



Article

Spatiotemporal Dynamics of Carbon Footprint of Main Crop Production in China

Jianling Fan ^{1,*} , Dengwei Guo ¹, Lu Han ¹, Cuiying Liu ², Chuanhong Zhang ¹, Jianan Xie ³, Junzhao Niu ¹ and Liwen Yin ¹

- ¹ Jiangsu Key Laboratory of Atmospheric Environment Monitoring and Pollution Control, Jiangsu Collaborative Innovation Center of Atmospheric Environment and Equipment Technology, School of Environmental Science and Engineering, Nanjing University of Information Science and Technology, Nanjing 210044, China
- ² Jiangsu Key Laboratory of Agricultural Meteorology, School of Applied Meteorology, Nanjing University of Information Science and Technology, Nanjing 210044, China
- ³ Reading Academy, Nanjing University of Information Science and Technology, 219 Ningliu Road, Nanjing 210044, China
- * Correspondence: jlfan@nuist.edu.cn

Abstract: As a major agricultural country, the comprehensive accounting of the dynamics and composition of the carbon footprint of major crops in China will provide a decision-making basis for environmental management and agricultural green development in the whole process of the major crop production system in China. To investigate the spatiotemporal dynamics of the carbon footprint for major crops in China, a life cycle-based carbon footprint approach was used to evaluate the carbon footprint per unit area (CF_A) and per unit yield (CF_Y) of eight crops for the period of 1990 to 2019. Our results showed that the CF_A for all major crops showed an increasing trend with time before 2016 but slowly decreased afterward, while the CF_Y decreased by 16–43% over the past 30 years due to the increase in crop yield. The three main grain crops, rice ($4871 \pm 418 \text{ kg CO}_2\text{-eq} \cdot \text{ha}^{-1}$), wheat ($2766 \pm 552 \text{ kg CO}_2\text{-eq} \cdot \text{ha}^{-1}$), and maize ($2439 \pm 530 \text{ kg CO}_2\text{-eq} \cdot \text{ha}^{-1}$), showed the highest carbon footprint and contribution to the total greenhouse gas (GHG) emissions, mainly due to their larger cultivated areas and higher fertilizer application rates. CH_4 emission was the major component of the carbon footprint for rice production, accounting for 66% and 48% of the CF_A and CF_Y , respectively, while fertilizer production and usage were the largest components of carbon footprint for dryland crops, making up to 26–49% of the CF_A and 26–50% of the CF_Y for different crops. The present study also highlighted the spatial and temporal patterns of the carbon footprint for major crops in China, which could serve as references for the development of best management practices for different crop production in China, to mitigate agricultural GHG emission and to pursue low-carbon agriculture.



Citation: Fan, J.; Guo, D.; Han, L.; Liu, C.; Zhang, C.; Xie, J.; Niu, J.; Yin, L. Spatiotemporal Dynamics of Carbon Footprint of Main Crop Production in China. *Int. J. Environ. Res. Public Health* **2022**, *19*, 13896. <https://doi.org/10.3390/ijerph192113896>

Academic Editor: Paul B. Tchounwou

Received: 29 September 2022

Accepted: 22 October 2022

Published: 26 October 2022

Publisher's Note: MDPI stays neutral with regard to jurisdictional claims in published maps and institutional affiliations.



Copyright: © 2022 by the authors. Licensee MDPI, Basel, Switzerland. This article is an open access article distributed under the terms and conditions of the Creative Commons Attribution (CC BY) license (<https://creativecommons.org/licenses/by/4.0/>).

Keywords: agricultural carbon footprint; life cycle assessment; greenhouse gas emission; crop production; mitigation

1. Introduction

Agriculture generates around a quarter of the global greenhouse gas (GHG) emissions [1], including 52% and 84% of global anthropogenic CH_4 and N_2O emissions, respectively [2]. The agricultural sector, however, is vital to human beings by providing a large number of products and services. With 18% of the global population, China holds the world's largest agriculture market, with a 46.3% increase in crop yields from 2000 to 2015 [3,4]. Agricultural development in China has involved a high input of chemical fertilizers, pesticides, and agricultural film in the process of pursuing the increase in productivity, which lead to serious negative environmental effects, including the damage to natural resources, decrease in land productivity, accelerated spread of pests and diseases, and reduction in biodiversity [5,6]. Therefore, it is of great importance to analyze GHG emissions from crop production to make climate-resilient strategies for agricultural development in China.

Life cycle assessment (LCA) is a methodology for assessing the environmental aspects and potential impacts associated with a product [7]. Agricultural LCA comprehensively summarizes the impact of agriculture on the environment, which is an effective tool to promote the sustainability and green development of agriculture. In recent years, agricultural LCA has been widely used in agroecosystems for resource and environmental impact analysis, such as in intensive agriculture systems [8–10] and in organic agriculture [11–13]. The impact categories of the environmental load covered in LCA include climate change, ozone depletion, acidification, freshwater eutrophication, marine eutrophication, human toxicity, photochemical oxidant formation, particulate matter formation, terrestrial ecotoxicity, freshwater ecotoxicity, marine ecotoxicity, ionizing radiation, urban land use, and fossil energy consumption [14].

Recently, carbon footprint has been widely adopted as an efficient environmental indicator for climate change feedbacks, which is defined as the direct and indirect CO₂ emission of a product or service in the life cycle process [15]. The carbon footprint calculation includes all GHG emissions as CO₂ equivalents (CO₂-eq), which permits the quantification of the whole life cycle of a product manufactured within a certain system [16]. Cheng, et al. [17] used national statistical data to evaluate the carbon footprint of crop production in China from 1993 to 2007, and found that the carbon emission of crop production in China was up to 119.5 Mt in 14 years, and the carbon footprint per unit area (CF_A) was 0.78 ± 0.08 t CO₂-eq ha⁻¹. Yan, et al. [18] calculated the carbon footprint of major grain crops in China by a questionnaire survey of farmers in the representative region of eastern China. The CF_A of rice, wheat, and maize were 6.0 ± 0.1 , 3.0 ± 0.2 , and 2.3 ± 0.1 t CO₂-eq ha⁻¹, respectively, while the carbon footprint per unit yield (CF_Y) were 0.8 ± 0.02 , 0.66 ± 0.03 , and 0.33 ± 0.02 kg CO₂-eq kg⁻¹, respectively. Hillier, et al. [19] reported the carbon footprint for major crops in the UK and found that over 75% of the carbon footprint resulted from fertilizer application. Heidari, et al. [20] estimated the carbon footprint of durum wheat production in Iran and found that the variation in crop yield accounted for the majority of the variation in the carbon footprint. However, there are limited studies on the long-term spatial dynamic change of the agricultural carbon footprint and its composition assessment.

To quantify the mitigation potential of management strategies in China's agriculture, we evaluated the spatiotemporal dynamics of the carbon footprint for major crops in China from 1990 to 2019, based on national and local statistical data. In order to provide a decision-making basis for the whole process of environmental management of China's main crop production system, the influencing factors on the carbon footprint for major crop production were further analyzed. The specific objectives were: (1) to quantify the temporal and spatial variation in the carbon footprint of major crop production in China over the last 30 years and (2) to identify the main influencing factors on carbon footprint for different crop production systems in different regions.

2. Materials and Methods

2.1. The Study Area and Main Crops in Different Provinces

The carbon footprints of major crop production were calculated for 31 provinces of China from 1990 to 2019. Over the past 30 years, the averaged total sown area of crops in China has been 156,800 ha, among which eight major crops, including rice, wheat, maize, soybean, rapeseed, peanut, cotton, and highland barley, accounted for 62% to 96% of the total sown area for different provinces (Table S1). Therefore, we took the main crops from each province (Table S1) as the research objects and calculated the spatiotemporal dynamics of their carbon footprint.

2.2. Inventory Data Collection

Agricultural input data of major crops mainly include chemical fertilizers, seeds, pesticides, fuel, agricultural machinery and tools, and irrigation electricity consumption. Crop yields were calculated from the sown area and total yield from the China Statistical Yearbook series throughout the period of 1990–2019 [21]. The input data of chemical fertilizer and seeds

for each province were available from the National Agricultural Product Cost and Income Data Compilation [22]. Pesticides, fuels, and irrigation power consumption were calculated indirectly through the National Agricultural Product Cost and Income Data Compilation [22] and the Price Yearbook of China [23]. Agricultural machinery and related parameters were obtained from the agricultural machinery network (<https://toutiao.nongjitong.com/>) (accessed on 20 April 2022). The emission factors of agricultural inputs mainly came from Ecoinvent database [24] and related references [25] (Table S2).

