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Manure to insect farming for feed mitigates nitrogen and greenhouse gas emissions

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Livestock manure is nutrient-rich and represents a promising feedstock for insect farming, with insect protein offering a high-quality alternative to conventional crop-based livestock feed. This pathway has substantial potential to mitigate nitrogen (N) and greenhouse gas (GHG) emissions from livestock systems. Here, we integrate global livestock models with trade, land-use, and cost–benefit analysis to quantify the environmental and economic implications of manure-to-insect conversion while ensuring the recycling of manure to croplands to maintain soil fertility across 166 countries. We show that insect farming could yield approximately 4.2 (3.4 to 5.5) million tonnes (Mt) N of insect protein to substitute crop-based feed and release up to 45 million hectares of cropland. This transition could reduce N and GHG emissions by 6.0 (5.5 to 6.8) Mt N and 1,277 (1,081 to 1,634) Mt CO_2-e, respectively, during feed cultivation and manure storage. The residual manure, after insect processing, can be returned to fields as a high-quality organic fertilizer, offsetting 25 Mt N of synthetic fertilizers. With an estimated implementation cost of 25 billion USD, this strategy could deliver combined economic and environmental benefits of 226 billion USD. These results position manure-based insect farming as a cost-effective and circular solution to enhance global food system resilience under environmental pressures.

nutrient recycling | nitrogen management | alternative protein | sustainable livestock production

Rising living standards have driven a growing global demand for livestock products (1), accompanied by escalating pollution, greenhouse gas (GHG) emissions, and the accumulation of large quantities of manure that require urgent sustainable management (2–4). Excess manure exerts substantial pressure on air, water, and soil systems because protein-rich livestock feeds such as soybean meal are only partially digested, leaving large residues of nitrogen (N) and other nutrients in manure (5–7). While a portion of this manure can be recycled to croplands to enhance soil fertility, timely application is often constrained by the spatial separation of livestock farms from croplands, and seasonal mismatches in fertilizer demand (5, 8, 9). These spatiotemporal mismatches have exacerbated the decoupling of livestock production from cropland cultivation (10). Meanwhile, rising demand for animal products has intensified competition between feed and food crops, amplifying concerns over global food security (11). These challenges underscore the urgent need for stable, efficient, and environmentally sustainable manure management strategies that restore the coupling between crop and livestock systems.

Insects are increasingly recognized as a promising alternative protein source (12–14). While their direct use as human food remains limited (15, 16), their adoption as animal feed has already achieved industrial-scale application (17). Among the diverse insect groups, saprophagous species such as the black soldier fly (*Hermetia illucens*) are of particular interest (18). These insects not only provide protein independent of crops but also offer an effective pathway for the bioconversion of organic wastes, including livestock manure. Livestock manure, rich in N and organic matter content, provides a sustainable substrate for insect farming (19). With bioconversion, insect farming can reduce organic waste by up to 60% (20) while stimulating microbial communities that suppress methane (CH_4) and nitrous oxide (N_2O) production, thereby lowering GHG emissions (12, 21). This process also reduces ammonia (NH_3) emissions (22). The resulting larvae, rich in protein and fat (23, 24), can be used as livestock feed and substitute conventional feed ingredients such as soybean meal, reducing feed costs and alleviating the food–feed competition that exacerbates food security pressures. In addition, insect bioconversion accelerates manure stabilization, yielding compost with lower pathogen loads, thereby reducing zoonotic risks and facilitating safer recycling of manure to croplands (25–27).

Significance

Livestock production is a major source of nitrogen pollution and greenhouse gas emissions and depends heavily on cropland for feed protein. Converting livestock manure into insect protein offers a circular strategy that recovers waste nitrogen as high-quality feed while producing organic fertilizer. This transition could reduce nitrogen losses and greenhouse gas emissions while easing food-feed competition. The benefits are particularly promising in Africa, where favorable climatic conditions make manure-to-insect systems well-suited to reduction of feed imports, lowering fertilizer dependence and improving food self-sufficiency. Achieving this potential requires coordinated efforts across supply chains, manure logistics, safety control, market acceptance, and regulatory frameworks governing substrate eligibility and traceability.

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Maintaining soil fertility requires that a certain amount of the manure be returned to croplands, but the remaining manure can be used as substrate for insect production and the resulting insect protein can partially substitute for crop-based animal feed. Recycling the stabilized manure residues from insect treatment can sustain cropland productivity, promote nutrient recycling, and help mitigate the challenges posed by the spatiotemporal separation of livestock and cropland systems (20). Moreover, compared with insect-derived food for human consumption, insect-derived feed generally faces fewer acceptance barriers, making it a more immediately viable solution for sustainable and circular food systems (17, 28, 29). However, the global potential of manure-based insect farming in terms of reducing crop feed demand, mitigating N losses and GHG emissions, and delivering environmental, economic, and societal benefits remains poorly quantified, particularly under intensifying global emission pressures. This knowledge gap in the socioeconomic and environmental assessment of insect protein from manure has hindered policy development and the design of operational circular production systems.

Compared with other insects such as the housefly, the black soldier fly shows higher production stability, feed conversion efficiency, and more flexible adaptability to diverse substrates and environmental conditions (25). Black soldier fly production is already well-established and has been deployed in many regions (13, 30). Here, we address this gap by quantifying the environmental potential of utilizing livestock manure to produce black soldier fly larvae as a feed substitute. We focused on the remaining manure after recycling to croplands, excluding grazing manure directly deposited on grasslands. Using data from 166 countries combined with the Global Livestock Environmental Assessment Model (GLEAM) (31) and the Coupled Human and Natural Systems (CHANS) model (32), we evaluate the potential of an insect farming scenario relative to a business-as-usual (BAU) baseline. Specifically, we estimate reductions in N and GHG emissions, substitution potential for synthetic fertilizers, and cropland released through lower crop feed demand over both feed cultivation and manure storage stages. We further integrate the FAO trade matrix (33) to calculate feed imports and domestic feed production in each country, thereby assessing changes in feed import dependence under the insect farming scenario. Finally, we conduct a comprehensive cost–benefit analysis to evaluate feasibility. Our findings provide insights into manure management, demonstrating that insect farming on livestock manure is a scalable, cost-effective, and circular strategy to strengthen agricultural sustainability under mounting environmental pressures.

Results and Discussion

Total N and GHG Emission Reduction. Global reactive nitrogen (Nr) emissions from the livestock manure storage stage and feed crop production totaled 33.2 Mt N in 2020, including losses of NH_3, NO_x, and NO_3^- (Fig. 1). After recycling 17.8 Mt N of manure to croplands for feed production and 8.6 Mt N for food production, 26.7 Mt N of manure remained available for insect farming. This residual manure could generate 4.2 (3.4 to 5.5) Mt N of insect-based feed, substituting 8% of conventional crop-based feed (Fig. 2*A*). Under this scenario, referred to as the Insect scenario (*SI Appendix*, Fig. S1), total N emissions from feed production and manure management could be reduced by 6.0 (5.5 to 6.8) Mt N (−18%), with NH_3 and NO_3^- losses declining by 4.9 Mt N (−21%) and 1.1 Mt N (−11%), respectively.

