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# Toward Low-Carbon Rice Production in China: Historical Changes, Driving Factors, and Mitigation Potential

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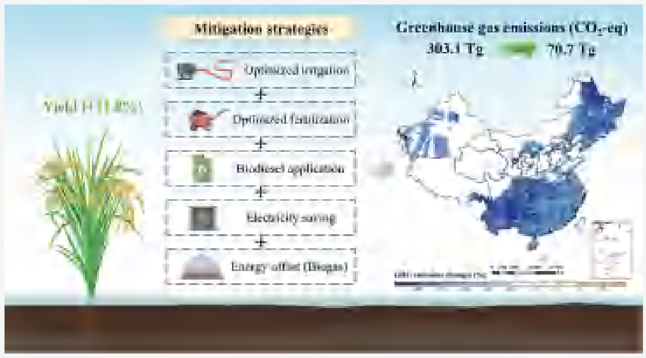
**ABSTRACT:** Under the "Double Carbon" target, the development of low-carbon agriculture requires a holistic comprehension of spatially and temporally explicit greenhouse gas (GHG) emissions associated with agricultural products. However, the lack of systematic evaluation at a fine scale presents considerable challenges in guiding localized strategies for mitigating GHG emissions from crop production. Here, we analyzed the county-level carbon footprint (CF) of China's rice production from 2007 to 2018 by coupling life cycle assessment and the DNDC model. Results revealed a significant annual increase of 74.3 kg $CO_2$-eq ha$^{-1}$ in the average farm-based CF (FCF), while it remained stable for the product-based CF (PCF). The CF exhibited considerable variations among counties, ranging from 2324 to 20,768 kg $CO_2$-eq ha$^{-1}$ for FCF and from 0.36 to 3.81 kg $CO_2$-eq kg$^{-1}$ for PCF in 2018. The spatiotemporal heterogeneities of FCF were predominantly influenced by field $CH_4$ emissions, followed by diesel consumption and soil organic carbon sequestration. Scenario analysis elucidates that the national total GHG emissions from rice production could be significantly reduced through optimized irrigation (48.5%) and straw-based biogas production (18.0%). Moreover, integrating additional strategies (e.g., advanced crop management, optimized fertilization, and biodiesel application) could amplify the overall emission reduction to 76.7% while concurrently boosting the rice yield by 11.8%. Our county-level research provides valuable insights for the formulation of targeted GHG mitigation policies in rice production, thereby advancing the pursuit of carbon-neutral agricultural practices.




**KEYWORDS:** *GHG emissions, life cycle assessment, DNDC, carbon footprint, rice production, mitigation strategies*



## 1. INTRODUCTION

The rapid acceleration of climate change has propelled the issue of greenhouse gas (GHG) emissions resulting from human activities to the forefront of global attention. Agricultural production, as a crucial manifestation of human activity, contributes to one-third of global anthropogenic GHG emissions.[1] China is the foremost global producer of rice.[2] However, rice production is responsible for 22% of the country's total GHG emissions from agricultural activities.[3] The demand for rice in China is expected to increase to 218 Mt year$^{-1}$ by the year 2030.[4] To reach this yield target, it is anticipated that rice production activities will be further intensified in the coming years. This presents a significant challenge for China in its pursuit of carbon neutrality in the agri-food sector. Hence, it is imperative to mitigate GHG emissions from rice production for the development of low-carbon agriculture.

Life cycle assessment (LCA) is a systematic methodological framework for evaluating the consumption of resources and environmental impacts associated with various stages of a product.[5] Over the past few decades, the LCA has been widely employed to enhance the understanding of GHG emissions from the production of agricultural products.[6−8] Based on the LCA, carbon footprint (CF) was utilized as a comprehensive indicator to encompass all aspects of carbon emissions and carbon sequestration during agricultural production, such as indirect emissions from agricultural materials, direct emissions from field soil, and carbon sequestered into the soil.[9,10] Currently, numerous studies have quantified the CF for rice at the provincial or national level,[11−13] which provide valuable insights into the GHG emissions from rice production. However, environmental conditions and farming practices exhibit significant variations across diverse rice cropping regions. The assessment at a coarse spatial scale is inadequate in accurately capturing the spatial heterogeneity in the CF of rice production. Furthermore, there exists a notable paucity of

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research focusing on the historical changes in CF of rice production due to the lack of long-term spatial explicit data on emitting sources from various processes, particularly the direct emissions from paddy ecosystems.[14,15] Consequently, the assessment of CF for rice production at both finer spatial (i.e., county scale) and temporal scales is urgently needed for accurately analyzing the spatiotemporal pattern, disclosing the potential driving factors, and facilitating the formulation of localized mitigation policies.

Throughout the entire process of rice production, the direct methane ($CH_4$) and nitrous oxide ($N_2O$) emissions from paddy fields contributed to over 70% of CF.[11,16] At the same time, paddy fields can act as a carbon sink to reduce GHG emissions by sequestrating the atmospheric carbon dioxide into the soil.[17] The precise estimation of GHG emissions and soil organic carbon (SOC) sequestration from paddy fields is therefore of utmost importance in determining the spatiotemporal pattern of CF. In previous studies, the Tier 1 and Tier 2 methods outlined in the Intergovernmental Panel on Climate Change (IPCC) guidelines were commonly used to estimate $CH_4$ emissions,[18] $N_2O$ emissions,[19] and SOC sequestration[20] in paddy ecosystems. When making estimates in countries with expansive territories, such as China, these methods may prove insufficient to reflect the combined effects of local climate, soil, and management conditions (e.g., fertilization, irrigation, and straw return) on field GHG emissions and SOC sequestration.[19,21,22] To take such finer-scale impacts into account, it is necessary to couple LCA with the process-based model explicated in the Tier 3 method, such as the Denitrification−Decomposition (DNDC) model,[23] the Environmental Policy Integrated Climate (EPIC) model,[24] and the Dynamic Land Ecosystem Model (DLEM).[25]

Furthermore, the specific driving factors behind the spatiotemporal variation in the CF of rice production, particularly those related to GHG emissions and SOC sequestration in paddy fields, have not been precisely identified, which hinders the development of effective mitigation measures. Based on large spatiotemporal data samples, machine learning models [e.g., random forest (RF) and boosted regression trees] are capable of efficiently describing the complex predictor−response relationships.[26] These approaches provide a pathway for exploring the relative influence of various emission sources on CF of rice production, as well as the relative influence of various environmental and management factors on the GHG emissions and SOC sequestration in paddy fields.