2.3. System Boundaries and Functional Units

Total GHG emissions throughout the whole crop production process were quantitatively assessed in this study. The system boundary of crop production in this study was defined as the whole life cycle from the process of mining, production, and transportation of agricultural materials (chemical fertilizers, seeds, pesticides, etc.), to the completion of crop harvesting (Figure 1). Therefore, the sources of GHG emissions considered in this study, included the production, storage, transportation, and application of agricultural inputs, the energy consumption for machinery operation, CH₄ emission from paddy fields, and N₂O emission from N fertilizer application. The functional units (FU) were defined based on both the area and product in this study, i.e., the kg CO₂-eq ha^{−1} of sown area and kg CO₂-eq kg^{−1} of crop yield. FU based on area were used to compare the environmental effects, while FU based on the product were used to compare the production efficiency.

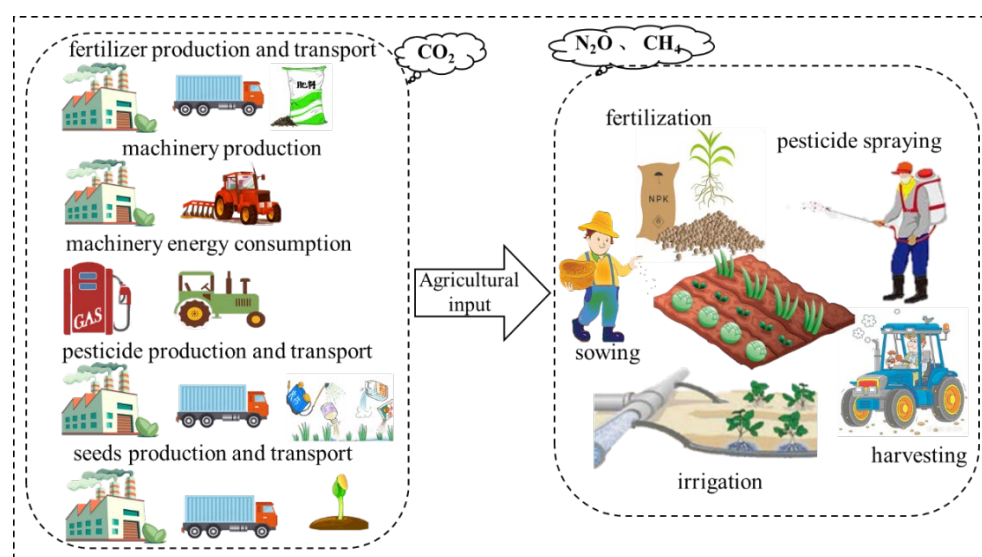


Figure 1. The system boundary of major crop production in China.

2.4. Carbon Footprint Calculation

Total GHG emissions from agricultural inputs in the carbon equivalent ($CE_{input,j}$, kg CO₂-eq · ha^{−1}) for crop j were estimated using the equation:

$$CE_{input,j} = \sum_{i=1}^n (D_{i,j} \times EF_{i,j}) \quad (1)$$

where $D_{i,j}$ is the input of the i th agricultural material for crop j (kg ha^{−1} or kWh ha^{−1}) and $EF_{i,j}$ is the emission factor of the i th agricultural material for crop j (kg kg^{−1} or kg GJ^{−1}).

In particular, total N₂O emission, including direct and indirect emissions, from agricultural land in the carbon equivalent for crop j ($CE_{N_2O,j}$, kg CO₂-eq · ha^{−1}) were calculated using the equation:

$$CE_{N_2O,j} = (F_{D,N_2O,j} + F_{A,N_2O,j} + F_{L,N_2O,j}) \times 265 \quad (2)$$

where $F_{D,N_2O,j}$ ($\text{kg N}_2\text{O ha}^{-1}$) is the direct N_2O emission from chemical N fertilizer application for crop j ; $F_{A,N_2O,j}$ and $F_{L,N_2O,j}$ ($\text{kg N}_2\text{O ha}^{-1}$) are the indirect N_2O emission caused by atmospheric deposition, leaching, and runoff, respectively, for crop j ; and 265 is the global warming potential value of N_2O at the 100-year time horizon [26].

Direct N_2O emission $F_{D,N_2O,j}$ for crop j ($\text{kg N}_2\text{O ha}^{-1}$) was calculated using the equation:

$$F_{D,N_2O,j} = N_{\text{input},j} \times F_{1,j} \times 44/28 \quad (3)$$

where $N_{\text{input},j}$ (kg ha^{-1}) is the amount of N fertilizer application for crop j in farmland; $F_{1,j}$ ($\text{kg N}_2\text{O-N kg}^{-1}$) is the N_2O direct emission factor caused by N fertilizer application for crop j ; and 44/28 is the ratio of $\text{N}_2\text{O-N}$ to N_2O molecular weight.

Indirect N_2O emission $F_{A,N_2O,j}$ and $F_{L,N_2O,j}$ ($\text{kg N}_2\text{O ha}^{-1}$) for crop j were calculated using the equations:

$$F_{A,N_2O,j} = N_{\text{input},j} \times R_{A,j} \times F_{2,j} \times 44/28 \quad (4)$$

$$F_{L,N_2O,j} = N_{\text{input},j} \times R_{L,j} \times F_{3,j} \times 44/28 \quad (5)$$

where $R_{A,j}$ is the volatilization rate of ammonia and NOx from agricultural land for crop j ; $R_{L,j}$ is the N leaching and runoff rate of farmland for crop j ; $F_{2,j}$ ($\text{kg N}_2\text{O-N kg}^{-1}$) is the indirect emission factor of N_2O caused by nitrogen deposition for crop j ; and $F_{3,j}$ ($\text{kg N}_2\text{O-N kg}^{-1}$) is the indirect emission factor of N_2O caused by leaching and runoff for crop j .

Furthermore, CH_4 emissions from paddy fields in the carbon equivalent (CE_{CH_4} , $\text{kg CO}_2\text{-eq} \cdot \text{ha}^{-1}$) were estimated using the equation:

$$CE_{\text{CH}_4} = F_{\text{CH}_4} \times 28 \quad (6)$$

where F_{CH_4} ($\text{kg CH}_4 \text{ ha}^{-1}$) is the CH_4 emission factor for paddy fields and 28 is the global warming potential of CH_4 at the 100-year time horizon [26].

Therefore, total GHG emissions in the carbon equivalent for crop j ($CE_{t,j}$, $\text{kg CO}_2\text{-eq} \cdot \text{ha}^{-1}$) were calculated using the equation:

$$CE_{t,j} = CE_{\text{input},j} + CE_{N_2O,j} + CE_{\text{CH}_4} \quad (7)$$

Carbon footprint per unit area ($CF_{A,j}$, $\text{kg CO}_2\text{-eq} \cdot \text{ha}^{-1}$) and carbon footprint per unit yield ($CF_{Y,j}$, $\text{kg CO}_2\text{-eq} \cdot \text{kg}^{-1}$) for crop j in different provinces were calculated using the equations:

$$CF_{A,j} = CE_{t,j} \quad (8)$$

$$CF_{Y,j} = \frac{CE_{t,j}}{Y_j} \quad (9)$$

where A_j is the sown area of crop j (ha) and Y_j (kg ha^{-1}) is the yield of crop j .

Furthermore, the total mean carbon footprint per unit area (mCF_A , $\text{kg CO}_2\text{-eq} \cdot \text{ha}^{-1}$) and CF per unit yield (mCF_Y , $\text{kg CO}_2\text{-eq} \cdot \text{kg}^{-1}$) for each province were calculated using the equations by considering all the crops together:

$$mCF_A = \frac{\sum_{j=1}^8 CE_{t,j} \times A_j}{\sum_{j=1}^8 A_j} \quad (10)$$

$$mCF_Y = \frac{\sum_{j=1}^8 CE_{t,j} \times A_j}{\sum_{j=1}^8 Y_j \times A_j} \quad (11)$$

2.5. Statistical Analysis

One-way ANOVA and the least significant difference test (LSD) at the level of 5% were used to compare the differences in the CF_A and CF_Y among the different crops. All statistical analyses were performed using R software [27].

