PNAS RESEARCH ARTICLE | SUSTAINABILITY SCIENCE

Investing in mini-livestock production for food security and carbon neutrality in China

Zhaohai Bai[a,1], Xiaofei Wu[a,1], Luis Lassaletta[b], Alexander Haverkamp[c], Wei Li[d], Zengwei Yuan[e], Eduardo Aguilera[b,f], Aimable Uwizeye[g], Alberto Sanz-Cobena[b], Nannan Zhang[a], Xiangwen Fan[a], Feng Zhu[a], Marcel Dicke[c], Xuan Wang[a,2], and Lin Ma[a,2]

Edited by B. Turner, Arizona State University, Tempe, AZ; received March 23, 2023; accepted August 18, 2023

Future food farming technology faces challenges that must integrate the core goal of keeping the global temperature increase within 1.5 °C without reducing food security and nutrition. Here, we show that boosting the production of insects and earthworms based on food waste and livestock manure to provide food and feed in China will greatly contribute to meeting the country's food security and carbon neutrality pledges. By substituting domestic products with mini-livestock (defined as earthworms and insects produced for food or feed) protein and utilizing the recovered land for bioenergy production plus carbon capture and storage, China's agricultural sector could become carbon-neutral and reduce feed protein imports to near zero. This structural change may lead to reducing greenhouse gas emissions by 2,350 Tg CO_2eq per year globally when both domestic and imported products are substituted. Overall, the success of mini-livestock protein production in achieving carbon neutrality and food security for China and its major trading partners depends on how the substitution strategies will be implemented and how the recovered agricultural land will be managed, e.g., free use for afforestation and bioenergy or by restricting this land to food crop use. Using China as an example, this study also demonstrates the potential of mini-livestock for decreasing the environmental burden of food production in general.

insect | greenhouse gas emission | protein | food security | mitigation

Significance

The sustainable production of sufficient proteins to feed the growing human population is a global challenge. Investing in China's mini-livestock production will contribute to the country's food security, carbon neutrality pledges, and reduction in resource use. Substituting domestic products with mini-livestock protein and utilizing the recovered land for bioenergy production with carbon capture and storage allows China's agricultural sector to become carbon neutral and reduce feed protein imports to almost zero. This structural change toward the full potential of using insect protein may lead to a reduction in GHG emissions by 2,350 Tg CO_2eq at the global level, under a strategy that substitutes both domestic and imported products with protein from mini-livestock.

Agriculture, including crops and livestock production, faces the huge challenge of producing enough food while reducing its contribution to climate change (1). The sector has dramatic impacts on several planetary boundaries, including cropland use (2), greenhouse gas (GHG) emissions (3–5), and nitrogen (N) use (6, 7).

Livestock production provides quality food but requires substantial amounts of land and other resources such as synthetic N fertilizer to produce feed (8, 9); additionally, the associated GHG emissions and reactive N losses are high (10, 11). In many cases, a significant proportion of feed production occurs abroad, and large livestock producers such as Europe and China, depend on a significant net importation of feed (12, 13). This external dependency leads to food security risks, which is becoming an increasing global concern (14).

Reducing animal-sourced food consumption and using alternative feed sources, such as swill, have been recognized as efficient strategies for reducing both fertilizer demand and environmental impacts (15, 16). The benefits include a reduction both of the land needed to produce livestock feed and reduction of soybean and cereal imports, increasing the N-use efficiency of the agrifood systems (10). The recovered land could be used for afforestation, bioenergy production, and carbon capture and storage (BECCS), thus reducing GHG emissions (11).

Future food farming technology, such as the mini-livestock production, including insects and earthworms, is an efficient way to produce protein, and reduce the environmental impacts of agricultural output while boosting regional self-sufficiency without reducing food quality (17, 18). Mini-livestock protein for human consumption can be produced by food waste, decreasing the demand for conventional animal-sourced products (19). Mini-livestock can also be raised on manure and used as feed for conventional livestock such as swine and poultry, diminishing the external dependency on imported feed (20–22).

In this paper, we systematically quantify the effects of mini-livestock production in China on improving food self-sufficiency, while reducing the use of land and synthetic N fertilizer use and GHG emissions (Fig. 1). China's food supply increasingly relies on imports (total food imports rose to a record level of 160 Tg in 2021) (23) and the country recently highlighted that food security and carbon neutrality are its leading priorities (24, 25). Nevertheless, the impacts of adopting mini-livestock production technology in

Author contributions: Z.B., X. Wang, and L.M. designed research; X. Wu performed research; X. Wu analyzed data; and Z.B., X. Wu, L.L., A.H., W.L., Z.Y., E.A., A.U., A.S.-C., N.Z., X.F., F.Z., M.D., X. Wang, and L.M. wrote the paper.

The authors declare no competing interest.

This article is a PNAS Direct Submission.

[1]Z.B. and X. Wu contributed equally to this work.

[2]To whom correspondence may be addressed. Email: wangxuan@sjziam.ac.cn or malin1979@sjziam.ac.cn.

This article contains supporting information online at https://www.pnas.org/lookup/suppl/doi:10.1073/pnas.2304826120/-/DCSupplemental.

Published October 16, 2023.

China on reducing GHG emissions for the country's major agricultural product trading partners have not been identified (Fig. 1). The present study aims to identify the spillover effects of agricultural trade on the mitigation of GHG emissions by reducing the import of agricultural products through adopting mini-livestock production (26).

Results and Discussion

Mini-Livestock Protein Production Capacity. In China, an abundance of food waste and livestock manure are available that could be used as feed for mini-livestock (27, 28), indicative of the ample opportunities to produce mini-livestock as a protein source. Here, we focus on food waste as feed for house crickets, mealworms, and black soldier flies, to convert products unsuitable for human consumption into mini-livestock protein for human consumption after proper sanitation and drying treatments. The total potential production of mini-livestock protein for food is 4 to 7 Tg (Fig. 2*B*), based on an input of 102 ± 14 Tg fresh food waste for their production in 2018 (*SI Appendix*, Tables S1 and S2). Approximately 17 to 24 Tg of mini-livestock protein for feed can be produced (Fig. 2*C*), via fully collecting and feeding 1,661 ± 174 Tg solid livestock manure to black soldier flies, houseflies and earthworms, as these mini-livestock species are well adapted to use this substrate as feed (*SI Appendix*, Tables S1 and S2). The relatively lower efficiency of producing mini-livestock protein on manure compared to the current commercial insect production (*SI Appendix*, Table S2) is mainly due to the poor feed quality. After proper sanitation and defatting treatments, these mini-livestock proteins can be used as feed for livestock, especially poultry and swine (20).

