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RESEARCH ARTICLE

Greenhouse gas emissions during the COVID-19 pandemic from agriculture in China

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KEYWORDS

Agricultural systems, emission factors, greenhouse gases, soil carbon sequestration

HIGHLIGHTS

- Methane led China's growth in net greenhouse gas emissions over the pandemic.
- N_2O was linked to fertilizers and waste management.
- CO_2 emissions varied by region, calling for tailored mitigation approaches.
- COVID-19 boosted methane from pig farming disruptions.

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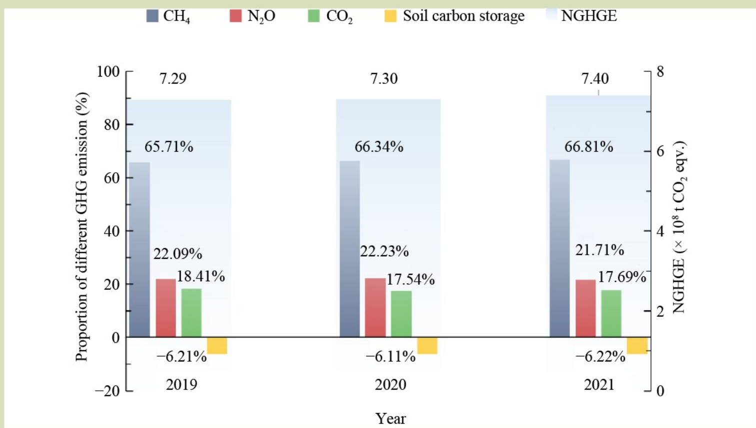
GRAPHICAL ABSTRACT



ABSTRACT

To study the impact of the COVID-19 pandemic on agricultural carbon emissions in China, the greenhouse gas emissions generated by crop and livestock production, and agricultural material and energy inputs in China from 2019 to 2021 were systematically calculated. It was found that from 2019 to 2021, Net greenhouse gas emissions (NGHGE) from agriculture in China had an increasing trend. Methane emissions ranked first in NGHGE, with an annual proportion exceeding 65% and an increasing annual trend. CH_4 emissions were primarily influenced by enteric fermentation and rice production. Nitrous oxide emissions accounted for around 22% of annual NGHGE and decreased from 2019 to 2021. The main sources of N_2O emissions were the use of nitrogen fertilizers and manure management. Carbon dioxide emissions accounted for about 18% annually, with diesel and agricultural electricity use contributing to over 60% of CO_2 emissions. Soil carbon sequestration represented about a 6.1% lowering of NGHGE. The combined proportion of CH_4 emissions from enteric fermentation and rice production accounted for over 50% of total GHG emissions. The changes in NGHGE were mainly caused by disturbance of the livestock industry during the pandemic.

1 Introduction

Greenhouse gas (GHG) emissions exacerbate the global warming trend, with IPCC data indicating that GHG emissions from agricultural production account for 23% to 30% of the total GHG emissions from human activities[1]. It is estimated that more than 40% to 47% of global anthropogenic CH_4 and 60% to 82% of N_2O emissions from 2007 to 2016 originated from agricultural production. It is projected that by 2070, these emissions will increase by 47% and 50%, respectively, compared to the levels in 2010[1]. GHG emissions from agricultural production result from various sources, including crop and, livestock production, agricultural inputs, and energy usage. China, being a major agricultural nation, has a wide distribution of crop production, a diverse range of livestock (including poultry), and a significant increase in agricultural inputs and energy usage in recent years. Additionally, the vast geographical expanse of China and its significant variations in climate and geological conditions between different regions have led to distinct crop and livestock production structures in each region, resulting in significant variations in GHG emissions. Therefore, understanding the total GHG emissions, structural patterns, and regional characteristics of agricultural GHG emissions in China is a primary task for formulating rational and effective agricultural emission reduction policies. Additionally, studying carbon sequestration measures in agriculture is an effective approach to reducing net greenhouse gas emissions (NGHGE) from agriculture.

Current research on agricultural carbon emissions often focuses on specific perspectives. Only certain agricultural carbon sources were selected to calculate their carbon emissions. This mainly includes carbon emissions from land use change in agriculture[2–4] carbon emissions from agricultural energy[5], carbon emissions from livestock production[6,7], emissions of one of the GHGs, for example, N_2O[8], carbon emissions from regional farmland[9,10] and carbon emissions under specific crop production patterns[11,12]. Undoubtedly, these research achievements have greatly enriched the research on agricultural carbon emissions, laying a solid foundation for subsequent related studies. However, current research primarily focuses on agricultural carbon emissions, with relatively limited research on carbon sequestration in agriculture, and even fewer studies that bridge the two. Also, most studies are mostly limited to crop production and have not expanded into the livestock production sector, or vice versa. These results in research conclusions that are somewhat one-sided, with limited practical guidance. Therefore, there is a pressing need to conduct a comprehensive assessment of agricultural carbon footprints.

According to the latest United Nations report, major GHG concentrations continued to increase during the COVID-19 pandemic[13]. However, there have been no reports on the spatiotemporal changes in carbon emissions from Chinese agriculture during the COVID-19 pandemic. We address the shortcomings of previous research and follow the guidelines from the 2011 "Guidelines for Provincial Greenhouse Gas Inventories" to calculate the NGHGE from the agricultural systems of the 31 provinces, municipalities, and autonomous regions in Chinese mainland from 2019 to 2021[14]. We hypothesized that there have been significant changes in the carbon source structure and emissions of Chinese agricultural GHGs during the pandemic period (2019–2021), influenced by factors such as altered agricultural practices, external material and energy inputs, and variations in soil carbon pools. We aimed to understand the carbon source structure and emissions of Chinese agricultural GHGs during the COVID-19 pandemic. We also calculated the GHG produced by external materials and energy inputs in agriculture. By including the soil carbon pool, we assessed the NGHGE from agriculture in Chinese mainland and analyzed the trends during the pandemic. Additionally, we analyzed the spatial variations in GHG emissions from agriculture at the provincial level. The study aimed to provide a scientific reference for the current status and future development of GHG emissions in agricultural systems and to offer theoretical support for the formulation of low-carbon agricultural production policies in China and beyond.

2 Accounting scope and method

We employed the accounting methods outlined in the 2011 Guidelines[14] to calculate the GHG emissions caused by agricultural production in various provinces (municipalities, autonomous regions) of Chinese mainland from 2019 to 2021. The scope of the calculation, methods, and data sources are given in Table 1.

GHGs produced by agricultural production include methane, nitrous oxide, and carbon dioxide. CH_4 emissions calculations include rice production, enteric fermentation in livestock (including poultry), and manure management. N_2O emissions calculations include direct and indirect emissions from agricultural land use. Direct N_2O emissions from agricultural land use include the use of nitrogen fertilizers, straw incorporation and manure return. Indirect N_2O emissions include atmospheric deposition and nitrogen leaching and runoff. The livestock calculations include various animal species, including pigs, cattle (beef, draft, and dairy cattle),

Table 1 Accounting ranges and data sources of GHG emissions from agriculture in China

GHG	Carbon sources	Emission factor reference	Emission factor unit	Formula	Data composition of the formula
CH_4	Rice production	Recommended CH_4 emission factors in rice field in China regions in 2005[14]	$kg \cdot ha^{-1}$	$E_{CH_4,Rpro} = \sum EF_i \times AD_i \times 10^{-4}$	$E_{CH_4,Rpro}$: CH_4 emission of rice production per 10 kt; EF: emission factor; AD: sowing area per kha; *i*: rice field types
	Enteric fermentation	Animal enteric fermentation CH_4 emission factors[14]	kg per unit per year	$E_{CH_4,Ent} = \sum EF_{CH_4,Ent,j} \times AP_j \times 10^{-3}$	$E_{CH_4,Ent}$: CH_4 emission of animal enteric fermentation per 10 kt; $E_{CH_4,Manu}$: CH_4 emission of manure management per 10 kt; AP: number of standing stocks of livestock production at the end of the year/ten thousand; *j*: types of livestock, pigs, dairy cattle, non-dairy cattle, goats, sheep, horses, donkeys, mules, camels
	Manure management (CH_4 emission)	Manure management CH_4 emission factors[14]	kg per unit per year	$E_{CH_4,Manu} = \sum EF_{CH_4,Manu,j} \times AP_j \times 10^{-3}$	
CO_2	Pesticides	4.9341[14]	$kg \cdot kg^{-1}$ C eqv.	$E_{CO_2,Pest} = M_{Pest} \times 4.9341 \times 44/12$	$E_{CO_2,Pest}$: CO_2 emission of pesticide usage per 10 kt; M_{Pest}: pesticide usage per 10 kt
	Plastic mulching	5.18[14]	$kg \cdot kg^{-1}$ C eqv.	$E_{CO_2,Plastm} = M_{Plastm} \times 5.18 \times 44/12$	$E_{CO_2,Plastm}$: CO_2 emission of plastic mulching per 10 kt; M_{Plastm}: plastic mulching usage per 10 kt
	Agricultural Electricity	2019 annual emission reduction project China regional power grid baseline CO_2 emission factors[15]	t CO_2 per MWh	$E_{CO_2,Agrielect} = M_{Agrielect} \times 4.77\% \times EF_{CO_2} \times 10$	$E_{CO_2,Agrielect}$: CO_2 emission of agricultural electricity per 10 kt; $M_{Agrielect}$: electricity use in rural area per TWh[16]
	Diesel	0.5927[1]	$kg \cdot kg^{-1}$ C eqv.	$E_{CO_2,Dies} = M_{Dies} \times 0.5927 \times 44/12$	$E_{CO_2,Dies}$: CO_2 emission of diesel usage per 10 kt; M_{Dies}: diesel usage per 10 kt
	Irrigation	25[17]	$kg \cdot ha^{-1}$ C eqv.	$E_{CO_2,Irrig} = S_{Irrig} \times 25 \times 44/12 \times 10^{-4}$	$E_{CO_2,Irrig}$: CO_2 emission of irrigation per 10 kt; S_{Irrig}: irrigation area per kha
N_2O	Fertilizer	Parameters for estimating nitrogen input of crop straw turnover to field[14], the default values of N_2O direct emission factor for different regions[14] and nitrogen, phosphorus, potassium ratio of complex fertilizer in different regions[18]	kg N_2O-N per kg N input	$E_{N_2O,N\text{-}fert} = N_{N\text{-}fert} \times EF$ $E_{N_2O,C\text{-}fert} = N_{C\text{-}fert} \times EF$	$E_{N_2O,N\text{-}fert}$: N_2O emission of nitrogen fertilizer usage per 10 kt $N_{N\text{-}fert}$: total equivalent nitrogen in nitrogen fertilizer usage per 10 kt $E_{N_2O,C\text{-}fert}$: N_2O emission of complex fertilizer usage per 10 kt $N_{C\text{-}fert}$: total nitrogen from complex fertilizer usage per 10 kt
	Straw turnover			$E_{N_2O,Stwto} = \sum N^k_{Stwto} \times EF$ $N_{Stwto} = N_{Ground\ field} + N_{Underground\ root}$ = (Crop grain yield/Economic coefficient – Crop grain yield) × Straw turnover rate × Nitrogen content in straw + Crop yield/Economic coefficient × Root shoot ratio × Nitrogen content in straw or root $N_{Sugarcane}$ = (Crop yield – Crop yield × Economic coefficient) × Straw turnover rate × Nitrogen content in straw + Crop yield × Root shoot ratio × Nitrogen content in straw turnover rate $N_{Vegetable}$ = Crop yield × 0.1 × Nitrogen content in straw × Straw turnover rate	$E_{N_2O,Stwto}$: N_2O emission of straw turnover per 10 kt N_{Stwto}: nitrogen content of straw turnover per 10 kt; *k*: rice, wheat, corn, sorghum, millet, soybean, rapeseed, peanut, sesame, seed cotton, sugar beet, sugarcane, hemp, potato, vegetables, tobacco

