

Optimizing crop production toward agricultural carbon neutrality in China

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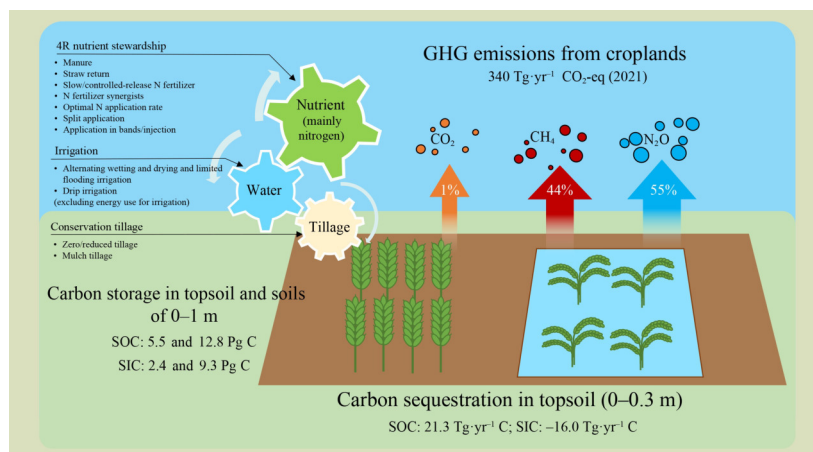
KEYWORDS

Carbon-neutral crop production, field management practices, greenhouse gas emissions, soil carbon sequestration

HIGHLIGHTS

- Chinese cropland is key for achieving carbon neutrality but with challenges to overcome
- Chinese cropland functions as a weak carbon sink but remain major GHG sources.
- GHG emission intensity in crop production has significantly improved.
- Improved management practices have great potential for emission reduction.

GRAPHICAL ABSTRACT



ABSTRACT

Crop production is strategic for food security and climate change mitigation, and can provide a temporary soil carbon sink. There is an ongoing debate about how to optimize crop production in China toward carbon-neutral agriculture. This paper summarizes major carbon budgets in staple crop production in China over recent decades, synthesize reported impacts of available and developing field management practices on greenhouse gas emissions reduction and carbon sink increase. According to recent studies, cropland-based GHG emissions (55% N₂O and 44% CH₄) increased at a rate of 4.3 Tg-yr⁻¹ CO₂-eq from 1990 to 2015 and peaked at 400 Tg CO₂-eq in 2015. Subsequently, there was a substantial decrease of 11.6 Tg-yr⁻¹ CO₂-eq between 2015 and 2021. A similar bell-shape trend has been observed in yield-scaled GHG emission intensity over the years for cereals excluding rice, as rice exhibited a steady decline in yield-scaled emission intensity since 1961. For soil

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C in Chinese cropland, topsoil C represents a huge C pool, containing 5.5 Pg of soil organic carbon (SOC) and 2.4 Pg of soil inorganic carbon (SIC). However, these densities are relatively low globally, indicating a high C sequestration potential. Soil C in cropland has been a weak sink of 5.3 Tg·yr⁻¹ C in China since the 1980s, resulting from the net effect of SOC sequestration (21.3 Tg·yr⁻¹ C) and SIC loss (−16 Tg·yr⁻¹ C), which only offsets 5.7% of simultaneous cropland GHG emissions. Hence, cropland remains consistent and significant GHG sources, even when considering soil C sequestration and excluding related industrial and energy sectors. Fortunately, many reliable management practices have positive effects on emission intensity of crop production, in terms of fertilizer application, irrigation and tillage. However, the path to achieving carbon neutrality in China's cropland is still uncertain and requires further quantitative assessment. Nonetheless, this synthesis highlights that the huge potential, and strong scientific and technical support in low-carbon crop production, for modifying China's food system.

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1 Introduction

To maintain global warming within 1.5 °C above preindustrial levels, the international community has called for the ambitious goal of carbon neutrality (also known as net zero emission of greenhouse gases; GHGs) as part of global sustainable development^[1,2]. As the largest grain producer and GHGs emitter, China's arable and permanent cropland has dual functions in food security and climate change mitigation. GHGs emission from crop production is about 465 Tg CO₂-eq in 2018, and contributes 50.3% of GHG emissions in agriculture and 3.6% of total national GHG emissions (excluding land use, land use change and forestry)^[3]. However, biogeochemical responses to in-field management practices can vary widely across China, basing on site-dependent factors including local climate, topography, soil properties and vegetation. Hence, it is still difficult to optimize crop production toward C neutrality and predict agroecosystems responses at a national scale.

Decarbonization of agriculture through promoting zero- or low-C farming and agricultural management technology innovation has been widely supported in policies and actions toward C neutrality targets. Many technological innovations have been tested for identifying the possible pathways for accomplishing C neutrality via scaling up available cost-effective and climate-smart practices such as improved cropland nutrient management, for example, 4R nutrient stewardship, improved manure application technology, precision agriculture (digitization and smart technologies), using nitrification inhibitors and other sustainable land

management practices such as agroforestry, cover crops, zero/reduced tillage and residue management.