3. Results

3.1. Temporal Dynamics of Carbon Footprint for Major Crops

The carbon footprint per unit area (CF_A) showed significant differences for different crops regardless of interannual variation (Figure 2a), which were highest for rice (4333–5195 kg CO₂-eq · ha^{−1}), followed by wheat (2123–3091 kg CO₂-eq · ha^{−1}), maize (1838–2766 kg CO₂-eq · ha^{−1}), highland barley (1653–2182 kg CO₂-eq · ha^{−1}), rapeseed (1378–1984 kg CO₂-eq · ha^{−1}), cotton (1109–1516 kg CO₂-eq · ha^{−1}), peanut (878–1178 kg CO₂-eq · ha^{−1}), and soybean (753–940 kg CO₂-eq · ha^{−1}). Generally, the CF_A for all major crops showed an increasing trend with time, which increased rapidly from 1990 to 1998, slowed down after 1999, and slowly decreased from 2016 to 2019.

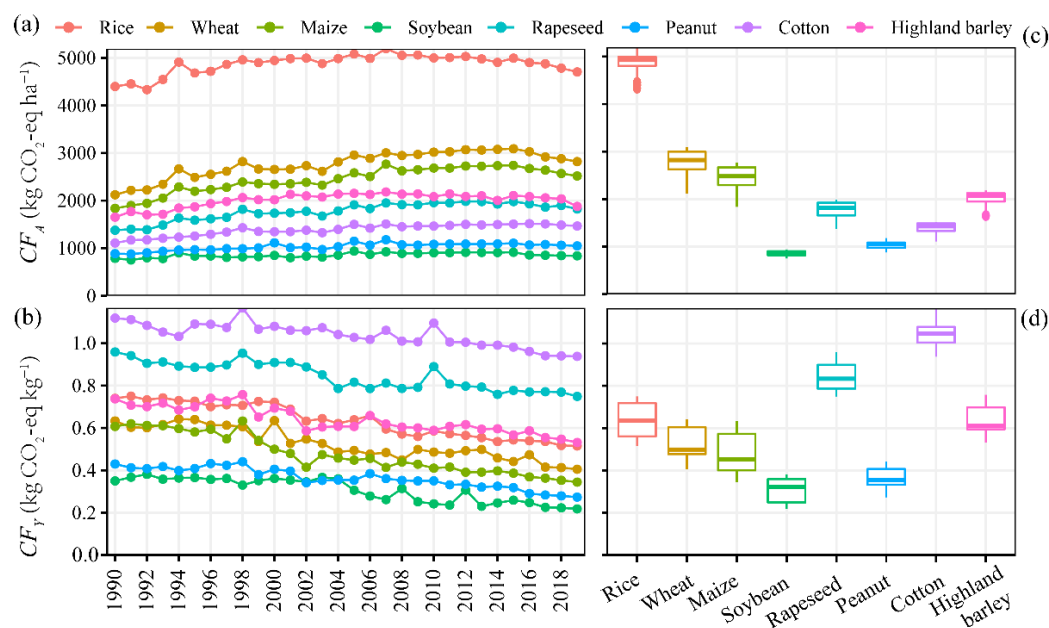


Figure 2. Temporal dynamics of the carbon footprint (a) per unit area and (b) per unit yield of major crops in China from 1990 to 2019, and the averaged carbon footprint (c) per unit area and (d) per unit yield of major crops in China.

In contrast, the carbon footprint per unit yield (CF_Y) showed an overall decreasing trend with fluctuation in some years (Figure 2b), which was mainly due to the increase in crop yield with time. In 1998, however, some areas of China suffered from catastrophic floods, which resulted in an extremely low yield and then in turn, a higher CF_Y . In 2003 and 2010, high temperature and drought led to a dramatic reduction in crop yield, and thus a higher CF_Y for several crops. As influenced by crop yield, the rank of CF_Y for major crops was not consistent with that of CF_A , where cotton showed the highest CF_Y (0.94–1.17 kg CO₂-eq · kg^{−1}), followed by rapeseed (0.75–0.96 kg CO₂-eq · kg^{−1}), highland barley (0.53–0.76 kg CO₂-eq · kg^{−1}), rice (0.51–0.75 kg CO₂-eq · kg^{−1}), wheat (0.40–0.64 kg CO₂-eq · kg^{−1}), maize (0.22–0.96 kg CO₂-eq · kg^{−1}), peanut (0.27–0.44 kg CO₂-eq · kg^{−1}),

and soybean ($0.22\text{--}0.38 \text{ kg CO}_2\text{-eq} \cdot \text{kg}^{-1}$). The higher CF_Y of cotton and rapeseed could mainly be attributed to their lower yield but higher fertilizer requirements.

3.2. Distribution of Carbon Footprint in Different Regions

The annual mean CF_A and CF_Y for major crops from 1990 to 2019 showed different spatial patterns due to the distributions of cropping area and yield across China (Figures 3 and 4). For instance, the spatial distribution pattern of the annual mean CF_A of wheat was higher in the northern provinces but lower in the southwest provinces, where the highest value of $3740 \text{ kg CO}_2\text{-eq ha}^{-1}$ was obtained in Shanxi province and the lowest value of $2361 \text{ kg CO}_2\text{-eq ha}^{-1}$ was obtained in Guizhou province (Figure 3). However, the annual mean CF_A of soybean was higher in the eastern and southwest provinces but lower in the middle provinces of China, where the highest value of $1272 \text{ kg CO}_2\text{-eq ha}^{-1}$ was obtained in Jiangsu province and the lowest value of $966 \text{ kg CO}_2\text{-eq ha}^{-1}$ was obtained in Chongqing province.

In order to take all major crops together into consideration, the annual total mean carbon footprints (mCF_A and mCF_Y) for cropping lands in different provinces were calculated by averaging the values from 1990 to 2019 (Figure 5). The higher mCF_A values of crop production were mainly obtained in Xinjiang province in the west, Zhejiang, Shandong, Jiangsu, and Zhejiang in the east, and Hubei, Guangdong, and Guangxi in the middle of China, where the mCF_A was more than $2800 \text{ kg (CO}_2\text{-eq) ha}^{-1}$. Lower mCF_A values of crop production were obtained in Qinghai, Guizhou, Sichuan, and Chongqing, where the mCF_A was below $2000 \text{ kg (CO}_2\text{-eq) ha}^{-1}$. In general, the mCF_A values were higher in the eastern and western provinces but lower in the middle provinces (Figure 5a). On the other hand, the mCF_Y for major crops in China ranged from 0.34 to $0.78 \text{ kg (CO}_2\text{-eq) kg}^{-1}$ (Figure 5b). The mCF_Y of 15 provinces, including Guangxi, Sichuan, Guizhou, and Yunnan, were lower than the national average of $0.61 \text{ kg (CO}_2\text{-eq) kg}^{-1}$, which were mainly obtained in the southwest and northeast regions of China. However, other provinces with a higher annual mean mCF_Y were distributed in the northwest and eastern coastal regions of China.

3.3. Composition of Carbon Footprint for Different Crops

According to the defined system boundary (Figure 1), the carbon footprint of crop production in this study consisted of the energy consumption by machinery operation, fertilizer production, pesticides, seeds, CH_4 emission from paddy fields, N_2O emission from N fertilizer application, and electricity consumption by irrigation. The relative contribution of these components to the CF_A and CF_Y for different provinces were calculated and their annual mean values from 1990 to 2019 are shown in Figures S1–S8 for different crops. Although the composition of the annual mean carbon footprint (both CF_A and CF_Y) of different crops showed slight differences among different provinces, the rank of these components was consistent for each crop (Figures S1–S8). Therefore, composition values of the CF_A and CF_Y were averaged across provinces to show differences among crops at a national scale. For rice production, CH_4 emission was the major component of carbon footprint, accounting for 66% and 48% of CF_A and CF_Y , respectively (Figure 6). Fertilizer production and usage as well as N_2O emission from N fertilizer application made up 11% and 7% of CF_A for rice production, but 15% and 16% of CF_Y , respectively.

