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# Mitigation of Multiple Environmental Footprints for China's Pig Production Using Different Land Use Strategies

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**ABSTRACT**

Pig production contributes considerably to land use, greenhouse gas (GHG), and reactive nitrogen (Nr) emissions. Land use strategies were widely proposed, but the spillover effects on biological flow are rarely explored. Here we simultaneously assessed carbon (C), nitrogen (N), and cropland footprints of China's pig production at the provincial scale in 2017. The environmental impacts of land use strategies were further evaluated. Results show that one kg live-weight pig production generated an average of 1.9 kg $CO_2$-eq and 59 g Nr emissions, occupying 3.5 $m^2$ cropland, with large regional variations. A large reduction in GHG (58-64%) and Nr (12-14%) losses and occupied cropland (10-11%) could be achieved simultaneously if combined strategies of intensive crop production, improved feed-protein utilization efficiency, and feeding co-products were implemented. However, adopting a single strategy may have environmental side-effects. Reallocating cropland that pigs used for feed to plant food alternatives would enhance human-edible energy (3-20 times) and protein delivery (1-5 times) and reduce C and N footprints, except for rice and vegetables. Reallocating cropland to beef and milk production would decrease energy and protein supply. Therefore, a proper combination of land use strategies is essential to alleviate land use changes and nutrient emissions without sacrificing food supply.

## INTRODUCTION

The livestock sector is a key consumer of natural resources (e.g., land, water, energy) and also a major contributor to climate change, reactive nitrogen (Nr) pollution, and land use change.[1] Global livestock production is estimated to contribute 13-18% of the total anthropogenic greenhouse gas (GHG) emissions,[2] 40% of the global anthropogenic ammonia ($NH_3$),[3] and occupy nearly 70% of global cropland area for producing animal feed.[4] The pork sector contributed to 48% of the worldwide meat supply in 2017.[5] China, as the world's largest producer, supplies 46% of the world's pork production in 2017, via a production growth rate of 1% per year over the past three decades[5] and a rapid transition towards intensive production.[6] But this rapid expansion in pig production in China has come at the expense of the environment, and the development of more sustainable production systems, therefore, needs to be prioritized.

Among these impacts, land use is a central concern as it is a major driver of environmental change at local and global scales, with important impacts on biogeochemical cycling, ecosystem function, and GHG emissions.[7, 8] About 75% of the Chinese pigs are currently raised in medium and large-scale intensive farms, relative 22% in 2000.[9] Intensification improves productivity, primarily via an increased proportion of grain feed in pig diets.[6] Producing human-edible grain feed crops in China relies on the overuse of synthetic N fertilizer, which may increase the N emission intensity of pig production. The increasing cultivation of feed crops is also associated with land use change domestically and overseas.[10] China has gradually increased the import of high-protein feed, such as soybean, which contributes to deforestation and associated GHG emissions in soybean-exporting countries (i.e., Brazil and Argentina).[11] Changes in feed composition can also affect nutrient excretions and emissions from the entire manure management chain.[6] Environmental assessment of the whole pork supply chain is therefore essential, and has been previously conducted at the farm, country (e.g., European countries, Brazil, and China), [12-15] and global level.[16] However, Country-level evaluations often overlook variations among regions. There has been a lack of environmental assessment at sub-national and provincial scales, limiting the opportunity for governments to develop region-specific policies.

To reduce ecological burdens related to land use, three groups of efficient land use strategies in agricultural systems are usually proposed: 1) to close yield gaps by increasing land productivity, which is in line with intensive crop production;[10, 17] 2) to improve feed-protein utilization efficiency (e.g., lowering dietary crude protein content) at animal level;[10, 18] and 3) to use ecological leftovers, such as co-products of food processing and food waste, to reduce cropland used for feed production.[10, 19] However, these previous studies often focused on either a specific strategy only or a particular environmental pollutant (mainly GHG emissions), with limited understanding on changes in land use. The environmental consequences of these strategies can be interlinked. Assessing the broader synergistic and antagonistic impacts (e.g., GHG emissions, Nr emissions, and land use) of a full range of land use strategies is therefore needed to better inform future policy.[20]

The ‘saved’ cropland could be reallocated to produce more efficient alternative food items (food that require less environmental resource per unit protein or energy) or conserved, negating further environmental degradation. The sacrifice of unchosen alternatives is the so-called “opportunity cost.” [7] One recent study indicated the possible potential of resource (fertilizer, land, and water) saving when repurposing cropland used for beef feed to other plant- and animal-based alternatives in the United States (US).[8] Redirection of intensively utilized cropland to natural forest may increase biodiversity and soil carbon sequestration.[21] If some cropland shifted from grain feed to human-edible food alternatives in China, what would be the key implications for food supply and environmental impacts? There is as yet little information about these implications in China, in particular when GHG emissions, Nr emissions, and food (energy and protein) supply are considered simultaneously.

The objectives of this study were to i) assess carbon (C), nitrogen (N), and cropland footprints of pig production at the provincial scale in China for the year 2017 (the latest year for which most activity data from statistics are available) using a life cycle assessment (LCA) approach, and to ii) evaluate how alternative land use scenarios affect these environmental footprints. Further, the implications of reallocation of 'saved' cropland to produce plant- and animal-based food alternatives were also explored.

## MATERIALS AND METHODS

**System boundary of life cycle assessment.** We used an LCA approach to evaluate the C, N, and cropland footprints of China's pig production system in 2017. The system boundary and main components of this life cycle, in terms of ancillary inputs production, feed production, feed processing and transportation, manure storage (indoor and outdoor) and treatment, and manure application to cropland, are presented in Figure 1 & S2. The functional unit (FU) was defined as one kg of pig live weight (LW). Carbon, nitrogen, and cropland footprints per FU were expressed as kg $CO_2$-eq $FU^{-1}$, kg Nr $FU^{-1}$, and $m^2$ $FU^{-1}$, respectively. Three main pig breeding systems were categorized in this study because of the differences in feed composition and manure management used in, i) smallholder farm (< 50 slaughtered pigs per farm), ii) medium farm (50-3000 slaughtered pigs per farm), and iii) industrial farm (>3000 slaughtered pigs per farm).[6, 22] Information on herd and management characteristics of these three breeding systems is provided in Table S1-2. The ratio of the three breeding systems in each province of China in 2017 is presented in Figure S1.