Actually, China has focused on improving crop productivity and reducing environmental costs for many years, introducing policies such as “Soil testing for formulated fertilizer application” (in 2005), “Zero-growth” action plan for mineral fertilizers and pesticides use (in 2015), the promotion of “Integrated management of water and fertilizer” (in 2016–2020), “Conservation tillage action plan for black soils in Northeast China” (in 2020–2025). Recent studies have provided insights into achievable ways to reduce GHG emissions from crop production systems and that will improve knowledge of how it can be accomplished cost-effectively.

Thus, to better understand the role of crop production systems in achieving C neutrality in China, this review summarizes recent major progress in the C management of crop production, and clarifies GHG emissions, soil C storage and the C sink in crop production in China. We describe the available and viable practices to reduce GHG emissions and enhance soil C storage, and then discuss the challenges and requirements for achieving optimized crop production for C neutrality, food security and environmental sustainability in China.

2 Carbon budget of crop production in China

Considering that cropland acts both as a carbon sink and a

carbon source, storing photosynthetic carbon produced by crops in the soil while emitting GHGs into the atmosphere, we assessed the carbon budget of cropland in China from two key perspectives: greenhouse gas emissions and soil carbon sequestration. We define the cropland-based sectors contributing to GHG emissions to include manure applied to soils, burning-crop residues, crop residues, rice cultivation, synthetic fertilizers, and drained cropland with organic soils. All data are from the Food and Agriculture Organization Corporate Statistical Database (FAOSTAT^[4]). Then, GHG emission intensities of crop production, defined as cropland-based GHG emissions per unit of crop yield, were recalculated for rice (mainly paddy) and non-rice cereals (upland) by dividing GHG emissions by grain yield. Non-rice cereals include barley, maize, millet, oats, rye, sorghum and wheat. National-scale studies were also collected to estimate soil carbon storage and the rates of change in SOC and SIC within cropland. In addition, published integrative research and meta-analyses were used to evaluate the impacts of management practices on GHG emissions and SOC. The main data are presented in Table 1.

2.1 GHG emissions from crop production

China is the most populous nation and largest food producer in the world, with increasing agricultural GHG emissions over the past decades, over half of which are attributed to crop production (Fig. 1). According to FAOSTAT, China’s GHG emissions from crop production increased almost 2.2 times between 1961 and 2021, from 158 Tg·yr^{−1} CO₂-eq in 1961 to 340 Tg·yr^{−1} CO₂-eq in 2021, while CH₄ and N₂O contributed 40%–52% and 47%–60% of the GHG emissions during 1990–2021, respectively (Fig. S1). Particularly cropland-based

GHG emissions increased, with a rate of 4.3 Tg·yr^{−1} CO₂-eq from 1990 to 2015 and peaking at 400 Tg·yr^{−1} CO₂-eq in 2015. Subsequently, from this source there was a large decrease of 11.6 Tg·yr^{−1} CO₂-eq between 2015 and 2021, mainly as N₂O (55%) and CH₄ (44%).

Of the cropping sectors, rice production and mineral fertilizers are the top two GHG emitters, contributing for > 80% of the total GHG emissions from all cropping sectors and being responsible for its changes along years (Fig. S2(a)). Including industrial and energy sectors, total GHG emissions was 594.5 Tg·yr^{−1} CO₂-eq in 2021, with machinery use and fertilizer manufacturing also important GHG sources being comparable to rice cultivation and mineral fertilizers (Fig. S2(b)).

2.2 GHG emission intensity of crop production

Over the past 10 years (2012–2021), the average emissions-intensity of cereals excluding rice (upland) was 0.21 kg CO₂-eq per kg grain (denoted as kg·kg^{−1} CO₂-eq) in China, which is 1.14, 1.19 and 1.30 times those of World, USA and EU27, respectively (Fig. 2 and Fig. S3). Meanwhile, the rice emissions-intensity was 0.95 kg·kg^{−1} CO₂-eq in China, much lower than those of World, USA and EU27.

For cereals excluding rice, there were unimodal historical trends of GHG emissions intensity for global and top agricultural countries/group (China, USA and EU27) while the yield kept increasing (Fig. S3). Bell-shaped relationships between emissions-intensity and yield (Fig. 2) indicated that emissions-intensity increased and then decreased as agricultural development proceeded. Specifically, emissions-intensity for cereals excluding rice increased at a rate of

Table 1 GHG emissions and soil C in China’s cropland

Items			Values (soil depth, period)	Reference
GHG emissions* (Tg CO ₂ -eq)		CO ₂	2.7	[4]
		CH ₄	153.6	
		N ₂ O	183.3	
Soil carbon	Organic carbon	Storage	12.8 ± 2.3 (0–1 m)	[5–13]
		(Pg C)	5.5 ± 1.3 (0–0.3 m)	
		Sequestration rate (Tg·yr ^{−1} C)	21.3 ± 5.1 (0–0.3 m, the 1980s–2000s)	[7,14–22]
	Inorganic carbon	Storage	9.30 ± 0.08 (0–1 m, the 2010s)	[23]
		(Pg C)	2.43 ± 0.02 (0–0.3 m, the 2010s)	
		Sequestration rate (Tg·yr ^{−1} C)	−16.0 ± 1.4 (0–0.3 m, the 1980s–2010s)	

Note: *GHG emissions from cropland-based sectors (2021), including manure applied to soils, burning-crop residues, crop residues, rice cultivation, mineral fertilizers and drained cropland with organic soils. GHG emissions largely depend on the definition of the agricultural sector and crop production, leading to differences in results of different studies.