For dryland crops, however, fertilizer production and usage were the largest composition of carbon footprint, making up 26–49% of CF_A and 26–50% of CF_Y for different crops, respectively (Figure 6 and Table S3). N_2O emission from N fertilizer application was the second largest component of carbon footprint, representing 21–35% of CF_A and 16–25% of CF_Y for different crops, respectively. In contrast, the proportion of other components to the carbon footprint was comparatively small, accounting for 5–13% of CF_A and 6–14% of CF_Y for energy consumption by machinery operation, 2–9% of CF_A and 2–10% of CF_Y for pesticide application, 4–20% of CF_A and 4–20% of CF_Y for seeds, and 4–17% of CF_A and 4–19% of CF_Y for electricity consumption by irrigation, respectively.

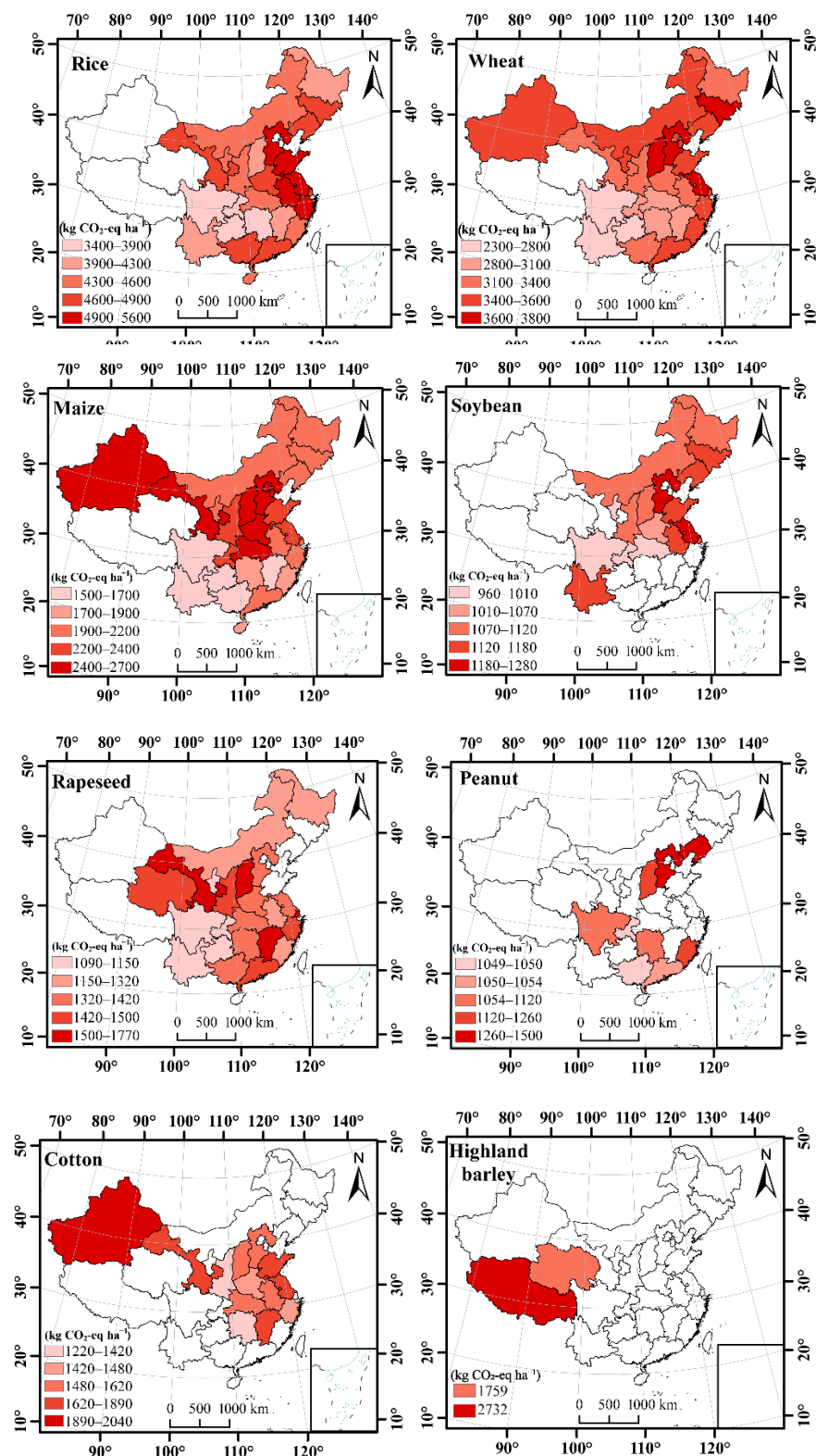


Figure 3. Spatial distributions of the annual mean carbon footprint per unit area (CF_A) for different crops in China from 1990 to 2019.

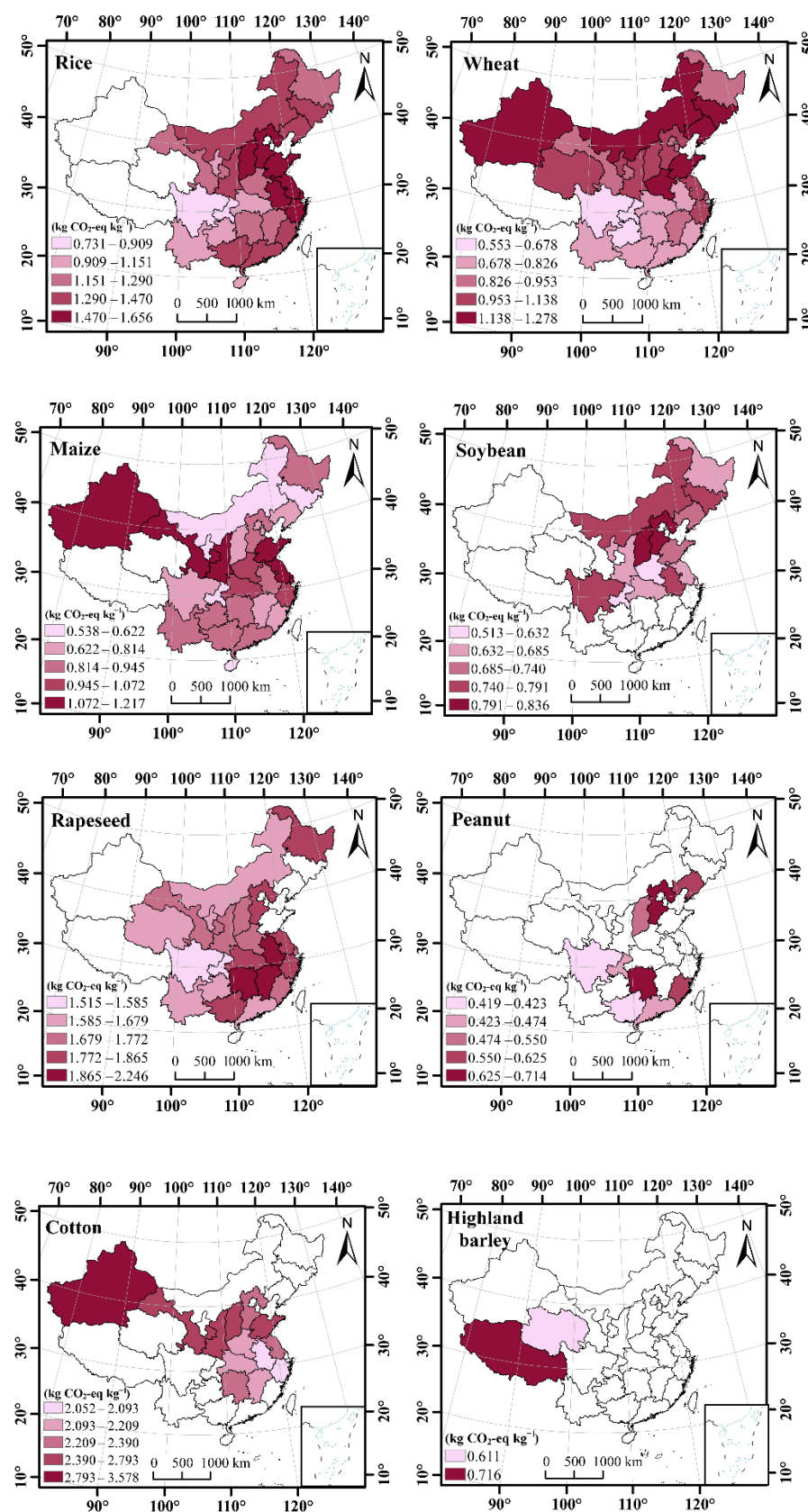


Figure 4. Spatial distributions of the carbon footprint per unit yield (CF_Y) for different crops in China from 1990 to 2019.