Reaching the full potential of mini-livestock protein production is possible in China because the mini-livestock production industry is well developed and expanding (Fig. 2). Currently, there are 9,500 earthworm, 3,500 mealworm, 480 black soldier fly, 320 housefly, and 120 cricket production plants in operation across different climatic and agricultural ecological zones (Fig. 2 *D*–*F*). The current reinforced policies on municipal solid waste separation and collection (29), and mandatory solid and liquid manure separation of industrial livestock farms (30), have ensured the supply of feed for mini-livestock. In addition, these mini-livestock production plants geographically overlap with the distribution of municipal food waste collection plants and industrial livestock farms (Fig. 2 *D*–*F*), which further ensures the success of the uptake of mini-livestock protein production. However, the upscaling of mini-livestock production will need various investments and policies to support the construction of insect production plants (31), to effectively collect and redistribute food waste and solid manure (32), for sanitation and processing (33), and to package and sell the final products (33).

Mini-Livestock Protein Substitution Strategies. Different strategies of using mini-livestock protein to substitute conventional protein sources result in contrasting effects on protein self-sufficiency and environmental impacts for China and its major trading partners (Fig. 3). Here, we designate mini-livestock protein as food to substitute five different animal-sourced foods and mini-livestock protein as feed to substitute three different plant-sourced feeds (*Methods*). This substitution is feasible due to their similar roles in the diets of humans and livestock and, in part, to the larger environmental footprints and greater dependence on the importation of traditional livestock and feed products (*SI Appendix*, Fig. S1 and ref. 34).

The impacts of different substitution priorities, namely substitution of imported products only (SIP), substitution of domestic products only (SDP), and substitution of domestic and imported products in hybrid form (SDIH), have been quantified to identify the best-performing strategies for China and its major trading partners due to recent geopolitical concerns. Substitution of animal-sourced food or feed was carried out proportionally for

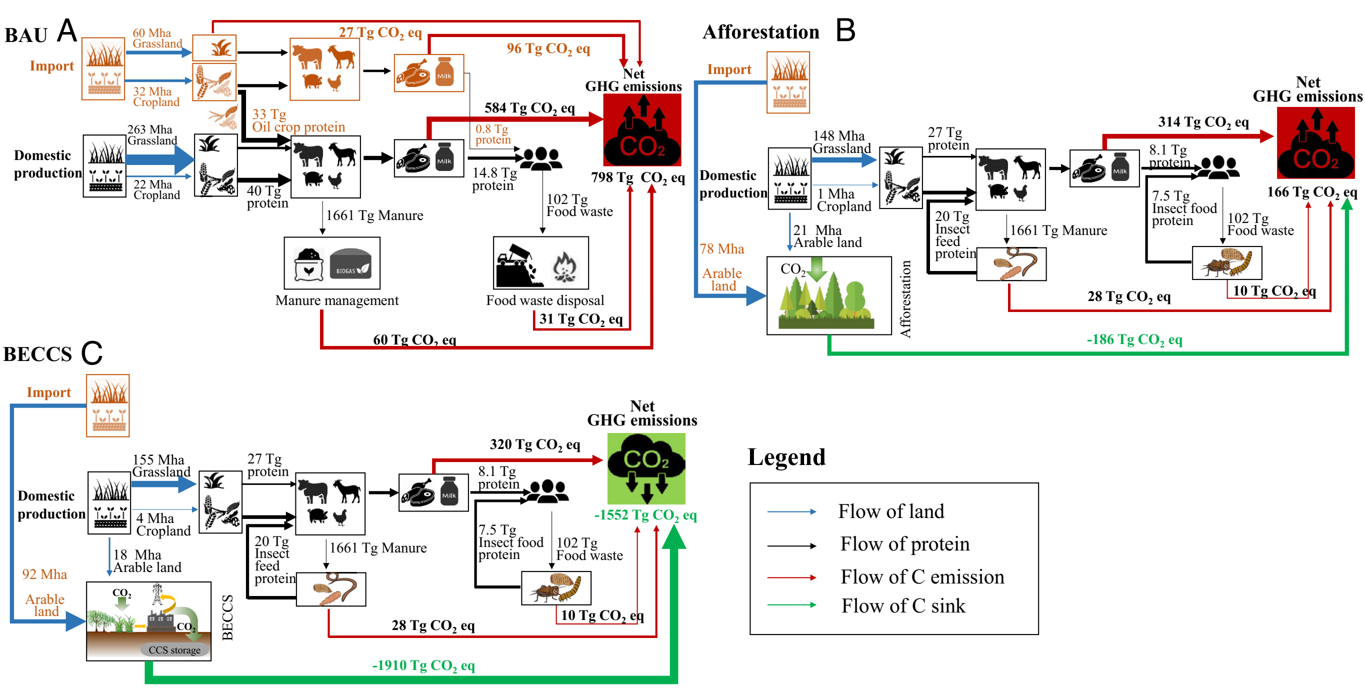

Fig. 1. The role of mini-livestock production in improving food security and mitigating GHG emissions from the agricultural sector in China and its major trading partners under a business-as-usual scenario (BAU) (*A*), a recovered land used for afforestation scenario (*B*), and a BECCS, (*C*) at the highest GHG mitigation rate with no land use restriction.

