use (Mha) in China and MTP in scenarios (**a**) S1, (**c**) S2, (**e**) S3, and (**g**) S4 with respect to the baseline (S0). Changes in land-use change greenhouse gases emissions (Tg CO_2-eq) in China and MTP in scenarios (**b**) S1, (**d**) S2, (**f**) S3, and (**h**) S4 with respect to the baseline (S0). Definitions of scenarios (S1 - Food scenario: A dietary shift in China; S2 - Land scenario: An afforestation policy in China; S3 - Climate scenario: A global uniform carbon tax with carbon tax revenue recycling; S4 - Combined scenarios: S1+S2+S3) are described in **Supplementary Table 1**.
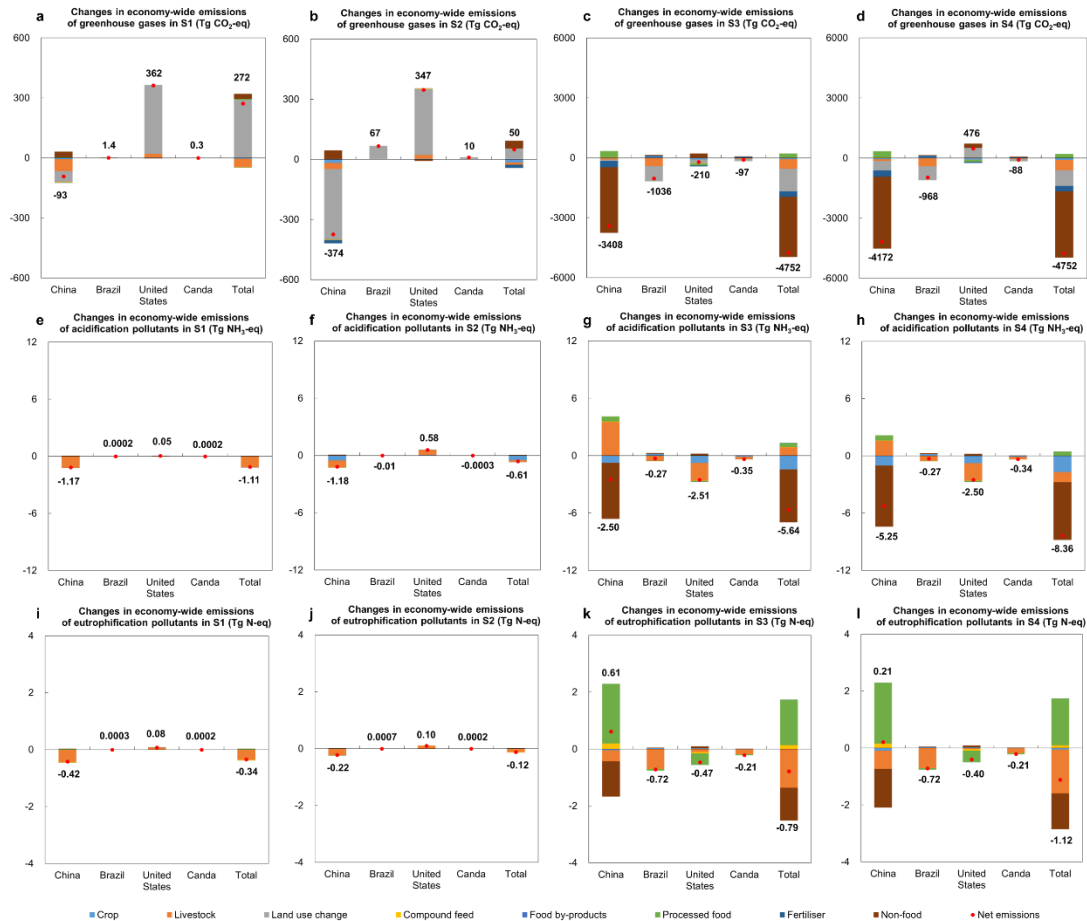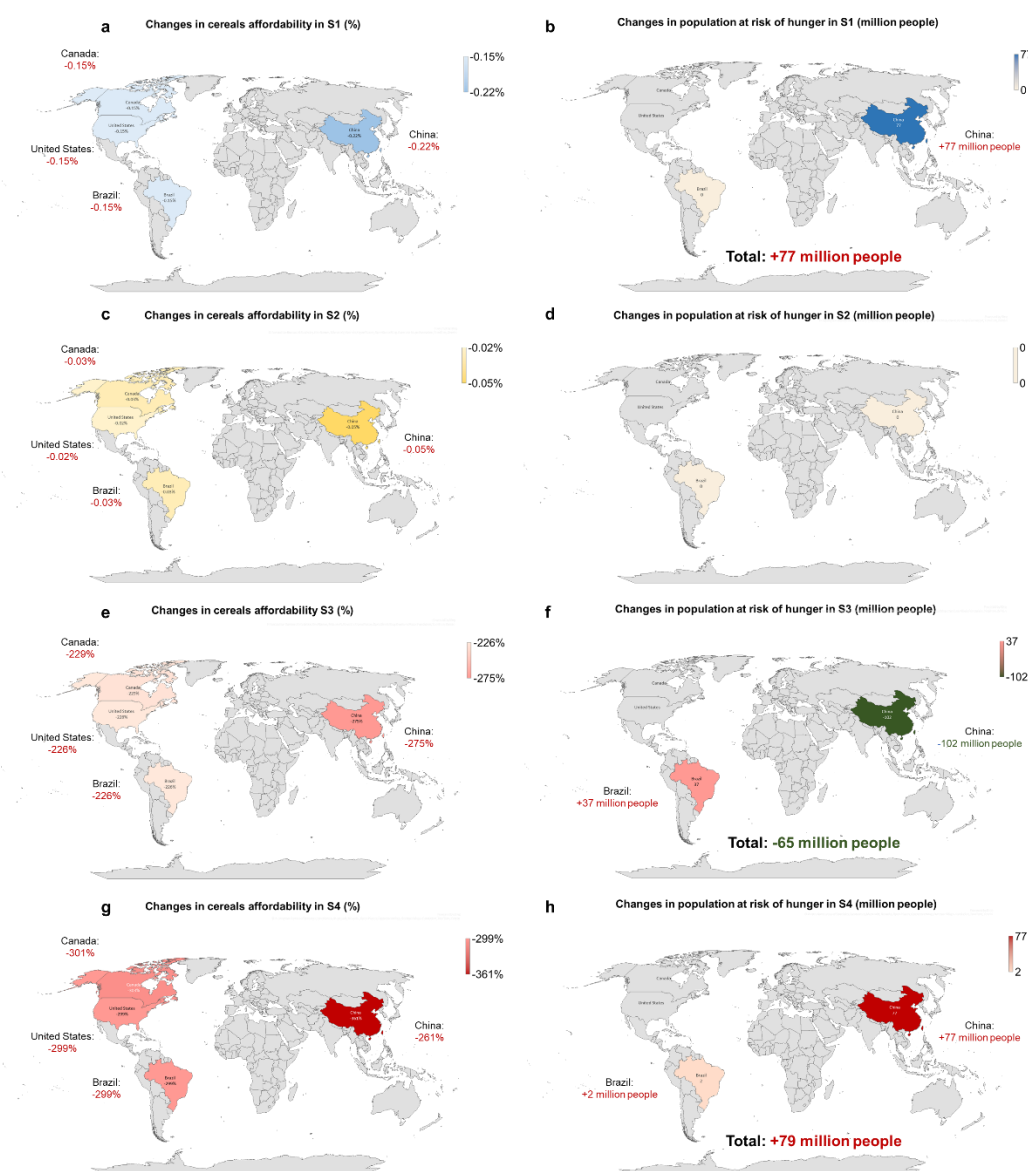
Fig. 3 | Impacts of mitigation measures on economy-wide emissions in China and its main food and feed trading partners (MTP, including Brazil, the United States, and Canada). Changes in economy-wide emissions of greenhouse gases (GHGs) (Tg CO_2-eq) in China and MTP in scenarios (**a**) S1, (**b**) S2, (**c**) S3, and (**d**) S4 with respect to the baseline (S0). Changes in economy-wide emissions of acidification pollutants (Tg NH_3-eq) in China and MTP in scenarios (**e**) S1, (**f**) S2, (**g**) S3, and (**h**) S4 with respect to the baseline (S0). Changes in economy-wide emissions of eutrophication pollutants (Tg N-eq) in China and MTP in scenarios (**i**) S1, (**j**) S2, (**k**) S3, and (**l**) S4 with respect