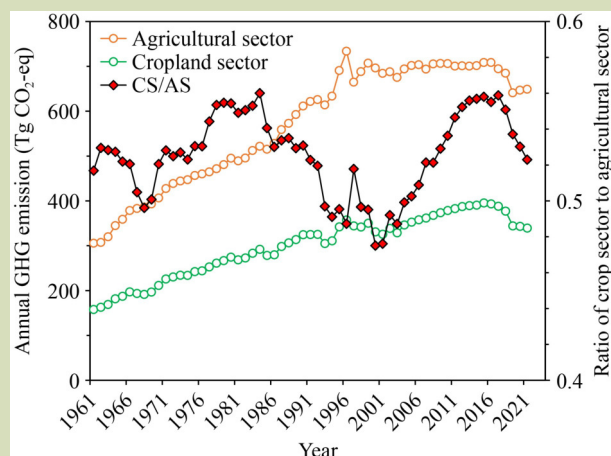


Fig. 1 Historical trends of GHG emissions from agricultural sector (AS), cropping sector (CS) and ratios of GHG emissions from CS to total GHG emissions from AS during 1961–2021. Data from FAOSTAT^[4]. Agricultural sector includes manure applied to soils, burning-crop residues, crop residues, rice cultivation, mineral fertilizers, drained cropland with organic soils, enteric fermentation, manure management and manure left on pasture. Cropping sector includes manure applied to soils, burning-crop residues, crop residues, rice cultivation, mineral fertilizers and drained cropland with organic soils.

3.6 g·kg⁻¹·yr⁻¹ CO₂-eq between 1961 and 2003, peaking at 0.30 kg·kg⁻¹ CO₂-eq in 2003. Subsequently, it decreased at a rate of 6.6 g·kg⁻¹·yr⁻¹ CO₂-eq, reaching 0.16 kg·kg⁻¹ CO₂-eq by 2021. China's emissions-intensity of cereals excluding rice peaked 15 years later and 24% higher than the world. Even though current emissions-intensity of cereals excluding rice in China is comparable to the USA, further progress is still needed in terms of yield improvement.

For rice, the emissions-intensity in China decreased from 2.49 to 0.88 kg·kg⁻¹ CO₂-eq in 2021 at a rate of 17.1 g·kg⁻¹·yr⁻¹ CO₂-eq during 1961–2021, while rice yield increased smoothly (Fig. 2 and Fig. S3). Similar patterns were observed in the world, USA and EU27. The rice yield in China was 1.5 t·ha⁻¹ lower than that in USA, but China's rice emissions-intensity was at a relative low level, being only 65.5% of the USA. Nevertheless, the emissions-intensity of rice were more than 5-fold greater than those of cereals excluding rice, indicating an urgent need to cut rice CH₄ emission globally.

Mineral fertilizers have been the major contributor to the rapid increase in yield in China, while fertilizer overuse is also common and serious, leading to particularly high GHG emissions and other environmental costs^[24]. In context of

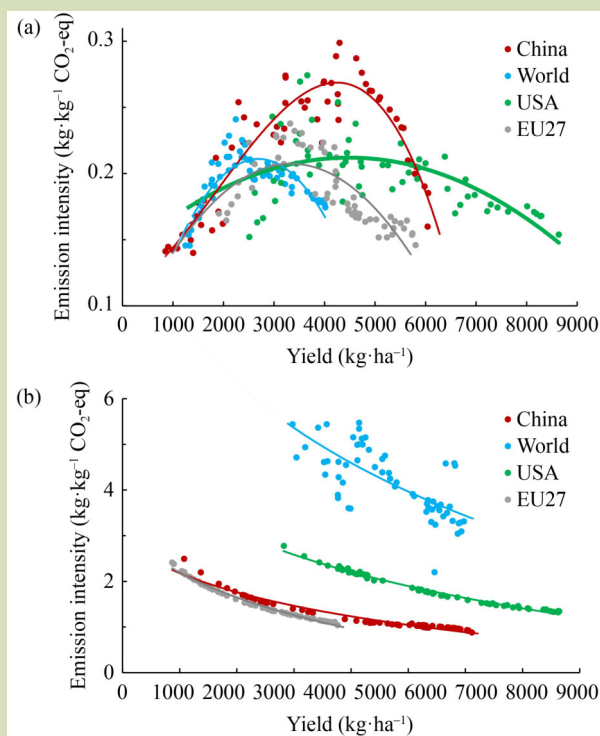


Fig. 2 Historical trends of the relationship between cereal emission intensity and yield for cereals excluding rice (a) and rice (b). Data from FAOSTAT^[4]. EU27 refers to the 27 European Union countries.