(Continued)

GHG	Carbon sources	Emission factor reference	Emission factor unit	Formula	Data composition of the formula
	Manure return			$R>1,\ \mathrm{AAP}=\left(\frac{Q_{napa}}{365}\right)\times T$ $R<1,\ \mathrm{AAP}=(Q_{pre}+Q_{cur})/2$ $P_{faeces(urine)}=\mathrm{AAP}\times T\times q_{faeces(urine)}/1000$ $E_{N_2O,Manurt}=30\%\sum(P^j_{faeces}\times r^j_{faeces,N}\%$ $+P^j_{urine}\times r^j_{urine,N}\%)\times \mathrm{EF}$	R: off-take rate of livestock including poultry; AAP: average livestock production per year per ten thousand; Q_{napa}: annual livestock production per ten thousand; Q_{pre}: standing stock of previous year; Q_{cur}: standing stock of current year per ten thousand; T: feeding cycle of livestock per day; P: annual excretion per 10 kt; q: daily excretion per kg; r: nitrogen content of faeces and urine[18] $E_{N_2O,Manurt}$: N_2O emission of manure return per 10 kt
	Atmospheric deposition	0.01[14]	kg N_2O-N per kg N input	$E_{N_2O,Sed}=[N_{Manu}\times 20\%+(N_{N\text{-}fert}+N_{C\text{-}fert}+N_{Stwto})\times 10\%]\times 0.01$	$E_{N_2O,Sed}$: N_2O emission of atmospheric deposition per 10 kt N_{Manu}: total nitrogen excretion from livestock; N_{Stwto}: total nitrogen input of straw turnover;
	Nitrogen leaching and runoff	0.0075[14]	kg N_2O-N per kg N input	$E_{N_2O,Lea}=(N_{N\text{-}fert}+N_{C\text{-}fert}+N_{Stwto}+N_{Manurt})\times 20\%\times 0.0075$	$E_{N_2O,Lea}$: N_2O emission of nitrogen leaching and runoff per 10 kt N_{Stwto}: total nitrogen input of straw turnover; N_{Manurt}: total nitrogen input of manure return
	Manure management (N_2O emission)	Manure management N_2O emission factor [14]	kg per unit per year	$E_{N_2O,Manu}=\sum EF_{N_2O,Manu,i}\times AP_j\times 10^{-3}$	$E_{N_2O,Manu}$: N_2O emission of manure management per 10 kt

sheep (including goats), poultry, rabbits, horses, donkeys, mules, and camels. CO_2 emissions calculations encompass GHG emissions resulting from the use of external auxiliary materials in agricultural production, such as diesel, electricity, pesticides, and plastic mulching. It is important to note that the carbon emissions generated during the upstream production and transportation of nitrogen fertilizers were not included in this calculation in this study, and neither were carbon emissions from the production of potassium and phosphate fertilizers.

From 2005 to 2014, China implemented the Soil Testing and Formulated Fertilization (STFF) project. The contents of soil organic matter (SOM) were presented in a monograph, “Soil Basic Nutrient Data Set for Soil Testing and Formulated Fertilization (2005–2014)”[19]. In 1980, China conducted the Second National Soil Survey (SNSS) on major cropland soils, which covered typical soil types and cropping systems. The SNSS data, including SOM, were mainly collected from a series of monographs contained in the China Soil Series Vols. 1–6 (National Soil Survey Office, 1993–1996). The SOM contents in both STFF and SNSS data sets were determined by the potassium dichromate oxidation method. The SOC content was calculated by multiplying SOM by 0.58. The annual changes in SOC (dSOC) were calculated by the difference in the SOC divided by the years between the STFF and SNSS. The details of dSOC calculation can be found in Zuo et al.[20].

Based on the calculation formula mentioned in Table 1, N_2O and CH_4 emissions are calculated and then converted to CO_2 eqv. based on their global warming potential. The conversion factors used are 273 for N_2O and 27.9 for CH_4 (as defined by IPCC[21]). By including the soil carbon pool, agricultural NGHGE for different regions and the net total national emissions are computed. The calculations were as follows with all resulting in tons CO_2 eqv.

$$\mathrm{NGHGE}=\mathrm{GWP}-\mathrm{GWP}_{dSOC} \tag{1}$$

$$\mathrm{GWP}=E_{CO_2}+E_{CH_4}\times 27.9+E_{N_2O}\times 273 \tag{2}$$

$$\mathrm{GWP}_{dSOC}=\mathrm{dSOC}\times S\times 44/12 \tag{3}$$

where, NGHGE is the net greenhouse gas emissions from agricultural systems, GWP is the total global warming potential from agricultural systems (including CH_4, N_2O and CO_2), GWP_{dSOC} is the global warming potential due to changes in soil carbon pool, dSOC is the annual change of soil carbon pool content (kg·ha^{-1} C), and S is the agricultural production area of the year (kha).

Carbon foot intensity (CFI) was calculated by dividing NGHGE by the gross agricultural production (GAP).

The accounting data came from the China Statistical Yearbook and the online version of the statistical yearbook of various provincial regions.

3 Results

3.1 Characteristics of agricultural NGHGE in China from 2019 to 2021

Figure 1 shows the results of the calculation of NGHGE from agriculture in China from 2019 to 2021 (as CO_2 eqv.). In 2019, NGHGE was 7.29×10^8 t, while in 2020 and 2021, NGHGE was 7.30×10^8 and 7.40×10^8 t, respectively. There was a 0.14% annual increase in 2020 and a 1.34% increase in 2021, indicating a gradual upward trend.

Regarding the structure of GHG emissions within the agricultural system, CH_4 emissions accounted for the largest proportion of NGHGE from agriculture in China. Over the 3 years, they constituted 65.7%, 66.3%, and 66.8% of the total, respectively, showing a continuous increase. N_2O emissions, in contrast, accounted for 22.1% in 2019, increasing slightly to 22.2% in 2020 and decreasing to 21.7% in 2021, indicating a fluctuating trend. CO_2 emissions were the smallest proportion, with 18.4% in 2019, decreasing to 17.5% in 2020, with a slight increase to 17.7% in 2021. The change in the warming potential due to variations in soil carbon storage ranged from −6.11% to −6.22%. It is evident that the combined impact of CO_2, N_2O, and CH_4 emissions is the primary reason for the fluctuations in NGHGE from agriculture in China.

The structure of carbon sources is shown in Fig. 2, the contributions of the carbon sources to GHG emissions, from highest to lowest, were: enteric fermentation > rice production > fertilizer use > N_2O emissions from manure management > CH_4 emissions from manure management > diesel > agricultural electricity > plastic mulching > pesticide use ≈ indirect emissions from agricultural land > straw incorporation > manure return. Of these sources, CH_4 emissions from enteric fermentation and rice production were the highest, accounting for an annual average of 28.2% and 22.8%, respectively, with their combined contribution exceeding 50% over the 3-year period. Straw incorporation and manure return had the lowest contribution, with annual averages of 1.5% and 0.5%, respectively. The order of contributions of different carbon sources to total GHG emissions remained relatively consistent over the 3 years.