Total GHG emissions from manure storage and feed production reached 1,351 Mt CO_2-e in 2020, encompassing emissions from fertilizer manufacture, field processes (including CH_4 emission from rice cultivation), feed processing, blending and transport, as well as CH_4 emissions from manure. Insect treatment of manure

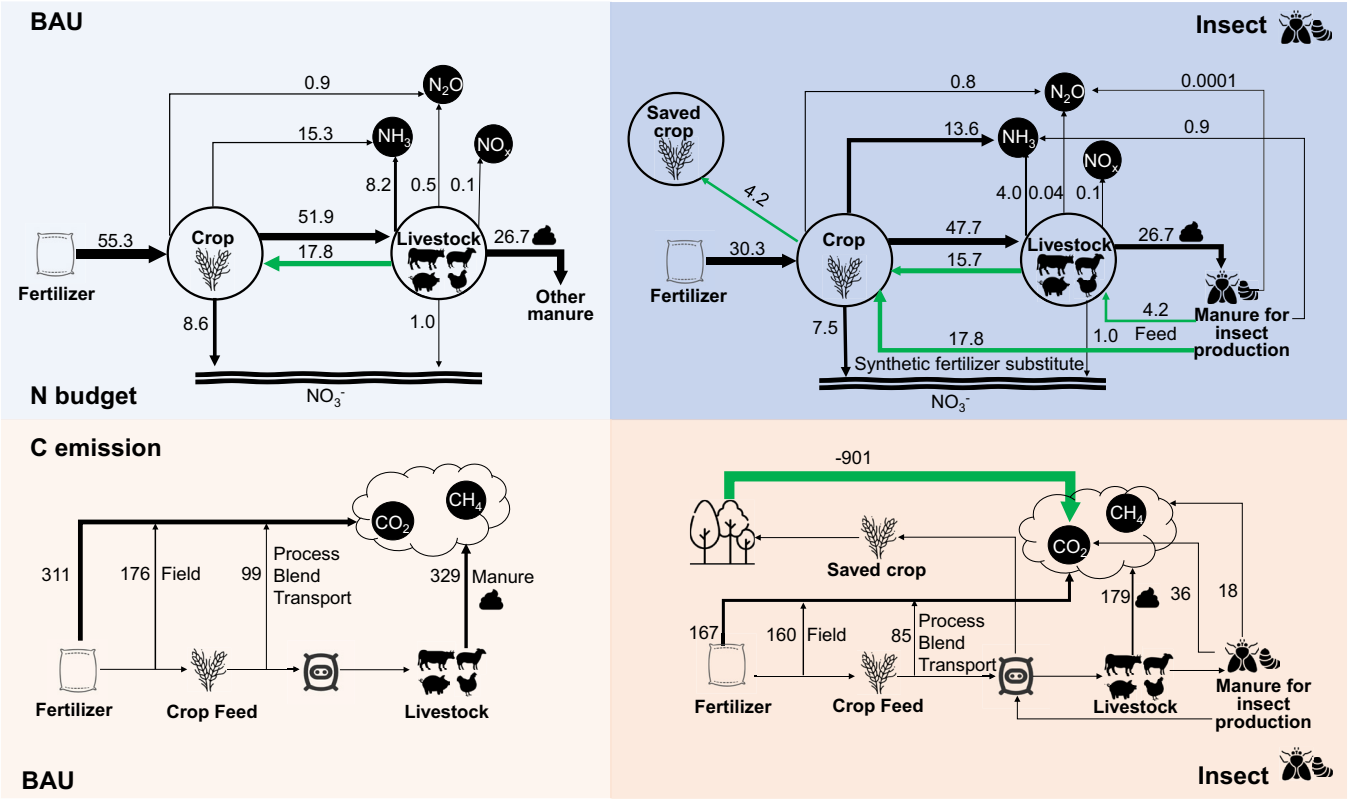

Fig. 1. N budget and carbon emissions under BAU and Insect scenarios. Unit, Mt N for N budget and Mt CO_2-e for C emissions. Black arrows represent material flows, while green arrows indicate green flows associated with sustainable recycling or reducing emissions.

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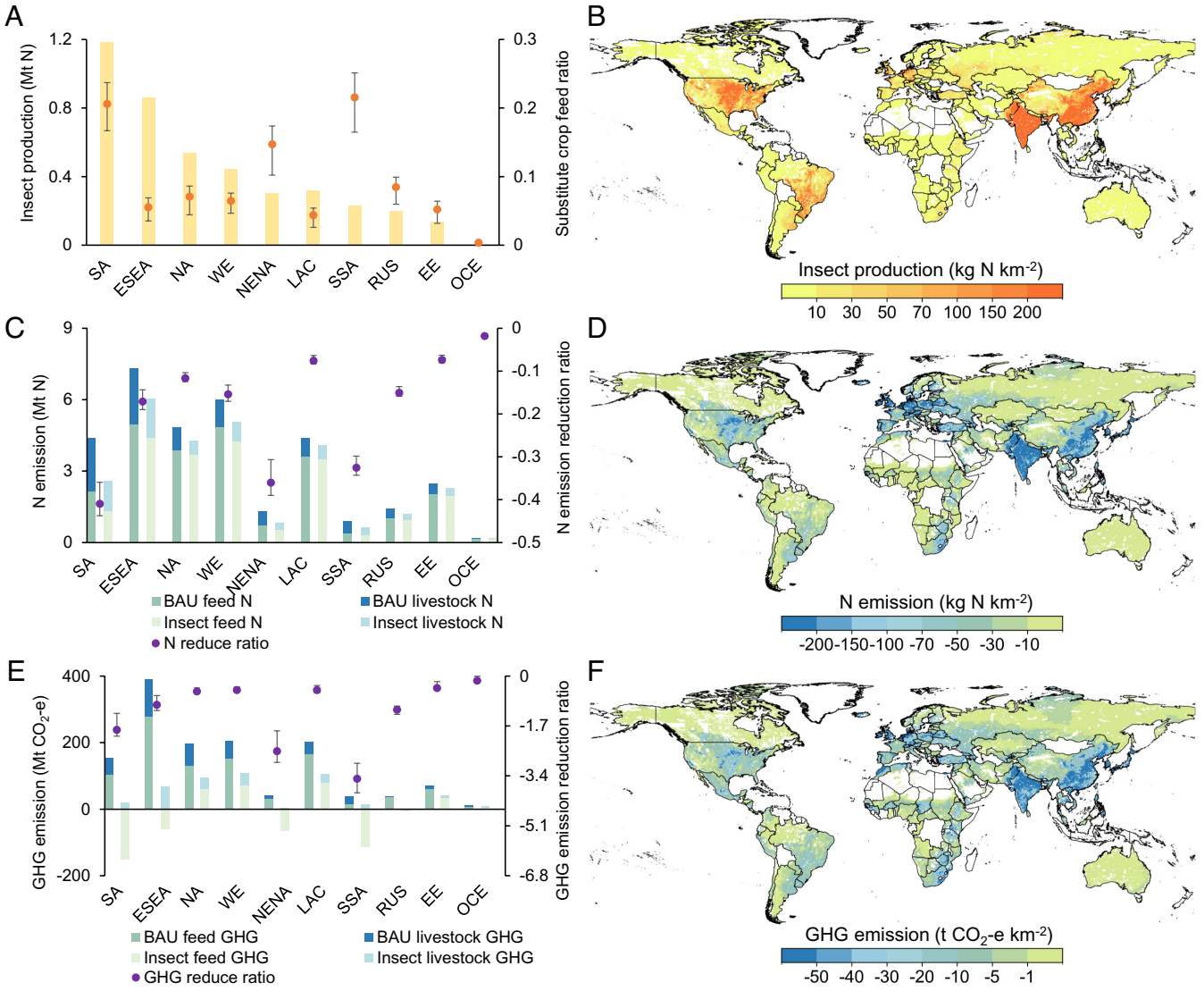

Fig. 2. Insect production and reductions in N and GHG emissions under the Insect scenario. (*A*) Regional insect production. (*B*) Spatial insect production, kg N km^{-2}. (*C*) Regional N emission reduction. N emissions include NH_3, NO_x, and NO_3^-. (*D*) Spatial N emission reduction, kg N km^{-2}. (*E*) Regional GHG emission reduction. GHG emissions include N_2O, CH_4, and CO_2. (*F*) Spatial GHG emission reduction, t CO_2-e km^{-2}. The division of the regions is based on the GLEAM (https://www.fao.org/gleam/en/). WE, Western Europe. SSA, Sub-Saharan Africa. SA, South Asia. RUS, Russian Federation. OCE, Oceania. NENA, Near East and Northern Africa. NA, North America. LAC, Latin America, and Caribbean. ESEA, East and Southeast Asia. EE, Eastern Europe. The base map was applied from the Database of Global Administrative Areas (GADM; https://gadm.org/).

reduced CH_4 emissions by 132 Mt CO_2-e (–40%) and N_2O emissions by 18 Mt CO_2-e (–50%), while substituting crop-based feed with insect protein lowered GHG emissions from feed production by 226 Mt CO_2-e (–23%). If cropland released from feed production were reforested, it could sequester an additional 901 Mt CO_2-e, bringing the total reduction in GHG emissions to 1,277 (1,081 to 1,634) Mt CO_2-e.