these clear trade-offs, the Chinese government has released many strategies toward green agriculture without sacrificing grain yield, including developing high-standard farmland, implementing water-saving projects and extending technologies for soil testing and formulated fertilizer application. Over the last decade, the application rates of mineral fertilizers and pesticides decreased rapidly in China (Figs. S3–S4). Consequently, the cropland nitrogen use efficiency (NUE) increased and cereal GHG emissions intensity decreased gradually, while the grain yield kept increasing.

Nevertheless, this improvement of cropland management still cannot meet the grain demand in China, leaving a gap of 162 Mt that was mainly filled by importing maize (17%), wheat (7%), barley (7%), and soybeans (64%) in 2023^[25]. This makes it difficult to reduce cereal GHG emissions while ensuring the national food supply.

2.3 Soil organic carbon in China's cropland

Enhancing and preserving soil organic matter yields many benefits for both humanity and the environment^[26,27]. SOC

sequestration, which is persistent, has been widely considered to be a cost-effective solution to offset anthropogenic GHG emissions, as it holds about 2–3 times more C than the atmosphere globally^[28,29]. Based on previous studies, current SOC storage is 12.8 ± 2.3 (mean $\pm$ S.D.) Pg C and 5.5 ± 1.3 Pg C in China’s agricultural soils of 0–1 and 0–0.3 m depth, respectively (Table 2), over 70% of which were attributed to upland soils^[7,12]. However, SOC density in China is only $70 \text{ Mg}\cdot\text{ha}^{-1} \text{ C}$, much lower than those in the USA ($94 \text{ Mg}\cdot\text{ha}^{-1} \text{ C}$), Europe ($105 \text{ Mg}\cdot\text{ha}^{-1} \text{ C}$), and Canada ($138 \text{ Mg}\cdot\text{ha}^{-1} \text{ C}$), indicating a large opportunity for increasing SOC sequestration in China^[31], but which will require more efforts to reduce soil carbon decomposition and increase carbon inputs, for example, by planting crops with a high carbon sink and/or adding organic matter.

The international *4-per-1000* initiative for increasing SOC stocks in agricultural soils was launched by France during COP 21 in 2015. It ambitiously suggested that, if the 0.4% goal was achieved to a depth of 0.4 m of all global soils, topsoil C sequestration (estimated to be $3.4 \text{ Pg}\cdot\text{yr}^{-1} \text{ C}$) would offset three-fourths of the net annual C increase of the atmosphere ($4.7 \text{ Pg}\cdot\text{yr}^{-1} \text{ C}$). If SOC increased by 0.4% per year in China, C sequestration as SOC would be $\sim 22.2 \text{ Tg}\cdot\text{yr}^{-1} \text{ C}$ ($\sim 81.2 \text{ Tg}\cdot\text{yr}^{-1} \text{ CO}_2$) in agricultural soils to a depth of 0.3 m, being almost a quarter of total GHG emissions from the cropland sector in 2021 (Fig. 1). Actually, an comparable SOC sequestration rate of $21.3 \text{ Tg}\cdot\text{yr}^{-1} \text{ C}$, with a range of $9.6\text{--}26.0 \text{ Tg}\cdot\text{yr}^{-1} \text{ C}$, was observed in topsoil to 0.3 m deep across China’s cropland

during the 1980s–2000s^[22]. It is clear that China’s agricultural soils are already providing an increasing C sink in recent decades, and the desired annual growth rate of 0.4% in SOC could be achievable in China’s cropland with relative low SOC density compared to other regions globally. However, it is important to note that there is SOC saturation, which should be seriously considered in the assessment of the potential contribution of SOC sequestration to climate change mitigation, otherwise the benefits of SOC sequestration will be overestimated^[32].

2.4 Soil inorganic C in China’s cropland

Since SIC has a slow exchange rate at the air-soil interface and also forms slowly, SIC has usually been ignored in the calculation of the C sink, whether as a source or sink, in spite of its very long turnover time^[33]. However, agricultural activities especially N fertilizer application and liming strongly accelerate SIC cycling, together with acid deposition and climate change^[23,34,35]. SIC storage was $\sim 9.3 \text{ Pg C}$ in China’s cropland soil to 1 m deep in the 2010s, comparable to SOC (12.4 Pg C). Meanwhile, topsoil (to 0.3 m deep) inorganic C storage decreased by 0.48 Pg C during the 1980s–2010s ($16.0 \text{ Tg}\cdot\text{yr}^{-1} \text{ C}$, offsetting $\sim 75\%$ of SOC sequestration)^[23]. Water balance (i.e., surplus of water) and N fertilizer application are major drivers leading to dramatic losses of SIC density in cropland^[36].