3.2 Structural characteristics of different GHG emissions from agriculture in China from 2019 to 2021

To further understand the source structure and emissions of

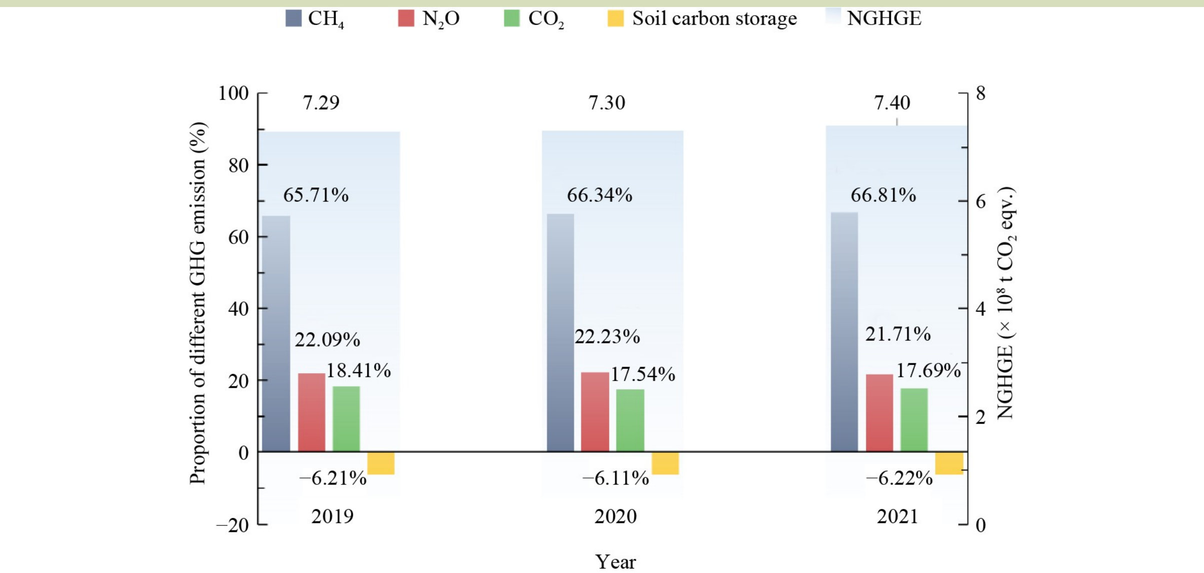


Fig. 1 Proportion of GHG emissions and NGHGE from agriculture in China from 2019 to 2021.

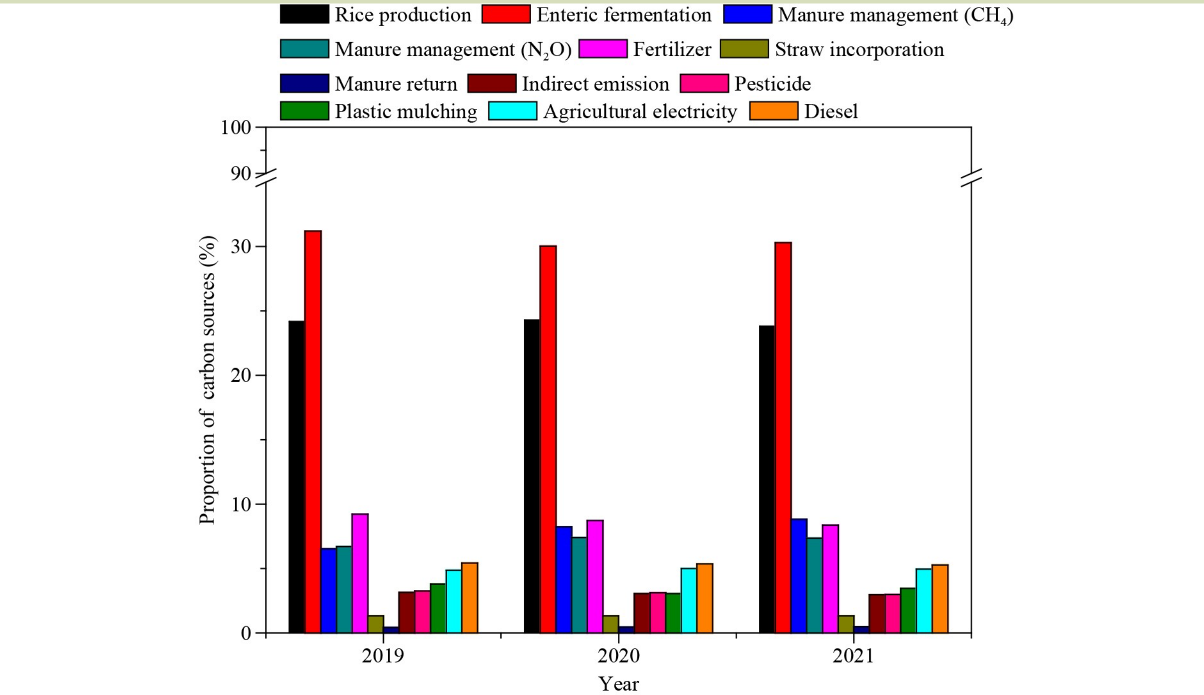


Fig. 2 Proportion of carbon sources from agriculture in China in 2019 to 2021.

agricultural CO_2, N_2O, and CH_4 in 2019 to 2021, Fig. 3 compares the impacts of different emission sources on GHG emissions in agricultural production (the line chart is the change chart of CO_2, N_2O, and CH_4 emissions, all calculated as CO_2 eqv.).

CH_4 emissions increased from 4.79 × 10^8 t in 2019 to 4.85 × 10^8 t in 2020 and reached 4.95 × 10^8 t in 2021 (Fig. 3). The annual growth rates were 1.1% and 2.1%, respectively, indicating a slowing trend in the rate of increase. In terms of emission sources, CH_4 emissions from enteric fermentation

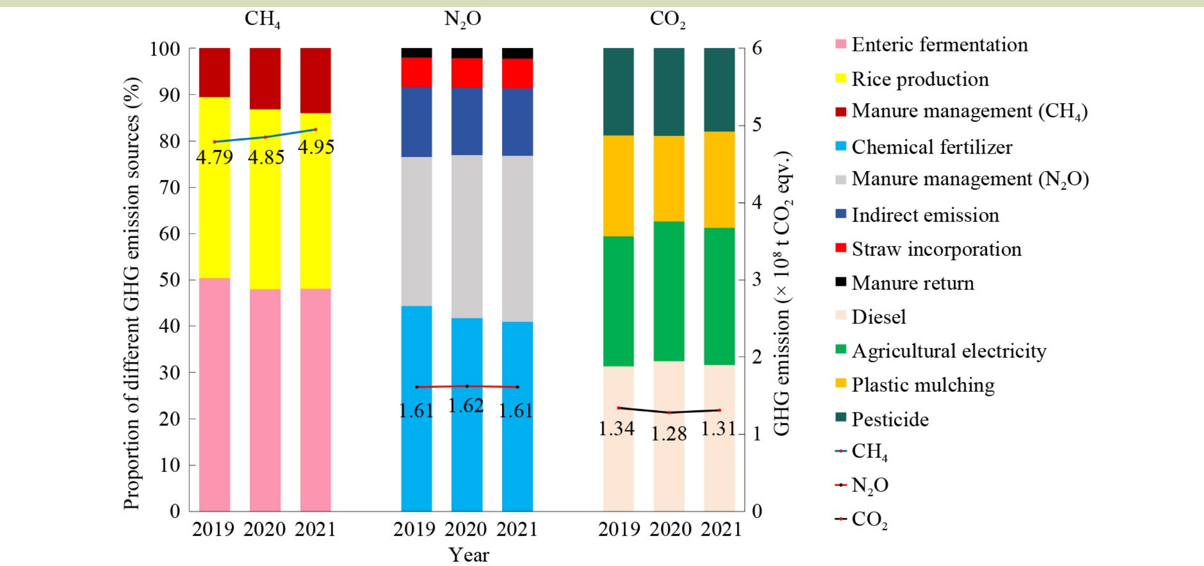


Fig. 3 Proportion and total amount of each GHG from different sources from agriculture in China in 2019 to 2021.

and rice production were significant. In 2019, these sources accounted for 50.4% and 39.1% of CH_4 emissions, respectively. Over the 3 years, the proportion of CH_4 emissions from enteric fermentation remained above 48%. The proportion of CH_4 emissions from rice production had a decreasing trend, as 38.8% and 37.8% in 2020 and 2021, respectively, representing decreases of 0.6% and 2.5% compared to the previous year. In 2019, CH_4 emissions from manure management accounted for 10.5% of total CH_4 emissions, which increased to 13.2% in 2020 and further to 14.0% in 2021. This trend indicated continuous growth in manure production due to an increase in livestock production.

CH_4 emissions from different types of manure management are shown in Fig. 4. CH_4 emissions from manure management increased from 5.05×10^7 t in 2019 to 6.38×10^7 t in 2020, marking a 26.3% increase. In 2021, CH_4 emissions from manure management further increased to 6.92×10^7 t, with an annual growth rate of 8.6%. Of these emissions, CH_4 emissions from pig manure management were the largest, increasing from 75.3% to 81.0% over the 3 years. However, the proportion of CH_4 emissions from non-cow manure management decreased from 12.1% in 2019 to 10.9% in 2021, showing a declining trend over the 3 years. It was also evident that the CH_4 emissions from manure management of animals, such as poultry, cows, goats, and sheep, had different degrees of annual decline. This suggests that changes in pig production practices are a significant factor influencing CH_4 emissions from manure management.

In 2019, N_2O emissions amounted to 1.61×10^8 t, increasing to 1.62×10^8 t in 2020, representing an annual increase of 0.62% (Fig. 3). However, in 2021, N_2O emissions decreased to 1.61×10^8 t, being a 0.62% annual decrease. Regarding N_2O emissions sources, it is notable that the use of nitrogen fertilizers and N_2O emissions from manure management are significant contributors. Nitrogen fertilizer use accounted for the largest proportion and had a decreasing trend over the 3 years. In 2019, it accounted for 44.3% of N_2O emissions, which decreased to 41.7% in 2020 and further to 40.9% in 2021. The average annual reductions were 6.0% and 1.9%, respectively.