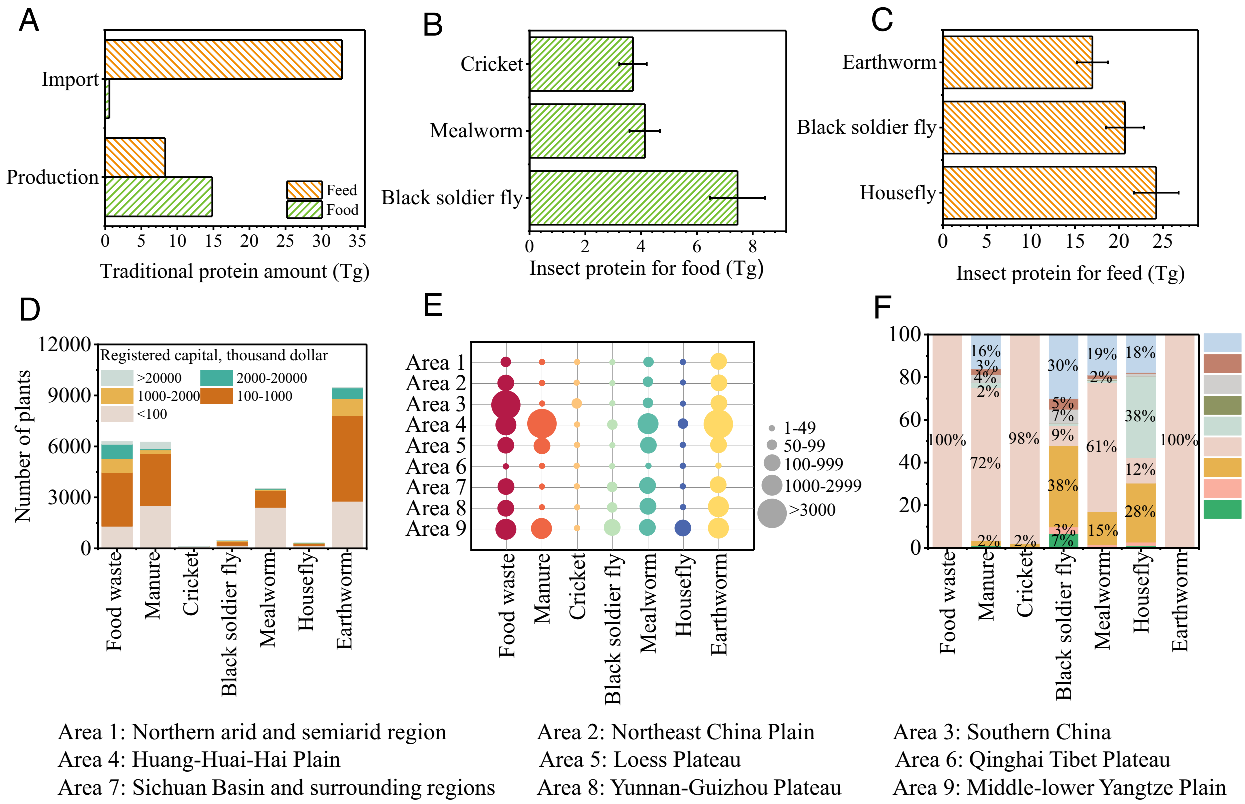

Fig. 2. Food and feed protein production and imports (*A*), mini-livestock protein production capacity as food by black soldier fly larvae or mealworms or crickets (*B*), and mini-livestock protein production capacity as feed by house fly larvae, black soldier fly larvae or earthworms (*C*) in China in 2018. Number and the size of registered food waste collection and mini-livestock production plants (*D*), distribution of registered food waste collection plants and mini-livestock production plants in different agroecological area (*E*), and the contribution of different agroecological areas to total registered capital of different plants in China (*F*). Note: Error bars represent the SD for protein produced by the different insect species under various rearing conditions.

the selected conventional animal-sourced food or feed in all strategies to avoid significant disturbances of the diet structure. The imported products with larger GHG emission rates per unit of protein production were set with a high substitution priority to maximize GHG reduction in the context of mitigating climate change (*SI Appendix*, Figs. S2 and S3).

Recovered agricultural land via a decrease in the demand for agricultural products within and outside of China is designated for afforestation or BECCS to reduce carbon emissions (Fig. 1). These two different land use scenarios have been developed separately, due to their significant differences in the reduction of GHG emissions and N fertilizer use. The highest mini-livestock protein production capacity by different types of insects or earthworms has been used for each scenario and substitution strategy (See *SI Appendix*, Figs. S4–S7 for more information on other species, and *SI Appendix*, Table S3 for the selection of best-performing mini-livestock of each situation).

Effects on Food Security and Environmental Performance.

Improvement of protein self-sufficiency. The SIP strategy could reduce China's demand for protein by 25 Tg which accounted for 75% of total imports in 2018 (Fig. 3 *A* and *B*), although it had a negligible reduction in the conventional animal-sourced food protein consumption rate (*SI Appendix*, Fig. S8). This was because the direct import of animal-sourced food is relatively low in China (Fig. 2*A*), except for dairy products (35). Targeting only the SDP strategy yields a 29% higher replacement of conventional livestock protein than the SIP strategy (Fig. 3*A*). Together with directly substituting imported feed, SDP strategy may reduce feed protein demand by 25 Tg, which equaled to 78% of total feed protein import in China in 2018 (Fig. 2*A*). Deploying the SDP strategy requires replacing 48% of China's domestically produced conventional animal-sourced food with mini-livestock protein (*SI Appendix*, Fig. S8). The strategy of SDIH, which aims to replace products with higher GHG emission intensity beyond geopolitical priorities or concerns, leads to twice the replacement of conventionally sourced protein compared with the SIP strategy (Fig. 3 *A* and *B*).

Relying on feed protein imports could be substantially reduced via different levels of mini-livestock protein production without reducing the nutritional protein intake level per capita, especially under the SDIH strategy. The total replacement of conventionally sourced animal protein in the SDIH strategy was 25% higher than China's total feed protein import in 2018 (Figs. 2*A* and 3 *A* and *B*). This indicates that the SDIH strategy not only makes China's protein feed consumption fully self-sufficient but also that it replaces some domestically produced feed crops, releasing more agricultural land for nature conservation and/or bioenergy production.

Land use reduction. Land use reduction ranges from 86 to 214 Mha for all substitution strategies at the global level, considering the different requirements of grassland and cropland areas for the different substitution products, and crop indexes of different countries (Fig. 3 *C* and *D*). The highest land use reduction is under the SDIH strategy, followed by SDP and SIP. Approximately 92% of the saved land area occurred China in the SDP strategy, and 64% for SDIH and 0% for SIP, respectively (Fig. 3 *C* and *D*). Most of the agricultural land saving comes from grassland reduction, due to the lower consumption of beef and mutton (*SI Appendix*, Fig. S9). The freed-up agricultural land is used for afforestation or