Regional differences were pronounced. South Asia exhibited the largest potential for insect protein production, yielding 1.2 Mt N and replacing 21% of conventional crop-based feed protein (Fig. 2*A*). This reflects the region's dual role as a major livestock producer and net feed importer, which generates large amounts of manure that cannot be fully recycled to croplands. South Asia also exhibited the highest N emission mitigation potential, with reductions of 1.8 Mt N (–41%). Favorable temperature and light conditions supported insect farming in Sub-Saharan Africa (34), where insect protein could replace up to 22% of crop-based feed protein (Fig. 2*B*). High emission intensity in the region implies substantial mitigation (*SI Appendix*, Fig. S2), with projected reductions of 0.3 Mt N (–33%) and 138 Mt CO_2-e (–350%) when land-use change emissions were considered. The reduction rate exceeded 100% because the assumed reforestation of released cropland led to net negative GHG emissions. Oceania showed the lowest reduction potential with N and GHG emission reductions of 2% and 16%, respectively, due to its already low emission intensity and dominance of ruminant systems (35, 36), in which manure is mostly deposited directly onto grasslands. In some regions, net GHG emissions from feed production became negative (Fig. 2*C*), as croplands associated with displaced crop feed were assumed to be reforested, leading to additional carbon sequestration.

Emission Reductions Along Livestock Production Chain. Across the livestock production chain, N emissions from feed production substantially exceeded those from manure storage stage. Under the Insect scenario, N emissions were reduced by 2.7 Mt N (–11%) from feed production and 3.3 Mt N (–36%) from manure storage (Fig. 1). India exhibited the largest N emission mitigation, with

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reductions of 0.5 Mt N from feed production and 0.7 Mt N from manure storage (*SI Appendix*, Fig. S3), reflecting its highest potential for insect protein production (0.9 Mt N). China had the second-largest potential for insect protein production (0.7 Mt N), with reductions of 0.4 Mt N from feed production and 0.5 Mt N from manure storage. Despite producing more manure N (9 Mt N) than India (7 Mt N), China showed lower insect protein potential, largely because of its higher existing manure N recycling rate (52% in China versus 31% in India, *SI Appendix*, Fig. S4). These results indicate that insect farming can serve as a targeted mitigation strategy in regions with low recycling efficiency, such as India.

Similarly, GHG emissions were reduced by 1,127 Mt CO_2-e from feed production and 150 Mt CO_2-e from manure storage (Fig. 2*E*). The highest GHG reduction rates from feed production were observed in Near East and North Africa (NENA) and Sub-Saharan Africa (SSA) region (Fig. 2*E*), reflecting their high replacement rate of crop-based protein with insect protein. In these regions, relatively low crop yields mean that replacing crop protein with insect protein both reduces emissions and releases substantial cropland. Reforesting this could further sequester up to 83 Mt CO_2-e in NENA and 123 Mt CO_2-e in SSA, respectively. Alternatively, reallocating the released cropland to food crop production could reduce reliance on imported feed, strengthen domestic crop self-sufficiency, and enhance regional food security.

Change in Trade Flow and Resource Utilization. Livestock production requires large quantities of crop-based feed (Fig. 3*A*), prompting some major livestock-producing countries to import feed from exporting countries and thereby increasing the risk of land-use change from forests to croplands in the exporting regions (37, 38). In 2020, for example, China imported substantial amounts of feed (Fig. 3*B*), including 2.9 Mt N of feed from Brazil, 1.2 Mt N from the United States, and 0.3 Mt N from Argentina (*SI Appendix*, Fig. S5). Under the Insect scenario, manure-derived insect feed increases national feed self-sufficiency and reduces crop feed demand, thereby releasing croplands in exporting countries and easing land-use pressures.

Globally, the Insect scenario saves 118 Mt dry matter (DM) of grain, corresponding to 46 million hectares of cropland (Fig. 4). China showed the greatest potential for grain savings because China is the largest pork producer and a major chicken producer and consumes large amounts of soybean meal and other high-protein feed grains. Approximately 25 Mt DM of crop feed could be substituted under Insect scenario in China, substantially increasing feed self-sufficiency by recycling locally produced manure into insect protein, and thereby reducing dependence on imported feed. For instance, China's feed imports from Brazil could fall by 409 Gg (1 Gg = 10^9 g) N under this scenario (Fig. 3*B*). In some smaller importing countries, such as Kenya and

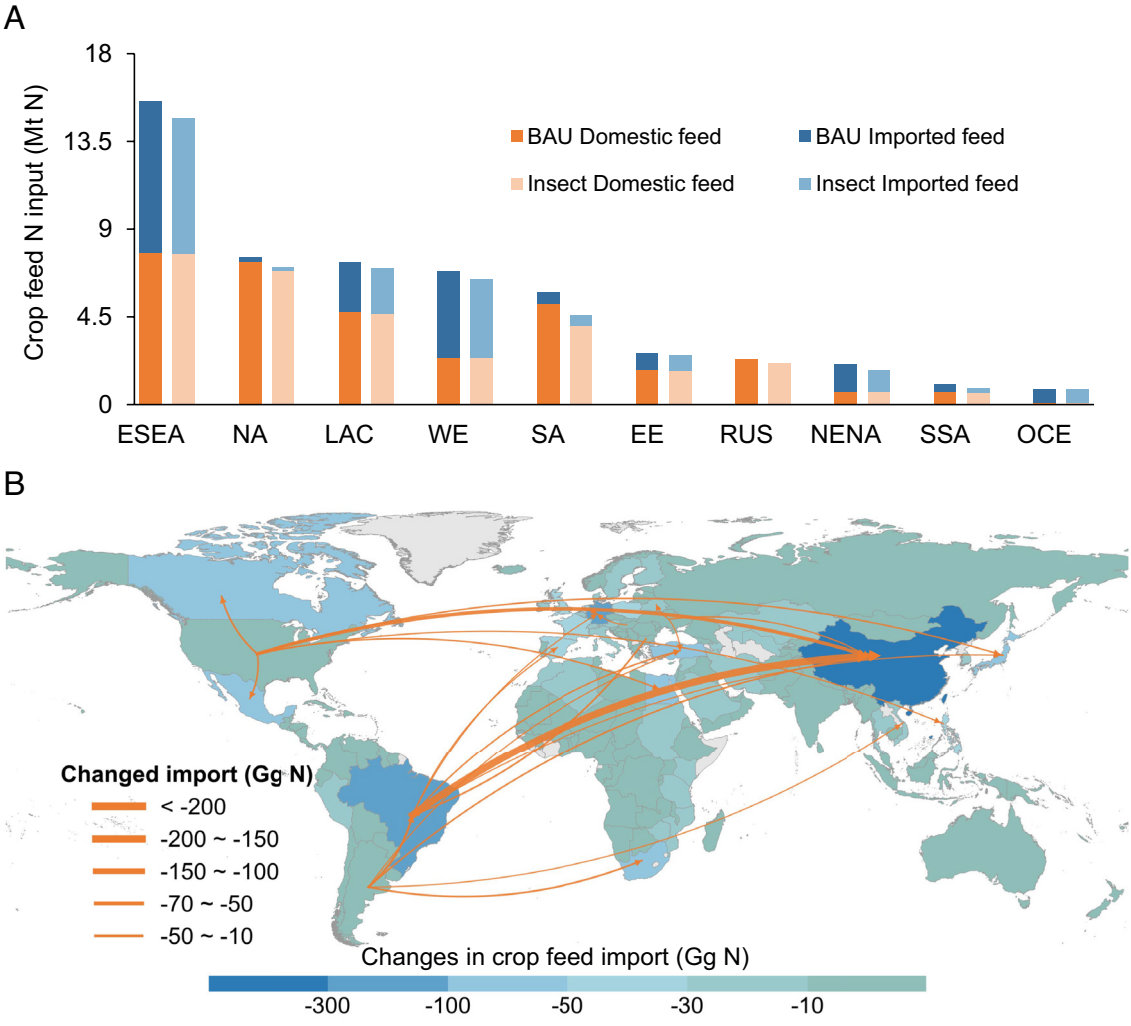

Fig. 3. Changes in feed imports under the Insect scenario. (*A*) Domestic and imported crop-based feed N under BAU and Insect scenarios. (*B*) Changed crop-based feed N import under Insect scenario (Gg N). Map shows reductions in total feed imports under Insect scenario, with arrows indicating the 20 trade routes with the largest decreases. The base map was applied from the Database of Global Administrative Areas (GADM; https://gadm.org/).