However, N_2O emissions from manure management had an increasing trend. In 2019, it accounted for 32.25% of total N_2O emissions, and in 2020, it experienced significant growth, reaching 35.4%. In 2021, it increased further to 36.0%. The annual increases for the two years were 9.6% and 1.8%, respectively. N_2O emissions from indirect agricultural land sources due to atmospheric deposition and nitrogen leaching and runoff decreased over the 3 years, dropping from 15.1% in 2019 to 14.5% in 2020 and slightly further to 14.5% in 2021. Contributions from straw incorporation and manure return to total agricultural N_2O emissions were relatively small, with their combined proportion not exceeding 10%. Straw

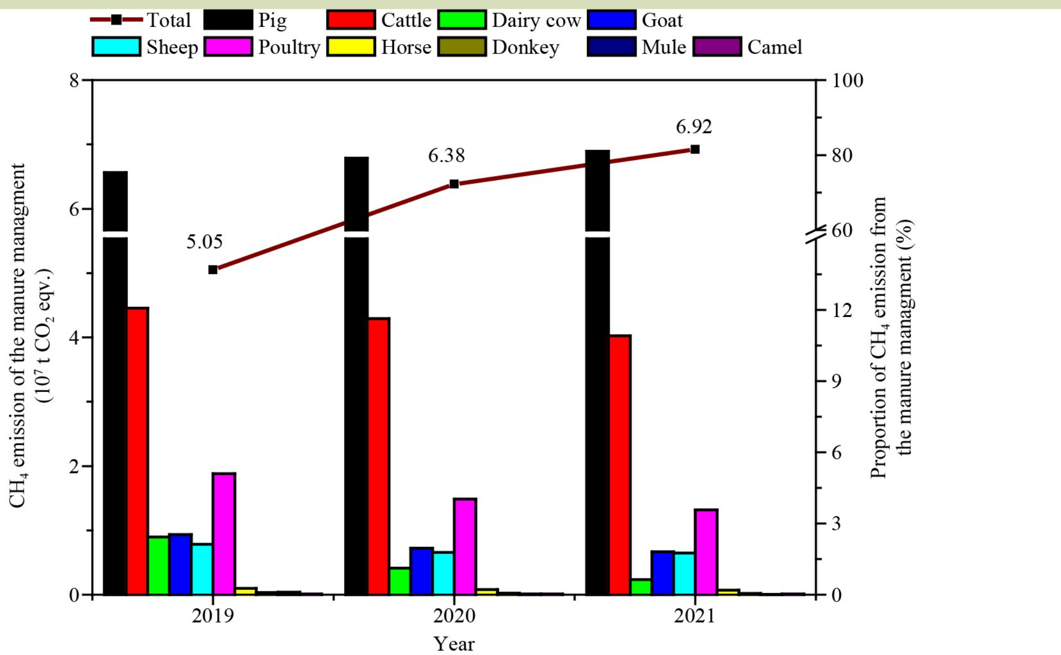


Fig. 4 Total CH_4 emissions and proportion of CH_4 emissions from manure management of livestock in China in 2019 to 2021

incorporation had an average annual contribution of 6.4%, showing a slightly increasing trend. Manure return had an average annual proportion of 2.1%, having an increasing trend, which was related to the increase in livestock production in the past 2 years.

CO_2 emissions in 2019 amounted to 1.34×10^8 t, and decreased to 1.28×10^8 t in 2020, representing an annual decrease of 4.6% (Fig. 3). In 2021, CO_2 emissions increased to 1.31×10^8 t, marking a 2.2% annual increase. Notably, diesel use and agricultural electricity consumption were the primary sources of CO_2 emissions, averaging 32% and 29% of total CO_2 emissions, respectively. Following these, the use of plastic mulching and pesticides contributed to CO_2 emissions, accounting for an average of 20% and 19%, respectively.

3.3 Spatial characteristics of NGHGE from agriculture in China from 2019 to 2021

Agriculture in China has significant regional differences in crop and livestock production, resulting in distinct regional characteristics in NGHGE from agriculture. Table 2 shows the distribution of NGHGE from agriculture in China from 2019 to 2021. Heilongjiang Province was the highest emitter of agricultural NGHGE. Over the 3 years, its emissions were 6.16×10^7, 6.03×10^7, and 6.34×10^7 t (CO_2 eqv.), accounting for 8.44%, 8.25%, and 8.56% of NGHGE from agriculture in China, respectively. Hunan, Guangdong, and Sichuan Provinces follow closely, with annual NGHGE exceeding 4.00×10^7 t. Regionally, areas with higher NGHGE are mainly concentrated in the northeastern, central, southern, and southwestern, in significant agricultural regions. Conversely, less-developed agricultural regions in the western and central parts of the country had relatively lower net emissions. Within the same province, the trends in emissions from different sources within the total GHG emissions remain relatively stable. However, there were variations in the source contributions among different provinces in the same year.

Figure 5 shows the source contributions to total GHG emissions for the top four provinces in NGHGE in 2020: Heilongjiang, Guangdong, Hunan, and Sichuan. In Heilongjiang Province, rice production CH_4 emissions make up the largest proportion at 45%, followed by 20% from enteric fermentation in large ruminant animals. This suggests that in Heilongjiang Province, agricultural GHG emissions were primarily driven by rice production, with large ruminant production as the second-largest contributor. In Guangdong Province, rice production CH_4 emissions dominated at 45%, while agricultural electricity and fertilizer use consumption contributed 13% and 12%, respectively, emphasizing the significance of crop production, particularly rice, and the application of electricity in agriculture-related emissions. The source structure in Hunan Province resembles that in Heilongjiang, with both heavily reliant on CH_4 emissions from rice production and enteric fermentation. Additionally, attention is drawn to Hunan CH_4 emissions from manure management, indicating that GHG emissions in Hunan primarily stem from rice and livestock production. In Sichuan Province, CH_4 emissions from enteric fermentation were the highest at 39%, followed by 18% from rice production, and CH_4 and N_2O emissions from manure management each contribute 13% and 10%, respectively. This highlights that GHG emissions from agriculture in Sichuan were primarily generated by livestock production, with rice production having only a supplementary contribution. The source structure characteristics of these four provinces demonstrate that the magnitude of agricultural NGHGE is closely related to the scale of agricultural production, the scale of crop production, the geographical distribution of livestock production, and the degree of agricultural modernization.

Due to variations in agricultural production across different regions, distinct GHG emission structures were observed. Table 3 presents the regional characteristics of CH_4 emissions. Hunan had the highest CH_4 emissions, averaging more than 4.00×10^7 t CO_2 eqv. per year, accounting for 8% to 9% of the total CH_4 emissions. The second tier of agricultural CH_4 emissions includes Sichuan, Inner Mongolia, and Heilongjiang. Regions in northern, northeast, central and southern China also had relatively high CH_4 emissions. This suggests that CH_4 emissions were primarily concentrated in significant agricultural provinces. The different production structures in these agricultural provinces resulted in distinct primary sources of CH_4 emissions, adding to their uniqueness.

The emission source structures of provinces with higher CH_4 emissions indicate that CH_4 emissions primarily resulted from rice production in Hunan (Fig. 6(a)). The percentage of CH_4 emissions from rice production in Hunan was the largest. Over the 3 years, the CH_4 emissions from rice production in Hunan constituted 66.2%, 63.2%, and 61.6% of the total CH_4 emissions, respectively. From 2019 to 2021, there was a decreasing trend in the proportion of CH_4 emissions from rice production in Hunan, while emissions from enteric fermentation remained relatively stable, accounting for about 21% each year. The proportion of CH_4 emissions from manure management increased annually, rising from 12.7% in 2019 to 17.2% in 2021. In Inner Mongolia, the CH_4 emissions in 2019 were 2.91×10^7 t CO_2 eqv. CH_4 emissions decreased by 6.1% in

Table 2 NGHGE and carbon footprint intensity in agriculture in China

Region	NGHGE ($\times 10^6$ t C eqv.)			Gross agricultural production ($\times 10^{11}$ yuan)			Carbon footprint intensity ($\times 10^4$ t C eqv. per 10^8 yuan)		
	2019	2020	2021	2019	2020	2021	2019	2020	2021
Qinghai	8.05	4.13	4.15	0.181	0.189	0.205	4.44	2.19↓	2.03↓
Xizang	3.81	4.26	4.40	0.0949	0.104	0.115	4.01	4.10↑	3.82↓
Shanghai	1.54	1.52	1.50	0.146	0.138	0.145	1.06	1.10↑	1.03↓
Jilin	7.12	7.02	7.24	1.01	1.23	1.30	0.70	0.57↓	0.56↓
Inner Mongolia	10.2	9.52	10.0	1.61	1.70	1.88	0.64	0.56↓	0.53↓
Ningxia	1.67	1.68	2.10	0.331	0.398	0.413	0.50	0.42↓	0.51↑
Jiangxi	7.43	8.23	7.95	1.62	1.69	1.80	0.46	0.49	0.44↓
Heilongjiang	16.8	16.4	17.2	3.77	4.04	4.10	0.44	0.41↓	0.42↑
Gansu	5.59	6.10	6.31	1.31	1.43	1.62	0.43	0.43	0.39↓
Hunan	12.3	13.0	13.3	3.05	3.36	3.53	0.40	0.34↓	0.38↑
Guangdong	12.0	11.8	11.8	3.53	3.77	3.95	0.34	0.31↓	0.30↓
Anhui	7.85	8.07	8.11	2.37	2.53	2.80	0.33	0.32↓	0.29↓
Zhejiang	5.15	5.35	4.53	1.60	1.59	1.70	0.32	0.34↑	0.27↓
Yunnan	8.65	8.77	8.94	2.68	2.90	3.44	0.32	0.30↓	0.26↑
Xinjiang	8.11	7.17	9.17	2.62	2.94	3.49	0.31	0.24↓	0.26↓
Liaoning	5.42	5.90	5.60	1.91	2.06	2.22	0.28	0.29↑	0.25↓
Guangxi	8.17	8.48	8.06	3.10	3.27	3.69	0.26	0.26	0.22↓
Sichuan	11.2	11.8	11.7	4.40	4.70	5.09	0.26	0.25↓	0.23↓
Shanxi	2.33	3.07	2.64	0.937	1.08	1.22	0.25	0.29	0.22↓
Tianjin	0.497	0.489	0.589	0.203	0.229	0.258	0.25	0.21↓	0.23↑
Hainan	1.95	2.14	1.58	0.82	0.875	1.05	0.24	0.25↑	0.15↓
Hubei	7.73	7.66	8.15	3.26	3.49	3.91	0.24	0.22↓	0.21↓
Jiangsu	9.04	9.53	9.69	3.83	4.10	4.43	0.24	0.23↓	0.22
Fujian	3.81	3.87	3.90	1.77	1.82	1.91	0.21	0.21	0.20↓
Chongqing	2.94	3.18	3.17	1.40	1.60	1.76	0.21	0.20↓	0.18↓
Hebei	6.37	6.59	6.91	3.11	3.41	3.65	0.20	0.19↓	0.19
Beijing	0.197	0.224	0.230	0.102	0.108	0.123	0.19	0.21↑	0.19↓
Guizhou	4.33	4.53	4.45	2.54	2.78	3.12	0.17	0.16↓	0.14↓
Henan	8.58	8.55	8.91	5.41	6.24	6.56	0.16	0.14↓	0.14
Shandong	7.15	6.80	6.61	4.91	5.17	5.81	0.15	0.13↓	0.11↓
Shaanxi	2.86	3.37	2.91	2.45	2.81	3.04	0.12	0.12	0.10↓
Total	199	199	202	66.1	71.8	78.3	0.30	0.28↓	0.26↓

