

Cropland Displacement Drives Carbon Emission of Grain Transport in China

Chengchao Zuo (✉ c.zuo@mail.hzau.edu.cn)

Huazhong Agricultural University <https://orcid.org/0000-0002-1419-7989>

Cheng Wen

School of Geography, University of Leeds

Graham Clarke

School of Geography, University of Leeds

Andy Turner

School of Geography, University of Leeds

Liangzhi You

International Food Policy Research Institute <https://orcid.org/0000-0001-7930-8814>

Xinli Ke

College of Public Administration, Huazhong Agricultural University

Lanping Tang

College of Public Administration, Huazhong Agricultural University

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Title Page

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Main

Rapid urbanisation around the world has driven a greater need for grain transport. This could largely be attributed to two reasons. Firstly, the fast-growing urban population simply requires more grain to be transported from rural to urban areas. Secondly, to feed the growing cities, food production is being increased in areas further from these main areas of consumption as the growing cities swallow up former agricultural regions at the margins of those cities. Thus, in many countries, croplands have been displaced from land close to city-regions to much more remote, marginal areas due to that process of urbanisation (Meyfroidt et al., 2013; van Vliet, 2019). The increase in distance associated with the shifting production areas for food products is a trend expected to continue alongside urbanisation throughout the world (Garnett, 2011), leading to rising energy consumption and carbon emissions of grain transport.

Carbon emissions involved in food production and transportation, which contributes one-third of the total greenhouse gases emissions of human society (Clark et al., 2020, Crippa et al., 2021, Tukker, 2006), present a serious challenge to achieving carbon reduction targets set out in the Kyoto and Paris Agreements (Clark et al., 2020). Transport-related emission accounts for 11%-16% of carbon emissions of food supply chains (Weber and Matthews, 2008; Garnett, 2011; Wakeland et al., 2012; Crippa et al., 2021). Despite the merit of embedding transport-related carbon emission with the Life Cycle Analysis (LCA) of the food supply chain, previous studies usually oversimplified the transport process by applying emission factors to food-miles data from existing databases, without investigating the spatial flows of food between different areas via different transport networks (e.g. road, railway). As a result, these studies have rather limited capacity to adequately measure the impact of land-use changes on transport-related carbon emissions and provide evidence-based policy suggestions on mitigating the emission impact of agricultural and land-use policies.

This research focuses on cropland displacement and the rising carbon emission of grain transport in China during 1990 - 2015. China experienced rapid urbanisation in the last four decades. Its urban population nearly tripled from 310.02 million in 1979 to 875.08 million in 2019 (National Bureau of Statistics of China, NBSC, 2020). Massive cropland (3.31×10^4 km²) was occupied by urban expansion (Xu et al., 2016). Since 2000, China has implemented a series of cropland protection policies to ensure that the loss of cropland by urban development can be replenished with newly cultivated cropland in areas with lower population density. Such policies have generally stabilized the amount of cropland in China; however, they have increased cropland displacement from the core areas of consumption (Yang et al., 2020). Given the speed and intensity of urbanization and cropland displacement in China, carbon emissions associated with grain transport are believed to be growing fast. Although the issues of cropland displacement and transport-related carbon emissions in China have been studied separately (Feng et al., 2013; Yang et al., 2020; Wang et al., 2011; Chen et al., 2020), few studies have focused on the carbon emission associated with grain transport and the emission impact of cropland displacement. Thus the paper explores the nature and magnitude of the increase in carbon emission associated with grain transport in China.

This paper develops a model-based system first introduced by Zuo et al. (2013, 2018) to estimate the carbon emissions associated with the changing transportation of grain in China between 1990 and 2015. First, we developed a spatial interaction model to estimate the spatial flows of grain (which we define as rice, wheat, maize, and soybean) from the changing areas of production to the areas of consumption. These flows are disaggregated by transport modal choice. Second, based on these flow patterns, the total carbon emission produced by the transport system for grain is then estimated, measured by CO₂ emission equivalent (kgCO₂e), again based on transport mode and route. The changing carbon emission associated with grain transport is then revealed by comparing the two urbanisation stages (1990 and 2015). Once built, the model allows us to examine a series of what-if scenarios designed to look at the emission impact of cropland displacement, population growth, dietary change, and transport infrastructure development.

Results

Extended Distances between Grain Production and Consumption

Between 1990 and 2015, the population of China grew from 1.13 billion to 1.37 billion, while the cropland area grew from 177.15 million hm^2 to 178.51 million hm^2 (Cheng et al., 2018). Although neither population nor cropland area has increased much, their spatial distribution varied considerably. We adopted the Local Indicators for Spatial Association (LISA; Anselin, 1995) analysis to explore the spatial patterns of the changing distribution of population and cropland during the period (Fig 1a and b). Fig 1a indicates significant population growth in the southeast coastal areas and significant population decreases in the Northeast and central regions. Fig 2b indicates significant cropland expansion occurred in Northeast and Northwest China, and significant cropland shrinkages occurred in Central and Eastern China during the same period. The spatial shift of population and cropland extended the distance between grain production and consumption. We measured the distance between the mean centres (Methods Equation [2]) of the cropland and the population at both national and prefecture levels. At the national level, the mean centre of cropland moved 62.53km north, while the mean centre of population moved 17.91km south. As a result, the distance between the national mean centres of cropland and of population extended from 260.41km to 320.66km between 1990 and 2015 (Extended Data Fig. 1). At the prefectural level, the average distance between the mean centres of population and cropland increased from 10.89km to 12.59km between 1990-2015; and 251 out of 347 (approx. 72%) prefectures, which covers 80% of the population in China experienced a greater separation between population and cropland (Fig.1 c and d).

We estimated the grain consumption of each prefecture in 1990, 2000, 2005, 2010 and 2015 based on three types of grain consumption: staples, animal-feed, and industrial & other uses (e.g. grain-based snacks and alcoholic drinks; Extended Data Figure 2; Methods Equation [1]). Results indicate a transformed grain consumption pattern during this period due to the changing dietary structure of Chinese people. First, grain consumption as staples dropped nearly 30% from 266.52 million-ton in 1990 to 187.20 million-ton in 2015 despite the population growth. Second, grain consumption for feeding animals more than tripled from 71.27 million-ton to 237.72 million-ton, which outranked staples as the largest type of grain consumption in 2015. Lastly, industrial & other uses of grain also increased from 130.21 million-ton to 223.13 million-ton.

Overall, grain production in China grew from 468.01 million-ton, to 484.02 million ton, to 648.06 million-ton from 1990, 2005, to 2015. Spatial distributions of grain production at the prefecture-level show that, as a result of cropland displacement towards "marginal land" (Kuang et al., 2021), the increase in grain production occurred primarily in the north, Northeast and Northwest of China (Extended Data Fig. 3 b & c). In comparison, an increase in grain consumption mainly occurred in the east and southeast of China (Extended Data Fig. 3 e & f). Compared to 1990, the whole country was more reliant on grain supplies originated in northern China by 2015 (Fig. 1 e & f). The distance between national mean centres of grain production and consumption extended from 178.54km to 328.93km.