to the baseline (S0). Economy-wide emissions refer to total emissions of GHGs, acidification pollutants, and eutrophication pollutants from all sectors in the entire economies of China and MTP. Definitions of scenarios (S1 - Food scenario: A dietary shift in China; S2 - Land scenario: An afforestation policy in China; S3 - Climate scenario: A global uniform carbon tax with carbon tax revenue recycling; S4 - Combined scenarios: S1+S2+S3) are described in **Supplementary Table 1**.

Fig. 4 | Impacts of mitigation measures on food security in China and its main food and feed trading partners (MTP, including Brazil, the United States, and Canada). Changes in cereals affordability (%) in China and MTP in scenarios (**a**) S1, (**c**) S2, (**e**) S3, and (**g**) S4 with respect to the baseline (S0). Changes in population at risk of hunger (million people; S0 = 140.7 million people for China; S0 = 5.3 million people for Brazil) in China and MTP in scenarios (**b**) S1, (**d**) S2, (**f**) S3, and (**h**) S4 with respect to the baseline (S0). 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. Definitions of scenarios (S1 - Food scenario: A dietary shift in China; S2 - Land scenario: An afforestation policy in China; S3 - Climate scenario: A global uniform carbon tax with carbon tax revenue recycling; S4 - Combined scenarios: S1+S2+S3) are described in **Supplementary Table 1**.