Over the next 30 years (2020–2050), estimated topsoil SIC losses could be as high as 3.2 Pg C in China under SSP5-8.5-Low-Ambition-N (high mineral N inputs) policy scenario, owing to rapid N-induced soil acidification in cropland^[37]. However, soil acidification can be largely avoided by improving nutrient management, for example, optimized N application, crop residue return and manure application instead of mineral fertilizer^[38]. In addition, SIC could also be crucial for C sequestration in arid cropland independent of fertilizer use. For example, in North China, pedogenic carbonate contributed more than SOC to soil C accumulation^[39,40].

Clearly, SIC changes and regional-dependent responses to agricultural practices and climate changes should be seriously considered in C budgets and C sequestration in cropland.

3 Current field management practices to archive carbon neutrality in China’s cropland

Determining the potential of cropland to achieve C neutrality is

Table 2 SOC storage and density in cropland soils to 1 m deep in China

SOC stock* (Pg C)	SOC density* (Mg·ha ⁻¹ C)	Reference
15.1	87.1	[5]
12.2	92.2	[6]
13.0	83.3	[7]
15.2	81.6	[8]
12.8**	98.7**	[9]
11.1**	80.4**	[10]
8.7**	91.3**	[11]
11.3**	80.7**	[12]
15.8**	92.1**	[13]

Note: *C storage and density are for soils to 1 m deep. Where this was not measured, C storage in soils to 1 m deep was obtained by multiplying up the results for topsoil to 0.3 m deep, using literature information on the ratio of elements in 0.3 and 1 m deep soils. For C storage, the ratio was set as 2.174, based on global averaged data published by Chabbi et al.^[30]. The same ratio was used for C density. **Estimated values based on the results of topsoil to 0.3 m deep.

challenging^[41]. In part, this is because cropland related emissions appear to be an unavoidable environmental cost of feeding humanity, and reducing GHG emissions must not conflict with ensuring food security^[24,42]. Also, the fact that cropland generates GHG emissions and sequester C at multiple biotic and abiotic sources that are not easy to capture and quantify^[41]. In addition, GHG emission mitigation and C sequestration implementation are technologically difficult and not easily controlled. As discussed below, many papers have highlighted the technical potential to substantially reduce GHG emissions and increase C sequestration, and thereby achieve C neutrality^[43–47].

3.1 Nitrogen management

Given that terrestrial C cycle and GHG emissions are tightly coupled with N, optimal N management becomes a high priority among the many agricultural practices to reduce GHG emissions and increase soil C sequestration. Over the last four decades, N fertilizer use in China has grown threefold and now accounts for nearly one-third of global total N use^[48]. NUE in China's crop production systems is only 0.25–0.40 compared to 0.42 worldwide and 0.65 in North America^[49,50]. High inputs and low resource use efficiency is the dominant cause of agricultural GHG emissions, contributing about 50%^[46,47,51]. Hence, optimizing fertilizer applications to cropland holds significant potential for reducing GHG emissions in China's crop production. In particular the 4R principle for improved crop N management have been widely researched. In the following four subsections we discuss separately the potential of each factor to increase C sequestration and reduce GHG emissions.

3.1.1 Right source: best fertilizer type with appropriate enhanced fertilizer

(1) Nitrogen fertilizer types

GHG emissions from cropland, mainly N_2O , are logically affected by fertilizer types, as is SOC balance. Certainly, N transformations in agricultural soils were largely controlled by N availability (essential substrates), C availability (energy source), soil water content (O_2 diffusion and microbial activity), temperature, soil pH, and microbial community^[52,53]. Application of different N fertilizer types can differentially affect N transformations via changing soil pH, the ratio of NH_4^+ to NO_3^- , NH_4^+ levels (ammonium toxicity), the accumulation of NO_2^- and the solubility of soil organic matter^[54]. Actually, urea has always dominated mineral nitrogen fertilizer production in China, accounting for 78% of total production in 2023, according to the China Nitrogen

Fertilizer Industry Association. Nonetheless, variations in N_2O direct emission factors among three main staple crops across China were primarily influenced by climatic factors (humidity index), edaphic properties (SOC and soil pH) and irrigation practices, rather than the commonly emphasized fertilizer management practices (type, rate, frequency, and placement)^[55].

Organic fertilizer substitution has been widely promoted to reduce the negative effects of mineral fertilizer application and improve SOC sequestration, which can effectively enhance microbial activity and improve soil properties^[56,57], counteracts the process of microbial nitrogen mining, accelerates the consumption of soil oxygen and decreases the soil redox potential, and mitigates SOC decomposition^[58,59], but also leads to a high GHG emissions owing to the significant increase in CH_4 emissions^[60].