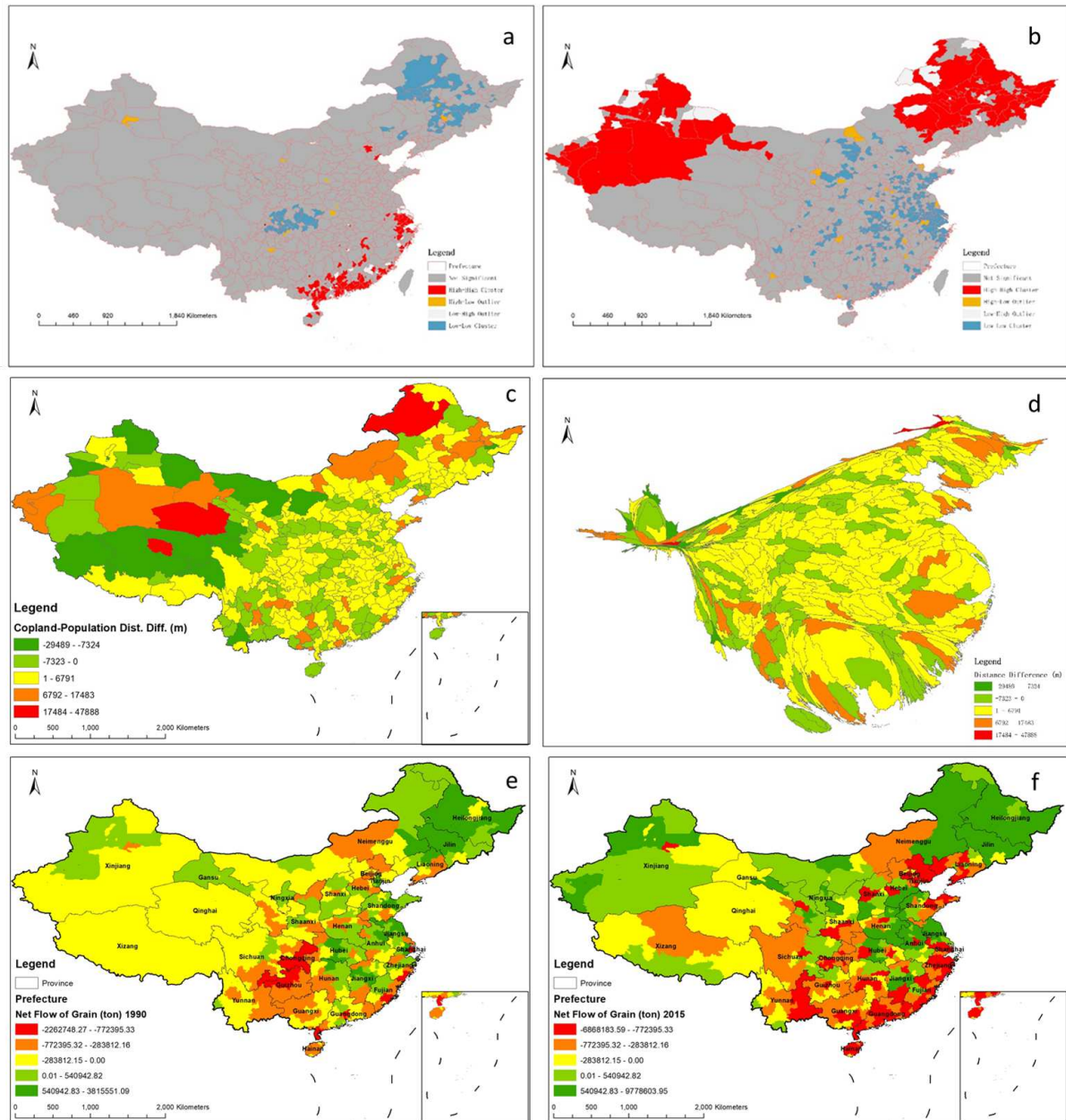


Fig. 1 Change of Grain Production and Consumption: a) The local indicators for spatial association (LISA; Anselin, 1995) map of population growth by county 1995-2015, b) LISA map of the change of cropland area by county 1990-2015, c) Change of distance between the mean centre of population and cropland at the prefectural level; d) Cartogram of change of distance between the mean centre of population and cropland (rescaled by the population of each prefecture); e) net-flow (production-consumption) of grain in 1990, green tones indicate net-exporters, while yellow to red tones indicate net-importers; and f) net-flow of grain in 2015.

Spatial Flows of Grain Transport

Harnessing the doubly constrained spatial interaction model (Methods Equation [3]-[8]) using three types of transport network data (road, railway, and waterway), we estimated the spatial flow of grain transport between each pair of prefectures of China in 1990, 2000, 2005, 2010 and 2015 (Extended Data Fig.6). Aggregated results at the provincial level show that the scale of inter-provincial grain flows grew from 64.33 million-ton to 168.03 million-ton, and the general pattern of grain transport turned more

north-to-south than east-to-west (Fig.3). In 1990, the northeastern provinces were the major grain suppliers for the North and Northwest of China, while the eastern provinces of China were the major suppliers for central and southern China. In 2015, Neimenggu (Inner-Mongolia) outranked Liaoning as the third-largest grain supplier in northern China after Heilongjiang and Jilin. Hunan and Hubei Provinces' role as major grain suppliers in central China declined, while Henan became the biggest supplier in the Central-South region. Provinces in the East of China (i.e. Zhejiang and Fujian) turned from net grain exporters to net importers. In Guangdong and Guangxi, two of the most southerly provinces in the Chinese mainland, the market share of northern-origin grain increased from less than 20% to approximately 50%. These changes echo the fact that grain production in China has been moving northward and the transport distances have been extending during this 25 year period.