To bridge the aforementioned knowledge gaps, this study conducted a systematic assessment of GHG emissions from rice production in China from the perspective of "historical changes, driving factors, and mitigation potential". Based on multisource data, we first quantified the county-level CF and total GHG emissions for different rice types from 2007 to 2018 by coupling the LCA with the DNDC model. Then, we used a RF model to identify the key emission sources that affect the spatiotemporal variation in CF, as well as the dominant environmental and management factors that influence the GHG emissions and SOC sequestration in paddy fields. Finally, we established multiple scenarios to comprehensively analyze the GHG mitigation potential of the widely discussed strategies. Our county-level assessment across multiple years accurately illustrates the spatially and temporally explicit CF of rice production. The findings from the scenario analysis can serve as a theoretical basis for the low-carbon development of rice production.

## 2. MATERIALS AND METHODS

**2.1. County-Level CF Quantification.** This study focused on the production of single rice, early rice, and late rice in two conventional cropping systems (single cropping rice and double cropping rice) in China. Excluding Hong Kong, Macao, and Taiwan, rice cultivation was recorded in 30 provinces in China from 2007 to 2018, encompassing 1733 counties for single rice and 700 counties for early and late rice. The cradle-to-farm gate LCA was employed to evaluate the county-level CF and total GHG emissions (TGHG) from the production of three rice types between 2007 and 2018. The CF was quantified using two functional units: the farm-based CF (FCF, kg $CO_2$-eq $ha^{-1}$) and the product-based CF (PCF, kg $CO_2$-eq $kg^{-1}$). The system boundary was divided into the agri-materials stage and the on-farm stage, which covered the indirect GHG emissions from supply chain processes of various agricultural materials and the direct GHG emissions from rice cultivation processes (Figure S1).

*2.1.1. Agri-Materials Stage.* During the agri-materials stage, the indirect GHG emissions were primarily attributed to the production and transportation of synthetic fertilizers, pesticides, seeds, plastic films, diesel, and irrigation electricity. These emissions were estimated by multiplying the quantity of each agricultural input by the corresponding GHG emission factor (EF) (detailed in Text S1). Due to the absence of survey-based spatial fertilizer application data for different rice types, the county-level application rates of nitrogen, phosphorus, and potassium fertilizer were estimated based on the data from Wu et al.[27] and the China Agricultural Product Cost−Benefit Data Compilation (CBD).[28] For other agri-materials, the China Rural Statistical Yearbook[29] and China Energy Statistical Yearbook[30] only provide the total amount of historical consumption without crop-specific differentiation. To downscale these data, this study initially applied the method outlined by Chen et al.[11] to calculate their provincial application rates for different rice types. Then, the county-level application rates were estimated based on the provincial application rates and the total consumption in each county. Specifically, there were no corresponding county-level data to downscale the provincial application rates of rice seed. Considering the relatively small CF produced by seed,[14] the provincial data from CBD were used. All data sources and processing methods are elaborated in Text S1. The GHG emissions from manure processing were excluded from the system boundary as we attributed them to animal husbandry systems.[7] To ensure the representativeness of EF, this study applied average EF values for each agricultural input, which were sourced from multiple literature sources and databases (Tables S1 and S2).

*2.1.2. On-Farm Stage.* During the on-farm stage, the direct GHG emissions from rice cultivation processes could be divided into three components: net GHG emissions from the biogeochemical processes in paddy ecosystems, emissions from diesel combustion, and emissions from straw burning in the households or fields. The DNDC model (version 9.5),[31] which can simulate crop growth and the soil C/N dynamics driven by microbial activities, was applied to calculate the net GHG emissions from paddy fields in each county. Multisource data, including meteorological data, soil data, and field management data, were integrated into the county-level database and used as the inputs for driving regional simulation of the DNDC model (detailed in Text S2). To improve the accuracy of the simulation results, the critical crop parameters in



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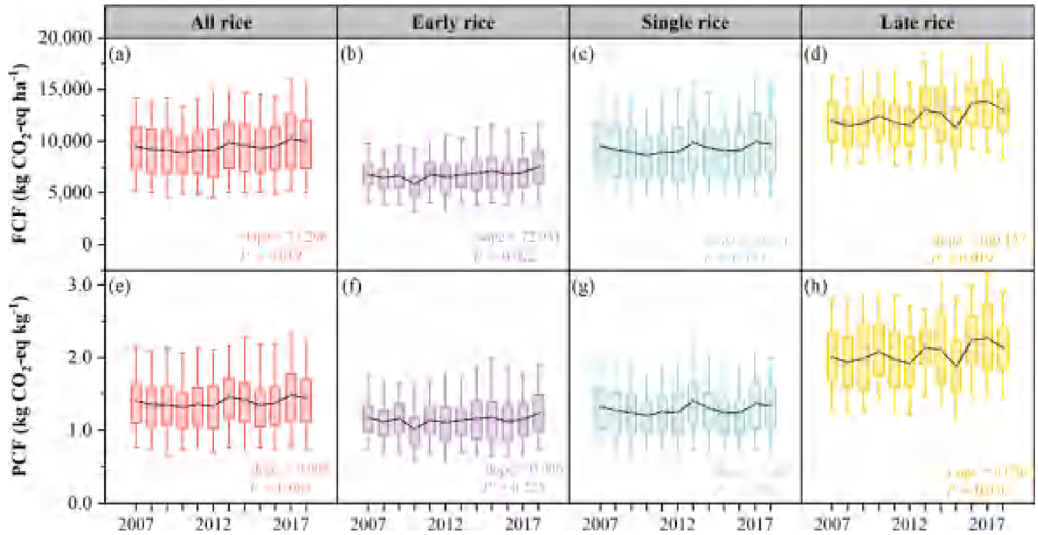


**Figure 1.** FCF and PCF for different rice types in China from 2007 to 2018. The gray points inside the box represent the mean value of county-level FCF and PCF, and the box boundary indicates the 25th and 75th percentiles. The slope and $p$ value represent the trend and significance of annual changes, respectively.

the DNDC model were calibrated and validated by using observed county-level yields. Additionally, field experimental data collected from peer-reviewed journals were used to validate the suitability of the DNDC model in simulating regional GHG emissions and SOC sequestration in paddy ecosystems (Figure S2 and Tables S4–S7). The results show that the simulated rice yields, $CH_4$ emissions, $N_2O$ emissions, $CO_2$ emissions (total soil respiration), and SOC contents were consistent with the recorded data and field experimental data ($R^2 > 0.70$, nRMSE $< 0.30$, and $p < 0.01$; Figure S3), which indicates that the DNDC model is capable of accurately simulating rice growth and soil C and N dynamics in paddy fields. We then utilized the calibrated DNDC model to quantify county-level $CH_4$ and $N_2O$ emissions as well as SOC sequestration from 2007 to 2018. To estimate direct GHG emissions from diesel combustion during machinery operations and straw burning, we adopted the method described in Text S1 (eq S1) and the approach outlined by Lu et al.,[32] respectively.