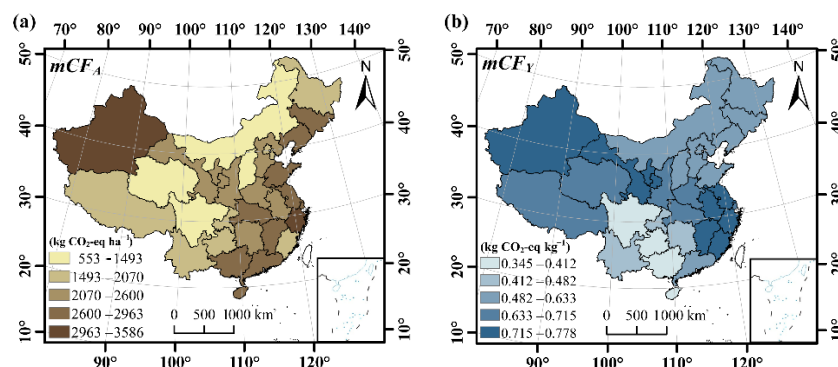


Figure 5. Spatial distributions of the annual total mean carbon footprint (a) per unit area and (b) per unit yield in China from 1990 to 2019.

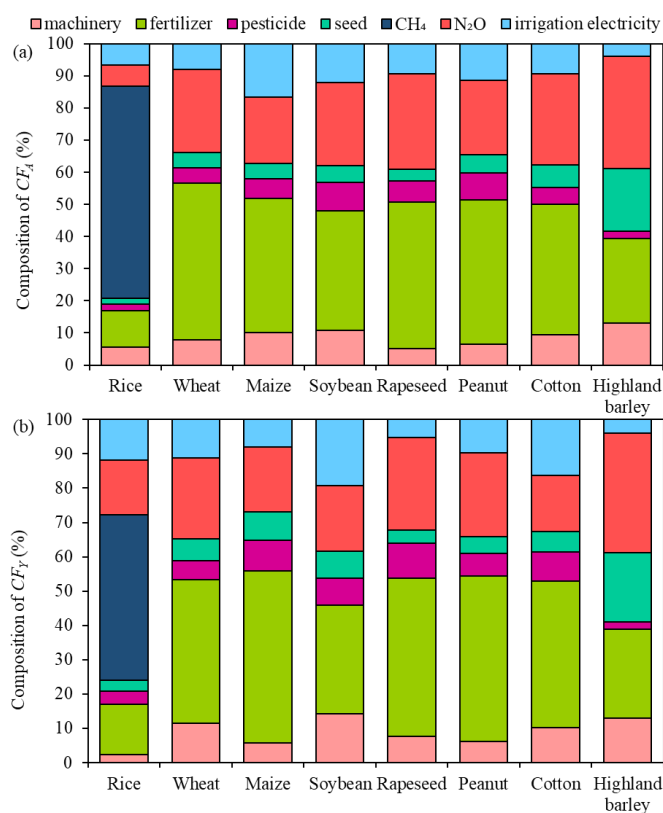


Figure 6. Compositions of the annual mean carbon footprints (a) per unit area and (b) per unit yield for different crops in China from 1990 to 2019.

4. Discussion

4.1. Dynamics of Carbon Footprint

The fossil fuel CO₂ emissions in China increased from 2.48×10^9 t CO₂-eq · in 1990 to 10.49×10^9 t CO₂-eq · in 2019 [28]. The carbon footprint of the eight major crop productions estimated in the present study corresponded to ~7% of the national total emissions, revealing the importance of GHG emissions from crop production in China. The CF_A for all major crops showed an increasing trend with time before 2016 but slowly decreased afterward (Figure 2), which fluctuated as the cultivated area changed (Figure S9). This trend is in agreement with Wu, et al. [29], who found that national GHG emissions per unit area increased from 980 kg CO₂-eq · ha^{−1} in 1998 to 1200 kg CO₂-eq · ha^{−1} in 2016. In contrast, the CF_Y decreased by 16–43% over the past 30 years for different crops (Figure 2b), which was significantly correlated with crop yield (Figure 7).

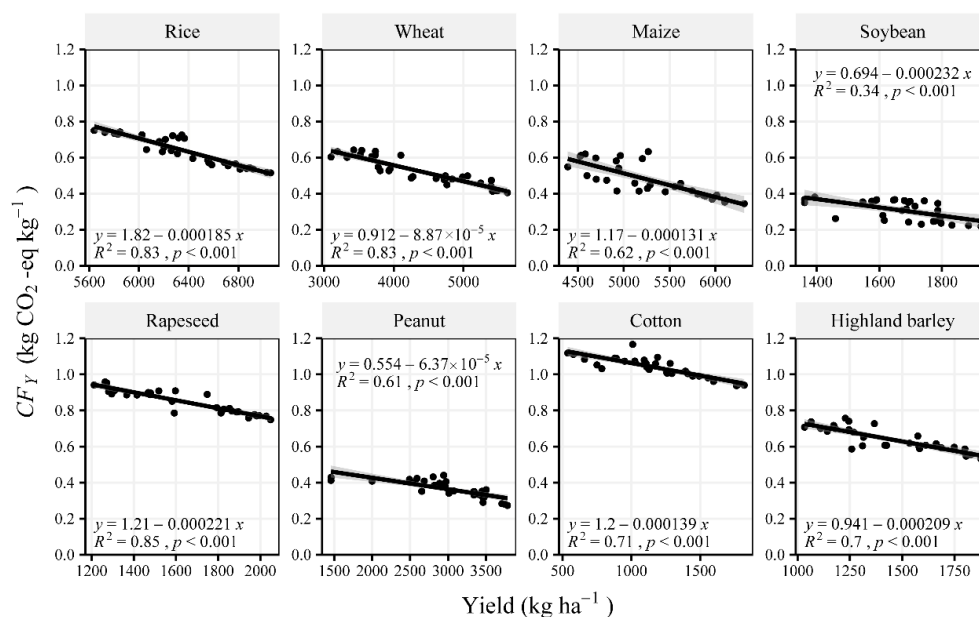


Figure 7. Correlations between the carbon footprint per unit yield (CF_Y) and crop yield for the study period.

The CF_A and/or CF_Y for all major crops in this study (Figure 2) were markedly higher than those reported for other countries. For instance, the CF_A of wheat was reported to be $350 \text{ kg CO}_2\text{-eq} \cdot \text{ha}^{-1}$ in the UK [19], $110 \text{ kg CO}_2\text{-eq} \cdot \text{ha}^{-1}$ in Denmark [30], and $90 \text{ kg CO}_2\text{-eq} \cdot \text{ha}^{-1}$ in Australia [31], while the CF_A of rapeseed was $220\text{--}350 \text{ kg CO}_2\text{-eq} \cdot \text{ha}^{-1}$ in Canada [32] and the CF_A of cotton was $350\text{--}370 \text{ kg CO}_2\text{-eq} \cdot \text{ha}^{-1}$ in Australia [33]. On the other hand, the CF_Y of rice was reported to be $0.07\text{--}0.10 \text{ kg CO}_2\text{-eq} \cdot \text{kg}^{-1}$ in the USA [34], while the CF_Y of wheat was $0.07\text{--}0.14 \text{ kg CO}_2\text{-eq} \cdot \text{kg}^{-1}$ in Canada [35]. The CF_Y of maize was $0.06 \text{ kg CO}_2\text{-eq} \cdot \text{kg}^{-1}$ in the USA [36] and the CF_Y of cotton was $0.43\text{--}0.48 \text{ kg CO}_2\text{-eq} \cdot \text{kg}^{-1}$ in Australia [33]. Higher CF_A and CF_Y values in China provide a greater potential for mitigating GHG emissions in Chinese cropland.