2020 but slightly increased by 3.9% in 2021 (Fig. 6(b)). In Inner Mongolia, CH_4 emissions were primarily from enteric fermentation, and the proportion of this source in total CH_4 emissions increased over the 3 years, reaching 91.3%, 91.4%, and 91.6%, respectively. In both Heilongjiang and Sichuan, CH_4 emissions from rice production and enteric fermentation contributed significantly to total CH_4 emissions. In Heilongjiang, rice production accounted for a larger proportion, especially in 2020 when they were 66.9% of total emissions, whereas emissions from enteric fermentation significantly decreased. In 2021, emissions from enteric fermentation and manure management increased compared to 2020. CH_4 emissions from rice production in Sichuan had minimal variation over the 3 years. In 2020, CH_4 emissions

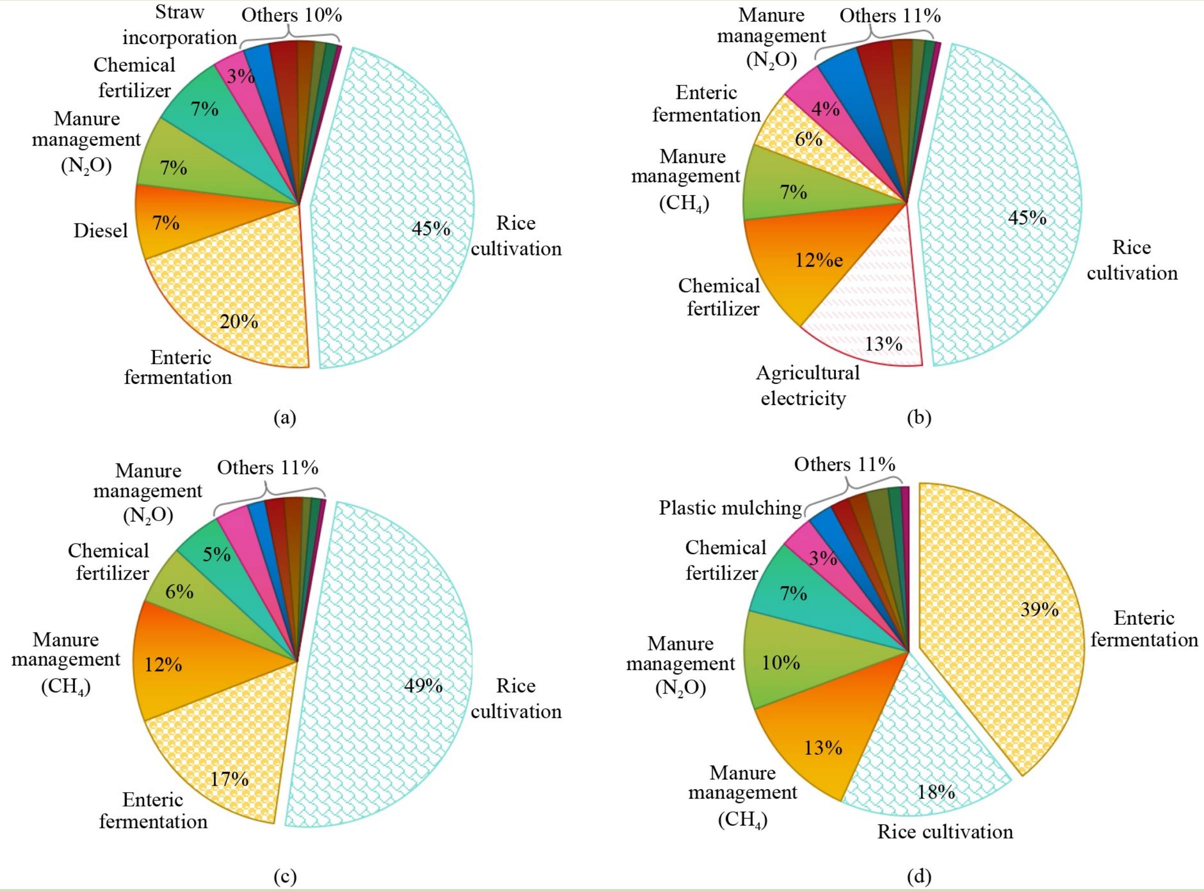


Fig. 5 Proportion of different GHG emission sources from agriculture for the top four NGHGE provinces in China in 2020. (a) Heilongjiang, (b) Guangdong, (c) Hunan, and (d) Sichuan.

from enteric fermentation significantly increased, but in 2021, they were comparable to the 2019 levels, while emissions from manure management had an increasing trend each year.

The regional characteristics of N_2O emissions are also evident from Table 3. Henan, Guangxi, Sichuan, and Shandong are among the top regions in terms of N_2O emissions, with annual emissions exceeding 9.00×10^6 t CO_2 eqv. each. These regions are also major agricultural provinces in central, southern, southwestern and eastern China. In contrast, Tianjin, Shanghai, and Beijing had annual N_2O emissions of less than 1.00×10^6 t CO_2 eqv. each, which is a consequence of the lower proportion of agriculture in these areas.

The emission source structure of N_2O in Henan, Guangxi, Sichuan, and Shandong for 2020 is shown in Fig. 7. The use of nitrogen fertilizers and manure management are the main sources of N_2O emissions. In Henan and Guangxi, the use of nitrogen fertilizers ranked first, while in Sichuan and Shandong, N_2O emissions from manure management were the highest. The proportion of different emission sources in the structure clearly reflects the differences in agricultural production structures in different provinces, where Henan relies on both crop and livestock production as the main sources of N_2O emissions, Guangxi N_2O emissions were primarily driven by crop production, and in Sichuan and Shandong, livestock production was the most significant source of N_2O emissions. Regions with high N_2O emissions are mainly concentrated in major agricultural provinces involved in both crop and livestock production.

Regional features of CO_2 emissions are given in Table 3. CO_2 emissions were primarily concentrated in the central and southern coastal regions of China, with Jiangsu being the most prominent, with emissions exceeding 1.00×10^7 t, accounting for approximately 9% of the total national CO_2 emissions. Next were Guangdong, Zhejiang, Shandong, and Hebei, with emissions all exceeding 8.00×10^6 t (except for Zhejiang in