**2.2. Driving Factors Analysis.** The RF model is proficient in capturing complex and highly nonlinear relationships between response variables and a set of predictor variables.[33,34] Based on the permutation-based method in the RF model, the relative importance of predictor variables to response variables can be quantified using the increased mean square error (% IncMSE). Given its advantages, we selected the RF model to clarify the primary driving factors for spatially and temporally explicit GHG emissions from two perspectives. First, the relative importance of each emission source was quantified to identify the key emission sources, driving the variation in the FCF of different rice types. Then, given the substantial contribution of the net GHG emissions from paddy fields to the CF, the primary drivers of $CH_4$ emissions, $N_2O$ emissions, and SOC sequestration were further disclosed by estimating the relative importance of climatic factors (the daily average temperature and cumulative precipitation during the growing season), soil properties (initial SOC content, clay content, pH, and bulk density), and field management factors (N fertilizer, manure-N, straw return amount, and flooding duration). All predictor variables were extracted from our county-level database (Texts S1 and S2). The "rfPermute" package in R (version 4.3.2) was employed to construct the RF model and calculate the significance level of each predictor variable. The % IncMSE values were standardized on a scale of 0 to 100% for comparison purposes.

**2.3. Mitigation Scenarios Establishment.** This study established five alternative scenarios to explore the mitigation potential of GHG emissions from rice production. The CF and TGHG in 2018 were set as the baseline scenario (baseline).

(1) Yield increase (YI). According to the finding of Chen et al.,[4] the average rice yield in China could be improved by 12–18% without the increase in N fertilizers through the implementation of advanced crop management (e.g., increasing planting density, maximizing the use of solar radiation, and optimizing nutrient and water resource). To investigate the effect of increased yield on the CF, the YI scenario was established by augmenting the potential grain yield of each county by 15% in the DNDC model while ensuring no increase in nitrogen stress on rice growth.

(2) Optimized irrigation (OI). The alternate wetting and drying (AWD) irrigation method, developed as an optimized water regime for rice production, is effective in conserving water resources and reducing $CH_4$ emissions simultaneously.[35,36] Based on the soil water table threshold, AWD involves multiple flooding and drainage cycles during the rice growth period.[37] The mitigation potential of GHG emissions under this OI scenario was evaluated by adjusting the irrigation method in the DNDC model.

(3) Optimized fertilization (OF) scenario. Given the overuse of synthetic N fertilizer and relatively low application rate of manure in Chinese paddy fields,[38,39] this study established the OF scenario to reduce the GHG emissions associated with synthetic N fertilizers. Under this scenario, we selected the N fertilizer level recommended by Wu et al.[27] (Text S1) as the target to reduce the synthetic N application rate in each county. Then, the contribution of manure N to total N fertilizers increased to 20% by partially substituting synthetic N fertilizer.[40,41]



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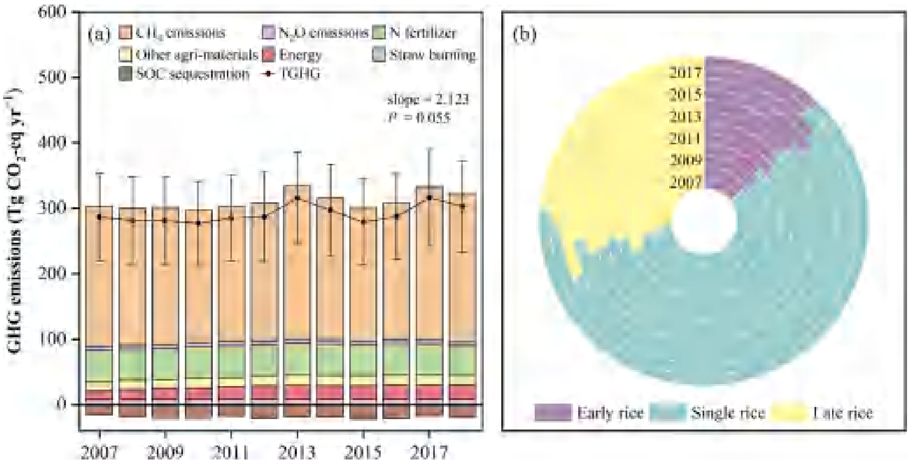


**Figure 2.** Composition of TGHG from rice production in China between 2007 and 2018. (a) Contributions of various sources to the TGHG. Other agri-materials indicate the total GHG emissions from the consumption of P fertilizers, K fertilizers, herbicides, insecticides, fungicides, rice seeds, and plastic films. Energy indicates the total GHG emissions from the consumption of diesel and irrigation electricity. Slope and *p*-value indicate the trend and significance of annual changes, respectively. Error bars indicate the uncertainty ranges. (b) Contributions of different rice types to the TGHG.

This optimization of fertilization was based on the premise that rice yield would not decrease.

(4) Energy substitution, saving, and offset (ES). This scenario was designed based on three aspects: substitution of diesel with biodiesel for agricultural machinery operations;[6] electricity saving through optimized irrigation practice;[42] and energy offset benefits generated by converting available straws (excluding straw used for returning and livestock feed) into biogas to replace natural gas.[43]

(5) Combined scenario (CS). This scenario was created to evaluate the comprehensive abatement potential of implementing the aforementioned management strategies. Considering the trade-off effect between YI and OF scenario, we prioritized the reduction and substitution of synthetic N fertilizers and then explored the potential for yield enhancement to maximize environmental benefits.[44] Details of the descriptions regarding the establishment of alternative mitigation scenarios can be found in Text S4.