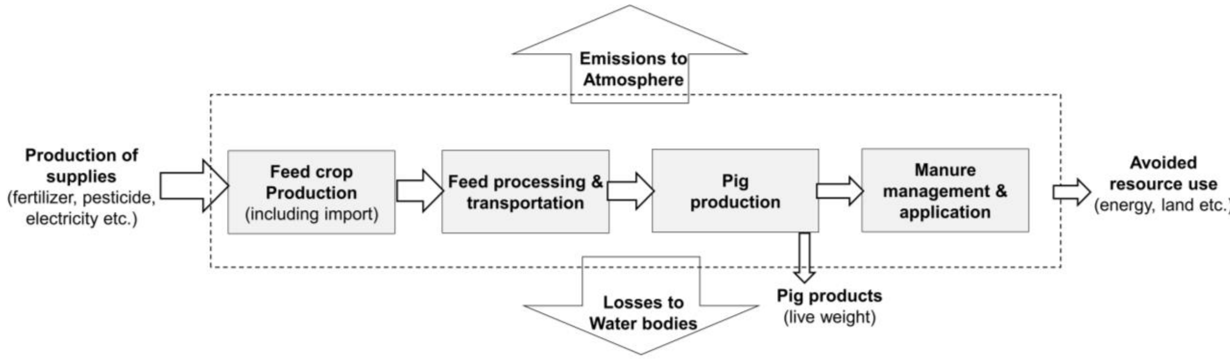


**Figure 1.** An illustration of simplified life-cycle system boundary of pig production in China, with the main processes for feed crop production, feed processing and transportation, pig production, and manure management and application

**Carbon and Nitrogen Footprint Analysis.** The C footprint includes the emission sources of GHG emissions, i.e., nitrous oxide ($N_2O$), methane ($CH_4$), and/or carbon dioxide ($CO_2$), from the production of chemical fertilizer, seed, agricultural film, and pesticide, application of chemical fertilizers and manure, feed processing and transportation, energy use, enteric fermentation, and manure management. Indirect $N_2O$ emissions resulting from the $NH_3$ volatilization, N leaching, and runoff from cropland were also included. $CO_2$ emissions from manure management were excluded. Emissions of $CH_4$ and $N_2O$ were converted to $CO_2$-eq, using the global warming potentials (GWP) of $CH_4$ and $N_2O$ of 25 and 298 times of $CO_2$, respectively.[23] The N footprint includes emissions of Nr (all nitrogen species except $N_2$) losses,[24] in terms of atmospheric emissions of $NH_3$, $N_2O$, and nitrogen oxides ($NO_x$), as well as N lost to water bodies. The emission sources of $NH_3$, $N_2O$, and $NO_x$ were almost the same as that of GHG emissions. Nitrogen lost to water bodies mainly included leaching, runoff, and erosion from cropland and manure management.[25, 26] The C and N footprints were quantified based on process-specific emission factors so that the effects of changes in specific processes could be addressed from the life cycle aspect. The calculations, parameters, activity data employed in our analysis, as well as the sensitivity analysis were detailed in Supporting Information (SI). The main calculation procedures are summarized below.

*Feed crop production:* Emissions embodied in agricultural inputs (e.g., chemical fertilizer and pesticide) were quantified as a function of emission intensity per unit product (Table S4-6) and the consumption of products applied for corresponding feed crops. Feed sources include both domestic production and imported production (i.e., 86% of soybean was imported).[5] Emissions of soil $N_2O$ and $CH_4$ (paddy rice), as well as $CO_2$ derived from urea application, were quantified according to the IPCC Tier 2 approach,[23] N losses from crop production in China were quantified using the mass flow model, NUFER (NUtrient flows in Food chains, Environment and Resources use).[25, 26] In China, cropland used for domestic feed production has been under cultivation for hundreds of years; therefore, these emissions associated with direct land use change were excluded.[23] GHG emissions of land use change from soybean production in Central West of Brazil (0.96 kg $CO_2$-eq $kg^{-1}$ soybean) and Argentina (0.72 kg $CO_2$-eq $kg^{-1}$ soybean) were accounted for analysis because of deforestation or the conversion of pasture and shrubland to cropland.[11, 27-29]

*Feed processing and transportation:* Emissions (e.g., $CO_2$) embodied in energy consumption during feed processing were quantified according to the emission intensity for a range of energy sources and the amount of energy used by feed types. Emissions from non-plant feed ingredients (e.g., minerals, fish meal, whey powder, amino acids) production were also considered. Emissions from national and international transport (truck and ship) of feed were calculated as a function of transportation distance and emission coefficients per product.

*Direct Energy use:* Farm energy consumption varied between the three pig breeding systems, and data were extracted from the National Data Compilation of Revenue and Cost of Agricultural Products.[30] Calculation of emissions from on-farm energy consumption adopted the same method for calculating the emissions embodied in energy consumption during feed processing.

*Pig production and manure management:* Manure management practices include indoor and outdoor storage of manure, manure treatment, and manure application to cropland. The leading treatment technologies in China include solid-liquid separation, anaerobic digestion, and composting.[31] The proportion of housing (slatted, solid, or litter-based floor) and storage types and the adaption of treatment techniques varied among different breeding systems (Table S7). Nr emissions from manure were analyzed according to manure N flows and process-specific emission factors (Table S8).[25, 26] Quantification of $N_2O$ and $CH_4$ emissions (including enteric fermentation) was based on IPCC guidelines.[23] The model calculations have been published previously[32] and are briefly described in SI.

**Cropland Footprint Analysis.** The cropland footprint here refers to the cropland occupation areas (CLO; measured in $m^2$ per year), which indicates the use of land cover for a certain period and is calculated by multiplying the occupied area by time.[33] In our study, we calculated the maximum land area occupied by the feed crops during the cultivation phase within a year; that is to say, we did not consider cultivation methods like intercropping or rotation. For feed ingredients as co-products (e.g., soybean meal), their CLO needs to be multiplied by the corresponding allocation coefficients.

**Allocation method.** Economic allocation is the most commonly used allocation method in environmental LCAs of livestock products,[34] and was, therefore, used in our study. Distribution was carried out twice in this study. The first allocation was only used for wheat production to partition environmental impacts associated with crop cultivation between wheat grain and wheat straw.[35] The second allocation was applied for maize, wheat, soybean, rice, peanut, and rapeseed, to divide environmental impacts related to crop cultivation, processing, and land use between feed ingredients and corresponding co-products. The allocation coefficients and mass ratio of different products are shown in SI (Table S3).