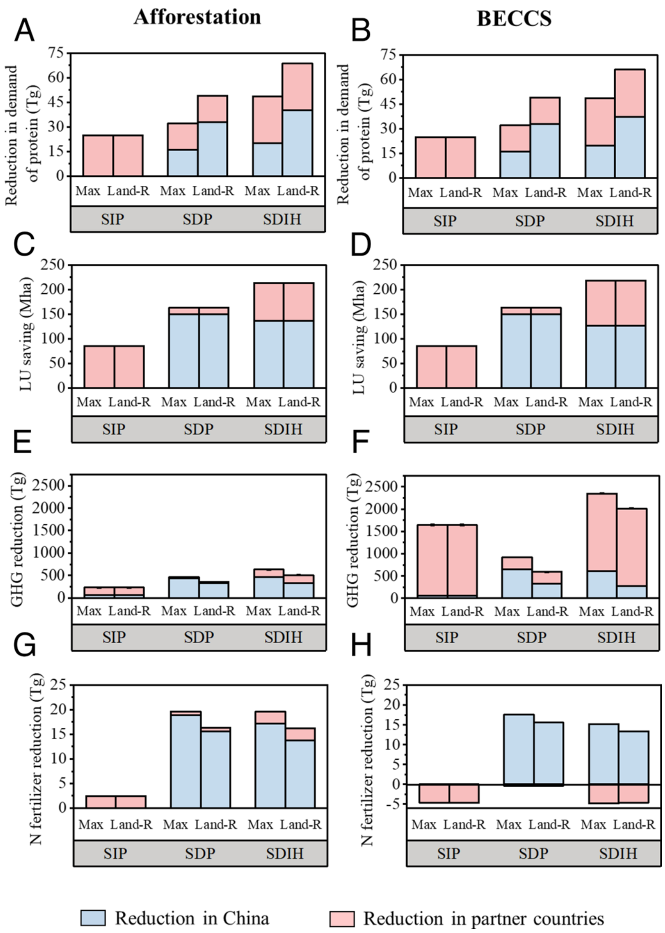

Fig. 3. The estimated reduction in protein demand (*A* and *B*), land use change (*C* and *D*), reduction in GHG emissions (*E* and *F*), and nitrogen (N) fertilizer use reduction (*G* and *H*) when adopting the strategies of substitution of imported products only (SIP), substitution of domestic products only (SDP), and substitution of domestic and imported products hybrid form (SDIH) in the Afforestation (*Left*) and BECCS scenario (*Right*) at maximum (Max) mini-livestock protein substitution rate and land use restriction scenario (Land-R). All results show the contribution by China (blue) and its major trading partners (red) in 2018. Note: Land-R represents a scenario that restricts recovered agricultural land for afforestation and BECCS. For more details about the contribution sources see *SI Appendix*, Figs. S4–S7 and S9. All analyses are based on the best performance of one specific mini-livestock species, i.e., the species with the most significant reduction in GHG emissions (*SI Appendix*, Fig. S4). For information on the other insect or earthworm species, see *SI Appendix*, Figs. S4–S7.

BECCS to remove CO_2 from the atmosphere, with the exception of grassland in China, which experience a higher land degradation rate and are mainly located in the arid regions with severe water limitations (36).

Mitigation of GHG emissions. In the Afforestation scenario, the total reduction in GHG emissions ranges from 231 to 632 Tg CO_2 equivalents (CO_2eq) y^{-1} at the global scale (Fig. 3*E*), including an increase in GHG emissions from mini-livestock protein production itself, a reduction in GHG emissions through improved food waste and manure management in China, a reduction of livestock and crop production, and carbon sequestration by afforestation on the saved agricultural land within and outside China's territory. Up to 2,350 Tg CO_2eq y^{-1} could be reduced when deploying the SDIH strategy in the BECCS scenario, which is 3.7 times that in the Afforestation scenario (Fig. 3*F*). This is in line with most studies showing that BECCS has a higher CDR potential than afforestation (Xu et al., 2022).

In the Afforestation scenario, China benefits most from the SDIH strategy, in which domestic GHG emissions can be reduced by 459 Tg CO_2eq y^{-1}. In the BECCS scenario, China can reduce its GHG emissions by 657 Tg CO_2eq y^{-1} under the SDP strategy (Fig. 3 *E* and *F*), which almost equals the total emissions from China's agricultural sector, i.e., 670 Tg CO_2eq in 2018 (34). This indicates that China's agricultural sector can become carbon neutral by fully and smartly empowering the production of mini-livestock protein (25, 37).

Although contributing less to the reduction in GHG emissions than BECCS, afforestation provides additional ecosystem values such as water provisioning, soil erosion control, and biodiversity protection in particular (38). Hence, the afforestation scenario would assist China in achieving ambitious biodiversity protection targets in addition to carbon neutrality (39). However, the recent increase in the frequency and intensity of forest fires resulting from extreme climate conditions threatens the effectiveness of afforestation (40, 41). For example, the 2019 to 2020 Australian forest fires emitted around 806 ± 70 Tg CO_2eq into the atmosphere (42), which could wipe out the entire GHG emission reduction efforts in the Afforestation scenario (Fig. 3 *E* and *F*).

Compared to afforestation, BECCS is more climate-risk resilient in reducing GHG emissions, although it is more expensive (43, 44, Xu et al., 2022). In addition, BECCS may also require more synthetic N fertilizer and blue water input to support the high productivity of primary biomass production (45), which may promote exceeding of the safe and just operating space of these two response variables of the Earth's system (46).

Reduction of N fertilizer use. Up to 20 Tg of N fertilizer could be reduced by adopting the SDIH strategy in afforestation (Fig. 3*G*). More than 88% of this reduction is due to reducing animal-sourced food and the related feed-crop reduction in China (Fig. 3 *G* and *H*) by using the most recently updated country-specific N footprint data for different livestock products (10). The total N fertilizer reduction will reach up to 50% of its 2018 usage (34). N fertilizer reduction is lower in the BECCS scenario, especially for the SIP strategy (Fig. 3 *G* and *H*), because it requires a large amount of additional fertilizer input (47).

Other benefits. Mini-livestock is more nutritious than traditional animal-sourced food (48). A high level of mini-livestock protein may lead to a healthy nutritious complement to the human diet in China, especially in terms of micronutrients such as zinc (49). Approximately, 150 to 200 million people in China suffer from zinc deficiency and related health problems (50). There is also considerable production of fat during the processing of insect larvae into protein-rich powder products; this fat can be used to produce biodiesel for energy (51, 52), which is paramount in the context of a worldwide energy shortage (53). The maximum biodiesel production is 20 Tg in the SDIH strategy (*SI Appendix*, Fig. S10), which also reduces GHG emissions when used to replace fossil fuels (54). Mini-livestock biotransformation of organic waste produces a by-product consisting of undigested feed and feces, commonly referred to as "frass," which improves crop yield and quality and, thus, may replace traditional fertilizers (55).

Spillover effects for the major trading partners. There are contrasting spillover effects in terms of GHG emission mitigation for the major trading partners under different substitution priorities and land use scenarios (Fig. 4). Deploying SDP would reduce GHG emissions by 21 Tg and 250 Tg CO_2eq y^{-1} for the major trading partners in the Afforestation and BECCS scenarios, respectively. Most reductions would occur in Brazil due to a decline in soybean demand (Fig. 4 *B* and *E*). The SDIH strategy leads to the greatest spillover effects of GHG emission reduction in trading partners in both scenarios of 144 Tg CO_2eq