**2.4. Uncertainty Analysis.** During the agri-materials stage, uncertainties of estimated indirect GHG emissions come from the quantity of different agricultural materials and their corresponding EF. The coefficient of variation (CV) of each agricultural input in each county was set to 30%,[45] while the CV of EF was calculated based on the cited papers in Table S1. It was postulated that both emission sources and the EF for each county conformed to a normal distribution. The Monte Carlo method with 10,000 times run was employed to calculate the uncertainty of GHG emissions from the consumption of agri-materials. During the on-farm stage, the Monte Carlo method was also used to calculate the uncertainties of GHG emissions from diesel combustion and straw burning (or energy offset). Uncertainties of net GHG emissions from paddy fields mainly arise from the simulations of the DNDC model. The most sensitive factor method[46] was employed to quantify the uncertainties of $CH_4$ emissions, $N_2O$ emissions, and SOC sequestration for each county (Text S5). The total uncertainty was finally estimated based on the principles and methods outlined in the IPCC 2006 guideline.[47]

## 3. RESULTS AND DISCUSSION

**3.1. Historical Variation in GHG Emissions.** Figure 1 illustrates the annual changes in the FCF and PCF for different rice types. Among the three rice types, late rice had the highest annual average FCF and PCF (12,413 kg $CO_2$-eq ha$^{-1}$; 2.06 kg $CO_2$-eq kg$^{-1}$), followed by single rice (9298 kg $CO_2$-eq ha$^{-1}$; 1.29 kg $CO_2$-eq kg$^{-1}$) and early rice (6808 kg $CO_2$-eq ha$^{-1}$; 1.15 kg $CO_2$-eq kg$^{-1}$). From 2007 to 2018, the annual FCF for early rice and late rice experienced a significant uptrend ($p < 0.05$), with an annual increase of approximately 73.0 and 160.2 kg $CO_2$-eq ha$^{-1}$, respectively. While the FCF for single rice similarly showed a parallel increase, the trend was not statistically significant. As for the PCF, the annual trend for all rice types remained largely consistent without a significant change. Overall, the average FCF of rice production has markedly increased at an annual rate of 74.3 kg $CO_2$-eq ha$^{-1}$ over the past 12 years, while the average PCF has leveled off (Figure 1a,e).

Although there was an increase from 286.7 Tg $CO_2$-eq in 2007 to 303.1 Tg $CO_2$-eq in 2018, the national TGHG exhibited no significant change over the past 12 years, nor did the contributions of three rice types (Figure 2). This trend was primarily attributed to the relatively stable $CH_4$ emissions from paddy fields in China.[18,21] Specifically, $CH_4$ emissions from paddy fields accounted for 67.9−70.7% of GHG emissions, which thereby significantly influenced the overall trend of TGHG. Approximately 17.3% of GHG emissions were associated with N fertilizers, including emissions from the production and transportation of N fertilizers and $N_2O$ emissions from paddy fields. Moreover, energy consumption in the fields accounted for 4.5−7.1% of GHG emissions. Due to the rapid development in agricultural mechanization levels, GHG emissions from field diesel consumption demonstrated a significant upward trend from 2007 to 2018, with an annual increment rate of 0.6 Tg $CO_2$-eq. SOC sequestration by the application of manure and straw return could decrease GHG emissions by 5.2−7.4%. Regarding distinct rice types, only the TGHG for single rice witnessed a significant increase of 16.1% from 2007 to 2018, attributable to the continuous expansion in its cultivation area (Figure S4).

Compared to previous studies, the multiyear average national FCF, PCF, and TGHG of rice production estimated in this study


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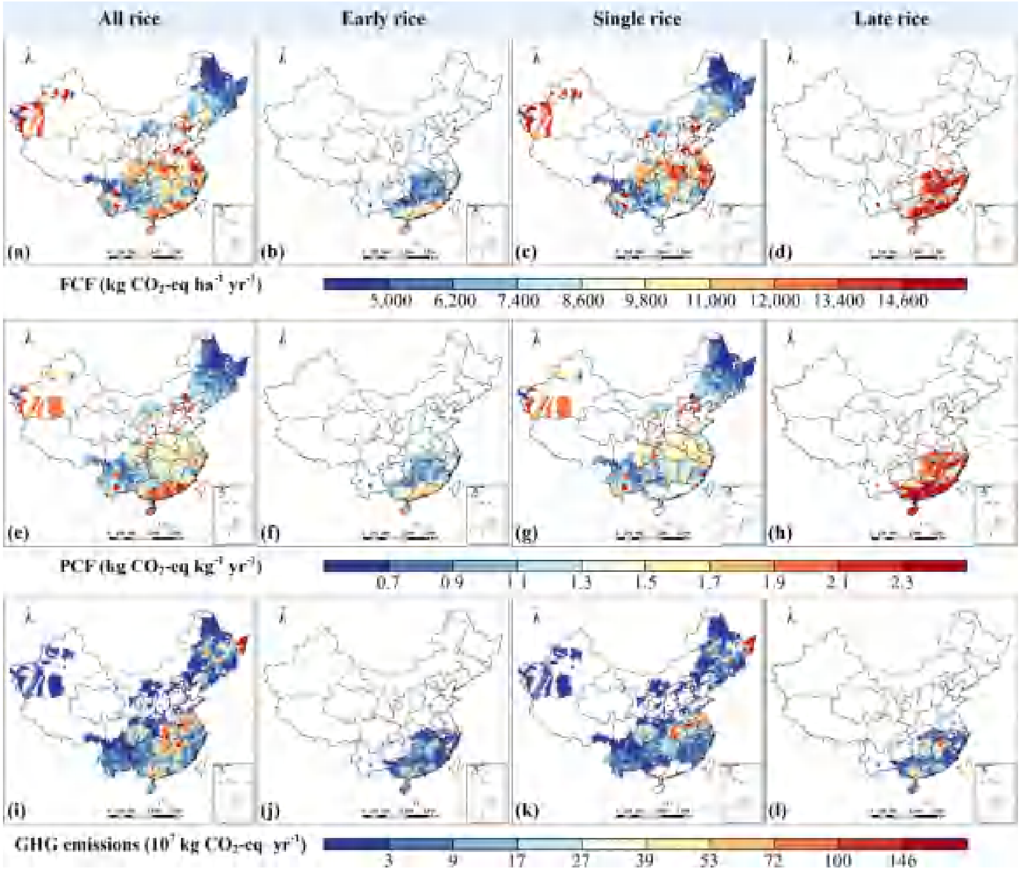


**Figure 3.** Spatial distributions of county-level FCF, PCF, and TGHG of rice production in 2018. (a−d) FCF for all rice, early rice, single rice, and late rice; (e−h) PCF for all rice, early rice, single rice, and late rice; and (i−l) TGHG from all rice, early rice, single rice, and late rice.