**Data Sources.** Data related to domestic feed crop production (i.e., crop yield and agricultural inputs) and pig production data were obtained from Chinese official statistics.[9, 30] Data concerning imported soybean cultivation was based on previous studies for exporting countries.[11, 28, 36] Literature data were used for the emission parameters of agricultural inputs production when available; otherwise, values from the Ecoinvent 3 database were used.[35] GHG and Nr emissions factors during feed crop production and manure management were derived from IPCC (2006), and the NUFER model.[23, 25, 26] Feed formulas (Table S10) and manure management data (i.e., housing and storage types, and the adaption of treatment techniques, Table S7) were obtained from a survey conducted for 531 medium farms and 219 industrial farms across China, a national study concentrated on manure treatment (> 42,000 farms), and relevant literature.[6, 37, 38]

**Defining Scenarios.** Effects of efficient land use strategies on the C, N, and cropland footprints of pig production in China were examined through scenario analysis. For the C footprint, the "opportunity cost of land (soil C sequestered by conversion of cropland into natural forest)" were also included in these scenarios (see SI for details).[39, 40] The assessment focuses on exploring the trade-offs and co-benefits between environmental impacts when strategies for alleviating feed-food competition are implemented. Seven scenarios with alternative measures were compared with the reference system, China's pig production system in 2017. The assessment was conducted at the provincial level. Scenarios include 1) intensive crop production with increased crop yields (S1); 2) intensive pig production with improved feed-protein utilization efficiency (S2); 3) feeding pigs on ecological leftovers, such as co-products of food processing (S3-4); and combinations of the strategies mentioned above (S5-7). Feed formulas of these scenarios are detailed in SI (Table S11).

*Intensive crop production (S1):* A set of Integrated Soil-Crop System Management (ISSM) practices were assumed to be adopted by all farmers growing the three main kinds of cereal (maize, wheat, and rice) in China. ISSM was designed based on the local environment, drawing upon appropriate crop varieties, sowing dates, densities, and advanced nutrient management.[41] The potential increase in yields (20.4-24.4%, 15.8-21.9% and 16.9-23.7% of increases for maize, wheat, and rice) and decrease in N fertilizer inputs (6.3-22.7%, 9.2-18.9%, and 3.2-16.0% of decrease for maize, wheat, and rice) for each crop at provincial levels, resulting from ISSM practices, were derived from > 13,000 field-experiment sites across China (Table S9).[41] Apart from ISSM implementation, all other practices are assumed to be unchanged in this scenario.

*Intensive pig production (S2):* Improvement in feed-protein utilization efficiency was achieved through a low crude protein (CP) feeding strategy with the inclusion of specific amino acids (AA). It aims to lower the CP level to better match pig growth demand and not impair growth performance.[42] Since soybean meal was the most commonly used protein feed in pig diets, reducing CP level in diet could help relieve pressure on imported soybean. This low CP strategy (lowering CP content by 1-2% in absolute value) was assumed to be only adopted by medium and industrial farms, in part due to the low CP content observed on smallholder farms (Table S11). This assumption is also in line with the scope of national action, released by China's Ministry of Agriculture in 2017, for promoting this strategy.

*Ecological leftovers (S3-4):* Here, the emphasis was on increasing the consumption of co-products from food processing, building on S2 (lowering dietary CP level). Assumptions include using maize dried distillers grains with solubles (DDGS) (S3) and a combination of oil seed meal (peanut and rapeseed meal) (S4) to reduce the amount of soybean meal in pig diets, which also only aims at medium and industrial farms (see feed formulas in Table S11). These low economic value co-products with low available energy density were chosen as representatives of "ecological leftovers" to reduce resource use and the cost of pig diets.[10, 43] DDGS (a co-product of maize ethanol production), with low starch, high protein, and high digestible fiber content, is generally considered to be a reasonable choice to replace soybean meal in pig and poultry feed.[44] Domestic oil seed meals were considered here because it is assumed to reduce transport distance and associated land use change of imported high-protein feed, such as soybean.

*Combination of measures (S5-7):* Three combinations of measures were designed to explore the upper boundaries of the potential impacts when implementing these proposed strategies. The combined scenarios S5-7 represent intensive crop production (S1) in combination with S2 (=S5), S3 (=S6), and S4 (=S7), respectively.

**Opportunity cost.** We further quantified opportunity magnitudes of reallocating the 'saved' land resulted from combinations of land use strategies (scenarios S5-7) to produce food alternatives, focusing on C and N footprints and food (energy and protein) delivery. For this opportunity cost analysis, we assumed that there are opportunities for using the saved land to produce food alternatives other than pig feed (see Figure S6). Parameters regarding these footprints for individual food alternatives were sourced from global-scale analyses (see Table S12). The adoption of global averages in the present study was because of the lack of country-specific data for all these food alternatives and an assumption of the food supply in a globalized context.[45, 46] To allow comparisons between indicators (C and N footprints and food delivery), we express these savings as changes in intake per capita on a weekly basis and percentages of the amount currently used for the production of pig feed on the reallocated cropland (see SI for details).

## RESULTS

**Carbon Footprint.** The total GHG emissions from China's pig production were 152 teragrams (Tg) $CO_2$-eq $yr^{-1}$ in 2017, with 89% being from domestic sources and 64% from medium-size farms (Figure 2a). Feed crop production and manure management accounted for 46% and 23% of total emissions, respectively. GHG emissions intensity varied nearly three-fold at the regional scale (from 1.2 kg $CO_2$-eq $FU^{-1}$ in Guizhou to 3.0 kg $CO_2$-eq $FU^{-1}$ in Xinjiang) (Figure S3a). The regions with relatively low GHG emission intensity, e.g., Sichuan, Yunnan, and Guizhou (in Southwest China), have a larger share of smallholder farms (0.6, 2.4, and 2.2 kg $CO_2$-eq $FU^{-1}$ for the smallholder, medium, and industrial farms, respectively; Figure S4a). The contribution of feed supply to the total C footprint in smallholder farms (28%) was less than that in medium (76%) and industrial (76%) farms. By-products and food waste were dominant in pig diets of smallholder farms (Table S10), which are often considered relatively lower environmental impacts than grain-based feed.[47]

**Nitrogen Footprint.** The total Nr losses from China's pig production were 4.9 Tg Nr $yr^{-1}$ in 2017, with 96% from domestic sources and 65% from medium-scale farms (Figure 2b). Manure management contributed 57% of total emissions, followed by feed production (33%). Nr emission intensity ranged from 38 g Nr $FU^{-1}$ in Qinghai to 77 g Nr $FU^{-1}$ in Jiangsu (Figure S3b). Most regions with relatively high emission intensity were in Eastern and Central China. Smallholder farms (20 g Nr $FU^{-1}$) had, on average, a lower Nr emission intensity than medium (76 g Nr $FU^{-1}$) and industrial (64 g Nr $FU^{-1}$) size farms (Figure S4b). Manure management contributed to higher Nr emissions in smallholder farms (78%) than in medium (53%) and industrial (50%) farms. Nitrogen flows (kg N per FU) in the aforementioned farming systems were presented in Figure S5.