A meta-study reported that replacing mineral fertilizers with manures increased grain yield as well as decreasing N_2O emissions by various amounts, but increasing CH_4 emissions from paddy rice^[57]. As with manure, straw returns could increase SOC at a rate of $72 \text{ Tg}\cdot\text{yr}^{-1} \text{ CO}_2\text{-eq}$ if the practice was adopted nationwide, but it would increase net GHG emissions, mainly due to increased CH_4 emissions from paddy rice^[61]. In terms of SOC sequestration, as observed in 20 long-term field experiments (> 20 years) in single-cropping, double-cropping and paddy-upland rotations across China, the application of manure combined with mineral fertilizers increased grain yield (by 6%–19%) and substantially increased SOC (9%–39%) compared to mineral fertilizers alone^[62]. Significant impacts of manure and straw returns on SOC were observed in 95 long-term field experiments on paddy soils in China, in which increases in SOC were twice as high as those with mineral fertilizers only^[63]. Hence, replacing all or some mineral N fertilizer with manure, and returning straw could increase SOC accumulation and reduce N_2O emissions, but the issue of CH_4 emissions from paddy soils remains, indicating that crop-specific organic fertilizer management for overall GHG emissions reduction is necessary.

Also, the pyrolysis of crop straw into biochar has shown significant advantages in both mitigating GHG emissions and enhancing SOC sequestration^[64–66]. In China, biochar application increased SOC storage at a rate of $0.91 \text{ Mg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ C and reduced CH_4 and N_2O emissions by 26.4% and 23.4%, respectively, while straw retention had an SOC sequestration rate of $0.16 \text{ Mg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ C and increasing CH_4 and N_2O emissions by 111% and 3.3%, respectively^[67].

(2) Enhanced nitrogen fertilizers

Compared with farmer practice (most commonly the application of urea-N), the single application of enhanced N fertilizer, that is, slow- or controlled-release N fertilizer, can increase crop yields and reduce the required rate of N fertilizer, and reduce N₂O emissions by 24% for maize^[68] and 35%–40% for rice as well as reducing CH₄ emissions by 20%–34%^[69].

To improve the *Right source*, N fertilizer producers and blenders have been developing enhanced-efficiency fertilizers using, for example, nitrification and urease inhibitors, and a combination of both to better synchronize N release with crop uptake. These offer the potential for enhanced NUE and reduced GHG emissions, and have proved to be promising tools for mitigating N₂O and NH₃ emissions (Table S1). This double reduction is essential as the IPCC reported that 1% of NH₃ is re-emitted as N₂O following NH₃ deposition.

Many studies have explored the effects of nitrification and urease inhibitors on N₂O mitigation across the North China Plain. We take DMPP (3,4-dimethylpyrazole phosphate a nitrification inhibitor) and LIMUS (a urease inhibitor) as examples. DMPP application increased NH₃ volatilization due to the retention of NH₄⁺-N in soil following the slow transformation of NH₄⁺-N into NO₃⁻-N^[70]. However, the presence of less NO₃⁻-N in soil leads to a reduced denitrification rate and a reduction in N₂O emissions by 51.5%. LIMUS is a urease inhibitor having a similar chemical structure to urea, causing it to react with urease before urea and so slow down urea hydrolysis^[71]. The resultant slow rate of NH₄⁺-N formation leads to reduced substrate for NH₃ volatilization and nitrification. Consequently, NH₃ and N₂O emissions were reduced by 62.5% and 17.1%, respectively, by using LIMUS. Also, the slow release of mineral N by using DMPP and LIMUS can satisfy crop demands and increase crop N uptake^[72,73]. A meta-analysis showed that using urease inhibitors in rice-paddy systems ($n = 100$), resulted in a 9% yield increase, 29% NUE improvement and 41% reduction in N-loss.

In summary, enhanced efficiency fertilizers (with nitrification and/or urease inhibitors) offer significant advantages for achieving C neutrality in agriculture by effectively reducing N₂O emissions.

3.1.2 Right rate: site-specific optimal N application rate

Nitrogen fertilizers can increase or decrease the GHG emissions intensity of cropland, depending on application rates and site-specific conditions^[74]. Generally, higher application rates cause greater GHG emissions^[75]. Although nitrogen

fertilizers can enhance SOC storage by improving crop productivity and residues^[76–78], but this might not offset the global warming potential in cropland arising from non-CO₂ GHG emissions.

In addition, soil acidification caused by excessive N input can significantly affect the C and N cycles in the soils. N-induced soil acidification can consume SIC if carbonates are present in soils via neutralization reaction^[35], but also promotes SOC accumulation by decreasing the decomposition of organic matter via suppressing microbial metabolism and increasing protection of SOC by mineral phases^[79]. Meanwhile, N₂O emissions increase significantly as soil pH decreases at a global scale^[80].