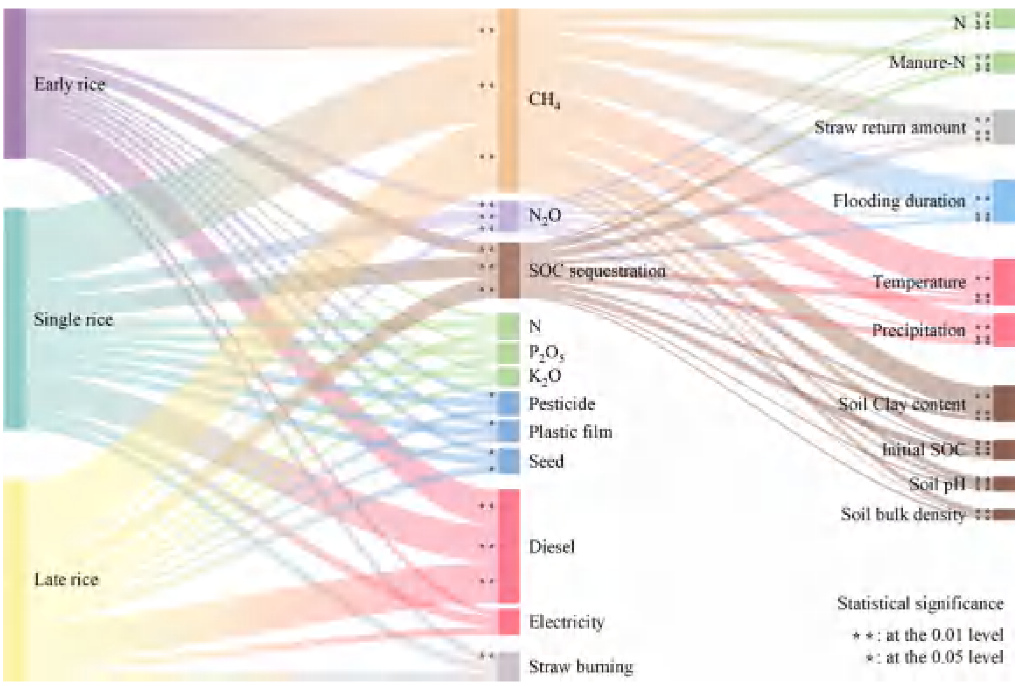


**Figure 4.** Relative importance of emission sources to the spatiotemporal variation in FCF for early rice, single rice, and late rice (left half), as well as environmental and management factors to the spatiotemporal variation in $CH_4$ emissions, $N_2O$ emissions, and SOC sequestration from paddy fields (right half). Comparisons are meaningful only for predictor variables related to the same response variable. A wider connecting line represents the greater relative importance of the predictor variable to the response variable.



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fall within the range of reported values (Figure S5). The dissimilarities in the results can be ascribed to discrepancies in the scope of LCA, spatiotemporal scales, and methods used for quantifying the CF. For example, existing studies generally concentrated solely on GHG emissions from rice production, whereas the effects of carbon sequestration by paddy ecosystems were overlooked. Then, the CF of rice production was generally estimated at a coarse spatial scale (e.g., national and provincial level).[15,48,49] Furthermore, the utilization of homogeneous field EFs based on Tier 1 or Tier 2 method may result in either underestimation or overestimation of localized GHG emissions from paddy ecosystems.[50,51] In this study, we selected counties as the basic unit, at which scale the DNDC model exhibits satisfactory performance.[52] This model-based estimate took into account the impact of environmental factors and management practices on $CH_4$ emissions, $N_2O$ emissions, and SOC sequestration under different rotation systems. Despite the limitation of some downscaled agri-materials data, our results could offer a more precise depiction of historical variations in CF and TGHG emissions for different rice types than previous studies.

**3.2. Spatial Pattern of County-Level CF.** In 2018, significant spatial heterogeneities were observed in FCF and PCF of rice production across various counties, with values ranging from 2324 to 20,768 kg $CO_2$-eq ha$^{-1}$ and 0.36 to 3.81 kg $CO_2$-eq kg$^{-1}$, respectively (Figure 3a,e). Counties with high FCF were mainly located in the North China Plain, Northwest China, and the Middle-Lower Yangtze Plain (especially Jiangsu and Shanghai) (Figure S6). These hotspots were characterized by higher $CH_4$ emissions and fertilizer application rates. Comparatively, counties exhibiting high PCF were mainly found in South China due to their low rice yields. For different rice types, the high FCF and PCF for early and late rice were primarily concentrated in South China, while the distributions of high FCF and PCF for single rice were predominantly located in the North China Plain and Northwest China. From 2007 to 2018, nearly 60% of counties experienced varying degrees of increase in CF (Figure S7). The increase in CF was primarily observed in Southern China.

In terms of the TGHG, the hotspots mainly appeared in the Middle-Lower Yangtze Plain, including Hunan, Jiangxi, Hubei, Jiangsu, and Anhui (Figure 3i). Approximately 78% of the national TGHG originated from nine provinces within the Sichuan Basin, the Middle-Lower Yangtze Plain, South China, and Northeast China (Figure S6). These regions collectively encompass nearly 80% of China's total rice cultivation area and production. Furthermore, the TGHG witnessed a substantial increase in Northeast China during 2007−2018 (Figure S7), primarily due to their rapid expansion of the rice cultivation area since 2007. Overall, it is imperative to identify the hotspot counties based on spatial explicit distributions of CF and TGHG. These counties should be given priority in future efforts for GHG mitigation in rice production.

**3.3. Relative Importance of Different Drivers to GHG Emissions.** For all rice types, the spatiotemporal variations in FCF were primarily influenced by $CH_4$ emissions from paddy fields, followed by field diesel consumption and SOC sequestration ($p < 0.01$) (Figure 4). Considering that $CH_4$ emissions represented the most substantial contributor to the FCF of rice production, any fluctuations in these emissions were likely to induce pronounced variations in FCF. From 2007 to 2018, the rapid popularization of agricultural mechanization has led to a continuous increase in the national average diesel consumption of rice production, rising from 91.5 to 164.1 kg ha$^{-1}$. This significant increase has positioned diesel consumption as the second major driving force behind the variation in FCF. Moreover, the spatiotemporal heterogeneity of climate conditions, soil properties, and field management across China could result in huge variations in SOC sequestration from paddy soil,[17] which further influences the carbon fixation benefits throughout the lifecycle of rice production. $N_2O$ emissions from paddy fields, also vulnerable to environmental and management conditions,[53] were identified as another contributor to significantly impact the variation of FCF for all rice types ($p < 0.01$).