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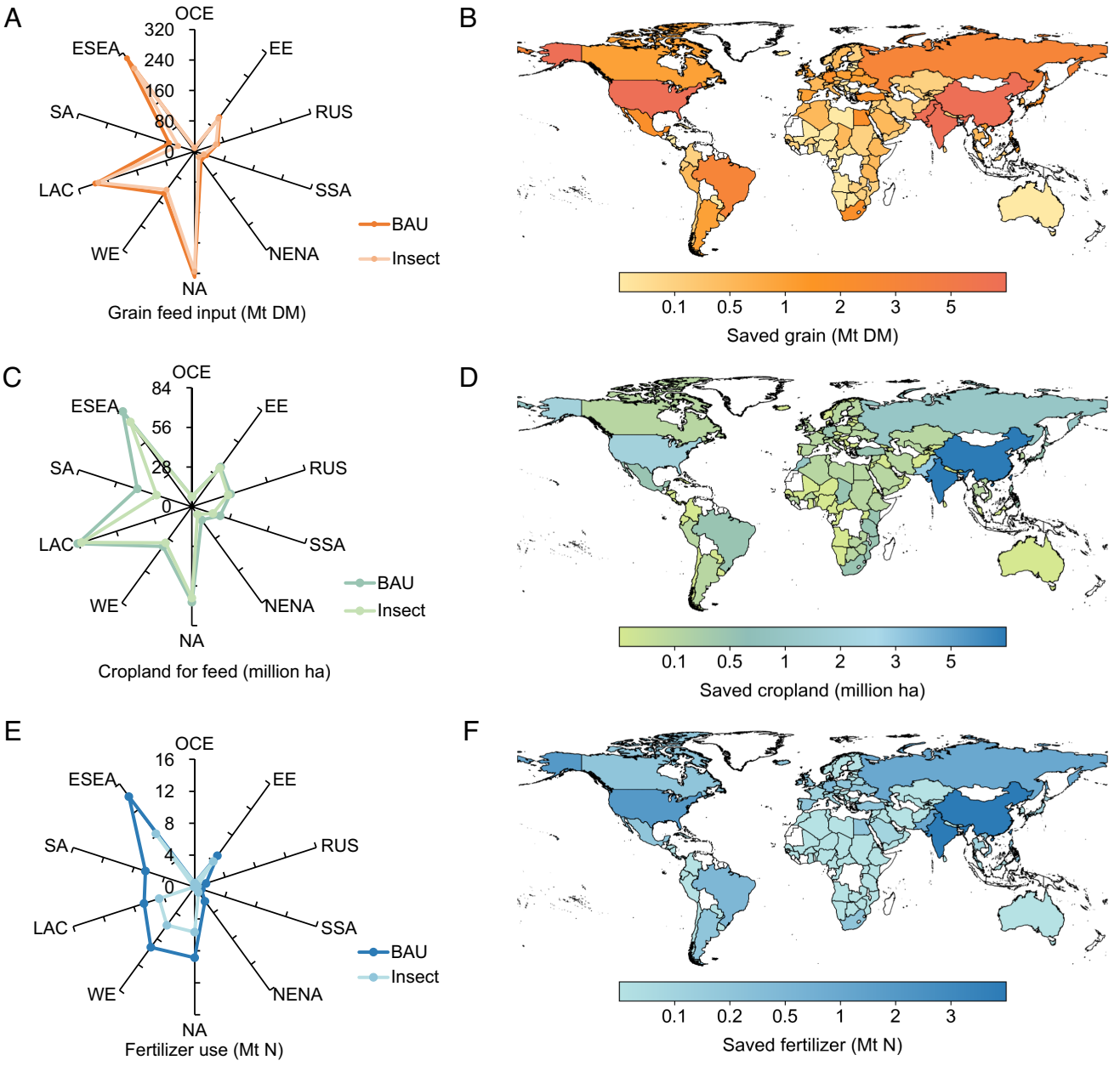

Fig. 4. Saved grains, cropland, and fertilizer under the Insect scenario. (*A*) Grain feed dry matter (DM) input under BAU and Insect scenarios. (*B*) Saved feed grain DM under Insect scenario. (*C*) Cropland use for feed production under BAU and Insect scenarios. (*D*) Saved cropland use for feed production under Insect scenarios. (*E*) Fertilizer use for feed production under BAU and Insect scenarios. (*F*) Saved fertilizer use for feed production under Insect scenario. The base map was applied from the Database of Global Administrative Areas (GADM; https://gadm.org/).

Bahrain, insect proteins fully replace imported feed (*SI Appendix*, Fig. S6).

Feed imports also exacerbate the decoupling of livestock production from crop cultivation, since manure generated in importing countries such as China cannot be recycled onto croplands in exporting regions (39, 40). Under the Insect scenario, however, the residual manure remaining after insect bioconversion serves as a stabilized organic fertilizer with lower pathogen and pollutant loads, replacing up to 17.8 Mt N of synthetic fertilizer for feed production (Fig. 1). This recoupling of crop and livestock systems reduces reliance on synthetic fertilizers and strengthens nutrient recycling. In addition, reduced demand for crop-based feed directly lowers synthetic fertilizer requirements by 7 Mt N (*SI Appendix*, Fig. S7), bringing the total synthetic fertilizer substitution for feed production to 25 Mt N (Fig. 4*E*). Meanwhile, due to reduced crop-based feed demand, the manure N directly returned to croplands decreases from 17.8 to 15.7 Mt N, of which 2.1 Mt N is applied to the 4.2 Mt of saved cropland (Fig. 1). South Asia achieved the largest fertilizer substitution, with 6.4 Mt N replaced at a rate of 99% (Fig. 4*E*), indicating near-complete coupling of cropland and livestock sectors and substantially lowering synthetic fertilizer use and associated production costs.

Cost and Benefit Analysis. To evaluate the feasibility of insect production as a feed strategy, we conducted a cost–benefit analysis comparing implementation costs with environmental and economic benefits. Under the Insect scenario, the total cost of insect protein production was estimated at 25 (20–33) billion USD, including expenditures for labor, larvae, water, depreciation, and energy (natural gas and electricity) (Fig. 5). This investment generated total benefits of 226 (182 to 267) billion USD, distributed across ecosystem (40 billion), health (69 billion), climate (57 billion),

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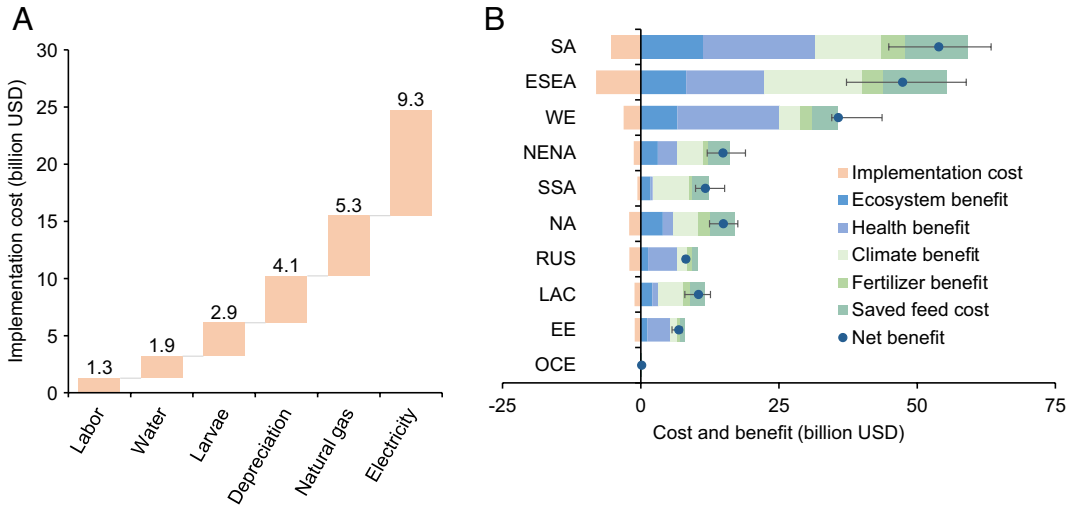