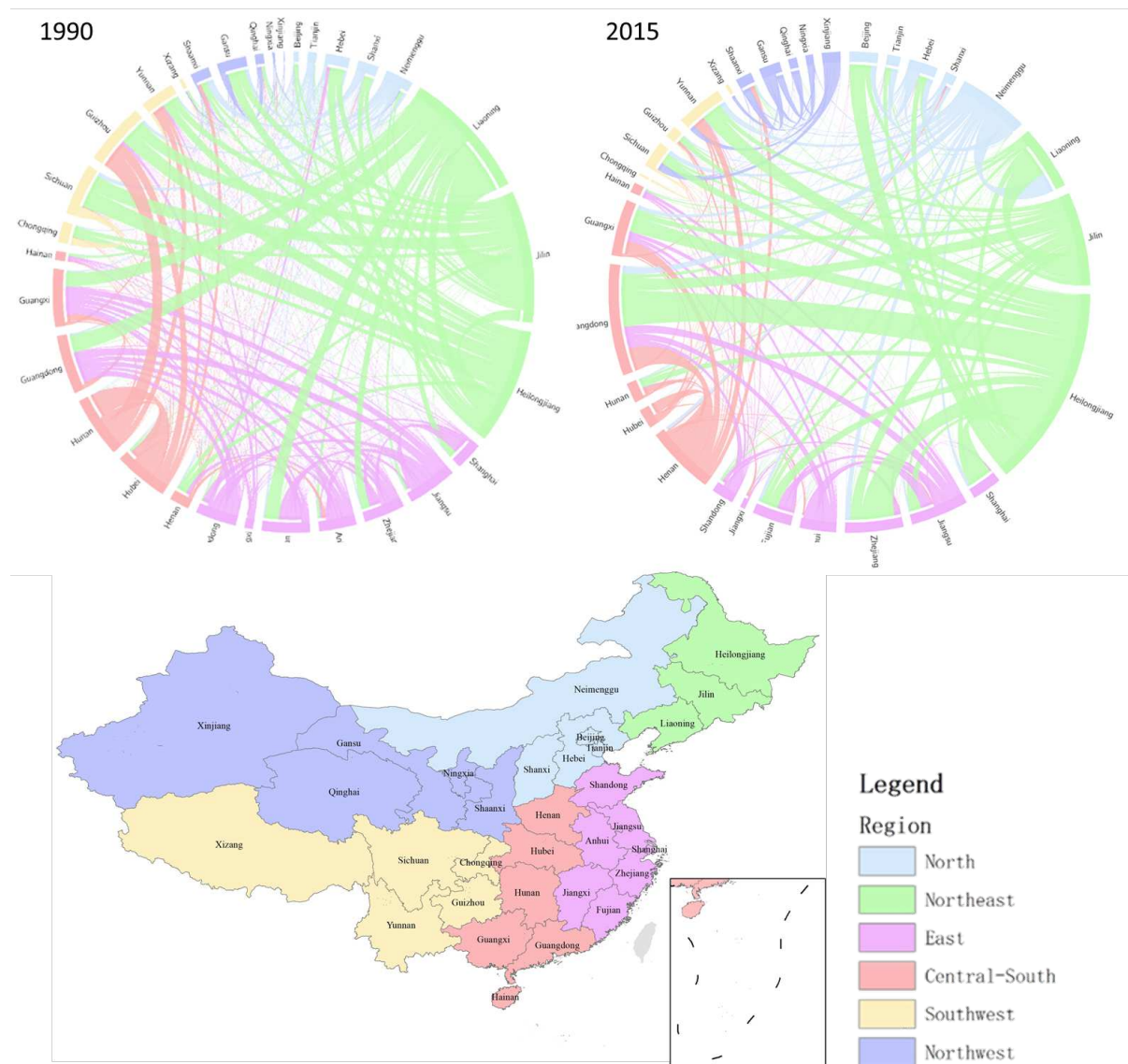


Fig.2 Inter-provincial Flow of Grain Transport in China (1990 and 2015): the arcs on the outer ring and the ribbons between the arcs. Each arc on the outer ring indicates a province, coloured by the region. The length of the arc represents the grain flux (inflow + outflow) of the province. The ribbons between the two arcs represent the grain flows between the export provinces and the import provinces, and the colour of each ribbon matches the colour export province. The width of the ribbon indicates the volume of the flow of grain.

A breakdown of the inter-provincial flow of grain by the three transport modes (Fig 4) shows that the railway has been the most critical transport mode for supplying grain to Southwest China and that grain from the Northeast dominated the market share of grain transported by railway. Due to the railway network development in Northwest China since the early 2000s, grain exported from Xinjiang Province has been one of the most significant changes in grain transport by railway between 1990 and 2015. The waterway mode of travel was mostly responsible for transporting grain from Northeast and East China to Central-South China; Guangdong Province was the largest destination for the waterway-transported grain (accounting for 30% in 1990 and 45% in 2015). Road transport was mainly for short-distance transport within all regions. Inter-regional grain transport by road was rare in 1990; however, it became more common in 2015 due to the increasing distances between grain production and consumption.