The CF_A and CF_Y for all dryland crops in the present study were dramatically dependent on chemical fertilizer, including chemical fertilizer production, transportation, and N_2O emissions from chemical fertilizer use, which together contributed to 61–74% of the CF_A and 51–73% of the CF_Y , respectively (Figure 6). These values were comparable with those reported by Cheng, et al. [17] (57%) from 1993 to 2007, and Liu, et al. [37] (50.7%) from 2000 to 2015 in China, but lower than the mean proportion of 75% for UK croplands in 2007 [19]. Therefore, the GHG emissions from fertilizer that were produced and then used on cropland contributed the most to the GHG emissions of dryland crop production [38]. During the study period, the CF_A reached the highest value in 2015 and decreased afterward (Figure 2a), resulting from the decrease in the chemical fertilizer application amount by 10% from 2015 to 2019 (Figure S10) by the implementation of the “action plan for zero growth in fertilizer use by 2020” at a national scale since 2015. Our results indicated that a cut in chemical fertilizer use and adoption of science-based fertilizer applications in crop production in China will offer a great option to reduce the national total GHG emissions. Furthermore, controlling N leaching, runoff, and volatilization losses after application by using deep placement and/or other techniques could also mitigate GHG emissions [39]. In addition, the use of organic fertilizer and other amendments, such as N-saving and slow-release biochar-based fertilizers could reduce GHG emissions and increase the crop yield, which further offer measures to cut down the carbon footprint from chemical fertilizer production and usage in China [37,40,41].

4.2. Carbon Footprint for Different Crops and GHG Mitigation Suggestions

Our results show that the CF_A values of rice, wheat, and maize are generally higher than those of other crops (soybean, rapeseed, peanut, cotton, and highland barley; Figure 2a). Lin, et al. [42] also found that rice, maize, and vegetables were the top three contributors to the grain carbon footprint in 2009. Compared with other dryland crops, CH_4 emission from paddy fields is the biggest contributor to the CF_A of rice (Figure 6). Rice is the crop with the highest fertilizer application rate, of $\sim 300 \text{ kg N ha}^{-1}$ in China [43], among the eight studied crops. However, it has been reported that N fertilizer application above 225 kg N ha^{-1} had little, or even a negative, impact on the rice yield and would increase GHG emissions [44]. Zhang, et al. [45] found that a 10% reduction in chemical N fertilizer in paddy fields relative to the current farmer's application rate (300 kg N ha^{-1}) could realize high yields and N use efficiency as well as low environmental impacts. In addition, rice production consumes more electricity per unit area than other crops for irrigation, which is also an important factor leading to a higher carbon footprint for rice. Solar energy-assisted electricity is suggested as a renewable energy source for irrigation [46].

Rice, wheat, and maize have the three largest cultivated areas in China (Figure S9), which further strongly affect the spatial distribution of the total carbon footprint (Figure 5). A reduction in the cultivated areas of these crops would remarkably reduce their carbon footprint [29], but would compromise national food supplies and food security in China. In 2018, the Ministry of Agriculture of China issued "Key Points of Planting in 2018", which plan to reduce the rice area by 2–3%, stabilize or reduce the wheat area by 1%, and reduce the maize area by 9–10%, to improve their production efficiencies. However, these plans are mainly being implemented in the regions with low productivity, high resource consumption, and insect damage.

In contrast, soybean, rapeseed, peanut, cotton, and highland barley contributed a relatively much lower carbon footprint during the study period, mainly due to the significantly smaller cultivated areas of these five crops (Figure S9). On the other hand, their fertilizer application rates were also much smaller, except for cotton. For instance, the fertilizer application rate of oil crops is only about $200 \text{ kg} \cdot \text{ha}^{-1}$. Zhang, et al. [47] found that the rapeseed production mode could be more efficient by adopting direct sowing, machine sowing, and harvesting. However, soybean is more energetically efficient than rapeseed due to its lower N fertilizer requirement [48]. China is the second largest consumer of oil crops in the world but is reliant on imports. In 2021, China produced 16.4 million tons of soybeans, which met the needs of edible consumption, but also imported 96.5 million tons, mainly used to make edible oils and animal fodder. Therefore, increasing the yield of oil crops, especially soybean, without increasing either fertilizer use or cultivated areas may be an important priority in their production to reduce the dependency on imports.

Although both the cultivated area (Figure S9) and yield of cotton were the smallest among all the crops, cotton had the highest CF_Y (Figure 2b) due to its high fertilizer application rate. Ma, et al. [49] suggested that plastic mulch and drip irrigation in cotton production could significantly decrease N_2O emissions as well as improve water use efficiency. Powell, et al. [50] found that rotating cotton with pulse crops, instead of wheat, could reduce GHG emissions by 8%. In China, 76% of cotton was cultivated in Xinjiang province in 2019, where the soil conditions such as low soil organic matter, light soil texture, and high soil pH, as well as climate factors of low precipitation and high evapotranspiration, restricted soil N_2O emissions. Therefore, decreasing fertilizer rates or replacing them with polymer-coated urea while improving cotton yields with more efficient production techniques should be considered by future cotton producers.

However, different strategies should be considered for different provinces. For instance, reducing chemical N fertilizer use and controlling the pathways of N loss were important GHG mitigation strategies across China [39,51]. Reduction in CH_4 emissions from paddy fields should be considered in the south, while conservation tillage measures should be implemented to mitigate future increases in GHG emissions in the north. Therefore, proper management could enhance carbon sequestration, reduce GHG emissions, as

well as increase crop yields simultaneously, and subsequently decrease the carbon footprint of crop production [37,52].

5. Conclusions

In this study, the spatiotemporal dynamics of the carbon footprint of major crops in China from 1990 to 2019 were investigated to quantify the mitigation potential of management strategies in Chinese agriculture. Our results showed that the carbon footprint of the eight major crop productions estimated in the present study corresponded to ~7% of the national total emissions, revealing the importance of GHG emissions from crop production in China. The CF_A and/or CF_Y for all major crops in this study were markedly higher than those reported for other countries, which provided greater potential for mitigating GHG emissions in Chinese cropland. CH_4 emission from paddy fields is the biggest contributor to the CF_A of rice, followed by fertilizer production and usage, while the CF_A and CF_Y for all dryland crops were dramatically dependent on chemical fertilizer. Therefore, cutting chemical fertilizer use and adopting science-based fertilizer applications, controlling N leaching, runoff, and volatilization losses after application by using deep placement and/or other techniques, as well as using organic fertilizer and other amendments, will offer a great option to cut down the carbon footprint of crop production in China.

Supplementary Materials: The following supporting information can be downloaded at: <https://www.mdpi.com/article/10.3390/ijerph192113896/s1>, Figure S1: Composition of the annual mean carbon footprint per unit area (CF_A) and per unit yield (CF_Y) of rice production for different provinces from 1990 to 2019; Figure S2: Composition of the annual mean carbon footprint per unit area (CF_A) and per unit yield (CF_Y) of wheat production for different provinces from 1990 to 2019; Figure S3: Composition of the annual mean carbon footprint per unit area (CF_A) and per unit yield (CF_Y) of maize production for different provinces from 1990 to 2019; Figure S4: Composition of the annual mean carbon footprint per unit area (CF_A) and per unit yield (CF_Y) of soybean production for different provinces from 1990 to 2019; Figure S5: Composition of the annual mean carbon footprint per unit area (CF_A) and per unit yield (CF_Y) of rapeseed production for different provinces from 1990 to 2019; Figure S6: Composition of the annual mean carbon footprint per unit area (CF_A) and per unit yield (CF_Y) of peanut production for different provinces from 1990 to 2019; Figure S7: Composition of the annual mean carbon footprint per unit area (CF_A) and per unit yield (CF_Y) of cotton production for different provinces from 1990 to 2019; Figure S8: Composition of the annual mean carbon footprint per unit area (CF_A) and per unit yield (CF_Y) of highland barley production for different provinces from 1990 to 2019; Figure S9: Changes in the cultivated area of major crops in China from 1990 to 2019; Figure S10: Changes in the chemical fertilizer application amount in China from 1990 to 2019; Table S1: Annual averaged sown area of major crops for different provinces in China during 1990–2019; Table S2: Greenhouse gas emission factors for agriculture inputs; Table S3: Composition of the annual mean CF_A and CF_Y for different crops in China from 1990 to 2019.