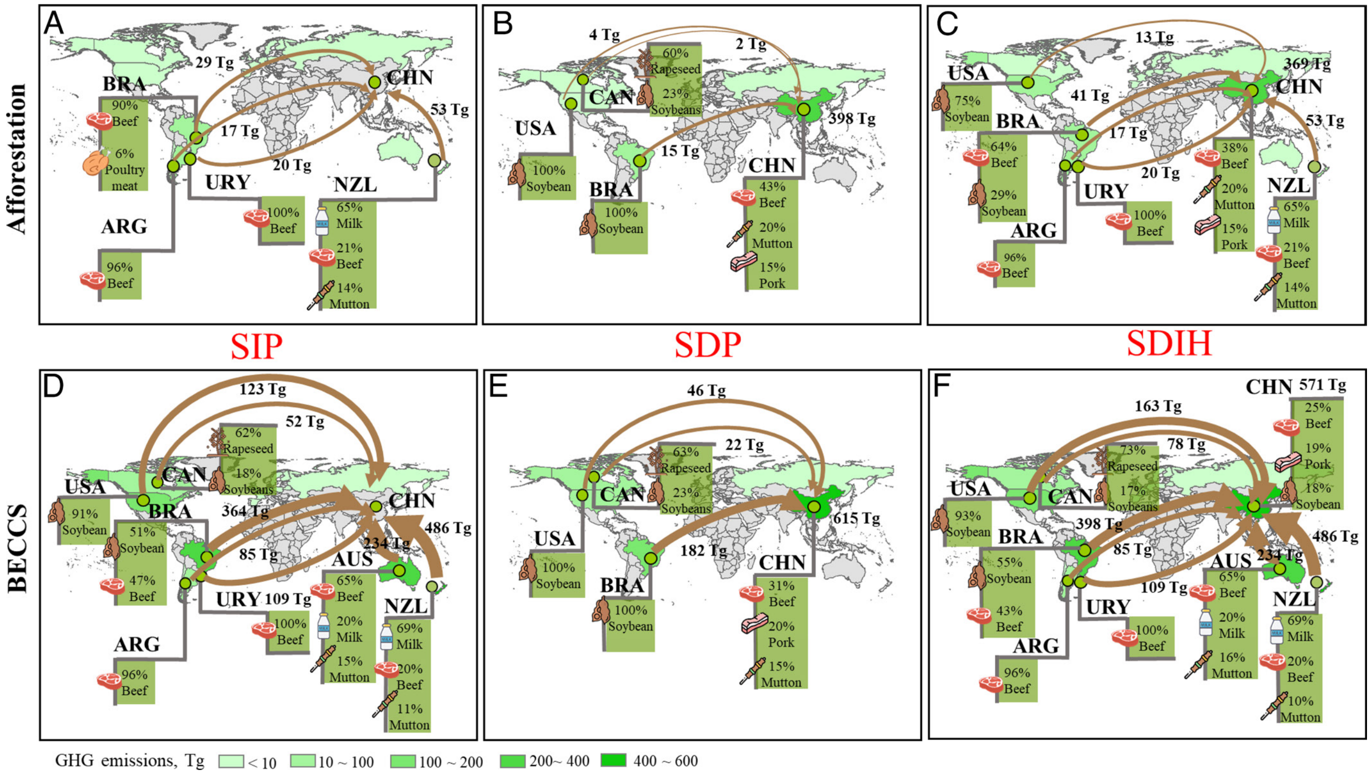

Fig. 4. The spillover effects of strategies of SIP only (*Left*), SDP only (*Middle*), and SDIH (*Right*) on reduction in GHG emissions in the main trading partner countries under the Afforestation (*A*–*C*) and BECCS (*D*–*F*) scenarios. The maps show the absolute reduction of GHG emissions from different countries; the green shaded boxes show the dominant contributors to reduction in GHG emissions. Note: Here, we only show the data from countries that contribute >95% of the total importation rate. The Tg values indicate reduction in GHG emissions. BRA, Brazil; CHN, China; URY, Uruguay; ARG, Argentina; NZL, New Zealand; CAN, Canada; AUS, Australia.

y^{-1} in the Afforestation scenario and up to 1,553 Tg CO_2eq y^{-1} in the BECCS scenario (Fig. 3 *E* and *F*).

There are more diverse spillover effects on the reduction of GHG emissions in the major trading partner countries in the SIP strategy than in the SDP strategy, due to China's diverse food and feed import (Fig. 4). For example, there will be a considerable reduction in emissions in New Zealand and Australia as a result of the reduced import of milk and beef products (Fig. 4*B*). This is due to the large grassland use footprints of ruminant production in these countries (34, 56) and, hence, the significant potential to convert these lands to produce biomass for BECCS. Information on the spillover effects of land use and N fertilizer can be found in *SI Appendix*, Figs. S11 and S12.

Barriers and Opportunities for Mini-Livestock Protein Production. Social acceptance is an obstacle to mini-livestock even with strict food quality assurance (48). This is especially true in emerging economies such as China, where the history of large amounts of animal-sourced food consumption is short (34). Therefore, we also explored additional scenarios with a 20% substitution, demonstrating that this reduced scenario can also lead to substantial environmental benefits (*SI Appendix*, Fig. S13). The replacement of conventional livestock production by mini-livestock production will reduce the amount of manure produced, thus may reduce the production capacity of insect feed protein in the next round. This will require a more sustainable design of mini-livestock production and spatial distribution in the future. Although a reduction in manure production will also reduce its availability for fertilization of cropland, this is not expected to substantially impact crop yields in China, due to over-fertilization, especially with synthetic fertilizers and because the transition will make insect frass available for crop fertilization (55, 57, 58).

Cropland protection policy. Another obstacle to adopting mini-livestock production is China's strict cropland protection policy. This policy established minimum cropland area of 120 million ha (59) and does not allow afforestation of recovered agricultural land. In addition, recent policies forbid converting cropland into the usage for production of nonfood products (60), which limits the opportunities to implement the BECCS scenario. Under these land use restrictions in China, the total reduction of GHG emissions at the global scale will decrease by 35% in the SDP and 14% in the SDIH strategy in the BECCS scenario and by 23% in the SDP strategy and 20% in the SDIH strategy in the Afforestation scenario, compared to a no land-use restriction situation (Fig. 3 *E*–*G*). All these reductions occur in China, especially for the SDIH strategy in the BECCS scenario in which the reduction of GHG emissions will decrease by 56% (338 Tg CO_2eq y^{-1}) compared to a no-restriction situation (Fig. 3 *E* and *F*). This implies a significant compromise of climate change control or carbon neutrality capabilities in China under the strict cropland use protection policy. However, the freed-up agricultural land may be used to produce food to meet additional food demands (61). The estimated increase in protein production will be 17 to 20 Tg for the SDP and SDIH strategies compared to a situation without restrictions (Fig. 3 *A* and *B*), ensuring China's long-term food security.

Similar limitations may exist for turning grassland into forest or cropland in Australia, due to concerns about grassland degradation. If there are also land use restrictions, the mitigation potential of

GHG emissions will decrease by 14% at the global level of the SDIH strategy under the BECCS scenario; however, the impact will be negligible under the Afforestation scenario (*SI Appendix*, Fig. S14). In addition, there should be policies and incentives for exporting countries to invest in afforestation and BECCS instead of looking for new export markets, especially in biodiversity hotspot areas.