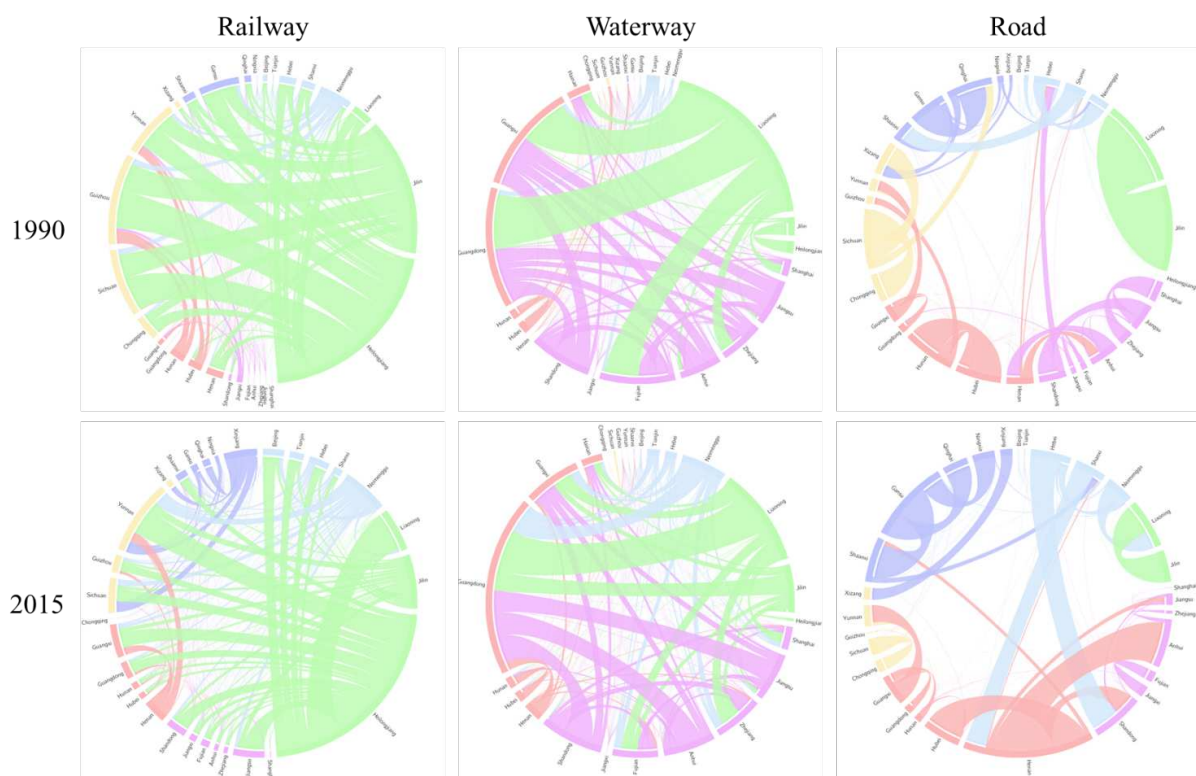


Fig. 3 Inter-provincial flow of grain by transport mode in 1990 and 2015

Carbon Emissions Associated with Grain Transport and Their Drivers

Based on the modelled spatial flows of grain transport and the carbon emission conversion factors of the different transport modes, we estimate that carbon emissions associated with grain transportation in China more than doubled from 4.46 million-ton in 1990 to 10.73 million-ton in 2015. In terms of different transport modes, the largest contributor has been grain transport by railway, which increased 2.5 times from 1.77 million-ton to 4.50 million-ton. The carbon emissions associated with grain transport by road nearly doubled from 2.16 million-ton to 4.18 million-ton. Grain transport by waterway contributed the least proportion of carbon emissions, which, however, almost tripled from 0.70 million-ton to 2.04 million-ton (Fig 4a). From the consumption type perspective, the carbon emission of staple grain increased by 29% (from 2.55 million-ton to 3.30 million-ton) during 1990 and 2015, which is much lower than feed grain (510.88%) and grain for industry and other uses (325.38%). These changes echo the shift of dietary change during this period.

We further break down grain increased transport-related carbon emissions into the provincial level. For most provinces or municipalities, grain transport-related carbon emissions increased from 1990 to 2015 (Fig 4b). Four provinces (or municipalities), including Henan, Gansu, Chongqing and Guizhou, reduced carbon emissions of grain transport (green coloured areas in Fig 4c) because increased grain production in northern-western China significantly reduced the transport distance of their grain supply.

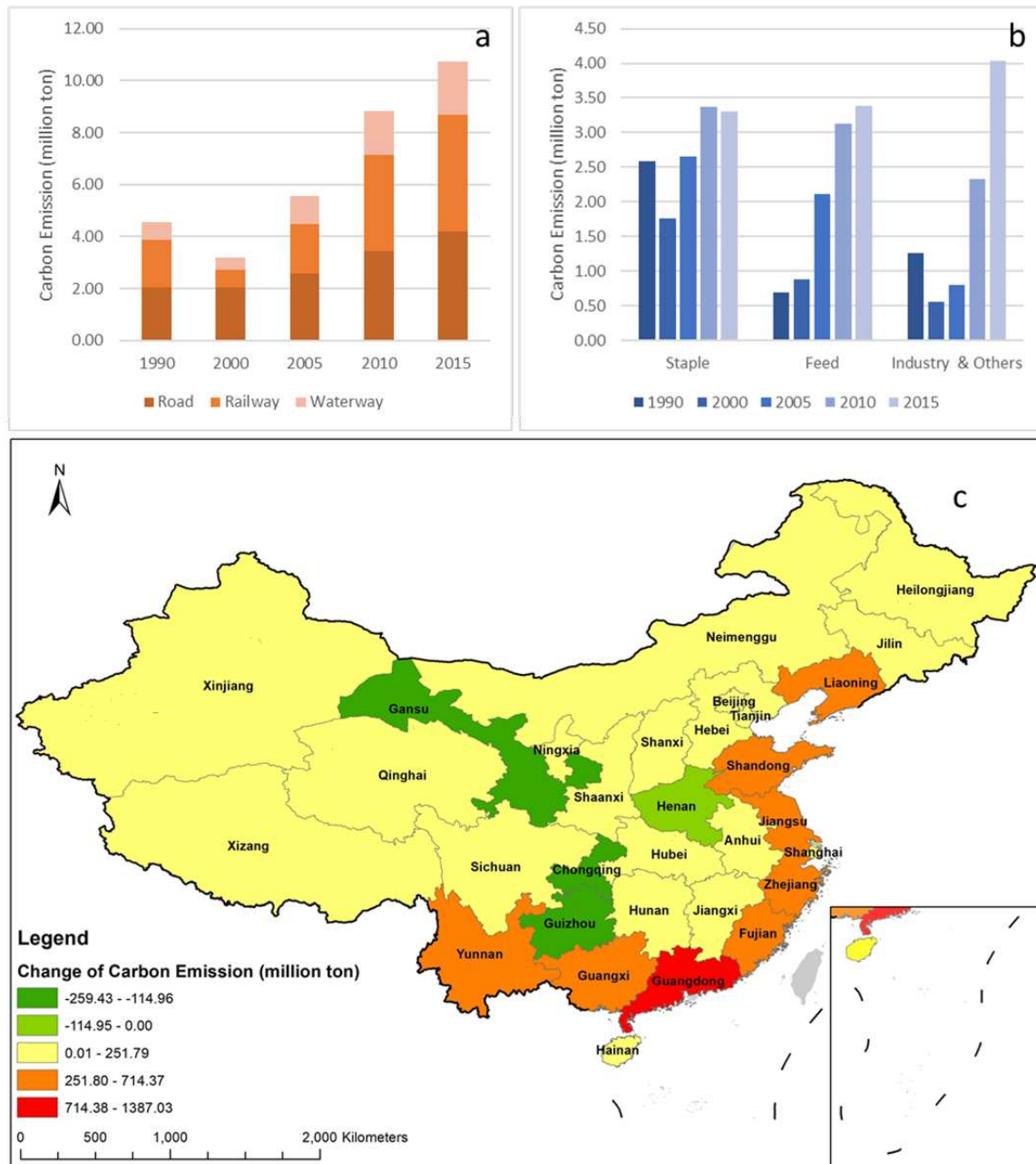


Fig 4.a) Grain Transport-related Carbon Emissions by Transport Mode 1990-2015; b) Grain Transport-related Carbon Emissions 1990-2015 by Use of Grain; c) Change of Grain Transport-related Carbon Emissions 1990-2015 by Province, China.

We applied a scenario analysis to identify the impacts of multiple factors on grain transport carbon emissions (see Fig 5a). We found that cropland displacement contributed 3.79 million-ton, accounting for 60.4% of the increased carbon emissions in China between 1990 and 2015. The change of dietary structure and population growth contributed 1.99 million-ton (31.7%) and 1.04 million-ton (16.6%),

respectively. However, the development of transport infrastructures, such as newly built highways and railways in western China, helped reduce 0.54 million-ton of carbon emissions associated with grain transport, equivalent to half of the increment related to population growth (see Fig 5a).

At the provincial level, there were 18 (out of 31) provinces or municipals where the cropland displacement contributed more than 50% of the increase in grain transport-related carbon emission (beige coloured areas in Fig 5b). Cropland displacement also drives the decline of grain transport-related carbon emissions in all four provinces where carbon emissions dropped. The dietary change primarily drove the increase in carbon emissions in 6 provinces (pink coloured areas in Fig 5b), where dietary change contributed more than half of the rise in grain transport-related carbon emissions. The increase in meat production in these provinces has led to a considerable growth in demand for grain, which greatly increased the carbon emissions associated with grain transport accordingly.