Author Contributions: Conceptualization, J.F. and C.L.; data curation, L.H., C.Z., D.G., J.N. and L.Y.; writing—original draft preparation, J.F. and J.X.; writing—review and editing, J.F. and C.L. All authors have read and agreed to the published version of the manuscript.

Funding: This research was funded by Special Funds for Carbon Peak and Carbon Neutral Science and Technology Innovation of Jiangsu Province, grant number BE2022302; the Six Talent Peaks Project of Jiangsu Province, grant number JNHB-061; and the Startup Foundation for Introducing Talent of NUIST, grant number 2018r038.

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: The data that support the findings of this study are available upon request.

Conflicts of Interest: The authors declare no conflict of interest.

References

1. IPCC. *Climate Change 2014 Mitigation of Climate Change: Working Group III Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*; IPCC: Geneva, Switzerland, 2014.
2. Smith, P.; Martino, D.; Cai, Z.; Gwary, D.; Janzen, H.; Kumar, P.; McCarl, B.; Ogle, S.; O'Mara, F.; Rice, C.; et al. Greenhouse gas mitigation in agriculture. *Philos. Trans. R. Soc. B Biol. Sci.* **2008**, *363*, 789–813. [\[CrossRef\]](#) [\[PubMed\]](#)
3. FAO. *FAOSTAT (Food and Agriculture Organization Statistical Data) Statistical Yearbook*; FAO: Rome, Italy, 2015.
4. Saeed, Q.; Zhang, A.; Mustafa, A.; Sun, B.; Zhang, S.; Yang, X.Y. Effect of long-term fertilization on greenhouse gas emissions and carbon footprints in northwest China: A field scale investigation using wheat-maize-fallow rotation cycles. *J. Clean. Prod.* **2022**, *332*, 643–656. [\[CrossRef\]](#)
5. Tilman, D.; Reich, P.B.; Knops, J.M.H. Biodiversity and ecosystem stability in a decade-long grassland experiment. *Nature* **2006**, *441*, 629–632. [\[CrossRef\]](#) [\[PubMed\]](#)
6. Tilman, D.; Fargione, J.; Wolff, B.; D'Antonio, C.; Dobson, A.; Howarth, R.; Schindler, D.; Schlesinger, W.H.; Simberloff, D.; Swackhamer, D. Forecasting agriculturally driven global environmental change. *Science* **2001**, *292*, 281–284. [\[CrossRef\]](#) [\[PubMed\]](#)
7. Brentrup, F.; Küsters, J.; Kuhlmann, H.; Lammel, J. Environmental impact assessment of agricultural production systems using the life cycle assessment methodology: I. Theoretical concept of a LCA method tailored to crop production. *Eur. J. Agron.* **2004**, *20*, 247–264. [\[CrossRef\]](#)
8. Baldini, C.; Gardoni, D.; Guarino, M. A critical review of the recent evolution of Life Cycle Assessment applied to milk production. *J. Clean. Prod.* **2017**, *140*, 421–435. [\[CrossRef\]](#)
9. Roy, P.; Nei, D.; Orikasa, T.; Xu, Q.; Okadome, H.; Nakamura, N.; Shiina, T. A review of life cycle assessment (LCA) on some food products. *J. Food Eng.* **2009**, *90*, 1–10. [\[CrossRef\]](#)
10. Notarnicola, B.; Sala, S.; Anton, A.; McLaren, S.J.; Saouter, E.; Sonesson, U. The role of life cycle assessment in supporting sustainable agri-food systems: A review of the challenges. *J. Clean. Prod.* **2017**, *140*, 399–409. [\[CrossRef\]](#)
11. Tricase, C.; Lamonaca, E.; Ingrao, C.; Bacenetti, J.; Lo Giudice, A. A comparative Life Cycle Assessment between organic and conventional barley cultivation for sustainable agriculture pathways. *J. Clean. Prod.* **2018**, *172*, 3747–3759. [\[CrossRef\]](#)
12. Cederberg, C.; Mattsson, B. Life cycle assessment of milk production—A comparison of conventional and organic farming. *J. Clean. Prod.* **2000**, *8*, 49–60. [\[CrossRef\]](#)
13. Meisterling, K.; Samaras, C.; Schweizer, V. Decisions to reduce greenhouse gases from agriculture and product transport: LCA case study of organic and conventional wheat. *J. Clean. Prod.* **2009**, *17*, 222–230. [\[CrossRef\]](#)
14. Guinee, J.B. Handbook on life cycle assessment operational guide to the ISO standards. *Int. J. Life Cycle Assess.* **2002**, *7*, 311. [\[CrossRef\]](#)
15. Wiedmann, T.; Minx, J. A definition of 'carbon footprint'. In *Ecological Economics Research Trends*; Pertsova, C.C., Ed.; Nova Science Publishers Inc.: Hauppauge, NY, USA, 2008; Volume 1, pp. 1–11.
16. Xue, J.-F.; Yuan, Y.-Q.; Zhang, H.-L.; Ren, A.-X.; Lin, W.; Sun, M.; Gao, Z.-Q.; Sun, D.-S. Carbon footprint of dryland winter wheat under film mulching during summer-fallow season and sowing method on the Loess Plateau. *Ecol. Indic.* **2018**, *95*, 12–20. [\[CrossRef\]](#)
17. Cheng, K.; Pan, G.; Smith, P.; Luo, T.; Li, L.; Zheng, J.; Zhang, X.; Han, X.; Yan, M. Carbon footprint of China's crop production—An estimation using agro-statistics data over 1993–2007. *Agric. Ecosyst. Environ.* **2011**, *142*, 231–237. [\[CrossRef\]](#)
18. Yan, M.; Cheng, K.; Luo, T.; Yan, Y.; Pan, G.; Rees, R.M. Carbon footprint of grain crop production in China—based on farm survey data. *J. Clean. Prod.* **2015**, *104*, 130–138. [\[CrossRef\]](#)
19. Hillier, J.; Hawes, C.; Squire, G.; Hilton, A.; Wale, S.; Smith, P. The carbon footprints of food crop production. *Int. J. Agric. Sustain.* **2009**, *7*, 107–118. [\[CrossRef\]](#)
20. Heidari, M.D.; Mobli, H.; Omid, M.; Rafiee, S.; Jamali Marbini, V.; Elshout, P.M.F.; Van Zelm, R.; Huijbregts, M.A.J. Spatial and technological variability in the carbon footprint of durum wheat production in Iran. *Int. J. Life Cycle Assess.* **2017**, *22*, 1893–1900. [\[CrossRef\]](#)
21. National Bureau of Statistics. *China Statistical Yearbook, 1990–2019*; China Statistics Press: Beijing, China, 2020.
22. National Development and Reform Commission Price Department. *National Agricultural Product Cost and Income Data Compilation, 1990–2019*; China Statistics Press: Beijing, China, 2020.
23. Editorial Committee of Price Yearbook of China. *Price Yearbook of China, 1990–2013*; People's Publishing House: Beijing, China, 2014.
24. Wernet, G.; Bauer, C.; Steubing, B.; Reinhard, J.; Moreno-Ruiz, E.; Weidema, B. The ecoinvent database version 3 (part I): Overview and methodology. *Int. J. Life Cycle Assess.* **2016**, *21*, 1218–1230. [\[CrossRef\]](#)
25. Yao, Y.; Chang, Y.; Huang, R.; Zhang, L.; Masanet, E. Environmental implications of the methanol economy in China: Well-to-wheel comparison of energy and environmental emissions for different methanol fuel production pathways. *J. Clean. Prod.* **2018**, *172*, 1381–1390. [\[CrossRef\]](#)
26. Myhre, G.; Shindell, D.; Bréon, F.M.; Collins, W.; Fuglestedt, J.; Huang, J.; Koch, D.; Lamarque, J.F.; Lee, D.; Mendoza, B.; et al. Anthropogenic and natural radiative forcing. In *Climate Change 2013: The Physical Science Basis. Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*; Stocker, T.F., Qin, D., Plattner, G.K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M., Eds.; Cambridge University Press: Cambridge, UK; New York, NY, USA, 2013; pp. 659–740.