**Cropland Footprint.** The total cropland footprint of China's pig production was 29 million hectares (Mha) $yr^{-1}$ in 2017, 35% of which occurred overseas (Figure 2c). Maize products and processed soybean products contributed to 55% and 36% of the total land footprint, respectively. The cropland footprint differed from 2.1 $m^2$ $FU^{-1}$ in Qinghai to 4.5 $m^2$ $FU^{-1}$ in Fujian (Figure S3c) due to the regional differences in feed production and manure management as a function of farm scales (Figure S4c). The low cropland footprint was observed in smallholder farms due to the large portion of feed sourced from by-products and food waste that have minute impacts on land use, which is in line with a global assessment.[3]

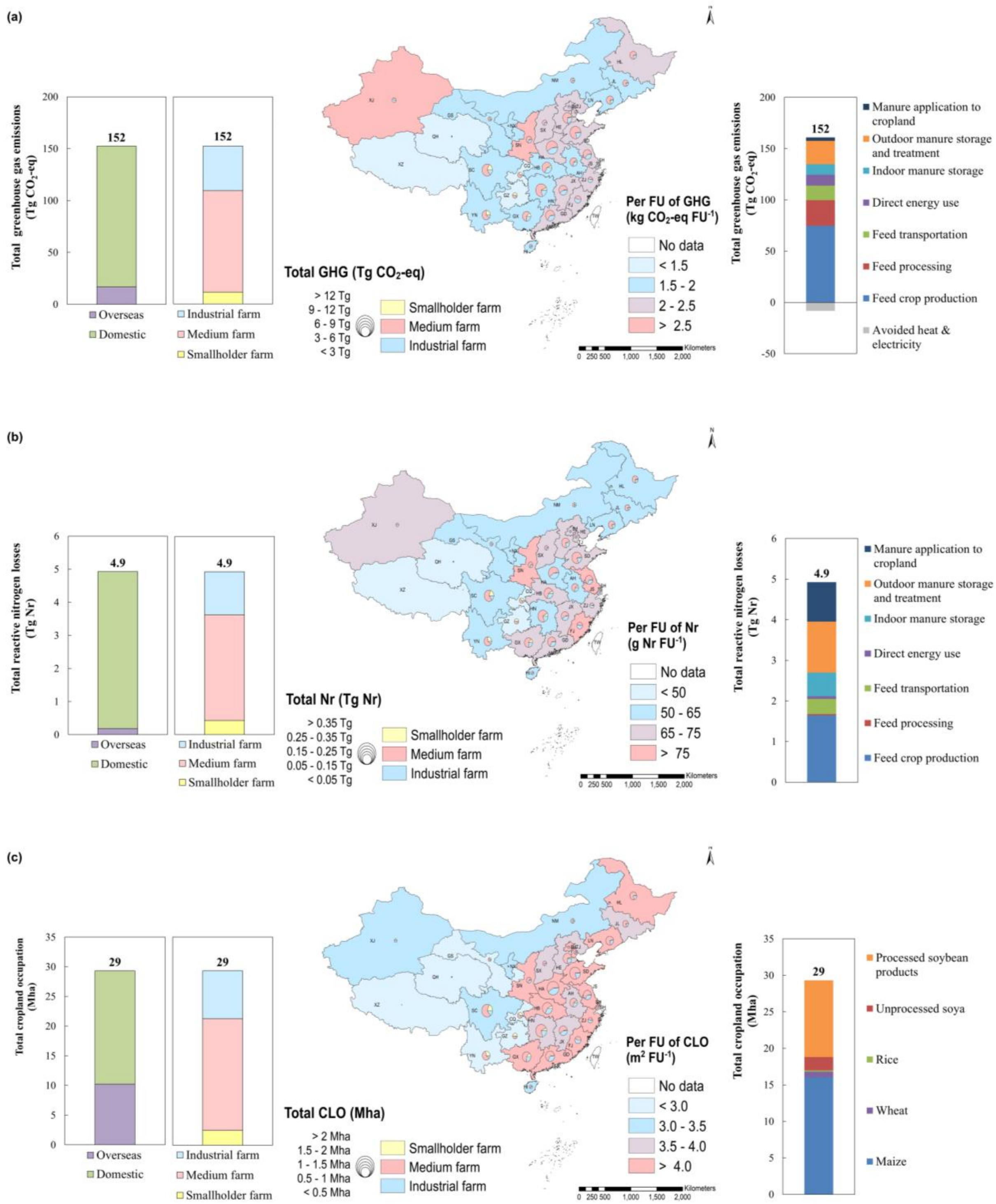


**Figure 2.** National environmental footprint characteristics of China's pig sector. The bar charts show the contribution of emissions sources (overseas or domestic), three main pig breeding systems in each province of China (excluding Hongkong, Macao, Taiwan, and Nansha Islands), and different processes of pig supply chain to (a) total greenhouse gas (GHG; Tg $CO_2$-eq) emissions, (b) total reactive nitrogen (Nr; Tg Nr) losses, and (c) total cropland occupation (CLO;

Mha). The background maps indicate (a) per FU of GHG (kg $CO_2$-eq $FU^{-1}$) emissions, (b) per FU of Nr (kg Nr $FU^{-1}$) losses, and (c) per FU of CLO ($m^2$ $FU^{-1}$) in corresponding provinces. The pie charts within maps represent the contribution of three main pig breeding systems in each province of China to (a) total GHG (Tg CO2-eq) emissions, (b) total Nr (Tg Nr) losses, and (c) total CLO (Mha), and their sizes indicate total GHG, Nr, and CLO amount. FU = Functional Unit. NM = Inner Mongolia. LN = Liaoning. JL = Jilin. HL = Heilongjiang. BJ = Beijing. TJ = Tianjin. HE = Hebei. HA = Henan. SD = Shandong. SX = Shanxi. AH = Anhui. HN = Hunan. JX = Jiangxi. GD = Guangdong. HI = Hainan. JS = Jiangsu. ZJ = Zhejiang. FJ =Fujian. SH = Shanghai. HB = Hubei. SN = Shaanxi. GS = Gansu. QH = Qinghai. NX = Ningxia. XJ = Xinjiang. CQ = Chongqing. SC = Sichuan. YN = Yunan. GZ = Guizhou. XZ = Tibet. GX = Guangxi. HK = Hongkong. MC = Macao. TW = Taiwan.