Fig. 5. Costs and benefits under the Insect scenario. (*A*) Implementation costs of the Insect scenario, including expenses from labor, larvae, water, depreciation, and energy (natural gas and electricity). (*B*) Total implementation costs and benefits under Insect scenario. Ecosystem benefit, Health benefit, and Climate benefit refer to the avoided ecosystem, health, and climate damage costs resulting from emission reductions, respectively. Fertilizer benefit refers to the value of fertilizer substitution. Saved feed cost refers to the economic benefit derived from replacing crop-based feed with insect-based protein.

fertilizer substitution (17 billion), and saved feed cost (43 billion). The resulting cost–benefit ratio of nine demonstrates substantial net gains from insect-based feed strategies.

Implementation costs varied geographically. High-latitude countries faced higher costs due to energy requirements for heating and lighting (*SI Appendix*, Fig. S8). By contrast, low-latitude regions such as much of Africa and Asia benefit from abundant sunlight and favorable temperatures, reducing the costs of insect farming. In Sub-Saharan Africa, for example, implementation costs of only 0.6 billion USD yielded benefits of 11 billion USD, distributed across ecosystem (2 billion), health (0.4 billion), climate (7 billion), fertilizer substitution (0.5 billion), and saved feed cost (3 billion). These findings highlight insect farming as a practical, low-cost strategy for many African countries to mitigate food security risks, reduce manure treatment costs, and deliver substantial environmental and societal benefits without large capital investment.

Policy Implication. Rising global demand for livestock products has heightened concerns over the environmental burdens of manure management and the threats to food security posed by competition between livestock feed and human diets (3, 7, 41). Alternative protein sources have emerged as promising options because of their high nutritional value and lower pollution footprint (42–44). However, the use of insect protein in human diets remains limited, constrained by psychological barriers (45, 46) and health concerns, including allergenicity (47). Consequently, the potential for insect protein to substantially replace animal-source foods in human diets remains modest. Across the supply chain, feed cultivation remains the dominant driver of both N pollution and GHG emissions, particularly when accounting for land use change from forests to croplands (36). Other feed alternatives, such as microbial protein (48, 49), as well as chemical and biological processes for producing food without conventional agriculture (50, 51), also offer mitigation potential but face higher implementation costs and are mainly used as feed additives at limited scales (52). By contrast, substituting insect protein for crop-based feed in livestock diets represents a more feasible and scalable pathway to reduce environmental impacts (17). This substitution thus offers a practical opportunity to address the dual challenges of pollution and food security in livestock systems.

Our results show that producing insect protein from livestock manure can reduce N and GHG emissions, save grain demand, release cropland, and substitute for synthetic fertilizer use. Benefits are particularly pronounced in Asian countries, where high demand for feed protein translates into substantial replacement potential. In African countries, although the absolute substitution of crop-based feed is smaller, the proportional replacement is the highest globally, underscoring its importance for alleviating food security pressures. Favorable environmental conditions in much of Africa, with high temperatures and abundant sunlight, also reduce energy demand for heating and lighting (53), thereby lowering the implementation costs of insect farming. At the same time, many African countries face limited fertilizer inputs and declining soil fertility (54), and the stable and pathogen-reduced manure residues from insect farming could substitute synthetic fertilizers and sustain soil productivity (55, 56). Accordingly, context-specific promotion of insect farming in Africa and Asia is especially important to achieve a triple win for sustainable agriculture, food security, and environmental protection.

The substituted fraction accounted for only 8% of total feed crops in the Insect scenario, of which approximately 87% were protein-rich soybeans and oilseeds (e.g., rapeseed and sunflower seed), and only 13% were carbohydrate-based cereals such as maize and wheat (*SI Appendix*, Fig. S6). Given this relatively small proportion, insect protein does not substantially alter the overall feed structure but rather diversifies the nutrient sources and helps stabilize nutrient supply (57–59). At this relatively low substitution level, the net effects for livestock production remain uncertain (60) and may be negligible (61, 62), or modestly positive (59, 63). Substitution may also reduce the intake of certain nutrients, but such imbalances are generally minor and manageable, and can be readily adjusted in practical feed formulations by supplementing additional ingredients (59, 64). Overall, our analysis primarily reflects the substitution potential at the protein level, while in practice it remains essential to ensure adequate provision of some micronutrients to sustain livestock productivity and health (65).

The use of black soldier fly larvae as animal feed to partially replace crop-based feed provides benefits beyond nutrient substitution, including enhanced disease resistance and reduced antibiotic usage (22, 66). Appropriate inclusion rates should be determined according to livestock species, production stage, and nutritional requirements (62, 67). In addition, black soldier fly bioconversion can alter the physicochemical properties and microbial composition

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of manure (68). Certain antibiotics, antibiotic resistance genes, and pathogens during larval treatment, potentially may be reduced through interactions among the feedstock environment, larval metabolism, and gut microbiota communities (68). Nevertheless, the underlying mechanisms remain insufficiently understood, and it remains unclear whether antibiotics, resistance genes, or pathogens may be transferred through larval biomass, frass, or associated microbial communities (69). These uncertainties highlight the need to consider biosafety risks during practical implementation, particularly because current production systems often prioritize larval biomass production while paying less attention to frass quality and its downstream use as fertilizer. Therefore, large-scale manure-based insect systems should be supported by standardized rearing protocols, improved production facilities, and whole-chain quality control systems covering feedstock pretreatment, bioconversion processes, larval products, and frass management, thereby ensuring both economic viability and environmental safety (70).

The decoupling of livestock and cropland systems arises from several factors. Rapid livestock expansion generates manure volumes that exceed the assimilative capacity of surrounding croplands (5). Moreover, livestock excrete manure continuously, whereas fertilizer application is seasonal, and manure requires composting to reduce pathogens and stabilize organic matter before field application (71, 72). Insect farming helps bridge these mismatches by converting manure into protein that can be dried and stored as a stable feed source, while simultaneously producing processed residues that function as safe organic fertilizers (20, 25). These requirements suggest that the manure-insect-feed system can reduce treatment costs for livestock farmers and generate stabilized residues in solid or semisolid form that are easier to store, transport, and market (73). Their relatively lower price enhances accessibility for crop farmers and supports the substitution for synthetic fertilizers (74). Together, these dynamics provide a dual benefit of sustainable alternative feed and organic fertilizers, thereby fostering closer recoupling of livestock and cropland systems.

Despite the substantial environmental and economic benefits estimated in this study, scaling manure-based insect farming faces practical barriers that require policy attention (12). Its implementation requires a supply chain that can coordinate manure collection, feedstock pretreatment, insect rearing, product processing, and the storage and transport of larvae and residues (70, 75, 76). These requirements depend on manure types and livestock production system (23, 24). For example, pig manure may require solid–liquid separation, whereas manure with high moisture content may need to be mixed with rice husk, sawdust, or other bulking materials to improve larval feeding and conversion efficiency (70, 77). Mixed feedstocks may also improve performance in some cases, such as combining dairy cattle and poultry manure at appropriate ratios (78). Large-scale rearing also requires control of temperature, light, contamination risks, and processing conditions to maintain larval performance, product safety, and residue quality. These requirements suggest that the manure-insect-feed systems should be tailored to local production systems and regional conditions (53). Adoption is therefore likely to be gradual and regionally uneven. It may begin where implementation conditions are most favorable, rather than where environmental benefits are greatest, including areas with stable manure supplies, short transport distances, established waste-management systems, manageable labor and energy costs, and supportive regulations (17, 75). In practice, supply chains should be built in different ways depending on farm size and spatial organization. Large livestock farms may be able to establish on-site insect-based manure treatment facilities. In regions dominated by small farms, centralized insect-processing hubs could be located according to farm distribution and proximity to cropland, thereby integrating manure collection and treatment, insect-based feed production, and the distribution or sale of frass fertilizer (79–81).