Beyond assessing the impacts of various emission sources on spatiotemporal variations of FCF, we also elucidated the relative contributions of climate, soil, and farming practices to $CH_4$ and $N_2O$ emissions, as well as SOC sequestration from paddy soil (Figure 4). As the major GHG during the rice growth period, the variations in $CH_4$ emissions were primarily driven by the temperature during the rice growth period. Elevated temperatures would enhance the substrate availability for methanogens and methanogenic activity,[54] thereby facilitating the release of $CH_4$ from paddy soil. Previous research has found that 1 °C warming will increase $CH_4$ emissions from paddies in China by 12.6%.[50] Another critical influential factor was flooding durations. For a long time, continuous flooding has traditionally been the prevailing irrigation practice in Chinese paddy fields. However, prolonged flooding establishes favorable anaerobic conditions conducive to $CH_4$ production. After the 1990s, the adoption of midseason drainage had significantly decreased $CH_4$ emissions.[46,55,56] Additionally, $CH_4$ emissions were also closely related to the soil clay content. Clay particles provide the majority of the adsorbent surface area in the soil and protect dissolved organic carbon from microbial degradation.[57] The reduction in the soil carbon substrate available to methanogens leads to the suppression of $CH_4$ production. Overall, within the context of global warming, effective field water management has emerged as a pivotal strategy for mitigating $CH_4$ emissions from paddy fields.

In terms of $N_2O$ emissions, the soil clay content, N fertilizer application rates, and soil pH were identified as the three most influential variables. An increased soil clay content correlates with higher porosity and improved water retention capabilities, leading to a greater reduction of soil N to $N_2O$ under anaerobic conditions.[53] However, paddy soils with high clay content are characterized by low permeability and prolonged $N_2O$ diffusion time, which is conducive to $N_2O$ uptake and consumption.[58] The mechanisms by which the soil clay content affects $N_2O$ emissions by determining the prevailing conditions warrant further exploration through soil biochemical analyses. As evidenced in previous studies,[19,59] N fertilizer application rates constituted an important factor driving the spatiotemporal variations in $N_2O$ emissions of paddy soil. The optimization of N fertilizer inputs is anticipated to yield cobenefits of reduced GHG emissions from N fertilizer supply and field $N_2O$ emissions. Furthermore, soil pH plays an important role in determining $N_2O$ emissions. Under high soil pH conditions, the expression of $N_2O^-$ reductase of denitrifying bacteria and electron-transfer efficiency may be promoted, which inhibits the formation of $N_2O$ in the denitrification process.[60]

The spatiotemporal variability of SOC sequestration was primarily driven by the straw return amount. SOC change is determined by the balance between carbon inputs and outputs. Straw return, serving as the primary exogenous C input for


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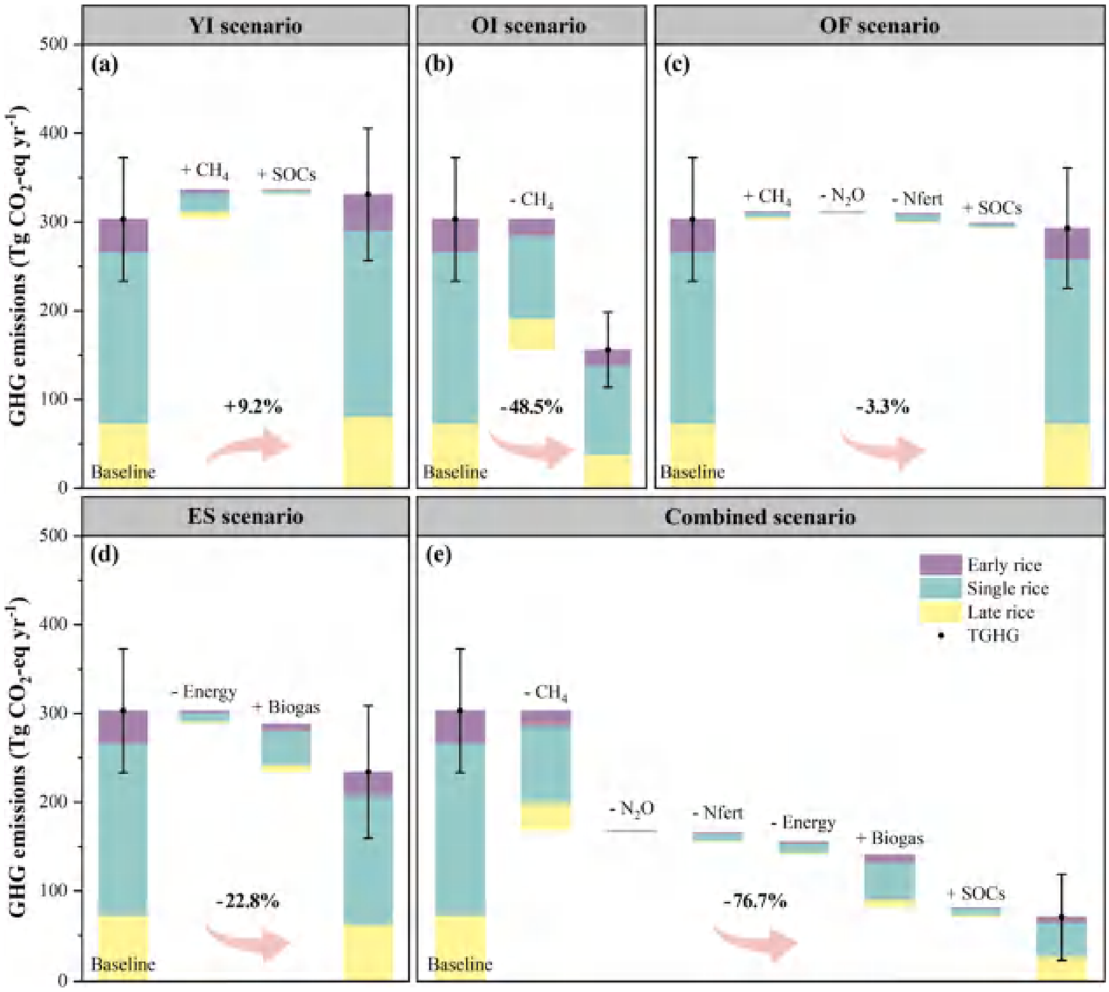