Table 3 GHG emissions from agriculture in China in 2019 to 2021

Region	CH_4 emission ($\times 10^7$ t CO_2 eqv.)			CO_2 emission ($\times 10^6$ t CO_2 eqv.)			N_2O emission ($\times 10^6$ t CO_2 eqv.)			GHG emission ($\times 10^7$t CO_2 eqv.)		
	2019	2020	2021	2019	2020	2021	2019	2020	2021	2019	2020	2021
Henan	2.26	2.41	2.54	8.43	6.67	6.70	12.4	12.5	12.5	4.34	4.32	4.46
Hubei	2.30	2.23	2.47	4.28	4.22	4.17	8.11	8.78	8.32	3.54	3.53	3.71
Hunan	4.10	4.40	4.49	4.48	4.41	4.18	8.17	7.81	8.60	5.37	5.63	5.77
Guangdong	2.62	2.56	2.60	9.46	9.45	9.05	8.80	8.79	8.64	4.45	4.39	4.37
Guangxi	2.06	2.17	2.09	3.53	3.58	3.75	9.97	10.1	9.23	3.41	3.54	3.38
Hunan	0.296	0.519	0.316	1.12	1.22	1.23	3.02	1.41	1.36	0.709	0.782	0.574
Shandong	1.53	1.45	1.45	9.06	8.71	8.64	9.98	9.77	9.23	3.43	3.30	3.23
Jiangsu	1.76	1.89	1.95	11.8	12.0	12.1	7.64	7.90	7.83	3.70	3.88	3.94
Shanghai	0.0832	0.0816	0.0883	4.56	4.49	4.37	0.244	0.245	0.216	0.563	0.555	0.547
Zhejiang	0.760	0.836	0.832	9.24	9.23	6.62	2.50	2.46	2.10	1.93	2.01	1.70
Anhui	1.99	2.08	2.11	4.79	4.73	5.14	7.72	7.74	7.17	3.24	3.32	3.34
Fujian	0.563	0.603	0.624	4.76	4.65	4.64	3.96	3.90	3.80	1.43	1.46	1.47
Jiangxi	2.64	2.95	2.83	2.88	2.74	2.88	4.04	4.23	4.27	3.34	3.64	3.54
Guizhou	1.34	1.41	1.36	1.26	1.29	1.46	3.61	3.62	3.56	1.83	1.90	1.86
Yunnan	2.43	2.47	2.56	3.73	3.65	3.69	8.61	8.73	8.48	3.66	3.71	3.77
Xizang	1.19	1.24	1.29	1.63	1.63	1.63	0.440	1.66	1.67	1.40	1.57	1.62
Chongqing	0.657	0.719	0.713	1.53	1.52	1.52	2.71	2.94	2.98	1.08	1.16	1.16
Sichuan	3.03	3.22	3.19	4.25	4.25	4.28	9.97	10.0	9.93	4.45	4.64	4.61
Beijing	0.0177	0.0209	0.0245	0.418	0.425	0.414	0.111	0.140	0.161	0.0706	0.0773	0.0820
Tianjin	0.106	0.0945	0.127	0.319	0.401	0.421	0.440	0.446	0.464	0.182	0.179	0.216
Hebei	1.026	0.959	1.16	8.59	8.63	8.48	5.33	6.14	6.11	2.42	2.44	2.62
Shanxi	0.493	0.766	0.565	2.09	2.11	2.33	2.38	2.37	2.58	0.94	1.21	1.06
Inner Mongolia	2.91	2.74	2.84	4.12	4.12	4.38	7.25	6.48	6.98	4.05	3.80	3.98
Liaoning	0.904	1.08	0.967	3.86	3.77	3.81	5.02	5.06	5.05	1.79	1.97	1.85
Jilin	1.06	1.04	1.13	3.12	3.13	3.21	6.62	6.43	6.23	2.03	1.99	2.07
Heilongjiang	2.87	2.71	2.99	5.07	5.03	4.91	8.09	8.27	8.51	4.19	4.04	4.33
Xinjiang	1.87	1.94	2.18	7.60	3.49	8.22	5.29	5.27	5.49	3.16	2.82	3.56
Ningxia	0.450	0.461	0.596	0.807	0.810	0.821	1.04	0.995	1.17	0.635	0.642	0.795
Gansu	1.24	1.47	1.44	3.99	4.26	4.28	2.62	2.95	2.86	1.90	2.19	2.15
Shaanxi	0.590	0.610	0.611	3.26	3.20	3.27	3.69	3.70	3.69	1.29	1.30	1.31
Qinghai	2.78	1.33	1.32	0.315	0.308	0.323	1.31	1.55	1.55	2.94	1.51	1.51
Total	47.9	48.5	49.5	134	128	131	161	162	161	77.5	77.5	78.6

2021). In 2021, Xinjiang CO_2 emissions also exceeded 8.00×10^6 t. In contrast, the northern, northwestern, central and southwestern regions of China, especially Beijing, Chongqing, Guizhou, Hainan, Ningxia, Qinghai, Tianjin, and Xizang had lower CO_2 emissions, all below 2.00×10^6 t.

Figure 8 shows the 10 regions that had the highest CO_2 emissions, which collectively accounted for over 50% of the total national emissions. It also shows the emission source structure. The top three regions, Jiangsu, Guangdong, and Shanghai, primarily emitted CO_2 from agricultural electricity usage. Diesel and agricultural electricity usage contributed significantly to the emissions in Zhejiang and Fujian. The main source of CO_2 emissions from Shandong is the use of

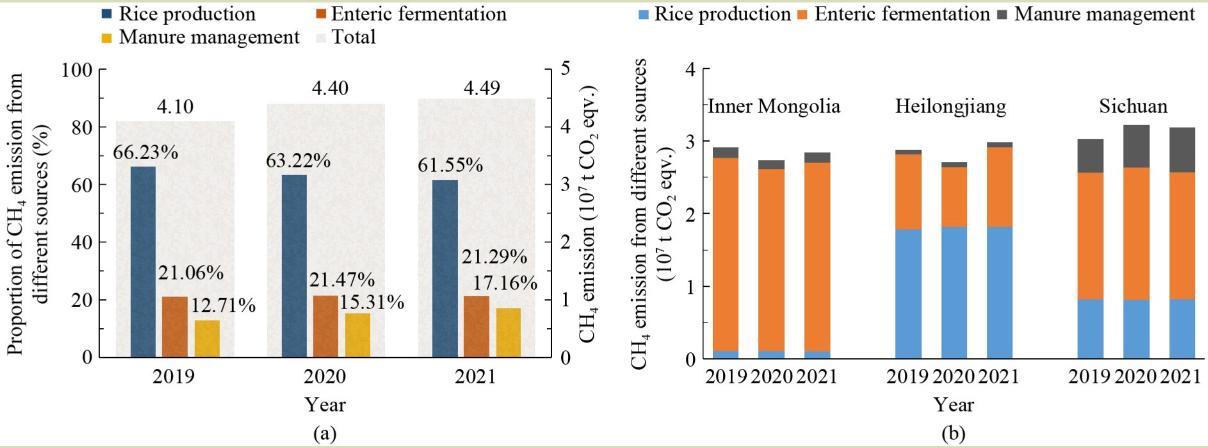


Fig. 6 The original characteristics of CH_4 emission in Hunan (a), Inner Mongolia, Heilongjiang, and Sichuan (b) from 2019 to 2021.

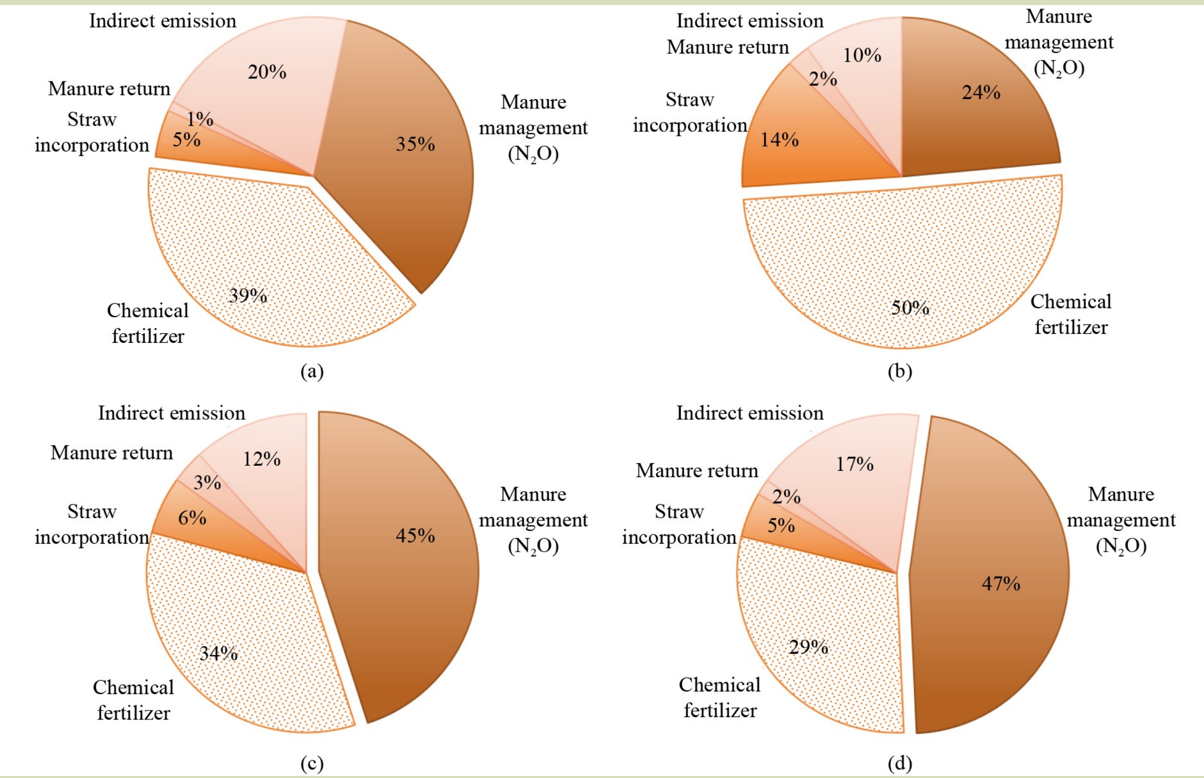


Fig. 7 Proportion of different N_2O emission sources of top four N_2O regions in China agriculture in 2020. (a) Henan, (b) Guangxi, (c) Sichuan, and (d) Shandong.

agricultural plastic mulching. Hebei and Heilongjiang relied on diesel usage as the primary contributor to CO_2 emissions. Henan and Anhui had a significant reliance on diesel and pesticide usage. It is evident from these data that the emission characteristics of CO_2 in the different provinces were closely related to their agricultural production structure, production methods, and level of modernization.