**Scenario Analysis.** Scenarios of different land use strategies were compared with the reference situation for the year 2017 (Table 1 and Figure 3).

*Carbon Footprint:* Implementing the single mitigation options (S1-4) was found to decrease the C footprint of China's pig production by 19-36%, with the largest reduction (19-30%)-due to the "opportunity cost of land" (Figure 3a). In addition to land use impacts, intensive crop production (S1) resulted in another 5% reduction from feed crop production. A combination of measures (S5-7) decreased the C footprint by 58-64% (or 7-8% when land use impacts were excluded).

*Nitrogen Footprint:* Intensive crop production (S1) decreased Nr emissions by 6% (mainly from feed crop production) relative to the reference (Figure 3b). Feeding strategies (S2-4) reduced Nr emissions by 5-7%, mostly from manure management. Increased emissions from feed crop production were observed in the ecological leftover – oil seed meal scenario (S4), compared with the DDGS scenario (S3), because of the increased emissions from growing the additional seed oil meal. Implementing combined measures (S5-7) resulted in a reduction of 12-14% in total Nr emissions.

*Cropland Footprint:* The cropland occupation area per FU decreased by 4-6%, resulting from the adoption of single measures (S1-4) (Figure 3c). Intensive crop production (S1) reduced cropland use for maize as feed by 6%. For all three feeding strategies (S2-4), land used for producing soybean decreased by 6-10% (mainly overseas) but was largely offset by increased land use for producing additional maize and processing domestic co-products. Implementing combined strategies (S5-7) decreased both domestic and overseas cropland used for feed, a decrease of 10-11% in total.

**Table 1. Environmental footprints (per functional unit) in the reference and land use scenarios.**

| Scenarios | C footprint [a] (kg $CO_2$-eq $FU^{-1}$) | N footprint (g Nr $FU^{-1}$) | Cropland footprint ($m^2$ $FU^{-1}$) |
|---|---|---|---|
| S0: Reference | 1.9 | 59 | 3.5 |
| S1: Intensive crop production | 1.2 | 56 | 3.3 |
| S2: Intensive pig production | 1.5 | 56 | 3.4 |
| S3: Ecological leftover (DDGS) [b] | 1.4 | 55 | 3.4 |
| S4: Ecological leftover (Oil seed meal) | 1.4 | 56 | 3.4 |
| S5: S1 + S2 | 0.8 | 51 | 3.2 |
| S6: S1 + S3 | 0.7 | 51 | 3.2 |
| S7: S1 + S4 | 0.7 | 52 | 3.2 |

[a] C footprint (kg $CO_2$-eq $FU^{-1}$) including the "opportunity cost of land." [b] DDGS = dried distillers grains with solubles, a co-product of maize processing into ethanol.