Regulation is another barrier to scaling manure-insect-feed systems (76). Current regulations differ across countries in feedstock eligibility, product safety standards, and traceability along the supply chain (82, 83). In the European Union, certain insect proteins are permitted in poultry and swine feeds, and properly processed insect frass can be used as an organic fertilizer. However, manure remains prohibited as a rearing feedstock for insects entering feed supply chains for food-producing animals (84). In contrast, several Asian countries, including China, have begun scaling insect production using organic waste streams, including manure (12, 82). This variation creates uncertainty over feedstock use, safety assessment, and market access for insect-derived feed products (85). Policymakers should establish feedstock eligibility criteria, safety thresholds for pathogens, heavy metals, and other contaminants, mandatory traceability systems, and certification pathways to reduce investment uncertainty and support market entry (82, 86). In many regions of Africa and Asia where insect farming is especially promising (34), public investment in supply chain infrastructure, technical training, and safety monitoring could help maximize environmental and food security benefits while ensuring product safety and traceability.

Uncertainty and Limitations. This study has several limitations. Owing to data and methodological constraints, we did not account for potential changes in feed supply–demand dynamics from substituting crop-based feed with insect-derived feed, nor the resulting effects on feed prices and crop production. Similarly, the potential impacts of replacing synthetic fertilizers with insect-derived residues on fertilizer prices were not considered. These omissions introduce uncertainty into the calculation of economic benefits, for which we conducted additional uncertainty analyses. We also did not fully assess potential effects of insect protein input on livestock growth performance, as current evidence remains inconclusive (17). It is important to emphasize that this study examined only the partial substitution of crop protein with insect protein, rather than a complete replacement of all feed components, thereby ensuring balanced nutrient inputs such as cellulose and amino acids. Moreover, biosafety implications, including potential reductions in antibiotics and pathogenic microorganisms, were not quantified due to limited data. Although manure-type-specific conversion parameters were incorporated, including feed conversion ratios, dry matter content, and insect protein yields, several real-world constraints remain outside the current modeling framework. These include supply-chain design, manure collection, storage and transport, infrastructure availability, regulatory conditions, and market demand. Therefore, the results presented here should be interpreted as the theoretical upper bound of potential benefits under the defined substitution scenario, rather than as benefits that can be fully achieved in the near term. Actual implementation depends on the supply-chain, regulatory, and market conditions discussed above, and is likely to proceed gradually and unevenly across regions. Future work should integrate these economic feedbacks, biosafety considerations, and nutritional interactions into global modeling frameworks to provide a more comprehensive assessment of the long-term implications of manure-based insect farming for sustainable food systems.

Methods

Data Sources. The FAOSTAT database (33) (https://www.fao.org/faostat/en/#data) was used to obtain livestock numbers and production, crop yields, producer prices for all crops, and crop trade matrix. Data on the distribution of all production systems

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for each livestock were derived from the Gridded Livestock of the World (GLW) database (35). Pigs are categorized into backyard, intermediate, and industrial systems. Chickens were classified as backyard, industrial layer, and industrial broiler systems. Dairy cattle, sheep, and goats all have grazing and mixed production systems. Beef cattle included grazing, mixed, and feedlot production systems. The Global Livestock Environmental Assessment Model 3.0 (GLEAM) (31) provided various crop feed input proportions for different livestock types, as well as the protein contents of crops. Data on major N inputs in cropland systems, including biological N fixation (BNF), irrigation, synthetic fertilizer, manure and N deposition, and crop harvest N were derived from Coupled Human and Natural Systems (CHANS) model (87). Parameters related to insect production, including feed conversion efficiency, protein content, dry matter content, emission factors, and land use, were collected from published literature, as detailed in *SI Appendix*, Table S1.

We used the GLEAM and CHANS to calculate N flows, N emissions, and GHG emissions during the livestock raising and feed production stages. Then, we used the trade matrix provided by FAO to calculate the feed imports between countries and determine the actual feed production in each country. Additionally, we determined the feed conversion efficiency, protein content, dry matter content, as well as the N and CH_4 emissions, and land requirements associated with insect production from livestock manure from literature (*SI Appendix*, Tables S1 and S2). Meanwhile, the manure available for insect production was restricted to the fraction remaining after field application, explicitly excluding manure deposited on grassland during grazing. Finally, based on the cost–benefit analysis, we calculated the feasibility of insect production.

N Budget for Whole-Chain Livestock Production. N budget for livestock systems was estimated using the GLEAM framework. GLEAM 3.0 model, developed by FAO, quantifies N and GHG emissions throughout the whole livestock production chain within a life cycle assessment (LCA) framework. The model covers emissions from crop and pasture production, manure management systems, feed processing and transport, and livestock raising and management and so on. A detailed description of GLEAM can be found at https://www.fao.org/gleam/en/. In this study, we quantified N budget and GHG emissions from feed production and manure management by applying the calculation methods and parameters provided by GLEAM, combining with production data from FAOSTAT and production-system distributions from GLW database.

First, the feed demand for each livestock species was calculated based on their protein production and feed conversion ratios derived from Mottet et al. (7). Feed N input was then estimated by feed ratio data and feed N content specific to each livestock production systems as defined in GLEAM. The N input for livestock systems is defined in Eq. **1**.

$$N_{input,i} = N_{grain,\,i} + N_{byproduct,\,i} + N_{crop\ residue,\,i} + N_{grass,\,i} + N_{swill,\,i} + N_{other\ feed,\,i}, \quad [1]$$

where $N_{grain,\,i}$, $N_{byproduct,\,i}$, $N_{crop\ residue,\,i}$, $N_{grass\ feed,\,i}$, $N_{swill,\,i}$, and $N_{other\ feed,\,i}$ are grain, crop byproduct, crop residue, grass, swill, and other feed N intake for livestock i, respectively.

Livestock N emissions were calculated using methodologies provided in GLEAM 3.0 and IPCC guidelines, incorporating emission factors and MMS (manure management systems). Detailed emission factors and calculation procedures are available in the GLEAM 3.0 model. In this study, the livestock N emissions mainly focused on the emissions during manure storage and did not include emissions during the housing storage, as insect production mainly comes from the manure storage stage.

The N budget for cropland systems in feed production stage was quantified using CHANS N cycling model (32, 87). We used CHANS model to calculate the synthetic N and manure N inputs in cropland systems. N inputs for cropland systems include crop biological N fixation (N_{CBNF}), synthetic fertilizer N ($N_{fertilizer}$), and manure N (N_{manure}), irrigation N ($N_{irrigation}$), and atmospheric N deposition ($N_{deposition}$). The nitrogen budget for feed crops was estimated using the N use efficiency (NUE), as shown in Eq. **2**. N emissions from feed crop production were calculated using emission factors and methodologies from the feed production stage in GLEAM.

$$NUE_{crop} = \frac{N_{crop\ products}}{N_{CBNF} + N_{fertilizer} + N_{manure} + N_{irrigation} + N_{deposition}}, \quad [2]$$

where $N_{crop\ products}$ is cropland harvest products.

GHG Emissions for Whole-Chain Livestock Production. Livestock C emissions include methane from manure management for all livestock production systems. Manure CH_4 emissions were calculated based on GLEAM and IPCC 2019 (88). Carbon emissions from feed production systems were calculated based on GLEAM, encompassing CO_2 emissions from field operations, feed processing, blending and transportation, as well as noncrop feed production (such as fish meal and limestone), and CH_4 emissions from rice cultivation. Based on GLEAM, emission for leaves and swill were assumed to be null.