To evaluate whether irrigation electricity usage is a good representation of the entire agricultural electricity usage, we calculated the GHG emissions resulting from agricultural irrigation electricity usage in various provincial regions. There

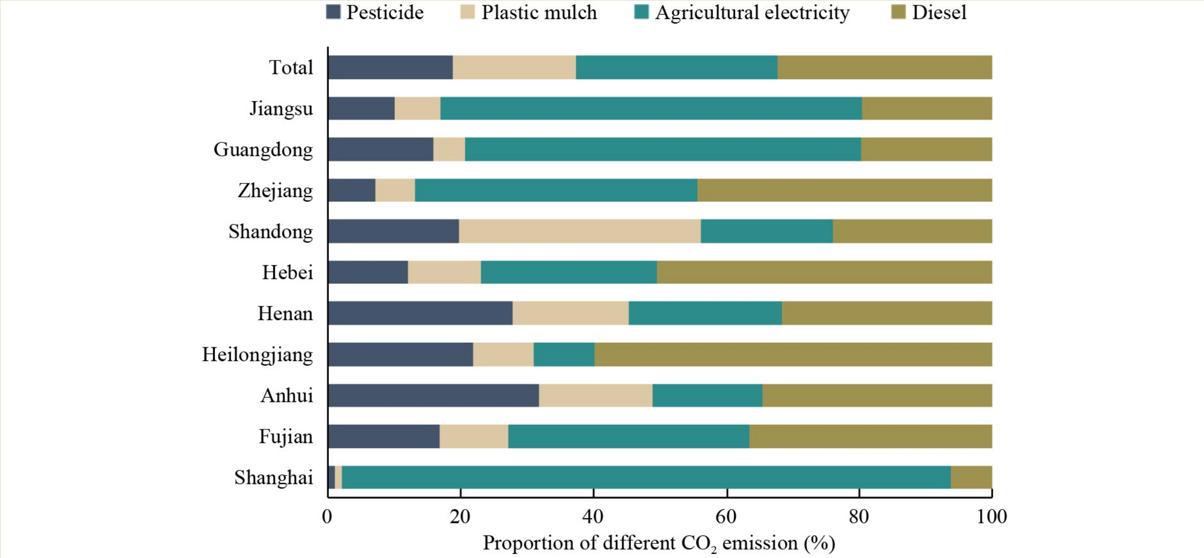


Fig. 8 Proportion of different CO_2 emission sources from agriculture in 10 regions of China with the greatest CO_2 emissions in 2020.

were significant differences in the proportion of irrigation in total agricultural electricity usage in different regions, with slight variations observed over the 3 years in most provinces. The results for Heilongjiang had a significant deviation, possibly due to inaccuracies in the algorithm used for calculating agricultural electricity usage. On a national average, irrigation electricity accounts for around 30% of agricultural electricity usage, indicating that a singular focus on calculating irrigation electricity usage is not comprehensive.

3.4 Comparison of carbon footprint intensity in China from 2019 to 2021

The carbon footprint intensity of the whole of China and 31 regions from 2019 to 2021 was calculated according to NGHGE (as C eqv.) and the total agricultural production value, and the results are shown in Table 2. The results indicate a decreasing trend in the overall carbon footprint intensity of the nation from 2019 to 2021, with decreases of 6.6% and 7.1% respectively. Qinghai, Xizang, and Shanghai were consistently the highest in terms of carbon footprint intensity at the national level over the 3 years. This is closely connected with regional variation in economic development in China, as well as the modernization of agriculture. The eastern and central regions had greater efficiency in agricultural production, whereas the western region relied predominantly on low-efficient agricultural production and material utilization. This discrepancy leads to a higher agricultural carbon intensity in the western region compared to the eastern and central regions. Compared to 2019, the carbon footprint intensity of the majority of regions had varying degrees of decline in both 2020 and 2021.

4 Discussion

4.1 CH_4 emissions

CH_4 emissions mainly originate from enteric fermentation, rice production and manure management. Enteric fermentation and rice production were the main sources of CH_4 emissions, accounting for over 85% of the total CH_4 emissions. The proportion of these emission sources in total methane emissions varies based on different agricultural production forms, scales, and structures in different regions. Hunan Province leads in terms of rice production area, CH_4 emission factors and CH_4 emissions of all regions. CH_4 emissions from rice production account for over 60% of the total CH_4 emissions in Hunan. Although the proportion of CH_4 emissions from rice production decreased from 2019 to 2021 in Hunan, CH_4 emissions from manure management increased each year (12.7%, 15.3%, and 17.2% in 2019, 2020, and 2021, respectively), keeping Hunan as the greatest emitter of CH_4 of all regions. This illustrates the close relationship between the sources of CH_4 emissions and the industrial structure of the region. Pig manure management contributes the most to CH_4 emissions, meaning that the scale of pig production has a significant impact on methane emissions from manure management. In 2020, there was a significant increase in methane emissions from manure management in Hunan due to the impact of African swine fever in 2019 and adverse weather conditions, leading to severe damage to pig farms. In September 2019, the national government implemented a series of pig production subsidies, which gradually led to the recovery and subsequent annual increase in the pig population over the next 2 years, thereby increasing the proportion of CH_4 emissions from manure management.

To assess the impact of pig and cattle production on GHG emissions, the carbon footprints per head for pig and cattle production processes were calculated, taking into account CH_4 emissions from enteric fermentation, N_2O emissions from manure return, CH_4 and N_2O emissions from manure management, as well as indirect N_2O emissions from manure, based on data from 2020. The results, as shown in Table 4, reveal that the carbon footprint of cattle production is over five times larger than that of pig production. Also, enteric fermentation emissions from cattle account for over half of the total GHG emissions in this species, with some provinces reaching 70% to 80%, and a few even reaching 90%. However, pig manure management contributes significantly to GHG emissions, with the maximum contribution reaching 87%.

4.2 N_2O emission

The use of nitrogen fertilizers and manure management were the two major sources of N_2O emissions. The proportion of emission sources in total N_2O emissions varied depending on the agricultural production structure and scale of different provinces. For example, in Henan, in 2020, N_2O emissions were 1.25×10^7 t CO_2 eqv., with nitrogen fertilizer use accounting for 39% and N_2O emissions from manure management contributing 35%. This indicates that both crop and livestock production in Henan had impacts on N_2O emissions. In Guangxi, nitrogen fertilizer use accounted for 50%, highlighting the need to focus on reducing N_2O emissions by addressing the use of nitrogen fertilizers in agriculture. Utilizing the advantages of straw production and manure management in Guangxi to increase the proportion of straw and manure incorporation into the soil and exploring the use of alternative fertilizer application techniques are potential strategies. In Sichuan and Shandong, N_2O emissions from manure management accounted for nearly half of the total emissions. Therefore, researching factors such as manure management practices, nitrification inhibitors, and their impact on reducing N_2O emissions, as well as improving manure management practices, are crucial methods for reducing N_2O emissions.

Table 4 Carbon footprint from pigs and cattle in agriculture in China

Region	CF from pigs (t per head)	CF from cattle (t per head)	Region	CF from pigs (t per head)	CF from cows (t per head)
Henan	0.1054	0.5835	Chongqing	0.0713	0.5290
Hubei	0.0964	0.4915	Sichuan	0.0653	0.4923
Hunan	0.0941	0.5247	Beijing	0.1605	0.4903
Guangdong	0.0824	0.5226	Tianjin	0.0739	0.4914
Guangxi	0.0946	0.5180	Hebei	0.0534	0.4416
Hainan	0.1116	0.8909	Shanxi	0.0632	0.7951
Shandong	0.0952	0.4951	Inner Mongolia	0.0637	0.5040
Jiangsu	0.0823	0.5482	Liaoning	0.0394	0.5747
Shanghai	0.0925	0.4644	Jilin	0.0452	0.4805
Zhejiang	0.1027	0.6925	Heilongjiang	0.0508	0.4709
Anhui	0.0722	0.5268	Xinjiang	0.0446	0.4835
Fujian	0.0771	0.5114	Ningxia	0.0543	0.5147
Jiangxi	0.0774	0.5829	Gansu	0.0557	0.5372
Guizhou	0.0774	0.4960	Shaanxi	0.0514	0.5298
Yunnan	0.0851	0.4904	Qinghai	0.0948	0.5243
Xizang	0.0631	0.4834			

Note: CF is the carbon footprint over the life span of livestock.

Although straw incorporation and manure return contribute to GHG emissions to some extent, their combined contribution to N_2O emissions was less than 10%. Additionally, during these processes, carbon is sequestered in the soil, thereby increasing soil carbon stocks and mitigating agricultural NGHGE. Gao and Serrenho[22] analyzed the global GHG emissions from fertilizer production and use in 2019 and found that the emissions from mineral fertilizer production and use amounted to 1.31 Gt CO_2 eqv. About one-third of these emissions were from the production process, and approximately two-thirds occurred during the application of fertilizers in agricultural fields. Since the GHG emissions per unit of nitrogen are the same, whether from synthetic fertilizers, straw incorporation or manure return, reducing the use of mineral fertilizers and replacing them with straw incorporation and manure return cannot only decrease carbon emissions from fertilizer production and transportation but also increase soil carbon stocks, promoting resource recycling. The key to successfully replacing mineral fertilizers lies in maximizing the efficiency of nitrogen utilization in straw incorporation and manure return. At the same time, converting straw combustion into electricity is also an effective way to achieve biomass energy conversion, which can produce significant economic benefits[23].

4.3 CO_2 emissions

CO_2 emissions account for nearly 20% of agricultural NGHGE in China. It is worth noting that different provinces have significant variations in the proportion of CO_2 emissions due to different inputs of agricultural materials and energy. Therefore, region-specific GHG reduction strategies can be developed based on the source contribution to emissions. In regions like South China and the coastal areas of eastern China, such as Jiangsu, Guangdong, and Shanghai, where agriculture is highly modernized, carbon emissions from agricultural electricity usage deserve particular attention. Additionally, since power generation in China still relies heavily on coal, an increase in agricultural electricity usage will inevitably lead to increased GHG emissions. Therefore, transitioning to cleaner energy sources like tidal power, wind energy, solar energy, and biomass energy to replace coal-fired power generation can significantly reduce CO_2 emissions.

Also, implementing measures such as tiered electricity pricing to control agricultural electricity consumption can contribute significantly to mitigating agricultural CO_2 emissions. Currently, specific electricity usage data for different stages of agriculture are challenging to find, except for the proportion of electricity consumed in irrigation for crops like maize, rice, and wheat[24]. Therefore, accumulating more detailed electricity usage data, especially for activities like irrigation, planting, harvesting, fertilization, plowing, and livestock production is essential for developing precise emission reduction policies. In provinces with advanced levels of mechanization and large-scale agriculture, such as Hebei and Heilongjiang, diesel usage becomes a major source of CO_2 emissions. In such regions, improving the thermal efficiency of diesel engines, coordinating agricultural production operations, and maximizing diesel utilization are key strategies to reduce CO_2 emissions. In vegetable-growing provinces like Shandong, the use of agricultural mulch film accounts for a significant portion of CO_2 emissions. Therefore, adopting low-carbon coefficient agricultural mulch films that are reusable and renewable holds practical significance for reducing agricultural CO_2 emissions. For provinces like Henan and Anhui, measures need to be taken to reduce the use of diesel and pesticides. For example, adopting automated physical laser weeding instead of pesticides not only allows for precise weed control but also reduces pesticide usage, enhancing food safety.