Economic competitiveness. To be accepted and implemented by industry, mini-livestock protein production must be at least as economically competitive as conventional food and feed proteins. Although feed costs are lower compared to livestock production, mini-livestock protein production requires large investments in facilities to provide a high and stable temperature to produce and grow the insect larvae, as well as for sanitation and drying treatments, such as converting insect larvae into protein-rich powder (20, 21). The production costs are higher in North China than in South China, as the colder climate requires additional natural gas and electricity to heat the mini-livestock production facility. On average, the production cost is around 1,370 US$ per ton of insect/earthworm dry matter in North China, three times higher than that in the South (*SI Appendix*, Fig. S15). However, the production cost of mini-livestock protein is only 12 to 40% of the total cost of conventional animal-sourced food and feed production (32, 34), depending on the substitution strategies and locations of mini-livestock protein production, except for the SIP strategy. While the cost of producing mini-livestock protein in North China is higher than the cost of soybean production in Latin America, the total production cost in South China is still half of the traditional food production cost of the SIP strategy (*SI Appendix*, Fig. S15).

Conclusions

Our results show that investing in mini-livestock production in China will contribute to achieving food security, meeting its carbon neutrality pledges, and reducing resource use. However, the outcomes of the three proposed strategies vary greatly. A minimum level of change in the Chinese diet, with a considerable reduction in GHG emissions and N fertilizer use at the global level, is achievable by replacing imported products; however, the reduction of GHG emissions in China would be negligible. Setting domestic products as the only substitution targets in combination with BECCS could help China's agricultural sector become carbon neutral and able to meet feed protein shortage. However, this would require a replacement of 48% of animal-sourced food with mini-livestock protein and largely offsetting climate change control capability at the global scale. The import and domestic product hybrid substitution strategy leads to balanced, high level of food security and GHG emission reduction improvement for China and its trading partners. Hence, achieving the full potential of climate control effects depends on how the substitution strategy is decided and how the recovered agricultural land is managed (Fig. 1). While our study focuses on the benefits of mini-livestock production in China as one of the world's most important markets for food and feed, it also sets an example for the potential adoption of mini-livestock in the agricultural sector of other countries resulting in a noteworthy global impact.

Methods

Mini-Livestock Protein Production Capability. We selected insects and earthworms with the ability to recycle food waste and solid livestock manure, such as black soldier fly (*Hermetia illucens*), mealworm (*Tenebrio molitor*), house cricket (*Aedes domesticus*), housefly (*Musca domestica*), and earthworm (*Pheretima*) (*SI Appendix*, Table S2). We divided these animals into two groups. The first group includes house crickets, black solider fly, and mealworms which can grow on organic source such as food waste. The mini-livestock protein produced by this group can be used for human consumption, after proper treatment, which entails drying, cleaning, and sanitation. The second group includes black soldier fly, housefly, and earthworm which can grow on organic waste such as livestock manure. Mini-livestock protein produced by the second group based on manure can be used as feed protein for livestock production, after proper treatment (20, 21). The yields of each type of mini-livestock protein can be calculated according to Eq. **1**

$$\mathrm{Pr} = PrC \times M \times FCR \times W, \qquad [\mathbf{1}]$$

where Pr is the yield of mini-livestock protein, *PrC*, *M*, and *FCR* are the protein content (dry basis), dry matter, and feed conversion ratio of each insect, respectively, and *W* is the amount of food waste or livestock manure. The reported food waste production rate and solid livestock manure production rate per year are listed in *SI Appendix*, Table S1. The reference year is 2018 due to the availability of data and parameters.

Mini-Livestock Protein Substitution Strategies. We targeted the use of edible protein from mini-livestock to substitute five different types of animal-sourced food, namely pork, eggs, poultry meat, beef and mutton, and feed protein from mini-livestock to substitute three different plant-sourced protein-rich feed types–soybean, rapeseed, and peas. These products are replaced at the same proportions, to minimize the disturbance of the diet composition. Three different substitution strategies were developed concerning the priority of domestically produced products or imported products, which will provide useful information for scientists and policymakers.

SIP. Products from China's trading partners with a higher GHG emission mitigation potential have been given a higher priority for substitution, until all the produced mini-livestock protein has been used or all targeted products have been replaced.

SDP. Only the five domestically produced animal-sourced products and three plant-sourced feed products are reduced, until all the produced mini-livestock protein has been used or all targeted products have been replaced.

SDIH. Products from either China or its major trader partners were replaced. The trading partners with the highest GHG emission mitigation potential were given a higher priority for substitution, without considering the location of agricultural production. Substitution of animal-sourced food or feed was carried out proportionally for the selected conventional animal-sourced food or feed in all strategies, to avoid large disturbances of diet structure.

Impacts on Protein Consumption. The related reduction of feed requirements due to the reduction of livestock production by the major trading partners has been tracked, to estimate the impacts on environmental performances. The impact of a reduction in China's livestock production on feed supply from the domestic and international markets has also been modeled using the NUFER-animal model, which distinguishes the feed requirement from domestic production and the international market (13).

Impacts on Land Savings. The arable land savings were calculated as the difference between the land use footprint of producing mini-livestock protein and that of conventional protein production. We divided the land use footprint into grassland and cropland. The land use footprints of different food and feed products of major trading partners were derived from previous studies (*SI Appendix*, Table S2). The land use footprints of the different insects are presented in *SI Appendix*, Table S2. Land use reduction was corrected by the multiple cropping index of different countries, defined as the total harvest area divided by the total cropland area in 2018 (34). Two different scenarios have been developed to use the recovered agricultural land, given their different roles in carbon emission reduction, sequestration, biodiversity protection, and N fertilizer use requirement.

Afforestation scenario. All recovered land will be used for afforestation, except for the recovered grassland in China, due to the serious degradation rate and lack of sufficient water supply for most grassland in China (36).

BECCS scenario. All recovered land will be used for bioenergy production, including carbon capture and storage technology, except for the recovered grassland in China (36). Five main lignocellulosic bioenergy crops were selected, namely eucalypt, *Miscanthus*, switchgrass, poplar, and willow, which have a wide range of possibilities to be used at the global scale (47).

Impacts on GHG Emission Reductions. Here, the integrated effects on changes in GHG emissions have been considered, including increases from the production of mini-livestock protein reduction in GHG emissions through improved food waste and manure management in China, reduced production of livestock and crops, and carbon sequestration by afforestation of saved agricultural land within and outside of China's territory border. Life cycle assessment has been applied, for example, considering the energy consumption related to indirect GHG emissions and the direct GHG emissions of N fertilizer manufacture and application (62). The CO_2 emissions from food waste disposal and manure were calculated based on a previous study (63) and the FAOSTAT database. Greenhouse gas emissions reduced by afforestation were calculated by multiplying the area of cultivated land saved by the amount of CO_2 removed by a unit area of forest. The amount of CO_2 removed per unit area of forest is the ratio of the total forest carbon removal to the total forest area of each country. The GHG emissions of net forest conversion in each country were derived from the FAOSTAT database. Net CO_2 sequestration from carbon capture and storage (SeqCCS) was calculated following the method by ref. (64).