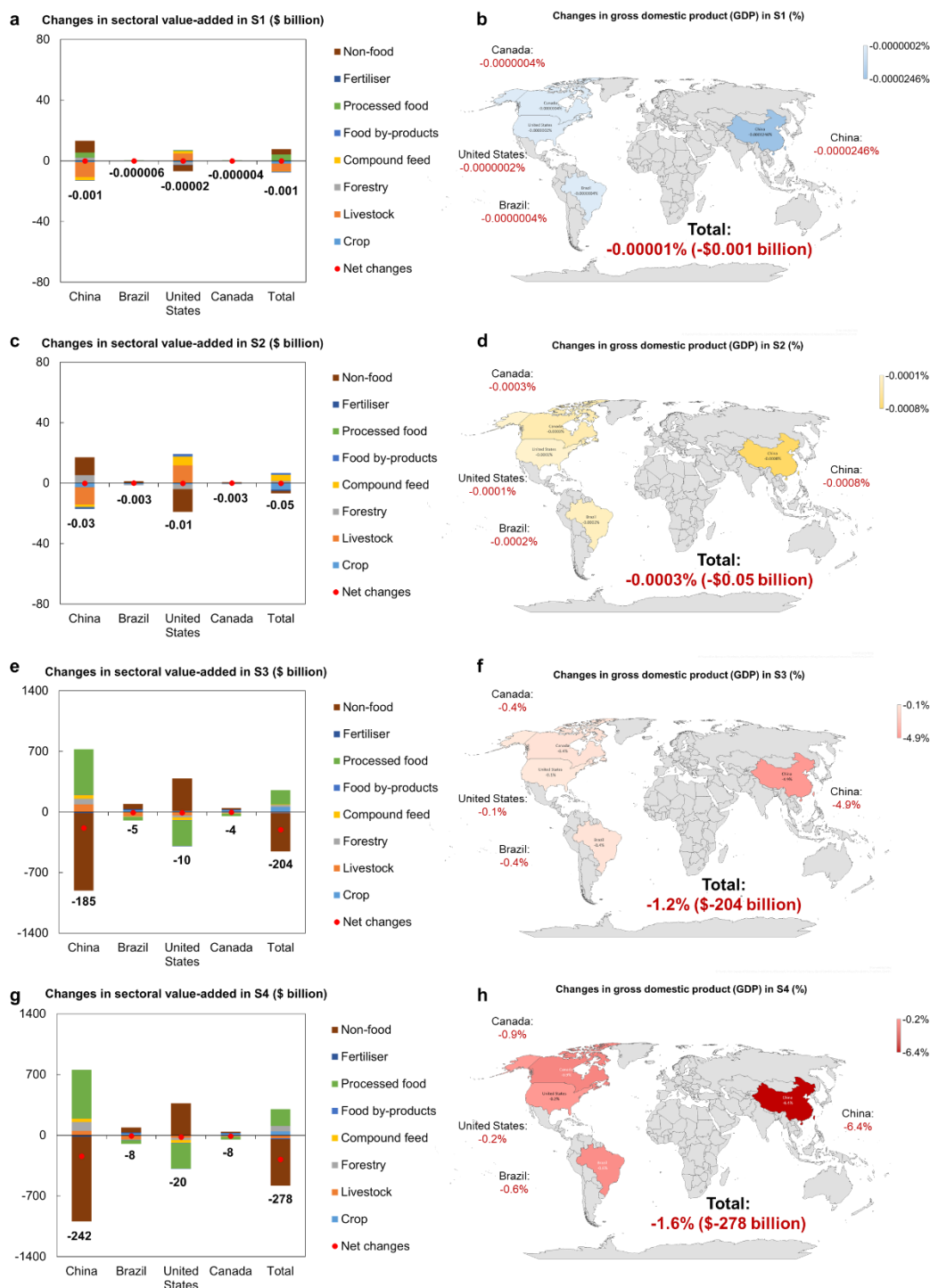
Fig. 5 | Impacts of mitigation measures on economic losses in China and its main food and feed trading partners (MTP, including Brazil, the United States, and Canada). Changes in sectoral vaule-added ($ billion) in China and MTP in scenarios (**a**) S1, (**c**) S2, (**e**) S3, and (**g**) S4 with respect to the baseline (S0). Changes in gross domestic product (GDP, %) in China and MTP in scenarios (**b**) S1, (**d**) S2, (**f**) S3, and (**h**) S4 with respect to the baseline (S0). GDP is calculated as the sum of sectoral vaule-added in the entire economy. Definitions of scenarios (S1 - Food scenario: A dietary shift in China; S2 - Land scenario: An afforestation policy in China; S3 - Climate scenario: A global uniform carbon tax with carbon tax revenue recycling; S4 - Combined scenarios: S1+S2+S3) are described in **Supplementary Table 1**.