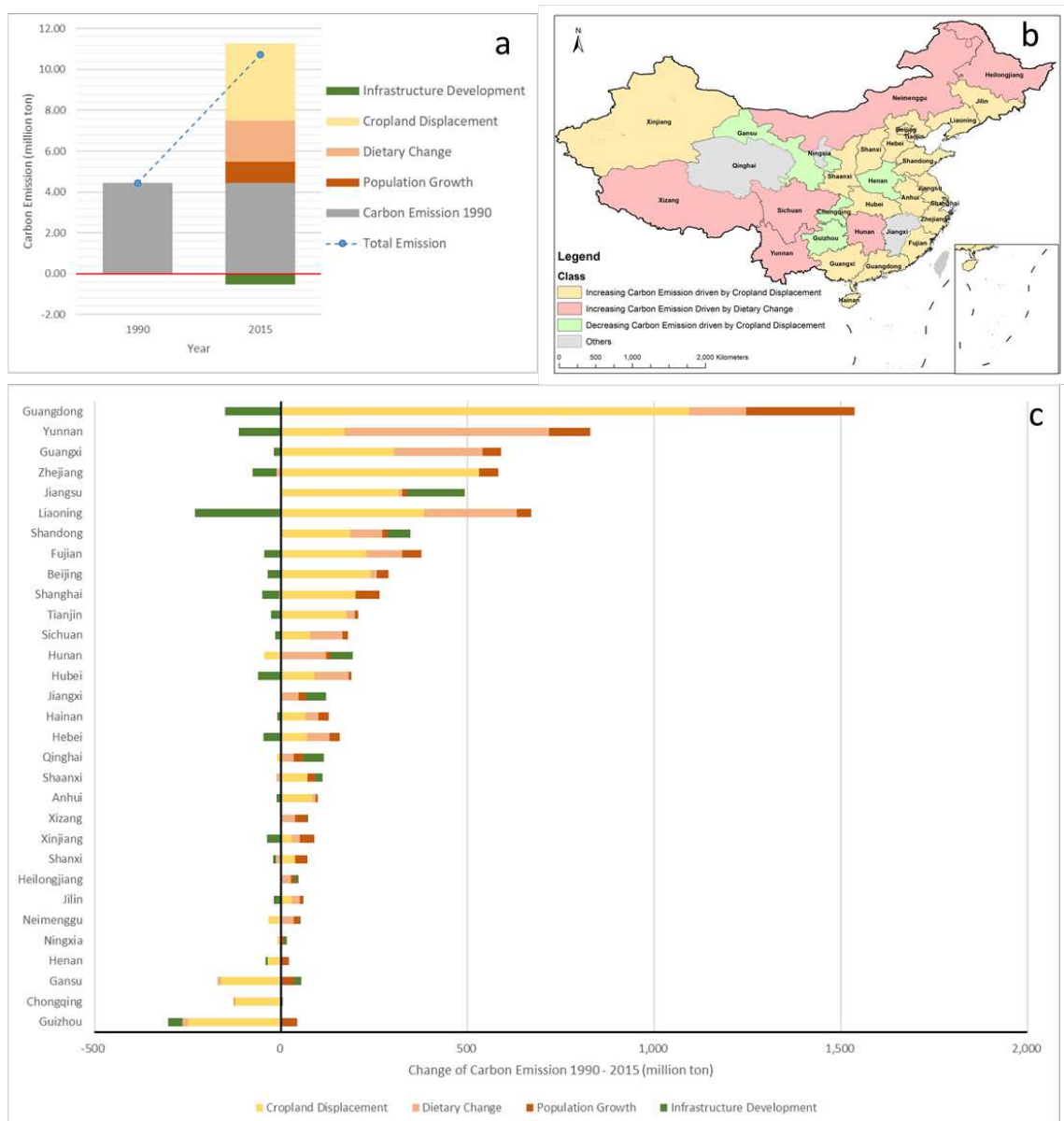


Fig.5 a) Drivers of increased carbon emission of grain transport at national level; b) primary drivers of the changes in grain transport-related carbon emissions; c) factors drove the changes in grain transport-related carbon emission by province

In order to explore the implications of the grain-transport-related carbon emissions on the land-use policies in China, we also estimated the carbon intensity (kg CO₂ emission equivalent per ton of grain) of grain transport at both national and prefecture levels. At the national level, grain for industrial & other uses had the highest transport-related carbon intensity, which grew from 9.64 kgCO₂e/ton to 18.11 kgCO₂e/ton between 1990 and 2015. The transport-related carbon intensity of grain for staples and animal-feed also grew from 9.52 kgCO₂e/ton and 9.31 kgCO₂e/ton to 17.63 kgCO₂e/ton and 14.26 kgCO₂e/ton respectively. We further disaggregated the grain transport-related carbon intensity to the prefecture-level on both the production and consumption sides (Methods equation [15] and [16]). The model results suggest that grain consumed in prefectures in the south and east coastal areas had higher transport-related carbon intensities than in other parts of China, except Xizang (Tibet) and Qinghai Provinces in the Qinghai-Tibet Plateau which has a harsh environment for grain production and relatively poor transport infrastructure (Fig. 6a). For instance, the average transport-related carbon intensity of grain consumed within the south-coastal Guangdong Province was almost six times higher than that in Hubei Province in Central China. On the production side, the transport-related carbon intensity of grain produced in prefectures in northern China is estimated to be generally higher than in southern China (Fig. 6b). For instance, the average transport-related carbon footprint of grain produced in the northeastern Province of Heilongjiang (42.87kgCO₂e/ton) was 20 times more than that in the southern province of Guangdong (2.05kgCO₂e/ton). Extended Data Table.1 shows the average transport-related carbon intensity of grain production and consumption in each province.

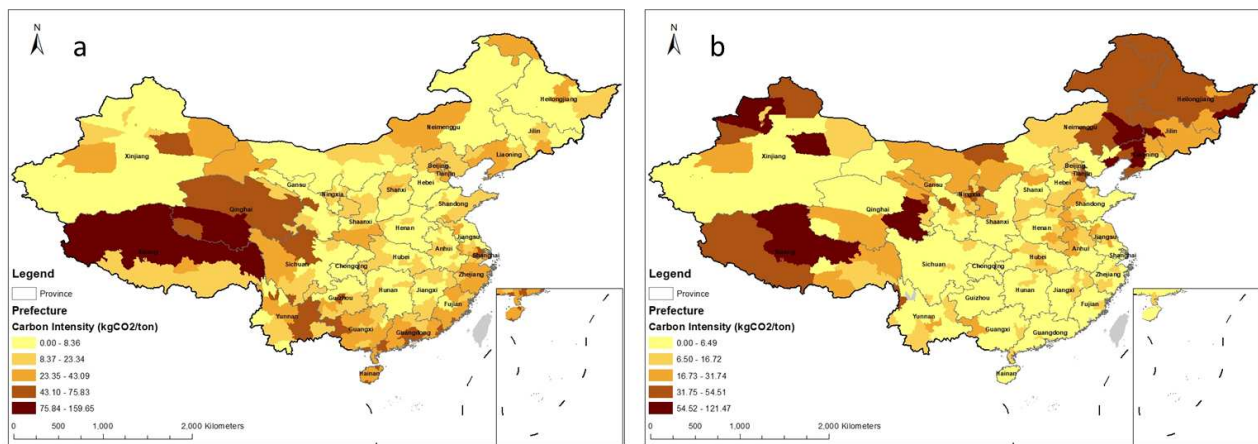


Fig.6 Transport-related Carbon Intensity (kgCO₂e/ton) at the Prefecture Level (2015) from: a) grain consumption perspective, and b) grain production perspective