Impact on N Fertilizer Use. N fertilizer use of different animal-sourced products of China's major trading partners, resulting from feed production, was estimated by using the most recently updated country-specific N footprint data for different livestock products (10). Synthetic N fertilizer use of different feed products was estimated following the methodology described in ref. 62 and applied to the main trading partners of China. The N fertilizer requirement of bioenergy crop production was derived from ref. 47.

Data, Materials, and Software Availability. All study data are included in the article and/or *SI Appendix*.

ACKNOWLEDGMENTS. This study was financially supported by Strategic Priority Research Program of the Chinese Academy of Sciences (CAS) (XDA28030302); National Natural Science Foundation of China (T2222016; 42001254; 41925004; 42175169); Youth Innovation Promotion Association of CAS (2019101; 2021095); L.L. acknowledges Ministry of Economy and Competitiveness and European Commission ERDF Ramon y Cajal grant (RYC-2016-20269), PP from UPM, and acknowledges the Comunidad de Madrid (Spain) and structural funds 2014–2020 (ERDF and ESF), the Spanish Ministry of Science, Innovation and Universities AgroScena-UP (PID2019-107972RB-I00) and the President's International Fellowship Initiative (PIFI) of CAS (2021VCA0012); E. A. acknowledges Juan de la Cierva grant from the Ministry of Economy and Competitiveness of Spain (IJC 2019-040699-I) Key Research and Development Program of Hebei (21327315D); Hebei Dairy Cattle Innovation Team of Modern Agro-industry Technology Research System (HBCT2023180202); Outstanding Young Scientists Project of Natural Science Foundation of Hebei (D2021503015).

Author affiliations: [a]Key Laboratory of Agricultural Water Resources, Hebei Key Laboratory of Soil Ecology, Center for Agricultural Resources Research, Institute of Genetic and Developmental Biology, Chinese Academy of Sciences, Shijiazhuang, Hebei 050021, China; [b]Research Centre for the Management of Agricultural and Environmental Risks, Escuela Técnica Superior de Ingeniería Agronomica, Alimentaria y de Biosistemas, Universidad Politécnica de Madrid, Madrid 28040, Spain; [c]Laboratory of Entomology, Wageningen University and Research, Wageningen 6700 AA, The Netherlands; [d]Department of Earth System Science, Ministry of Education Key Laboratory for Earth System Modeling, Institute for Global Change Studies, Tsinghua University, Beijing 100084, China; [e]State Key Laboratory of Pollution Control and Resource Reuse, School of the Environment, Nanjing University, Nanjing 210023, China; [f]Alimentta, Think Tank para la Transición Alimentaria, Andalucía 18320, Spain; and [g]Animal Production and Health Division, Food and Agriculture Organization of the United Nations, Rome 00153, Italy