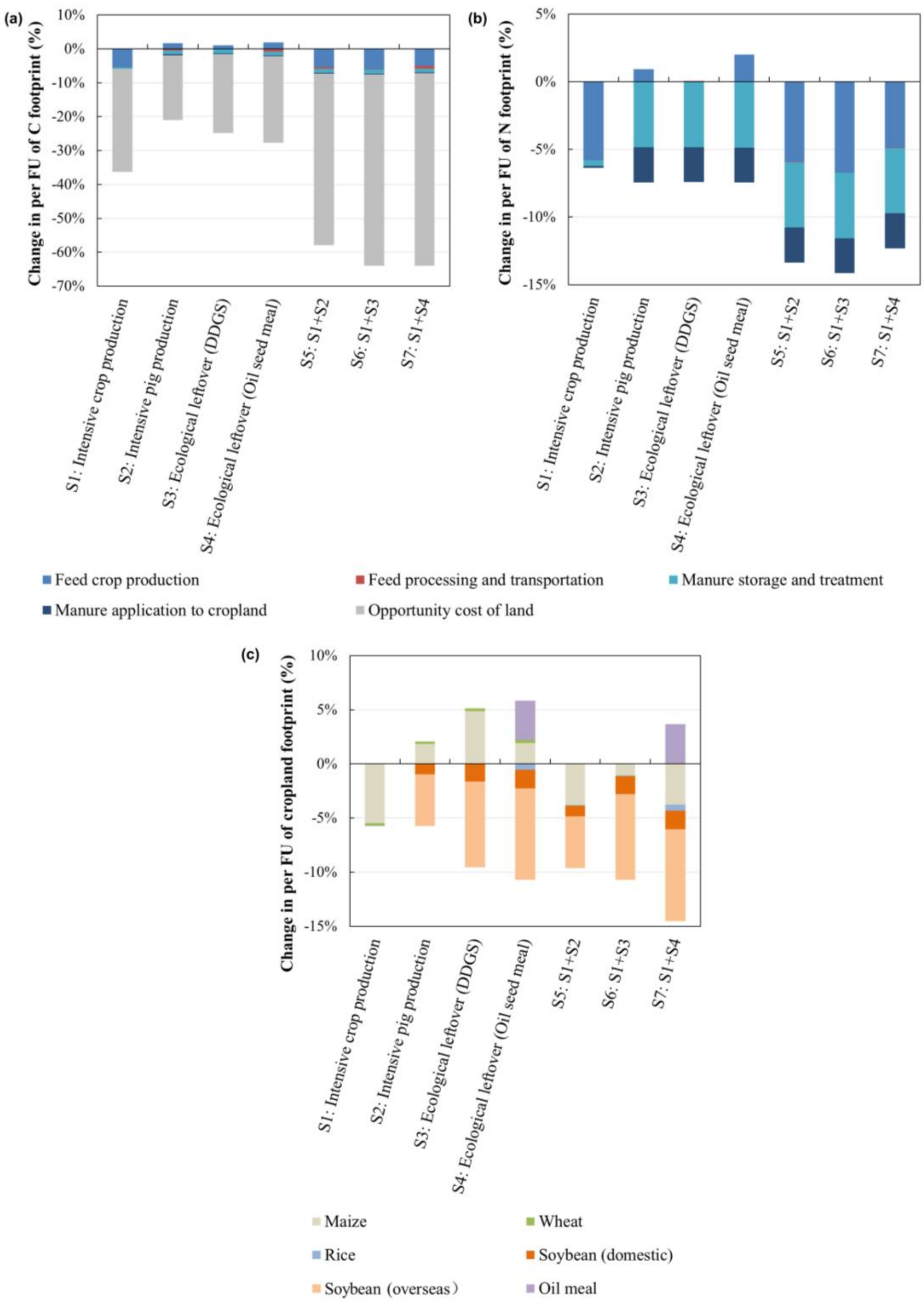


**Figure 3.** Changes in per FU of (a) C footprint (kg $CO_2$-eq $FU^{-1}$), (b) N footprint (kg Nr $FU^{-1}$), and (c) cropland footprint ($m^2$ $FU^{-1}$) from the whole chain following the implementation of feed manipulation scenarios, relative to a situation without feed manipulation. FU = Functional Unit. DDGS = dried distillers grains with solubles, a co-product of maize processing into ethanol.

**Opportunity cost.** The consequences of the 'saved' cropland reallocation were explored and shown in Figure 4. Reallocation to all the human food crop alternatives increased energy delivery (compared to pork) 3 to 20 times and protein delivery 1 to 5 times. This was accompanied by reduced environmental consequences for maize, wheat, and legumes and pulses (a reduction of 5% for C footprint and 5-12% for N footprint), or increased environmental consequences for rice and vegetables (an increase of 18-25% for C footprint, 17-21% for N footprint). Reallocating cropland that pig used for crop-based feed to beef and milk would decrease energy and protein supply, with a decline in the N (11-14%) footprints.

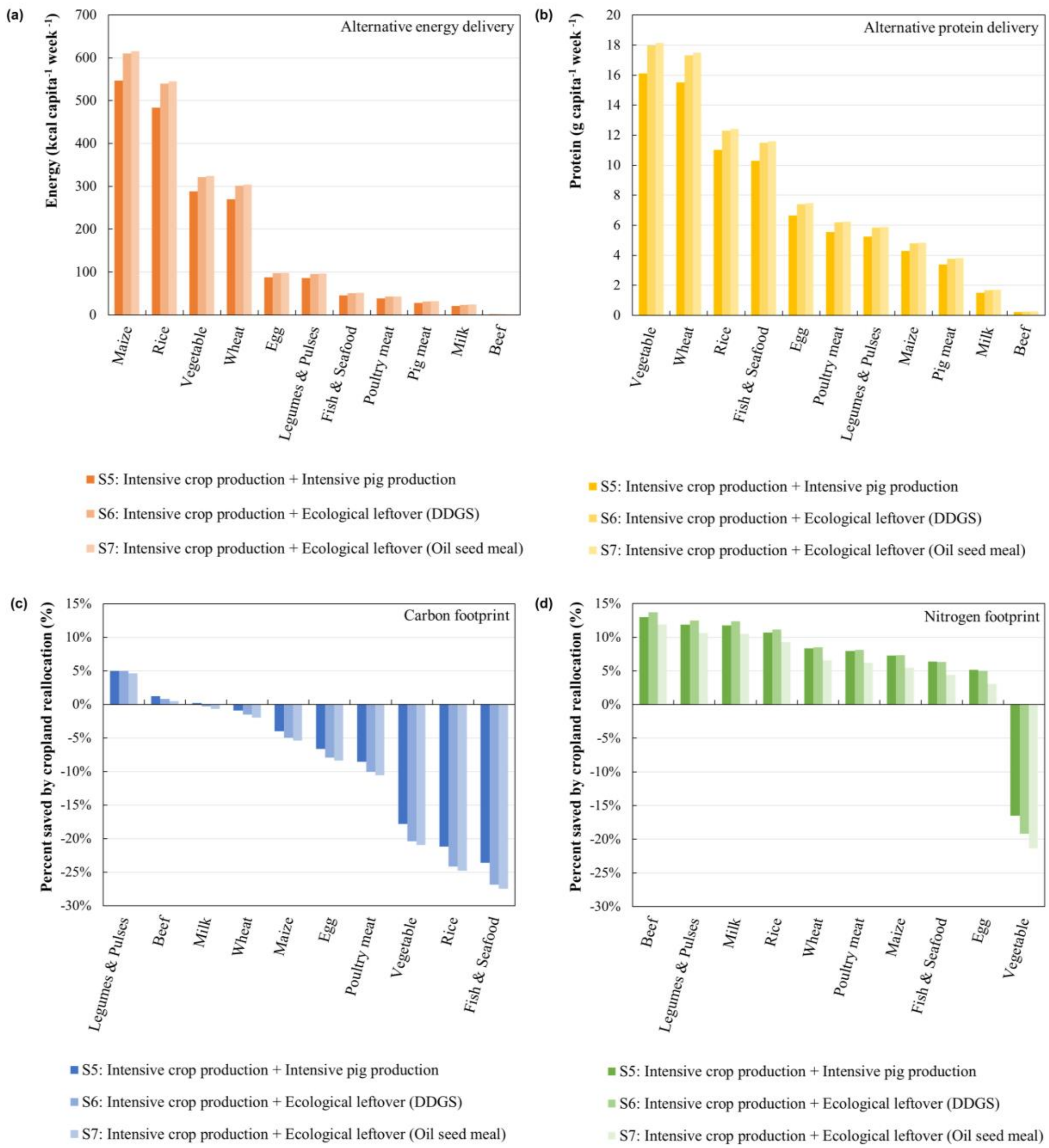


**Figure 4.** Opportunity cost for China's pig production calculated by global environmental footprints. (a) Alternative energy (kcal capita$^{-1}$ week$^{-1}$) and (b) protein delivery (g capita$^{-1}$ week$^{-1}$) associated with reallocating the saved land for the production of animal and plant food alternatives. The percentage (%) of (c) carbon (Tg $CO_2$-eq) and (d) nitrogen footprint (Tg Nr) saved by cropland reallocation to animal and plant food alternatives. The positive values indicate the environmental footprints of animal and plant food alternatives under the combination of measures (S5-7) are less than that in the reference scenario.