Discussion and Conclusion

China is facing dual challenges of reducing carbon emissions and feeding the largest population in the world. On the one hand, the country has announced its ambitious national strategy to achieve peak carbon emissions by 2030 and to be carbon neutral by 2060 (Mallapaty, 2020). A series of carbon emission targets have been set up in its 14th five-year plan (2021-2025) (Hou, 2020). The agriculture department plays a key role in achieving carbon emission targets. However, the extended distance of grain transport and the associated rising carbon emissions have not received sufficient attention. In this research, we found that 72% of prefectures in China experienced a greater separation between its population centres and the cropland centres that feed them. This distance between the national mean centres of grain production and consumption extended by 150.39km during 1990 - 2015. We estimate that the carbon emissions of grain transport consequently more than doubled during this same period.

On the other hand, securing the food supply to its growing population has been a long-standing challenge for China. To maintain self-sufficiency (as high as 95% in grain supply) China has endeavoured to keep its total amount of cropland despite the great pressure of rapid and massive

urbanisation. One of the consequences has been cropland displacement (van Vliet et al., 2017; Ke et al., 2017), which, as we estimate, contributed more than 60% of the increase in carbon emissions of grain transport between 1990 and 2015. In 2018, a revised cropland protection policy was introduced in China to allow cross-provincial cropland displacement (before that, displacement must be fulfilled within the same province). This new policy is expected to intensify cropland displacement across the country (Yang et al., 2020; Ke et al., 2020) and further increase carbon emissions from transporting grain from production to consumption areas. Here we argue for more synergic considerations of the emission impact of increasing the food-miles in agricultural policies. Based on this research, we provide the following policy suggestions on reducing the carbon emissions of grain transport in China.

First, we found that the east and south coast of China with the highest population density have almost the highest transport-related carbon intensity for grain consumption, whereas areas with the highest grain output have the highest transport-related carbon intensity for grain production (supply). In comparison, central China (e.g. Hubei Province) has relatively low transport-related carbon intensity on both supply and consumption side; however, both the proportions of grain output and population of central China to the whole country were declining during 1990 - 2015. We therefore suggest that encouraging and facilitating the development of central China as a major grain production and economic centre with a higher population density could help reduce the total grain transport distance and associated carbon emissions.

Second, our model results show that the change in dietary structure contributed 31.7% of the increase in grain transport carbon emissions in China between 1990 and 2015. Although the environmental impacts of dietary change in China is not a new topic (He et al., 2018; Xiong et al., 2020), we provide new insights from the different perspectives of its impact on grain consumption and transport. Our results show that the fast-growing grain consumption for feeding animals and industrial & other uses has been accompanied with rising transport-related carbon intensity, indicating that more meat and grain-based snacks are consumed in modern China. Given that meat, snacks, and alcoholic drinks are usually with much higher added value than staple grains, we suggest that building a more localised supply chain for those high added value commodities at the downstream value chain of grain production could both benefit local economies and help reduce carbon emissions related to grain transport. This echoes the 'local food' movement in some western countries (Smith et al., 2005; Engelhaupt, 2008).

Lastly, we found that the improvement of transport infrastructure, especially the development of the railway network in western China, had offset part of the increased grain transport carbon emissions driven by cropland displacement, change of dietary structure, and population growth. Since the carbon emission per ton-km of railway and waterway transport is much lower than that of road transport, we suggest that further increasing railway & waterway transport capacity is important for reducing the pressure of rising emissions from grain transport. Moreover, as the carbon-intensive road transport contributed nearly 40% of the grain transport emissions in 2015, and its market share of long-distance grain transportation has been increasing between 1990 and 2015, we also argue that technological evolution in clean energy and electric high gross vehicles must play an important role in reducing the grain transport-related emissions in the future.

Cropland displacement is not a unique phenomenon in China but a common issue in many countries across the world (Meyfroidt et al. 2013; van Vliet 2019). The negative environmental impacts of cropland displacement need to be taken better account of in land use and agriculture policy-making practices. This paper demonstrates a systematic evaluation framework to estimate the carbon emissions of grain transport and identifies the emission impact of cropland displacement as well as other factors. Although a few simplified assumptions were adopted in the modelling process, our estimation results of the increased food miles (ton-km) and related carbon emissions between 1990 and 2015 were validated through robust model calibration. Furthermore, a Monte Carlo simulation-based sensitivity analysis was conducted in this research, and the results also suggest that the modelled carbon emission is robust despite the considerable uncertainty introduced with the input parameters (reported in Supplementary Information). In a wider context, our research findings contribute to a better understanding of the life cycle environmental impact within the food supply chain. The methodology

proposed in this study could be applicable to a wide range of other commodities that involve trans-regional production and consumption, both in China and other countries.

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Methods

Modelling Framework

Modelling grain transport, in terms of flow and transport mode, based on the distribution of supplies and demands (land-use) is essential for understanding the impacts of urbanisation and cropland displacement on grain transport. Since the 1950s, many efforts have been made to model commodity or population flows between places, including multi-regional input-output models (Leonitif, 1963), linear programming (Reed et al., 2000; Dalin et al., 2014), spatial price equilibrium models (Samuelson, 1952; Ricci, 1999), and spatial interaction models (Wilson, 1974; Clarke and Birkin, 2018). In transport geography, spatial interaction models (SIMs) are the most commonly used approach to modelling the flows of freight or people between locations based on the distribution of supply and demand. Zuo *et al.* (2013, 2018) demonstrated a modelling framework, which adopted a doubly-constrained SIM to simulate the inter-regional flows of aggregates in the UK. With this modelling framework, they were able to evaluate the impacts of aggregate supply policies on carbon emissions.