1. D. Gerten *et al.*, Feeding ten billion people is possible within four terrestrial planetary boundaries. *Nat. Sustain.* **3**, 200–208 (2020).
2. M. Springmann *et al.*, Options for keeping the food system within environmental limits. *Nature* **562**, 519–525 (2018).
3. J. Poore, T. Nemecek, Reducing food's environmental impacts through producers and consumers. *Science* **363**, 939–939 (2018).
4. M. A. Clark *et al.*, Global food system emissions could preclude achieving the 1.5 and 2 C climate change targets. *Science* **370**, 705–708 (2020).
5. M. Crippa *et al.*, Food systems are responsible for a third of global anthropogenic GHG emissions. *Nat. Food* **2**, 198–209 (2021).
6. J. Chang *et al.*, Reconciling regional nitrogen boundaries with global food security. *Nat. Food* **2**, 700–711 (2021).
7. L. F. Schulte-Uebbing *et al.*, From planetary to regional boundaries for agricultural nitrogen pollution. *Nature* **610**, 507–512 (2022).
8. M. A. Sutton *et al.*, The nitrogen decade: Mobilizing global action on nitrogen to 2030 and beyond. *One Earth* **4**, 10–14 (2021).
9. D. K. Ray *et al.*, Crop harvests for direct food use insufficient to meet the UN's food security goal. *Nat. Food* **3**, 367–374 (2022).
10. A. Uwizeye *et al.*, Nitrogen emissions along global livestock supply chains. *Nat. Food* **1**, 437–446 (2020).
11. X. Xu *et al.*, Global greenhouse gas emissions from animal-based foods are twice those of plant-based foods. *Nat. Food* **2**, 724–732 (2021).
12. L. Lassaletta *et al.*, Food and feed trade as a driver in the global nitrogen cycle: 50-year trends. *Biogeochemistry* **118**, 225–241 (2014).
13. Z. Bai *et al.*, China's livestock transition: Driving forces, impacts, and consequences. *Sci. Adv.* **4**, eaar8534 (2018).
14. WFP and FAO, Hunger Hotspots. FAO–WFP early warnings on acute food insecurity: October 2022 to January 2023 Outlook. Rome (2022).
15. L. Lassaletta *et al.*, Future global pig production systems according to the Shared Socioeconomic Pathways. *Sci. Total Environ.* **665**, 739–751 (2019).
16. E. K. H. J. zu Ermgassen *et al.*, Reducing the land use of EU pork production: Where there's swill, there's a way. *Food Policy* **58**, 35–48 (2016).
17. P. Horton *et al.*, Technologies to deliver food and climate security through agriculture. *Nat. Plants* **7**, 250–255 (2021).
18. A. Tzachor *et al.*, Future foods for risk-resilient diets. *Nat. Food* **2**, 326–329 (2021).
19. A. K. Hazarika, U. Kalita, Human consumption of insects. *Science* **379**, 140–141 (2023).
20. A. van Huis, Manure and flies: Biodegradation and/or bioconversion? *J. Insects Food Feed* **5**, 55–58 (2019).
21. A. van Huis *et al.*, Insects as food and feed, a new emerging agricultural sector: A review. *J. Insects Food Feed* **6**, 27–44 (2020).
22. A. van Huis, L. Gasco, Insects as feed for livestock production. *Science* **379**, 138–139 (2023).
23. Ministry of Agricultural and Rural Affairs (MOARA). Import and export of agricultural products in China in 2021 (2022). http://www.moa.gov.cn/ztzl/nybrl/rlxx/202201/t20220127_6387781.htm. Accessed 9 July 2022.
24. H. Zhang, *Securing the 'Rice Bowl': China and Global Food Security* (Springer, 2018).
25. Z. Liu *et al.*, Challenges and opportunities for carbon neutrality in China. *Nat. Rev. Earth Environ.* **3**, 141–155 (2022).
26. C. Hong *et al.*, Land-use emissions embodied in international trade. *Science* **376**, 597–603 (2022).
27. W. Jia *et al.*, Evaluation of crop residues and manure production and their geographical distribution in China. *J. Clean. Prod.* **188**, 954–965 (2018).
28. L. Xue *et al.*, China's food loss and waste embodies increasing environmental impacts. *Nat. Food* **2**, 519–528 (2021).
29. M. H. Zhou *et al.*, New policy and implementation of municipal solid waste classification in Shanghai, China. *Int. J. Environ. Res. Public Health* **16**, 3099 (2019).
30. D. R. Chadwick *et al.*, Strategies to reduce nutrient pollution from manure management in China. *Front. Agri. Sci. Eng.* **7**, 45–55 (2020).
31. M. van der Spiegel *et al.*, Safety of novel protein sources (insects, microalgae, seaweed, duckweed, and rapeseed) and legislative aspects for their application in food and feed production. *Compr. Rev. Food Sci. Food Safety* **12**, 662–678 (2013).
32. D. Pleissner, S. Smetana, Estimation of the economy of heterotrophic microalgae-and insect-based food waste utilization processes. *Waste Manag.* **102**, 198–203 (2020).
33. G. Melgar-Lalanne *et al.*, Edible insects processing: Traditional and innovative technologies. *Compr. Rev. Food Sci. Food Saf.* **18**, 1166–1191 (2019).
34. Food and Agricultural Organization (FAO), FAOSTAT. https://www.fao.org/faostat/en/#data (2022). Accessed 9 July 2022.
35. Z. Bai *et al.*, Global environmental costs of China's thirst for milk. *Global Change Biol.* **24**, 2198–2211 (2018).
36. W. Zhou *et al.*, Grassland degradation remote sensing monitoring and driving factors quantitative assessment in China from 1982 to 2010. *Ecol. Indicators* **83**, 303–313 (2017).
37. X. Zhao *et al.*, Challenges toward carbon neutrality in China: Strategies and countermeasures. *Resour. Conserv. Recycl.* **176**, 105959 (2022).
38. F. Hua *et al.*, The biodiversity and ecosystem service contributions and trade-offs of forest restoration approaches. *Science* **376**, 839–844 (2022).
39. F. Wei, Toward post-2020 global biodiversity conservation: Footprint and direction in China. *Innovation* **2**, 100175 (2021).
40. L. Cui *et al.*, The influence of climate change on forest fires in Yunnan Province, Southwest China detected by GRACE satellites. *Remote Sens.* **14**, 712 (2022).
41. M. Senande-Rivera *et al.*, Spatial and temporal expansion of global wildland fire activity in response to climate change. *Nat. Commun.* **13**, 1–9 (2022).
42. T. Shiraishi, R. Hirata, Estimation of carbon dioxide emissions from the megafires of Australia in 2019–2020. *Sci. Rep.* **11**, 1–10 (2021).
43. A. B. Harper *et al.*, Land-use emissions play a critical role in land-based mitigation for Paris climate targets. *Nat. Commun.* **9**, 2938 (2018).
44. A. Krause *et al.*, Large uncertainty in carbon uptake potential of land-based climate-change mitigation efforts. *Global Change Biol.* **24**, 3025–3038 (2018).
45. Y. Cheng *et al.*, Future bioenergy expansion could alter carbon sequestration potential and exacerbate water stress in the United States. *Sci. Adv.* **8**, eabm8237 (2022).
46. J. Rockström *et al.*, Safe and just Earth system boundaries. *Nature* **619**, 102–111 (2023).
47. W. Li *et al.*, Bioenergy crops for low warming targets require half of the present agricultural fertilizer use. *Environ. Sci. Technol.* **55**, 10654–10661 (2021).
48. A. Parodi *et al.*, The potential of future foods for sustainable and healthy diets. *Nat. Sustain.* **1**, 782–789 (2018).
49. M. N. Mwangi *et al.*, Insects as sources of iron and zinc in human nutrition. *Nutr. Res. Rev.* **31**, 248–255 (2018).
50. D. B. Kumssa *et al.*, Dietary calcium and zinc deficiency risks are decreasing but remain prevalent. *Sci. Rep.* **5**, 1–11 (2015).
51. Q. Li *et al.*, From organic waste to biodiesel: Black soldier fly, Hermetia illucens, makes it feasible. *Fuel* **90**, 1545–1548 (2011).
52. S. Jung *et al.*, Biodiesel production from black soldier fly larvae derived from food waste by non-catalytic transesterification. *Energy* **238**, 121700 (2022).

53. J. Laherrère *et al.*, How much oil remains for the world to produce? Comparing assessment methods, and separating fact from fiction. *Curr. Res. Environ. Sustain.* **4**, 100174 (2022).
54. K. C. Surendra *et al.*, Bioconversion of organic wastes into biodiesel and animal feed via insect farming. *Renew. Energy* **98**, 197–202 (2016).
55. K. Y. Barragán-Fonseca *et al.*, Insect frass and exuviae to promote plant growth and health. *Trends Plant Sci.* **27**, 646–654 (2022).
56. Y. Du *et al.*, A global strategy to mitigate the environmental impact of China's ruminant consumption boom. *Nat. Commun.* **9**, 1–11 (2018).
57. X. Zhang *et al.*, Managing nitrogen for sustainable development. *Nature* **528**, 51–59 (2015).
58. T. Zou *et al.*, Global trends of cropland phosphorus use and sustainability challenges. *Nature* **611**, 81–87 (2022).
59. X. Kong, China must protect high-quality arable land. *Nature* **506**, 7 (2014).
60. The State Council of China (SCC), Preventing "Non grain" Farmland from Stabilizing Grain Production [in Chinese] (2020). http://www.gov.cn/zhengce/content/2020-11/17/content_5562053.htm?trs=1. Accessed 9 July 2022.
61. H. Zhao *et al.*, China's future food demand and its implications for trade and environment. *Nat. Sustain.* **4**, 1042–1051 (2021).
62. E. Aguilera *et al.*, Greenhouse gas emissions from Mediterranean agriculture: Evidence of unbalanced research efforts and knowledge gaps. *Global Environ. Change* **69**, 102319 (2021).
63. M. Liu *et al.*, Assessment and projection of environmental impacts of food waste treatment in China from life cycle perspectives. *Sci. Total Environ.* **807**, 150751 (2022).
64. S. V. Hanssen *et al.*, The climate change mitigation potential of bioenergy with carbon capture and storage. *Nat. Clim. Change* **10**, 1023–1029 (2020).