The *Right rate* can be determined from calculations of aboveground crop N uptake and soil N supply. For example, the in-season root-zone N management strategy, based on a soil NO₃⁻-N content test, aims to maintain the soil N supply in the root zone within a range that matches the quantity required by the crop^[81,82]. This has been shown to reduce the required N application rate by 61% from 325 to 128 kg·ha⁻¹ N compared to the farmer practice, with no loss in wheat grain yield, and GHG emissions decreased by 77% in 121 on-farm wheat experiments on the North China Plain^[83]. Wang et al.^[84] reported that using the ecologically optimal N rate (where net benefit was defined as yield benefit minus N fertilizer and environmental costs) maintained grain yield, increased net benefit by 53% and reduced N₂O emissions by 38% compared with current N management in maize on the North China Plain. Similar results were observed in wheat^[85] and rice^[86]. Total GHG emissions from China's maize production could be reduced by 18.6 Tg·yr⁻¹ CO₂-eq (from 110.2 to 91.5 Tg·yr⁻¹ CO₂-eq) if such site-specific N application methods are adopted in all maize-growing subregions of China^[87]. Using a long-term steady-state N balance approach, Yin et al.^[88] estimated the optimized N application range across 3824 cropping counties in China for the three main staple crops, maize, wheat and rice. They found that this could significantly reduce N losses by 23% to 29% while maintaining or even increasing yields by 6% to 7%.

3.1.3 Right timing: crop-specific split application at timings

Compared to a single application of mineral fertilizer, split applications at the right time could increase crop yield, N uptake and NUE by synchronizing soil N availability with crop requirement, contributing to the reduction in N surplus and further losses^[89–91].

For *Right timing*, split fertilizer application is regarded as an effective practice for improving N management. An intensive on-farm study showed that the highest N fertilizer recovery was achieved when ~65% of the N was applied during the rapid stem elongation growth stage of wheat^[81]. Compared to applying the fertilizer in few splits, increasing the number of splits increased grain yield by 5.9%, and reduced N₂O emissions by 5.4% for staple grain (rice, wheat and maize) production in China^[92].

3.1.4 Right place: fertilizer application in subsurface bands and injection

Compared to broadcast application, banded or point-injected N application can promote crop N uptake and NUE by improving the infiltration and movement of applied N to the roots, reducing the N runoff loss and N₂O emission^[93,94]. In addition, deep application favors root elongation, which improves root spatial structure that is more tolerant of stress conditions, for example, water, nutrient and lodging^[95]. Hence, compared to broadcasting, deep placement of fertilizer increased grain yield by 6.9% and reduced N₂O emissions by 14.6% for the main staple crops (rice, wheat and maize) in China^[92].

However, *Right place* needs to be designed with care, because those beneficial advantages are also largely depending on local rainfall and crop types with different root traits, as well as fertilizer source and application rate^[94]. For example, compared to the surface application of slurry, manure injection increased N₂O emissions, due to the increased N retained in soil as a result of the reduced NH₃ emissions^[96]. This suggests that trade-offs exist between GHG emissions and other benefits, for example, NH₃ emissions, NO₃⁻ leaching and agricultural input costs.

3.2 Water management

Irrigation can mitigate the negative effects of drought and heat stress on crop growth via providing water to meet crop demand and both canopy-level and local cooling. It is vital for maintaining optimal growing conditions to reduce crop loss risks, particularly in the face of climate changes^[97–99]. In addition, irrigation is also an important factor in agricultural GHG emissions^[100,101], as it is one of the controls of microbial activity and substrate supply in soil, potentially conflicting with cropland GHG mitigation.

Compared to continuous flood irrigation of paddy rice, optimized irrigation consisting of alternate wetting and drying

and limited flooding irrigation reduced GHG emissions by 37% due to reduced CH₄ emissions and the reduction in energy consumption used for irrigation^[102]. This was despite the increased N₂O emissions induced by water-saving irrigation^[103], thus being another trade-off. For irrigated wheat production, yield-scaled GHG emissions in optimized irrigation systems (optimized irrigation quantity and timing) decreased by 3% to 19% compared to standard irrigation practices^[44]. Of the possible irrigation systems, drip irrigation reduced N₂O emissions by 32% and 46% compared to furrow and sprinkler irrigation, respectively^[103]. Further, reducing the frequency and amount of drip irrigation reduced N₂O emissions in maize production^[104].

3.3 Tillage management

Tillage practices have a profound influence on soil physical properties and nutrient cycles^[105], including soil organic matter content, SOC sequestration^[106,107], GHG emissions^[105], and biological activity^[108].

Tillage is the key driver of SOC loss from cropland. Soil disturbance caused by tillage destabilizes soil aggregates in which soil C is protected from decomposition, resulting in emission of this C as CO₂ to the atmosphere^[109]. Hence, conservation tillage (e.g., no-tillage, reduced tillage, ridge tillage and mulch tillage) is being widely encouraged to replace standard tillage in cropland to reduce tillage-induced SOC loss, or potentially to increase SOC sequestration^[110].

In China, the SOC sink rate was calculated to be 0.3–1.3 Mg·ha⁻¹·yr⁻¹ C in topsoil under no-tillage^[111–113]. The total SOC sink from using no-tillage was calculated to be limited at 0.8 Tg·yr⁻¹ C in Chinese cropland, but this could be increased to 4.6 Tg·yr⁻¹ C (nearly one-fifth of the annual agricultural SOC sink) if no-tillage practices were adopted more widely^[114]. However, SOC has been found to increase in topsoil but decrease in subsoil under conservation tillage (i.e., no-tillage), which is important because SOC in subsoils is more stable than that in topsoil when considering C turnover time^[109,115].