Extended Data Fig.4 illustrates the modelling framework we applied to estimate the carbon emissions associated with grain transport in China. We first established data sets concerning grain production and consumption at the prefecture level in China. Then, we identify the shortest route between each pair of prefectures in China by three different transport modes: road, railway and waterway (including both inland waterway and coastal transport by sea). Since the number and the boundary of prefectures in China vary over time, to ensure consistency and to make the results comparable, we chose the 2010 prefecture boundary data as the basic unit. The road and railway transport network data were extracted from OpenStreetMap for 1990, 2000, 2005, 2010 and 2015. Table 1 summarises the key statistics about the data we used in this research. Next, we estimated the transport costs between each pair of prefectures based on the distance and the corresponding transport mode. As a full interaction matrix of real flows was not available as published data, a doubly constrained spatial interaction model was adopted to generate the spatial flows of grain between prefectures according to the transport cost, grain production levels at the origin prefecture and grain consumption levels at the destination prefecture. So, the ton-km of grain between each pair of prefectures in China was estimated. Then, in turn, we estimated the carbon emissions of grain transport for each prefecture origin-destination possibility based on the ton-km of grain, the corresponding transport modes and carbon emission conversion factors. Finally, we applied what-if scenario analysis to identify the impacts of multiple factors (e.g. cropland displacement, development of transport infrastructure) on carbon emissions of grain transport at the prefecture level. Details of each step are explained in more detail below.

Grain Consumption and Output by Prefecture in 1990 and 2015

The grain consumption of each prefecture was estimated based on three types of grain: staples, animal-feed, and industrial & other uses. The staple grain consumption was estimated based on the number of urban and rural residents in each prefecture. The animal-feed grain consumption was estimated by applying statistics on the production of meat to the corresponding grain-to-meat ratios. Due to the lack of relevant official statistics data, the grain for industrial and other uses was estimated by applying a fixed ratio factor to the population. Thus, the consumption of grain for each prefecture was calculated as equation [1].

$$D_j = {}^I GR * Pop_j + {}^U GR * {}_U Pop_j * U_j + {}^R GR * {}_R Pop_j + {}^P GR * {}_P Mt_j + {}^O GR * {}_O Mt_j \quad [1]$$

where Pop_j , ${}_U Pop_j$, and ${}_R Pop_j$, are the population, urban population, rural population in prefecture j ; ${}^U GR$ and ${}^R GR$ are the ratio grain consumption per capita for urban residents and rural residents, respectively; ${}^I GR$ represents the grain for industry (and other) uses per capita; ${}_P Mt_j$ and ${}_O Mt_j$ represent the meat outputs in prefecture j ; ${}^P GR$ and ${}^O GR$ represent the grain-to-meat ratios for pork and

other meat, respectively. The urban population rate and meat output of each prefecture in 2015 were obtained from the yearbook of each province. The corresponding numbers for 1990 were extracted from the 1990 census and 1990 agricultural survey.

The grain output of each prefecture in 1990 was aggregated from China County-Level Data on Population and Agriculture, Keyed 1:1M GIS Map (CITAS and CIESIN, 1997). The grain output of each prefecture in 2015 was extracted from the yearbook of each province.

We adopted the mean centre analysis to describe the centrality of grain production and population distribution, expressed as equation [2]:

$$\bar{s} = (\mu_x, \mu_y) = \left(\frac{\sum_{i=1}^n x_i W_i}{\sum_{i=1}^n W_i}, \frac{\sum_{i=1}^n y_i W_i}{\sum_{i=1}^n W_i} \right) \quad [2]$$

where \bar{s} represents the mean centre of grain production and population, the coordinates are denoted as μ_x and μ_y ; x_i and y_i are the coordinates of the county i , and weighting factor W_i represents the grain production or population at the county i .

Modelling the spatial flows of grain between prefectures in China

We adopted a doubly constrained spatial interaction model (SIM) to estimate the volume of grain transported between prefectures. This allocates the flows when both the grain output and consumption of each prefecture are known or estimated; hence the use of a doubly constrained SIM (Wilson, 1974). This can be expressed as the equations below:

$$\hat{m}_{ij} = A_i O_i B_j D_j f(d_{ij}), \quad (i, j = 1, 2, 3 \dots r) \quad [3]$$

$$\sum_j \hat{m}_{ij} = O_i, \quad (j = 1, 2, 3 \dots r) \quad [4]$$

$$\sum_i \hat{m}_{ij} = D_j, \quad (i = 1, 2, 3 \dots r) \quad [5]$$

$$A_i = \frac{1}{\sum_j B_j D_j f(d_{ij})}, \quad (i = 1, 2, 3 \dots r) \quad [6]$$

$$B_j = \frac{1}{\sum_i A_i O_i f(d_{ij})}, \quad (j = 1, 2, 3 \dots r) \quad [7]$$

$$f(d_{ij}) = \exp(-\beta \cdot d_{ij}) \quad [8]$$

where \hat{m}_{ij} represents the volume of grain transported from prefecture i to prefecture j ; O_i represents the output of grain at prefecture i , and D_j represents the consumption of grain at prefecture j . d_{ij} is the integrated transport cost of grain moved from prefecture i to prefecture j , which can be generated by the transport model (explained in the following section). Both the equations [4] and [5] show how the SIM is doubly constrained at both the supply and demand sides. A_i and B_j are balancing factors to ensure the equation [4] and [5] hold. Since A_i and B_j are dependent on each other, Equation [6] and [7] are solved iteratively.

Modelling the Transport Cost and Modal Choice

We considered three different transport modes for grain transport, i.e. road, railway and waterway.

For road transport, grain was assumed to be carried by High Gross Vehicles (HGVs) directly from the origins to the destinations. For grain transported by railway and waterway, the cost was measured by the distance equivalent between the origin and destination. We assumed that the grain was first transported by road from each origin to the closest railway station or wharf for loading. Then the loaded grain was carried by trains or ships to the railway station or wharf closest to the destination and unloaded for the final leg of the journey to the destination via the road network (illustrated in Extended Data Fig. 5). Relative cost converters were applied to convert the distance by railway or waterway to the road distance equivalent.

Thus d_{ij} in equation [8] can be rewritten as in Equation [9], where superscripts RD, RL and WT indicate road, railway, and waterway, respectively. The 'shortest path' analysis was applied for road transport to estimate the transport cost between each pair of origin and destination. For railway and waterway transport, the 'closest facility' analysis was first applied to identify the closest railway station or wharf of each origin and destination; then the 'shortest path' analysis was applied to calculate the transport cost between the railway stations or between the wharves (equation [10] and [11]). In practice, the ArcGIS software was implemented to conduct these analyses.

$$d_{ij} = \min (d_{ij}^{RD}, d_{ij}^{RL}, d_{ij}^{WT}) \quad [9]$$

$$d_{ij}^{RL} = d_{im}^{RD} + TC^{RL} + RC^{RL} * d_{mn}^{RL} + d_{nj}^{RD} \quad [10]$$

$$d_{ij}^{WT} = d_{ip}^{RD} + TC^{WT} + RC^{WT} * d_{pq}^{WT} + d_{nj}^{RD} \quad [11]$$

Note that RC^{RL} and RC^{WT} are the two relative cost converters for railway and waterway, respectively. Considering the transport cost per ton-km for railway and waterway are lower than for road transport, these two relative cost converters need to be smaller than 1. TC^{RL} and TC^{WT} are the two transshipment costs for railway and waterway, respectively, which were introduced to quantify the cost of transferring grain between origins/destinations and railway stations/wharfs. These two values were calibrated against the observed average distance and market-share of railway or waterway transport in practice. The calibration process is described in the Supplementary Information.