## DISCUSSION

**Carbon Footprint.** In this study, the estimated total GHG emissions (152 Tg $CO_2$-eq in 2017) from China's pig production amount to 11% of total agriculture GHG emissions (1410 Tg $CO_2$-eq in 2013) in China.[48] GHG emission intensities on most medium (2.0-3.4 kg $CO_2$-eq $FU^{-1}$ at the provincial level) and industrial (1.7-3.0 kg $CO_2$-eq $FU^{-1}$) pig farms in China were higher than the reported averages in European Union (EU) (2.0-2.4 kg $CO_2$-eq $kg^{-1}$ LW) and United States (US) (1.8-2.7 kg $CO_2$-eq $kg^{-1}$ LW).[49, 50] About 2.2 kg $CO_2$-eq $kg^{-1}$ LW was estimated for pig production by a study evaluating C footprints of 22 plant-based foods and 6 animal-based foods in China, close to our estimate.[51] Feed supply contributed over 71% of C footprint in the medium to industrial farms, which was much higher than that in the EU (45%) and the US (49%).[52, 53] This is mainly due to the overuse of chemical fertilizer during feed production in China.[54] Maize grain, as the dominant feedstuff, was produced with an average application of 305 kg N $ha^{-1}$ in China, about four times the worldwide average (74 kg N $ha^{-1}$).[55] Smallholder farms with residue-based feed had much lower GHG emissions than medium and industrial farms with grain-based feed. Smallholder farms still accounted for about 25% of current China's pig production,[9] so the national average C footprint was relatively low compared to developed countries. The large regional variations in GHG emission intensities were observed (Figure 2), attributing to differences in relative contributions of small and large size farms (Figure S1), fertilizer application rates, and irrigation practices during feed crop production. The amount of N fertilizer rate varied between 133-335 kg N $ha^{-1}$ for maize production among provinces in China.[30]

**Nitrogen Footprint.** The estimated total Nr emissions (4.9 Tg Nr in 2017) from China's pig production amounted to 22% of chemical N fertilizer use in China.[56] The N footprint largely varied among regions (38-77 g Nr $FU^{-1}$), attributing to regional differences in farm size and fertilizer application rates. The intensification levels of pig farms were high in East and Central China, relative to the Southwest (over 60% of pigs raised in small-sized farms in Sichuan and Guizhou).[30] Eutrophication is a key environmental consequence of Nr leakages to waterbodies. Surface water in large areas of the Huang-Huai-Hai Region and the Yangtze River basin of China are severely contaminated and identified as hotspots of eutrophication.[57, 58] The discharges and application of manure were found as the main sources.[59] Pig production in these regions accounted for about 40% of the national total, and therefore, mitigation of Nr losses from pig production needs more attention. Although it is difficult to consistently compare with other studies as a result of different system boundaries, methods, and assumptions, the relatively high N footprint (medium farms: 65-104 g Nr $FU^{-1}$ at the provincial level; industrial farms: 55-87 g Nr $U^{-1}$) of China's pig production is evident, compared to the EU average (68 g Nr $kg^{-1}$ LW).[60] A country-scale estimate indicates the N footprint of 136 g Nr $kg^{-1}$ LW for pork production in China,[61] which is slightly higher than that in our study where much detailed practices of feed production and manure management are considered. Both feed production (e.g., overuse of surface-applied urea) and poorly managed manures were identified as primary sources of Nr emission. The estimated recycling ratio of manure N to the field (32% of total N excretion) in China was far below the values reported in the US (75%)[62] and the EU countries (80%).[63] In these developed countries, environmental regulations were largely implemented. For example, the EU National Emission Ceiling Directive[64] and the Gothenborg protocol of the UNECE Convention on Long-range Transboundary Air Pollution[65] were established to decrease $NH_3$ and $NO_x$ emissions; the UN-FCCC Kyoto protocol[66] to reduce $N_2O$ emission; the EU Nitrates Directive[67] to minimize N leaching to groundwater and surface waters. Through the implementation of these policies, nitrogen mitigation measures for livestock manure have been adopted in many EU countries. For example, covering slurry stores and adopting low-$NH_3$ emission manure application methods have been adopted by >90% of farmers in the Netherlands and Denmark.[32] However, less than 20 % of pig farms have adopted low-$NH_3$ emissions measures, according to surveys taken in few regions of China (Table S7).

**Cropland footprint.** The overall cropland occupation area from China's pig production was estimated as 29 Mha in 2017, accounting for 21% of arable land (135 Mha in 2017) in China.[68] The land demand is mainly from cropland occupied by cereals (especially maize), which is suitable for food production for human consumption. Of that, nearly 35% (10 Mha) occurred overseas as a result of imported soybean. Medium (3.8-5.8 $m^2$ $FU^{-1}$ at the provincial level) and industrial (3.3-5.2 $m^2$ $FU^{-1}$) pig farms in China, despite large variations among provinces, occupied far more cropland than those case farms in the Netherlands (4.4 $m^2$ $kg^{-1}$ LW)[69] and France (3.98 $m^2$ $kg^{-1}$ LW).[18] Differences between regions and countries were attributed to variations in pig rearing, in terms of feed conversion ratio (FCR) and feed composition, and in crop production depending on how efficient land is used (e.g., crop productivity). The average FCR of pig production in China, especially for smallholder farms (3.4), was much higher than that in the EU (1.7-2.8)[18, 52] and US (2.75).[70] Maize grain yield in China (7.5 t $ha^{-1}$ in 2017) was lower than the EU average (8.9 t $ha^{-1}$) and US (11.8 t $ha^{-1}$).[5, 30] It is therefore evident that China's pig sector would require more land resources.

**Implications of Scenario Analysis.** Several policies have been implemented to reduce fertilizer use in China. The 'zero fertilizer growth' policy issued in 2015 has effectively slowed the increase in fertilizer use.[71] Field experiments have proved that reduced fertilizer use could go hand in hand with closing yield gaps in China. Integrated Soil–crop System Management (ISSM) practices have been tested and promoted for maize, wheat, and rice in China, which have been shown to be agronomically robust and economically acceptable.[41] Our results show that the pork supply chain can primarily benefit from this ISSM in reducing C and N footprints and saving land for food production. The promotion of the ISSM approach in smallholder farms that dominate the crop landscape can be constrained by social and logistical barriers beyond research-oriented experiments. This may require the implementation of appropriate incentives, such as establishing a national campaign network and scientific technology backyards in China, to allow farmers to adopt more knowledge-intensive techniques.[41]

It is worth noting that environmental impacts from feed crop production can be further reduced through improved fertilizer production technologies. Although a detailed analysis is beyond the scope of this study, here we provide a brief indication of the additional reduction in the C footprint by improving fertilizer production efficiency. Coal-based nitrogen fertilizer production is still predominant in China, where every ton of N fertilizer manufactured emits 8.3 t $CO_2$-eq,[54] compared with 4.8 t $CO_2$-eq in EU.[72] Mitigation opportunities, including improving methane recovery during coal mining and enhancing energy efficiency in fertilizer manufacture, can reduce N fertilizer-related GHG emissions by 30-43%.[54] Taking improved fertilizer manufacture into account, the ISSM scenario (S1) resulted in an additional 3-4% reduction in the C footprint of China's pig production.