For land-use change (LUC) emissions, we first estimated the cropland area required for feed production under the BAU and the Insect scenarios using Eq. **3** (7). The reduction in cropland demand, derived from the difference between the two scenarios, was then combined with the land area required for insect farming to calculate the net change in cropland use under the Insect scenario. This net change was multiplied by the land-use change emission factor (*LUC*) (89) to quantify the potential reduction in LUC-related emissions, according to Eq. **4**.

$$Land_c = \frac{DM_c}{DMYG_c \times FUE_c} \times \frac{EFA_c}{MFA_c}, \quad [3]$$

where $Land_c$ is the cropland area required for feed c; DM_c is the dry matter requirement of feed c; $DMYG_c$ is the dry matter yield of feed c (kg ha^{-1}), calculated from crop yield according to the GLEAM; FUE_c is feed use efficiency from GLEAM; and MFA_c and EFA_c are the mass and economic fractions, respectively, derived from GLEAM.

$$LUC_{Insect} = \left(Land_{Insect} - Land_{BAU} + Land_{insect\ production} \times LUC_{EF}\right), \quad [4]$$

where $Land_{Insect}$ and $Land_{BAU}$ are the areas of cropland required for crop feed production under Insect and BAU scenarios, respectively, and $Land_{insect\ production}$ is required land use parameter from insect production, derived from literature (12). LUC_{EF} is the emission factor for land use change, expressed as annual GHG emissions released from forest to cropland, in t CO_2·ha^{-1} y^{-1}. The LUC_{EF} value incorporated the long-term effects and was averaged over the assessment period to an average value for each year (89). Given the considerable uncertainty associated with GHG fluxes from deforestation and reforestation, this was explicitly addressed in the *Uncertainty Analysis*.

To estimate total GHG emissions, emissions of N_2O and CH_4 were expressed in terms of carbon dioxide equivalents (CO_2-e) using global warming potential (GWP) values from the IPCC 2019 assessment, based on a 100-y time horizon. The GWP values applied were 273 for N_2O and 27 for CH_4 emission, respectively (88).

Insect Production. The insect chosen for this study is the black soldier fly (*Hermetia illucens*). Compared to other insects such as the housefly, the black soldier fly has a higher maturity and stability in production, as well as a relatively higher feed conversion efficiency and more flexible adaptability to diverse substrates and environmental conditions (25). Black soldier fly production has been already well-established and deployed in various regions globally (13, 30). The calculation of insect protein ($Protein_{insect}$) is shown in Eq. **5** (12).

$$Protein_{insect} = \sum \frac{Mass_{manure,\,i}}{FCR_i} \times Protein_i \times DM_i \quad [5]$$

where $Mass_{manure,\,i}$ is the mass of manure from livestock species i, FCR_i is the feed conversion ratio of producing insect from livestock i manure, $Protein_i$ represents the protein content of black soldier fly production from livestock i manure, and DM_i is the dry matter content of black soldier fly producer from livestock i manure. The values and the literature sources of $Protein_i$, FCR_i, and DM_i for different livestock are listed in *SI Appendix*, Table S1.

The Insect scenario was developed under the assumption of a "manure-to-field priority," whereby insects are produced using the residual fraction of manure not returned directly to cropland. Manure excreted directly on pasture from grazing systems was excluded. Manure from backyard systems was considered, given its potential for centralized collection and management, particularly in regions such as Africa and Asia where it represents a substantial share. In terms of substitution, insect protein was assumed to replace crop-based feed, with priority given to high-protein sources such as soybean and other oilseeds, followed by cereals and other feed crops, in order to maintain nutrient stability and support livestock growth.

Emission factors for insect production were primarily derived from the literature (All literature are presented in *SI Appendix*, Table S2). Insect-based manure

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treatment can be considered as the manure management strategy, influencing NH_3, N_2O, and CH_4 emissions from manure storage. Based on a synthesis of published data (*SI Appendix*, Table S2), we determined the emission factors associated with insect treatment of manure. These parameters were applied to estimate emissions before and after insect processing, as well as the resulting mitigation potential, using Eq. **6**.

$$\nabla EM_j = RR_j \times EM_{j,\mathrm{BAU}}, \quad [6]$$

where ∇EM_j is the reduced emission indicator, including NH_3, N_2O and CH_4 emission, RR_j is the response ratio of emission indicator j under the Insect scenario (*SI Appendix*, Table S2), and $EM_{j,\mathrm{BAU}}$ is the emission of indicator j under *BAU* scenario.

Trade Flows. Feed trade was estimated by comparing feed production quantity (P_c) from the Food Balance Sheets provided by FAOSTAT with feed demand quantity (D_c) for each crop in country level. When demand was less than or equal to production ($D_{c,k} < P_{c,k}$), the feed crop j was considered domestically self-sufficient, and no imports were assumed. When demand exceeded production ($D_{c,k} > P_{c,k}$), the country was assumed to import the difference ($D_{c,k} - P_{c,k}$), which represents the feed crop import quantity. The calculation is given by ref. 90:

$$Feed_import_{c,k} = \max\left(D_{c,k} - P_{c,k}, 0\right), \quad [7]$$

where $Feed_import_{c,k}$ denotes the import volume of feed crop j in country k, $D_{c,k}$ is the feed demand, and $P_{c,k}$ is the domestic production.

To further allocate international feed trade flows, import data for crops and by products were extracted from FAOSTAT trade matrix. Because the feed-use portion of traded commodities is not explicitly reported, we first calculated the proportion of imports from each exporting country based on total crop trade volumes. These proportions were then applied to the estimated national feed crop import volume to allocate the feed crop imports among source countries. This approach allowed for the quantification of feed imports by origin as well as the corresponding export volumes from each producing country.

Implementation Cost Valuation. To assess the cost-effectiveness of insect farming, we calculated its implementation cost. The costs mainly include those associated with young larvae, labor, depreciation, water, and energy. Energy costs vary significantly between regions. Energy usage includes the operation of insect processing, cleaning, and other routine activities, as well as energy used for heating and maintaining lighting, as insect production requires specific lighting and temperature conditions. In regions with lower temperatures, energy costs for heating and maintaining lighting are higher. The cost calculation method in this study is based on Bai et al. (12), which differentiated the energy usage between southern China, located in a low-latitude area, and northern China, located in a high-latitude area. We calculated the specific implementation costs for each country based on their temperature conditions, as well as the prices for labor, water, and energy (natural gas price, electricity price). The specific young larvae costs for each country are adjusted based on the operational costs of labor, water, and energy. Unit depreciation costs are taken from the literature (12). The detailed costs for producing insect protein per unit for each country, including young larvae, labor, depreciation, water, and energy, are shown in *SI Appendix*, Table S3, which is derived from references.

$$\mathrm{Cost_{Insect}} = \mathrm{Cost_{labor}} + \mathrm{Cost_{water}} + \mathrm{Cost_{larvae}} + \mathrm{Cost_{depreciation}} + \mathrm{Cost_{energy}} \quad [8]$$

where $\mathrm{Cost_{labor}}$ represents the labor cost, derived from average monthly wages and actual working hours reported in the international labor organization Department of Statistics (ILOSTAT, https://ilostat.ilo.org/); $\mathrm{Cost_{water}}$ is the cost of freshwater consumption and wastewater services in insect cultivation, obtained from the World Bank International Benchmarking Network for Water and Sanitation Tariffs database (IBNET, https://newibnet.org/about-ibnet); $\mathrm{Cost_{larvae}}$ and $\mathrm{Cost_{depreciation}}$ denote the cost of starter larvae and the annualized depreciation of fixed assets (calculated using a 10-y straight-line method), respectively; and $\mathrm{Cost_{energy}}$ accounts for electricity and natural gas used for heating, ventilation, and lighting, with country-specific tariffs taken from the International Energy Agency's (IEA) World Energy Prices (WEP) dataset (https://www.iea.org/data-and-statistics/data-product/energy-prices).