Figure 5. Mitigation effects of TGHG from rice production in China under alternative scenarios. $CH_4$: $CH_4$ emissions from paddy fields; $N_2O$: $N_2O$ emissions from paddy fields. Nfert: N fertilizers; Energy: diesel and electricity; Biogas: biogas generated from straw with natural gas offset effect; and SOCs: SOC sequestration. Only components with changes exceeding 1 Tg $CO_2$-eq are presented. Error bars indicate the uncertainty ranges. The proportion above the arrow represents the change rate in TGHG.

paddy soil, can directly contribute to SOC accumulation through mineralization and humification processes.[61] At the same time, straw return increases available soil C content for methanogens, which will facilitate the soil C loss in the form of $CH_4$.[62] The initial SOC content was identified as the second most influential factor. A higher initial SOC content is typically associated with a greater SOC saturation degree, which tends to slow down the SOC sequestration rate.[63,64] For example, while rice paddies in Northeast China are characterized by the highest SOC content, SOC sequestration in these areas has remained relatively low. The temperature emerged as another pivotal factor influencing SOC sequestration. The soil C turnover is closely linked to the soil temperature. Warming can accelerate the processes of SOC decomposition, soil respiration, and methanogenesis, culminating in the release of soil C into the atmosphere as $CO_2$ and $CH_4$.[65,66]

### 3.4. Mitigation Potential of GHG Emissions under Alternative Scenarios.

Figure 5 illustrates the GHG mitigation potential of rice production under different scenarios. Compared with the baseline, the YI scenario increased the national average rice yield and TGHG by 14.4 and 9.2%, respectively (Table S9 and Figure 5a). Although increased SOC sequestration generated an additional climate benefit (5.6 Tg $CO_2$-eq), this was offset by the increased $CH_4$ emissions (33.2 Tg $CO_2$-eq). Spatially, the increase in rice yield has led to a rise in FCF and a concurrent decrease in PCF across China, with an average variation of +8.6 and −5.1%, respectively (Figure S8). The increased yield is generally accompanied by more root exudation, plant litter, and a higher straw return amount. As a result, this provides a greater abundance of soil carbon substrates for microbes to produce $CH_4$ and sequester C.[50,67] Generally, there is considerable potential to augment rice yields at existing N fertilizer levels through the implementation of advanced crop management. However, the associated risk of escalating $CH_4$ emissions should not be disregarded.

The OI scenario demonstrated a substantial GHG mitigation potential. The national FCF, PCF, and TGHG of rice production all decreased by about 48% without compromising rice yield (Table S9 and Figure 5b). This significant benefit was primarily attributed to the reduction in $CH_4$ emissions (146.8 Tg $CO_2$-eq). Furthermore, counties with relatively large mitigation potential were predominantly concentrated in the Sichuan Basin, the North China Plain, and South China (Figure S8). These areas were characterized by higher $CH_4$ emissions under the baseline. Unlike the continuous anaerobic conditions favored for $CH_4$ production, the AWD irrigation method intermittently enhances soil redox potential by altering soil water dynamics, which in turn inhibits $CH_4$ emissions. Previous


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research has also demonstrated that this intermittent wet−dry condition was conducive to the production of $N_2O$.[56,68] However, this study found a limited increase in $N_2O$ emissions, which could potentially be attributed to the limitations of AWD irrigation in the DNDC model.[37,52] Nonetheless, it appears that the underestimated $N_2O$ emissions under the OI scenario exert a minimal effect on the overall mitigation potential of GHG emissions.[35,36,42]

Under the OF scenario, there was a 24.8% reduction in the national average application rate of synthetic N fertilizer, whereas the application rate of manure N increased by 101.3%. Consequently, a total mitigation benefit of 18.6 Tg $CO_2$-eq could be achieved by reducing the GHG emissions associated with synthetic N fertilizer (12.3 Tg $CO_2$-eq) and enhancing SOC sequestration (6.3 Tg $CO_2$-eq) (Figure 5c). However, the increase in the proportion of manure not only improved the SOC sequestration efficiency but also supplied more manure-C for methanogens. The mitigation benefits could be partially offset by increased $CH_4$ emissions. Particularly, for late rice, increased $CH_4$ emissions counteracted 87.9% of the mitigation benefits. This is mainly because water management during the preceding early rice season establishes the favorable soil condition for methanogens to use manure C in the subsequent late rice season.[69,70] Overall, the reduction in the national FCF, PCF, and TGHG was modest, with a decrease of less than 5%.

Under the ES scenario, the national FCF, PCF, and TGHG all decreased by nearly 23% (Table S9 and Figure 5d). Among the three rice types, the FCF and PCF for early rice decreased the most (32.3%), followed by single rice (27.5%) and late rice (17.4%). Specifically, the usage of biodiesel instead of diesel for agricultural machinery operations could decrease GHG emissions by 13.6 Tg $CO_2$-eq. With the adoption of optimized irrigation, the GHG emissions from electricity consumption would be reduced by 21.7%. Furthermore, the utilization of available rice straws as bioenergy feedstock has demonstrated considerable potential in reducing GHG emissions. On the one hand, utilizing straw for biogas production could avert 8.1 Tg $CO_2$-eq of GHG emissions from straw burning. On the other hand, substituting natural gas with biogas could offset substantial GHG emissions (46.4 Tg $CO_2$-eq). Spatially, counties exhibiting greater GHG mitigation potential were chiefly situated in Northeast and Southwest China (Figure S8), which were identified as the coldspots for CF of rice production. Due to the relatively low $CH_4$ emissions in these areas, the mitigation potential was substantially influenced by variations in other emission sources.

Implementing all management practices simultaneously could augment the national average rice yield by 11.8% while decreasing the national FCF, PCF, and TGHG by 78.2, 80.2, and 76.7%, respectively (Table S9 and Figure 5e). Counties located in the North China Plain and Southwest China were found to have a greater potential for abating GHG emissions (Figure S8). Notably, a small number of counties in Yunnan, Guizhou, and Heilongjiang could achieve carbon-neutral rice production under the combined scenario.

**3.5. Implication for Management Strategies in Rice Production.** Among a variety of crop products, rice production stands as the largest GHG emitter in China. On average, the production of early, single, and late rice collectively leads to GHG emissions of 291.3 Tg $CO_2$-eq year$^{-1}$, which accounted for over 25% of China's total agricultural GHG emissions.[3,11] Furthermore, the demand for rice is further expected to ongoing increase until 2030.[44] Without the adoption of effective strategies to foster low-carbon rice production, the achievement of the "Double Carbon" goal in China's agricultural sector will face considerable challenge.