27. R Core Team. *R: A Language and Environment for Statistical Computing*; R Foundation for Statistical Computing: Vienna, Austria, 2022; Available online: <https://www.R-project.org/> (accessed on 20 April 2022).
28. Friedlingstein, P.; Jones, M.W.; O'Sullivan, M.; Andrew, R.M.; Bakker, D.C.E.; Hauck, J.; Le Quéré, C.; Peters, G.P.; Peters, W.; Pongratz, J.; et al. Global Carbon Budget 2021. *Earth Syst. Sci. Data* **2022**, *14*, 1917–2005. [\[CrossRef\]](#)
29. Wu, H.; MacDonald, G.K.; Galloway, J.N.; Zhang, L.; Gao, L.; Yang, L.; Yang, J.; Li, X.; Li, H.; Yang, T. The influence of crop and chemical fertilizer combinations on greenhouse gas emissions: A partial life-cycle assessment of fertilizer production and use in China. *Resour. Conserv. Recycl.* **2021**, *168*, 105303. [\[CrossRef\]](#)
30. Knudsen, M.T.; Meyer-Aurich, A.; Olesen, J.E.; Chirinda, N.; Hermansen, J.E. Carbon footprints of crops from organic and conventional arable crop rotations—using a life cycle assessment approach. *J. Clean. Prod.* **2014**, *64*, 609–618. [\[CrossRef\]](#)
31. Wang, W.; Dalal, R.C. Nitrogen management is the key for low-emission wheat production in Australia: A life cycle perspective. *Eur. J. Agron.* **2015**, *66*, 74–82. [\[CrossRef\]](#)
32. Shrestha, B.M.; Desjardins, R.L.; McConkey, B.G.; Worth, D.E.; Dyer, J.A.; Cerkowniak, D.D. Change in carbon footprint of canola production in the Canadian Prairies from 1986 to 2006. *Renew. Energy* **2014**, *63*, 634–641. [\[CrossRef\]](#)
33. Maraseni, T.N.; Cockfield, G.; Maroulis, J. An assessment of greenhouse gas emissions: Implications for the Australian cotton industry. *J. Agric. Sci.* **2010**, *148*, 501–510. [\[CrossRef\]](#)
34. Snyder, C.S.; Bruulsema, T.W.; Jensen, T.L.; Fixen, P.E. Review of greenhouse gas emissions from crop production systems and fertilizer management effects. *Agric. Ecosyst. Environ.* **2009**, *133*, 247–266. [\[CrossRef\]](#)
35. Gan, Y.; Liang, C.; Wang, X.; McConkey, B. Lowering carbon footprint of durum wheat by diversifying cropping systems. *Field Crop. Res.* **2011**, *122*, 199–206. [\[CrossRef\]](#)
36. Grassini, P.; Cassman, K.G. High-yield maize with large net energy yield and small global warming intensity. *Proc. Natl. Acad. Sci. USA* **2012**, *109*, 1074–1079. [\[CrossRef\]](#)
37. Liu, W.; Zhang, G.; Wang, X.; Lu, F.; Ouyang, Z. Carbon footprint of main crop production in China: Magnitude, spatial-temporal pattern and attribution. *Sci. Total Environ.* **2018**, *645*, 1296–1308. [\[CrossRef\]](#)
38. Xu, X.; Lan, Y. Spatial and temporal patterns of carbon footprints of grain crops in China. *J. Clean. Prod.* **2017**, *146*, 218–227. [\[CrossRef\]](#)
39. Ju, X.-T.; Xing, G.-X.; Chen, X.-P.; Zhang, S.-L.; Zhang, L.-J.; Liu, X.-J.; Cui, Z.-L.; Yin, B.; Christie, P.; Zhu, Z.-L.; et al. Reducing environmental risk by improving N management in intensive Chinese agricultural systems. *Proc. Natl. Acad. Sci. USA* **2009**, *106*, 3041–3046. [\[CrossRef\]](#) [\[PubMed\]](#)
40. Zhao, H.; Sun, B.; Lu, F.; Wang, X.; Zhang, G.; Wang, H.; Zhang, L.; Zhang, M. Meta analysis on impacts of biochar on trace greenhouse gases emissions from staple crops in China. *Trans. Chin. Soc. Agric. Eng.* **2017**, *33*, 10–16. [\[CrossRef\]](#)
41. Lu, F.; Wang, X.; Han, B.; Ouyang, Z.; Duan, X.; Zheng, H. Net mitigation potential of straw return to Chinese cropland: Estimation with a full greenhouse gas budget model. *Ecol. Appl.* **2010**, *20*, 634–647. [\[CrossRef\]](#) [\[PubMed\]](#)
42. Lin, J.; Hu, Y.; Cui, S.; Kang, J.; Xu, L. Carbon footprints of food production in China (1979–2009). *J. Clean. Prod.* **2015**, *90*, 97–103. [\[CrossRef\]](#)
43. Asseng, S.; Jamieson, P.D.; Kimball, B.; Pinter, P.; Sayre, K.; Bowden, J.W.; Howden, S.M. Simulated wheat growth affected by rising temperature, increased water deficit and elevated atmospheric CO₂. *Field Crop. Res.* **2004**, *85*, 85–102. [\[CrossRef\]](#)
44. Jiang, Z.; Zhong, Y.; Yang, J.; Wu, Y.; Li, H.; Zheng, L. Effect of nitrogen fertilizer rates on carbon footprint and ecosystem service of carbon sequestration in rice production. *Sci. Total Environ.* **2019**, *670*, 210–217. [\[CrossRef\]](#)
45. Zhang, G.; Wang, X.; Sun, B.; Zhao, H.; Lu, F.; Zhang, L. Status of mineral nitrogen fertilization and net mitigation potential of the state fertilization recommendation in Chinese cropland. *Agric. Syst.* **2016**, *146*, 1–10. [\[CrossRef\]](#)
46. Taki, M.; Soheili-Fard, F.; Rohani, A.; Chen, G.; Yildizhan, H. Life cycle assessment to compare the environmental impacts of different wheat production systems. *J. Clean. Prod.* **2018**, *197*, 195–207. [\[CrossRef\]](#)
47. Zhang, C.; Li, J.; Yu, L.; Li, F.; Ma, N. Input/output analysis on rapeseed production practices under different cultivation mode. *Chinese J. Oil Crop Sci.* **2010**, *32*, 57–64. (In Chinese)
48. Nikkhah, A.; Khojastehpour, M.; Emadi, B.; Taheri-Rad, A.; Khorramdel, S. Environmental impacts of peanut production system using life cycle assessment methodology. *J. Clean. Prod.* **2015**, *92*, 84–90. [\[CrossRef\]](#)
49. Ma, Z.; Gao, X.; Tenuta, M.; Kuang, W.; Gui, D.; Zeng, F. Urea fertigation sources affect nitrous oxide emission from a drip-fertigated cotton field in northwestern China. *Agric. Ecosyst. Environ.* **2018**, *265*, 22–30. [\[CrossRef\]](#)
50. Powell, J.W.; Welsh, J.M.; Eckard, R.J. An irrigated cotton farm emissions case study in NSW, Australia. *Agric. Syst.* **2017**, *158*, 61–67. [\[CrossRef\]](#)
51. Smith, P.; Haberl, H.; Popp, A.; Erb, K.-H.; Lauk, C.; Harper, R.; Tubiello, F.N.; de Siqueira, A.P.; Jafari, M.; Sohi, S.; et al. How much land-based greenhouse gas mitigation can be achieved without compromising food security and environmental goals? *Glob. Chang. Biol.* **2013**, *19*, 2285–2302. [\[CrossRef\]](#) [\[PubMed\]](#)
52. Qin, Z.; Huang, Y.; Zhuang, Q. Soil organic carbon sequestration potential of cropland in China. *Glob. Biogeochem. Cycles* **2013**, *27*, 711–722. [\[CrossRef\]](#)