All price data were standardized to constant 2020 USD. Local currency values were first deflated to 2020 levels using national consumer price indices (CPI) and subsequently converted to USD using market exchange rates or purchasing power parity (PPP) data obtained from the World Bank's World Development Indicators (WDI) database. When data were missing for industrial users, values from commercial or residential users were applied, with ±30% variation introduced in the uncertainty analysis. Conversion parameters, including feed conversion ratios, dry matter content, and protein fractions, followed the ranges reported in *SI Appendix*, Tables S5 and S6 to ensure consistency between cost and mitigation estimates.

Societal Benefit Assessment. These mainly include the crop feed substitution benefits derived from replacing intensive feed costs with insect production, the fertilizer substitution benefits from insect production, and the ecosystem ($E_{benefit}$), health ($H_{benefit}$), and climate ($C_{benefit}$) benefits from reduced nitrogen and greenhouse gas emissions. Ecosystem, health, and climate benefits primarily arise from avoided damage costs due to emissions reductions. The fertilizer substitution benefit ($F_{benefit}$) is calculated by multiplying the fertilizer amount saved by the unit fertilizer price. The crop feed substitution cost ($CR_{benefit}$) is derived by multiplying the crop feed amount saved by the producer prices of various crops (from FAOSTAT). The total societal benefits were calculated according to Eq. **9**.

$$\mathrm{SO_{benefit}} = \mathrm{E_{benefit}} + \mathrm{H_{benefit}} + \mathrm{C_{benefit}} + \mathrm{F_{benefit}} + \mathrm{CR_{benefit}}. \quad [9]$$

The unit N damage cost to the ecosystem was sourced from the European Nitrogen Assessment (91). Subsequently, we calculated BAU national-specific unit ecosystem damage cost of acidification and eutrophication, that is, unit ecosystem benefit ($Ecost_{\mathrm{BAU},j}$) by adjusting according to local willingness to pay (WTP) and purchasing power parity (PPP), derived from the literature (87, 92). Then, we calculated the national-specific unit ecosystem damage costs under Insect scenario ($Ecost_{\mathrm{Insect,NH_3}}$) based on Eq. **10**, since changes in NH_3 emission intensity under this scenario also alter the corresponding unit ecosystem damage costs. Finally, the avoided ecosystem damage cost by NH_3 emission mitigation compared to BAU scenario is regarded as the ecosystem benefits, as estimated in Eq. **11**. The WTP-based approaches could carry inherent limitations and uncertainties due to their dependence on socioeconomic context, the choice of valuation method (such as contingent valuation or revealed preference), and cultural attitudes toward environmental and health risks. Accordingly, the values reported here should not be interpreted as precise national estimates, but rather as indicative measures to facilitate comparative assessment of where mitigation may deliver relatively greater social benefits.

$$Ecost_{\mathrm{Insect},j} = Ecost_{\mathrm{BAU},j} \times \frac{Density_{\mathrm{Insect},j}}{Density_{\mathrm{BAU},j}}, \quad [10]$$

$$EH_{\mathrm{benefit,Insect},j} = E_{\mathrm{BAU},j} \times Ecost_{\mathrm{BAU},j} - E_{\mathrm{Insect},j} \times Ecost_{\mathrm{Insect},j}, \quad [11]$$

where $Density_{\mathrm{BAU},j}$ and $Density_{Insect,j}$ are emission j (including NH_3 and N_2O) density in kg km^{-2} under BAU and Insect scenario, respectively, $E_{\mathrm{BAU},j}$ and $E_{\mathrm{Insect},j}$ are emissions j (including NH_3 and N_2O) under BAU and Insect scenario, respectively, and $Ecost_{\mathrm{BAU},j}$ is the BAU ecosystem damage costs (*SI Appendix*, Table S4) of emissions j, in US $ per kg N.

The human health benefit (HH_{benefit}) was defined as the reduction in premature mortality with lower $PM_{2.5}$ exposure resulting from N emission abatement (92). We derived the BAU national-specific unit health damage costs ($Hcost_{\mathrm{BAU,NH_3}}$) of NH_3 emission based on the N-share methodology (92), which links the economic cost of avoided premature mortality attributable to NH_3 emissions with key factors, including $PM_{2.5}$ concentration, population density, per capita GDP (PGDP), and total years of life lost. Then, we calculated the national-specific unit health damage costs under Insect scenario ($Hcost_{\mathrm{Insect,NH_3}}$) based on Eq. **12**. Finally, the avoided health damage cost by NH_3 emission mitigation compared to BAU scenario is regarded as the health benefits, as estimated in Eq. **13**. These estimated health benefits should be interpreted as indicative magnitudes rather than exact valuations, owing to uncertainties in exposure-relationships, modeled $PM_{2.5}$ chemical fractions, and the use of PGDP-weighted monetization approaches.

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$$Hcost_{\mathrm{Insect,NH_3}} = Hcost_{\mathrm{BAU,NH_3}} \times \frac{Density_{\mathrm{Insect,NH_3}}}{Density_{\mathrm{BAU,NH_3}}}, \quad [12]$$

$$HH_{\mathrm{benefit,Insect,NH_3}} = E_{\mathrm{BAU,NH_3}} \times Hcost_{\mathrm{BAU,NH_3}} - E_{\mathrm{Insect,NH_3}} \times Hcost_{\mathrm{Insect,NH_3}}, \quad [13]$$

where $Density_{\mathrm{BAU,NH_3}}$ and $Density_{\mathrm{Insect,NH_3}}$ are NH_3 emission density in kg NH_3 km^{-2} under BAU and Insect scenario, respectively, $E_{\mathrm{BAU,NH_3}}$ and $E_{\mathrm{Insect,NH_3}}$ are NH_3 emission under BAU and Insect scenario, respectively, and $Hcost_{\mathrm{BAU,NH_3}}$ is the BAU health damage costs (*SI Appendix*, Table S4) of NH_3 emission, in US $ per kg N.

For the climate change mitigation benefits, we integrated avoided social cost of GHG (N_2O, CH_4, CO_2) emissions and the avoided cooling impact due to NH_3 emission abatement ($CI_{\mathrm{BAU,NH_3}}$ is negative), calculating as Eqs. **14**–**16**. This assessment did not quantify the potential greenhouse gas (GHG) cobenefits or trade-offs associated with NH_3 mitigation, as available data and current modeling frameworks remain insufficient.

$$CI_{\mathrm{Insect,NH_3}} = CI_{\mathrm{BAU,NH_3}} \times \frac{Density_{\mathrm{Insect,NH_3}}}{Density_{\mathrm{BAU,NH_3}}}, \quad [14]$$

$$CN_{\mathrm{benefit,Insect,NH_3}} = \mathrm{E}_{\mathrm{BAU,NH_3}} \times CI_{\mathrm{BAU,NH_3}} - \mathrm{E}_{\mathrm{Insect,NH_3}} \times CI_{\mathrm{Insect,NH_3}}, \quad [15]$$

$$\mathrm{C}_{\mathrm{benefit}} = \nabla E_{N_2O} \times SCC_{\mathrm{GHG},N_2O} + \nabla E_{CH_4} \times SCC_{\mathrm{GHG},CH_4} + \nabla E_{CO_2} \times SCC_{\mathrm{GHG},CO_2} + CN_{\mathrm{benefit,Insect,NH_3}}, \quad [16]$$

where SCC_{GHG,N_2O}, SCC_{GHG,CH_4}, and SCC_{GHG,CO_2} are the unit climate benefit of decreased N_2O, CH_4, and CO_2 emissions, from the literature (36, 93).

Uncertainty Analysis. We conducted 10,000 Monte Carlo simulations using MATLAB 2021b to assess the uncertainties of N and GHG emissions, and crop feed, fertilizer, and cropland use. The uncertainties of implementation costs and benefits under Insect scenario were also evaluated. The CI for all results was set at 95%, and the coefficient of variation (CV, %) values for the data and parameters can be found in *SI Appendix*, Tables S5 and S6. Given methodological and data limitations, our estimates should not be interpreted as precise values, but rather as indicative ranges that reflect the relative mitigation potential across regions.

Data, Materials, and Software Availability. Results are also available at Zenodo (94). Spatial analyses were performed using standard built-in tools in ArcGIS Pro 3.4. No proprietary or restricted datasets were used in this study. All other data are included in the article and *SI Appendix*.

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