From 2007 to 2008, the spatiotemporal variation in FCF was primarily driven by $CH_4$ emissions from paddy fields, field diesel consumption, and SOC sequestration. This finding further highlights the importance of reducing field $CH_4$ emissions and enhancing SOC storage as effective pathways for mitigating CF.[18,21] Concurrently, in the context of continuous advancements in agricultural machinery, the escalated GHG emissions from field energy consumption also merit attention. Despite the limited influence of N fertilizers on the historical variability of FCF, optimizing their utilization remains a crucial strategy, as the associated GHG emissions account for over 15% of total GHG emissions. In fact, both the application rates of N fertilizers and the associated EF in China are higher than those in Western countries.[38] Apart from the direct soil $N_2O$ emissions caused by N fertilizers, excessive N enters water bodies in the form of reactive N through runoff and leaching processes,[71] which indirectly results in $N_2O$ emissions from rivers.[72] This part of the emissions is not included in the scope of our study because of the unclear mechanism of how reactive N loss during rice cultivation affects $N_2O$ emissions in water bodies.

Based on the analysis of historical changes and driving factors, the GHG mitigation potential of widely discussed management strategies for rice production was evaluated at the county level. Our study reinforced previous findings that the optimized irrigation method (AWD) from rice straw demonstrated significant GHG mitigation potential.[43,45] Optimized irrigation can not only reduce $CH_4$ emissions by 64.7% but also significantly improve irrigation water productivity.[35] Under global warming, China's rice paddies are expected to emit an additional 0.73 Tg $CH_4$ for every 1 °C increase in temperature.[50] Additionally, the northward expansion of the rice cultivation belt[73] will further increase the irrigation demand for rice growth and irrigation-related GHG emissions. Therefore, the promotion of AWD irrigation technology is an urgent mitigation strategy to escape this vicious cycle caused by climate change. Despite the benefits in terms of GHG mitigation and water saving, the practices of AWD technology typically necessitates the utilization of soil water monitoring tools and elevated technical expertise among farmers.[35] To further motivate farmers in adopting this optimized irrigation, the Chinese government needs to exert more efforts to progressively overcome the socioeconomic barriers, such as augmenting field irrigation infrastructure,[74] offering specialized training to farmers, and instituting an irrigation water volume-based pricing system.[42]

Moreover, biogas production from rice straw can significantly avoid 54.5 Tg $CO_2$-eq from straw burning and natural gas consumption. According to the latest findings, the mitigation benefits would be further improved through the concurrent production of biochar and bioenergy from crop straw.[45] However, the utilization of straw is subject to numerous constraints, including available straw resources, environmental conditions, and farmers' willingness. More advanced straw collection and treatment technologies, coupled with well-developed market and policy mechanisms, are anticipated to accelerate the energy utilization of straw resources.[75] Additionally, to maximize the environmental and economic benefits, further surveys and research are required to determine the most


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effective straw utilization approach tailored to local socio-economic contexts.

By 2030, China's target for rice production could be met through prudent management of N inputs and intensive agriculture production.[44,76] This study also explored the impact of increased rice yield and optimized N inputs on the CF of rice production under the YI and MS scenarios, respectively. Although augmented rice biomass can transport more oxygen to the rhizosphere for $CH_4$ oxidation, our findings indicate that the promotional effect of more substrates on methanogenesis seems more pronounced. Consequently, the increase in rice yield would lead to a higher FCF, but a lower PCF. The adoption of new high-yielding rice cultivars characterized by low $CH_4$ emissions represents a promising solution to address this challenge.[18,54,77] Lowering the application rate of synthetic N fertilizers and increasing the proportion of manure also represent effective strategies for reducing GHG emissions associated with N fertilizers while simultaneously increasing SOC storage. Presently, due to the increased economic expenses and limited systematic knowledge, farmers demonstrate a relatively low propensity toward the usage of manure. Therefore, relevant policy support is essential to promote the implementation of this strategy, including providing targeted subsidies on manure to improve economic profits of different participants (e.g., enterprises, retailers, and farmers),[78] developing a transparent manure distribution market to ensure the availability of manure and fertilizer machinery,[79] enhancing farmers' awareness and practical ability in optimized fertilization through publicity and on-site guidance.[80]

Among all scenarios, the combined scenario could achieve the greatest GHG mitigation benefits (232.4 Tg $CO_2$-eq), while concomitantly enhancing rice productivity (11.8%). This implies that relying solely on individual mitigation measures falls short of achieving GHG emission reduction targets. While incorporating a multitude of measures, the trade-offs among these measures necessitate meticulous consideration.[44,45,54] Future endeavors aimed at advancing low-carbon rice production should fully leverage the benefits of various measures and develop systematic management strategies that are tailored to the economic and societal conditions of each locality.

In summary, despite the limitations and uncertainties of our downscaled data and DNDC model, our study conducted a comprehensive assessment of county-level CF for China's rice production by coupling the LCA and DNDC model. From 2007 to 2018, the average FCF of rice production has shown a significant annual increase of 74.3 kg $CO_2$-eq $ha^{-1}$, while the average PCF has remained relatively stable. Furthermore, significant spatial heterogeneity was observed in the CF across China, with the hotspots identified in the North China Plain, Northwest China, and the Middle-Lower Yangtze Plain for the FCF, as well as South China for the PCF. With the implementation of the optimized management strategies delineated in this study, the national TGHG from rice production could be reduced by a maximum of 76.7% while concurrently preserving rice yields. However, a shortfall of 70.7 Tg $CO_2$-eq remains toward China's carbon neutrality goal. Other management strategies, such as new high-yielding rice varieties,[56,77] knowledge-based N managements,[81] and integrated pyrolysis and electricity generation (IPEG) system,[45] can be explored to close this gap. The achievement of these strategies will require collaborative efforts among multiple stakeholders, including the government, agri-materials supply chain, and smallholder farmers.

## ■ ASSOCIATED CONTENT

### Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.4c00539.

Specific description of the system boundary, detailed data source and methods utilized to quantify county-level CF, carbon emission factors, establishment of the DNDC model, historical variations and mitigation potential for CF and TGHG, and comparisons among this study and previous studies (PDF)

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### Notes

The authors declare no competing financial interest.

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