Improving feed protein utilization efficiency within livestock production systems has been shown previously to increase the N use efficiency of the entire food system.[18] National standards on feed formulation for pigs and chicken, aimed at reducing protein levels in feed rations, were recently released by China's government.[73] For the first time, the multiple environmental impacts of such a policy (S2) were evaluated in our scenario analysis. The C footprint and cropland demand were generally decreased, mainly due to reduced imports of high-protein feeds (i.e., soybean), especially from South America with relatively high GHG emission from land use change .[11, 27] The

N footprint was reduced mostly through the reduction in the amount of manure N generation. Feeding pigs ecological leftovers from food processing (i.e., DDGS and oil meal; S3-4), combined with lowering feed protein content, play an essential role in further alleviating land use changes and associated GHG emissions that occur overseas. China is taking action to increase the domestic supply of soybean and oil crops mainly through crop redistribution and increased yield, which in part supports the increased demand for high-protein feed.[74]

These feeding options on GHG emissions were insignificant if the "opportunity cost of land" were not considered (Figure 3). The reason was that reduced GHG emissions from imports of feed were offset by emissions from the increased supply of domestic feed that had high emission intensity. These feeding strategies also lead to an increased domestic cropland footprint, and the reduction in Nr emissions from crop cultivation was small. Given the possible antagonistic effects, incorporating sustainable crop intensification with these feeding options is essential to achieve synergistic mitigation in China's pig production. It, therefore, calls for joint efforts between crop production and animal production sectors in policy decision-making.

Manure management strategies can also reduce GHG and/or Nr emissions along the pig production chain but may not much affect cropland use footprint,[6, 75] which is therefore not the focus of our study. To extend the discussion, we did additional scenarios to illustrate the impacts of manure management options on these emissions (see SI for details). Adopting low-emission measures during manure storage and banning manure discharge could reduce N footprint by 6-10%, with limited effects on C footprint. Promoting anaerobic digestion could reduce C footprint by 3%, yet slightly increase N footprint. A combination of these management options is expected to have a relatively large mitigation potential of Nr losses (22%; Figure S9). A previous estimate indicated a reduction of nearly two-thirds in manure-sourced N losses from China's pig production (excluding feed supply stage) if combined options of manure management were implemented in 2030.[6] Although the system boundary and timeframe of this early research were not exactly consistent with our study, it still confirms a large reduction potential in N losses via proper manure management. Achieving substantial abatement of C and N footprints from pork production chain requires a holistic policy package promoting efficient land use and manure management measures.

Implementing a combination of land use strategies considered in the present study would reduce the cropland footprint of China's pig production by 10-11% (Figure 3). This would allow repurposing high-quality croplands that are currently used to grow feed for pig production for other, more environmentally friendly, and nutritious food types. Our results indicate human food crop alternatives would deliver 3-20 and 1-5 folds more energy and protein, respectively, than the replaced pork on the basis of per ha cropland (Figure 4), which is because of their higher human-edible energy and protein conversion efficiency than animal products in general.[8] This strategy has implications for providing 18-40 million more population in terms of energy supply or 2-34 million more population in terms of protein supply (according to 1048 Mcal calories and 30 kg protein intake per capita per year) in China.[5] Reallocating land to vegetables, wheat, and rice would generate a relatively large increase in protein delivery, attributing to their high protein yield (420, 410, and 290 kg $ha^{-1}$, respectively) relative to pork (90 kg $ha^{-1}$). Although legumes and pulses are protein-dense crops, increased protein delivery may be marginal due to their low yields in China. The average yield of soybean was 2.1 t $ha^{-1}$ in China while it was 3.3 t $ha^{-1}$ in the US.[5, 30] For the animal food alternatives, the possible reduction in protein and energy delivery can be found when using the saved land to feed ruminants, such as beef cattle. This is because of the low human-edible energy and protein conversion efficiency for beef cattle (3%) relative to pork (9%).[76] These outcomes have implications for policy development regarding the promotion of more plant-based dietary patterns characterized by less consumption of red meat (e.g., pork and beef).

The environmental consequences of land reallocation are not consistent depending on food alternatives (Figure 4). Repurposing the saved cropland to the production of rice and vegetables was found to increase C and/or N footprints due to the high $CH_4$ emission intensity from paddy rice fields[77] and the overuse of fertilizers to grow vegetables.[61] An average increase of 17% in vegetable yields through improved soil and crop management was observed in experimental sites across China, with decreased N fertilizer use by 38%,[78] offering significant mitigation potential in case of cropland reallocation to grow vegetables. Repurposing cropland to legumes and pulse can alleviate both C and N footprints, which can be considered a targeted policy to tackle these interconnected environmental challenges. For animals (e.g., beef and milk), there is mitigation potential for Nr losses, but not for GHG emissions (Figure 4). Although C and N footprints per kg of animal products for beef and dairy cattle are found to be higher than that for pork, the differences may be varied (or reversed) on the basis of per ha of utilized agricultural land. This is because the global average land use of ruminants (e.g., 328 $m^2$ $kg^{-1}$ beef) is generally larger than that of pork (18 $m^2$ $kg^{-1}$ product). This tendency on emissions per ha land is in line with some previous studies in the US.[79, 80] For consistency with other studies, the global average footprints of food products were used in this study. However, we also recalculated the opportunity cost using country-specific information whenever available (see Figure S7). Despite absolute differences in final values, the general tendency was similar between the two estimates. These findings illustrate the importance of incorporating opportunity cost within LCA and have implications for national and region-specific policies for agricultural structure changes (including spatial reallocation of cropping). We believe that the policy options explored in the present study are also applicable to other industrial livestock systems in China and other nations facing similar challenges from livestock production.

## ASSOCIATED CONTENT

### Supporting Information

The Supporting Information is available free of charge on the ACS Publications website at DOI. Here you can find additional text, 9 figures, and 12 tables with details on the data, methods, and models used for calculating the multiple environmental footprints (PDF).

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### Notes

The authors declare no competing financial interest.


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## Abstract Art