No-till could also influence non-CO₂ GHG emissions^[116]. For rice paddies in China, no-tillage was found to reduce CH₄ emission by ~22% but increase N₂O emission by 42% compared to standard plowing. For uplands in China, no-tillage increase CH₄ uptake and only slightly changes N₂O emissions. Similar results for CH₄ emission/uptake have been widely observed, but no-tillage has been found to enhance^[117], decrease^[118] or have no effect^[119] on N₂O emissions. This is

probably due to factors such as temperature, precipitation and soil properties.

Mulch tillage (mulching with plastic film or crop residues) is a common practice in arid and semiarid regions to promote crop growth, mainly via increasing plant available water and soil temperature^[120]. Compared to no mulching, mulching decreased CH₄ emissions by 43% in China's crop production, but increased CO₂, N₂O and so total GHG emissions by 22%, 1.7% and 11%, respectively^[121]. However, mulching with straw or film could decrease GHG emissions by 20% to 106% compared with no mulching in a wheat-maize rotation at a specific site^[122].

4 Conclusions, future directions and recommendations

Cropland in China functions as both a carbon sink and source. Between the 1980s and 2020s, it sequestered an average of 21.3 Tg C (78.0 Tg CO₂) annually as soil organic carbon but simultaneously lost 16.0 Tg C (58.6 Tg CO₂) from soil inorganic carbon. Additionally, cropland emitted 341 Tg CO₂-eq of GHGs into the atmosphere, making it a significant net source of GHG emissions. Despite gaps in understanding, certain trends are clear, such as increased per-unit crop yield and reduced GHG emissions through optimized field management practices. Over time, yield-scaled GHG emission intensity for cereals has decreased, highlighting the potential for mitigation. This underscores the need for further research, particularly in the context of global climate change, to address existing knowledge gaps and enhance mitigation strategies.

Specifically, in China's crop production systems, many available mitigation practices have been shown to reduce GHG emissions and/or enhance soil C storage. However, many of these, including slow- and controlled-release fertilizers, enhanced N fertilizers, drip irrigation and manure injection, are taking time to be developed, despite national agricultural policies and actions. Also, some well-established practices are still not widely used, for example, manure application accounted for only ~10% of the total N application in cereal and fruit production^[123], while the proportion of straw returned increased only to ~40% from 1980 to 2010, largely due to the implementation of a straw retention policy^[124]. In 2017, the area under conservation tillage in China was 7.58 Mha, being ~10% of the total arable area, based on the report of the Ministry of Agricultural and Rural Affairs in 2020.

There is therefore an urgent need to promote the established practices while developing other effective practices, for example, new crop cultivars with high C sequestration potential^[125], low-C industrial biochar production^[126], cropping systems transformation and distribution (high C sink instead of low C sink).

Clearly, there are many individual practices that can move toward C neutrality in crop production that have identified across China. In summary, however, crop-specific integrated field management is the most effective way to improve practical crop production. Currently, implementing a set of integrated soil-crop system management practices based on current understanding of crop ecophysiology and soil biogeochemistry can increase maize, wheat and rice yields by 11%–12% and reduce GHG emissions by 14%–23% over the whole of China^[47]. However, more effort is needed to supplement this management approach with additional practices (e.g., biochar, biological nitrogen fertilizers) and crops (e.g., uncommon grain species with environmental advantages and economic benefits) that will create a nationally recognized standard for crop production that should be implemented across China, building close and effective cooperation between local government, agriculture companies, universities and research institutes, as in the now well-established Science and Technology Backyard model; a proven successful way of improving crop production that can be an innovative platform for delivering C neutrality in crop production^[127,128].

In the recent times, China has increasingly relied on imported agricultural products, mainly owing to the rising animal feed demand^[4,129]. This shows that China still gives food security a high priority, making the move toward agricultural C neutrality difficult in the short term. One possible pathway to achieve C neutrality would be improving crop productivity to reduce the C footprint per kg grain produced, freeing land for other uses such as reforestation with inherently high C sink benefits.

Additionally, China is making more effort outside of cropland to meet C neutrality targets in crop production, including planting structure adjustment^[130], low-carbon diets^[131], electrification of agricultural machinery^[132], and clean and efficient utilization of crop residues^[133]. As the largest developing country, China aims to be a responsible practitioner, moving toward C neutrality, and has plenty of valuable and successful experience of ways in which this might be achieved.

Supplementary materials

The online version of this article at <https://doi.org/10.15302/J-FASE-2025602> contains supplementary materials (Figs. S1–S4; Table S1).

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Compliance with ethics guidelines

Tianxiang Hao, Yangyang Zhang, Yulong Yin, Jingxia Wang, Zhenling Cui, Keith Goulding, and Xuejun Liu declare that they have no conflicts of interest or financial conflicts to disclose. This article does not contain any studies with human or animal subjects performed by any of the authors.

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