4.4 NGHGE

From 2019 to 2021, NGHGE from agriculture in China exceeded 7.00×10^8 t CO_2 eqv., showing an increasing trend between years. Among these gases, CH_4 emissions were significantly higher than the other two gases, increasing at a rate of more than 2% annually and reaching 4.95×10^8 t CO_2 eqv. in 2021. The use of nitrogen fertilizers and manure management were the main sources of N_2O emissions, accounting for over 75% of the total N_2O emissions. CO_2 emissions mainly came from the use of diesel and agricultural electricity, which together accounted for around 60% of the total CO_2 emissions. Changes in soil carbon stocks contributed to a reduction of NGHGE by approximately 6.5%.

The study found that animal intestinal fermentation, rice production, the use of nitrogen fertilizers, and manure management were significant contributors to agricultural GHG emissions. Hunan, due to its extensive rice production and high methane emission factors, was the province with the highest CH_4 emissions in the country, accounting for about 9% of national agricultural CH_4 emissions, with annual emissions exceeding 4.10×10^7 t CO_2 eqv. Henan, a major province in crop and livestock production, had the highest N_2O emissions due to the use of nitrogen fertilizers and manure management, with annual emissions exceeding 1.20×10^7 t CO_2 eqv. and showing an increasing trend, accounting for over 7.6% of national agricultural N_2O emissions. Jiangsu, with its higher agricultural electricity consumption, became the largest contributor to CO_2 emissions.

Hunan was the greatest gross GHG emitter, but soil carbon sequestration offset 18.3% of its GHG emissions. Heilongjiang, however, despite not being the greatest gross GHG emitter, it was the greatest net GHG emitter exceeding 6.00×10^7 t CO_2 eqv. This indicates a close relationship between the emissions of agricultural GHG and the structure of emission sources and regional variation. Therefore, it is essential to design emission reduction policies to the specific characteristics of GHG emissions for specific regions. Additionally, accurate calculation of soil carbon stocks is necessary to determine the net emissions of agricultural GHG.

Fan et al.[25] reported a national GHG emissions total of 8.56×10^8 t CO_2 eqv. in 2020, which differs from the 7.30×10^8 t CO_2 eqv. obtained in the present study. This discrepancy may be due to differences in accounting boundaries and factors, particularly as the study of Fan et al.[25] accounted for the impacts of straw burning and plowing, but did not include GHG emissions from agricultural electricity use. Previous research rarely considered carbon emissions from agricultural electricity use[25]), or only focused on irrigation electricity (such as Tian et al.[26]). Our study found that irrigation electricity accounted for an average of 26% to 36% of agricultural electricity use. Therefore, not including carbon emissions from the entire spectrum of agricultural electricity use could lead to inaccuracies in carbon accounting. Comprehensive statistics on electricity consumption for different production forms (e.g., irrigation, mechanized cropping, lighting, and livestock production) in agricultural statistical data would be highly beneficial for defining emission reduction directions and identifying effective mitigation measures.

4.5 Soil carbon sequestration

From 2019 to 2021, the proportion of agricultural carbon emissions that were sequestered and fixed in the soil through organic carbon accumulation was just over 6%. Previous studies on agricultural carbon emissions accounting have rarely considered the role of agricultural carbon sinks[2–6]. Even when considered, the focus has mostly been on estimating carbon storage in crops[22]. However, crops, due to their short harvest periods, often have their increased biomass decomposed and released back into the atmosphere shortly after, leading to the conclusion, in most studies, that crop biomass carbon sinks are negligible. However, agricultural soil carbon sinks are much larger than natural vegetation carbon sinks. Therefore, carbon sequestration in agricultural ecosystems primarily originates from soil carbon sequestration. Agricultural soil carbon sequestration occurs when crops absorb CO_2 through photosynthesis and store it in the form of organic matter in soil carbon reservoirs, thereby reducing atmospheric CO_2 concentrations. Additionally, increasing soil organic carbon can enhance soil fertility and crop yields. This study found that the role of soil carbon sequestration reached over 6% and still has ample scope to increase. This can be achieved through appropriate agricultural management practices that enhance soil carbon fixation, such as no-till cropping and fallow periods to preserve soil carbon reservoirs. Also, China has abundant straw and manure resources, which can be valuable for practices like mulching and replacing mineral fertilizers, turning waste into a valuable resource while simultaneously reducing GHG emissions[26].

4.6 Impact of the pandemic

Since the outbreak of the COVID-19 pandemic at the end of 2019, there have been far-reaching effects on agricultural production, particularly in the livestock industry. Over the course of 3 years, CH_4 emissions from pig manure management increased annually. In particular, in 2020, these emissions increased by 33% compared to the previous year, and in 2021, they increased by 11% compared to the previous year. The significant increase in 2020 can be attributed to the COVID-19 pandemic. Restrictions on the movement of personnel in the supply chain and a substantial decrease in pork procurement by the catering industry on the demand side led to disruptions in both supply and demand. This resulted in a larger pig inventory, with an annual increase of 30% in 2020, which also increased the emissions of CH_4 from manure management. In 2021, as the pandemic gradually eased, the growth rate of pig inventory slowed down, with an annual increase of 11%, leading to a relatively smaller increase in CH_4 emissions.

In comparison to the situation in 2019, before the pandemic, CH_4 emissions from Inner Mongolia, primarily driven by livestock production, decreased by 6% in 2020. This reduction may be attributed to the impact of the COVID-19 pandemic, which disrupted the trade, logistics, and supply chain for livestock, leading to a decrease in livestock production. This, in turn, resulted in reduced CH_4 emissions from both enteric fermentation and manure management, which decreased by 6% and 13%, respectively, compared to 2019. In 2021, as the pandemic situation eased, the livestock trade market reopened, and logistics gradually normalized. Livestock production also rebounded, leading to an increase in CH_4 emissions (a 4% increase for enteric fermentation and a 6% increase for manure management) compared to 2020. The changes in emissions from rice production in 2020 were relatively small, accounting for only 0.1% compared to 2019.

4.7 Implications for low-carbon agriculture

With agricultural modernization, the related GHG emissions caused by increased mechanical usage should receive more attention. The government should strongly promote the concept of agricultural energy conservation, raise awareness of energy conservation throughout society, enact energy conservation policies and incentives adapted to regional development, and encourage the development and application of non-fossil fuel power generation technologies. Efforts should be made to reduce the use of mineral fertilizers, increase the use of green manure and organic fertilizers for soil amendment, and enhance the development and application of soil carbon sequestration techniques. It is essential to establish a comprehensive agricultural statistical database based on artificial intelligence, expand the scope of data collection in agricultural statistical yearbooks and promote the application of big data technology for data collection, recording and analysis throughout the entire agricultural production process. Also, a unified, comprehensive, scientific and reasonable carbon emission factor database should be established based on national conditions in China and its regional characteristics. This database should be regularly updated according to agricultural production timelines to create a more precise agricultural carbon emission accounting system, providing targeted guidance for agricultural production.

5 Limitations and uncertainties

This research has certain limitations. The data on different aspects of agricultural electricity usage, including electricity for crop and livestock production, and various stages of crop production such as irrigation, drainage, seeding, and harvesting, were not detailed enough for analysis of different forms of electricity consumption. This limitation makes it challenging to provide precise policy guidance. The selection of emission factors for pesticides and plastic mulching was somewhat arbitrary. To reduce uncertainty, it would be beneficial to calculate emissions using emission factors specific to different types of pesticides and plastic mulching materials in different provinces. The study did not account for GHG emissions related to the production of fruit, tea, mushroom, and specialty agricultural products. Emissions from the production and transportation of pesticides and fertilizers were not considered, only emissions generated during their use in the fields were calculated. The precise data on the proportion of straw and manure returned to fields was not available, and the choice of 30% for this proportion introduced uncertainty. Additionally, some straw is used for purposes such as animal feed and biomass energy generation, for which there are no relevant carbon emission coefficients. Consequently, emissions related to the handling of straw that is not returned to the fields were not included in the calculations.

6 Conclusions

This analysis reveals some critical aspects of GHG emissions from agriculture in China, offering actionable insights for sustainable practices. The predominance of methane emissions from enteric fermentation, rice production, and pig manure management emerges, with Hunan Province having a close nexus between emissions and regional structures. Livestock dynamics reveal clear carbon footprint disparities, emphasizing the urgent need for targeted interventions. Nitrous oxide challenges demand innovative strategies, while carbon dioxide emissions underscore the importance of region-specific reduction policies and an energy transition toward cleaner sources. The comprehensive GHG assessment from 2019 to 2021 has an increasing trend, exceeding 7.00×10^8 t CO_2 eqv. Regional differences are apparent, with Hunan leading in gross GHG emissions but soil carbon sequestration offsetting a significant portion. Heilongjiang was the greatest net GHG emitter, necessitating specific emission reduction policies. Soil carbon sequestration proves a valuable offset, contributing over 6% to emissions. The study underscores the potential of practices like no-till cropping, fallow periods, and leveraging abundant straw and manure resources for sustainable soil carbon fixation. The ripple effects of the pandemic on pig manure management highlight the vulnerability of industry to external shocks. Implications for low-carbon agriculture center on energy conservation reduced mineral fertilizer use and the imperative of a comprehensive agricultural statistical database. The lessons learned from this analysis could serve as a useful guide for moving toward a greener and more resilient agricultural future.

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Compliance with ethics guidelines

Jianing Tian, Chuanhui Gu, and Yanchao Bai declare that they have no conflicts of interest or financial conflicts to disclose. All applicable institutional and national guidelines for the care and use of animals were followed.

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