Estimating Carbon Emissions

The carbon emission associated with the transport of grain was estimated based on the ton-km of grain transported by each transport mode multiplied by the corresponding carbon emission conversion factor (equation [12] and [13]).

$$C_{ij}^T = \hat{m}_{ij} * d_{ij}^{RD} * CF^{RD} + \hat{m}_{ij} * d_{ij}^{RL} * CF^{RL} + \hat{m}_{ij} * d_{ij}^{WT} * CF^{WT} \quad [12]$$

$$C^T = \sum_i^n \sum_j^n C_{ij}^T \quad [13]$$

where C_{ij}^T represents the carbon emission associated with the transport of grain from prefecture i to j ; and C^T represents the total carbon emission related to grain transport, CF represents the carbon emission conversion factors, the superscripts RD, RL and WT denote the transport modes of road, railway, and waterway. Due to the lack of officially published carbon emission conversion factors in China, the CF values were extracted from GHG Conversion Factors 2015, published by the Department of Environment and Food Affairs UK (Defra, 2016), and widely used in assessing CO₂ and other greenhouse gas emissions by different industry sectors. Considering that the auto emission standards adopted in China (National Standards IV in 2015) are equivalent to the UK standards (Euro IV in 2015) since 2000, it is reasonable to use the UK GHG Conversion Factors to approximate the corresponding

factors in China. Table 2 shows the conversion factors for each transport mode. Considering that considerable uncertainty might be introduced with the input parameters, we adopted a Monte Carlo simulation-based sensitivity analysis to investigate the robustness of the modelling results. The process of the sensitivity analysis is reported in the Supplementary Information.

Scenario Analysis

The difference in carbon emissions associated with grain transport in China between 1990 and 2015 could be attributed to three factors, i.e. cropland displacement, change of dietary structure and development of transport infrastructure. In order to identify the impacts of these three factors on carbon emissions, we constructed a baseline scenario and two alternative scenarios.

- Baseline Scenario: all the three factors, i.e. cropland displacement, dietary changes, and transport infrastructure development, were modelled based on the observed data of 1990 and 2015;
- Alternative Scenario 1 ("unchanged transport network" scenario): the transport infrastructure remained as in 1990; while the other two factors change as Baseline Scenario;
- Alternative Scenario 2 ("unchanged dietary structure" scenario): the dietary structure for urban and rural residents in China remains as in 1990, while the other two factors change as the Baseline Scenario.

Assuming the carbon intensity of grain remains as in 1990, the transport-related carbon emission of grain should increase 1.04 million tons by 2015, driven by population growth, which is much lower than 6.27 million tons as we estimated. To identify the different drivers of those increased carbon emissions, we further modelled the grain transport-related carbon emission under two alternative scenarios. According to the modelling results, the carbon emission associated with grain transportation was 11.27 million tons under Alternative Scenario 1 and 8.74 million tons under Alternative Scenario 2 (Figure 15).

The difference between the baseline scenario and the "unchanged transport network" scenario revealed the impact of transport infrastructure development on grain transport carbon emission, while the difference between the baseline scenario and the "unchanged dietary structure" scenario revealed the emission contribution of the dietary structure change. Then the rest part of the increment of grain transport carbon emission between 1990 and 2015 was attributed to cropland displacement.

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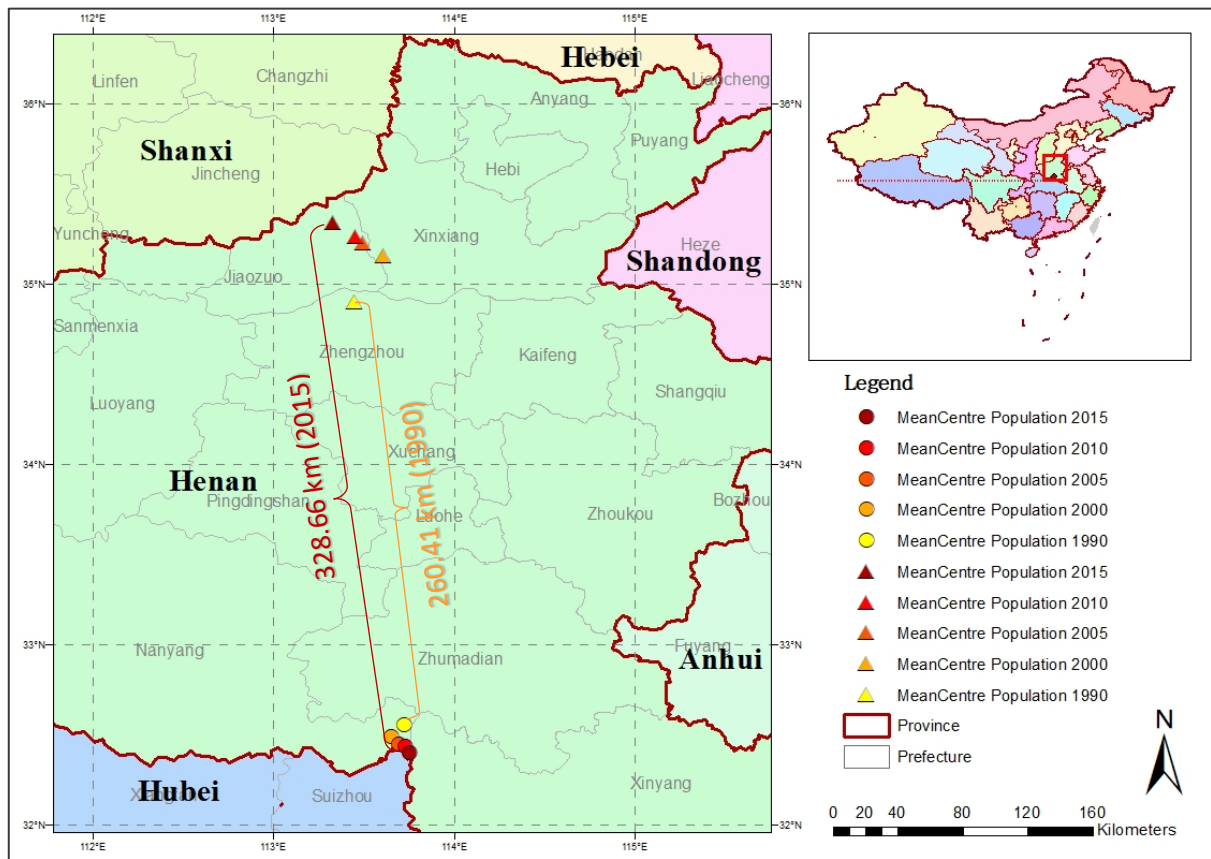
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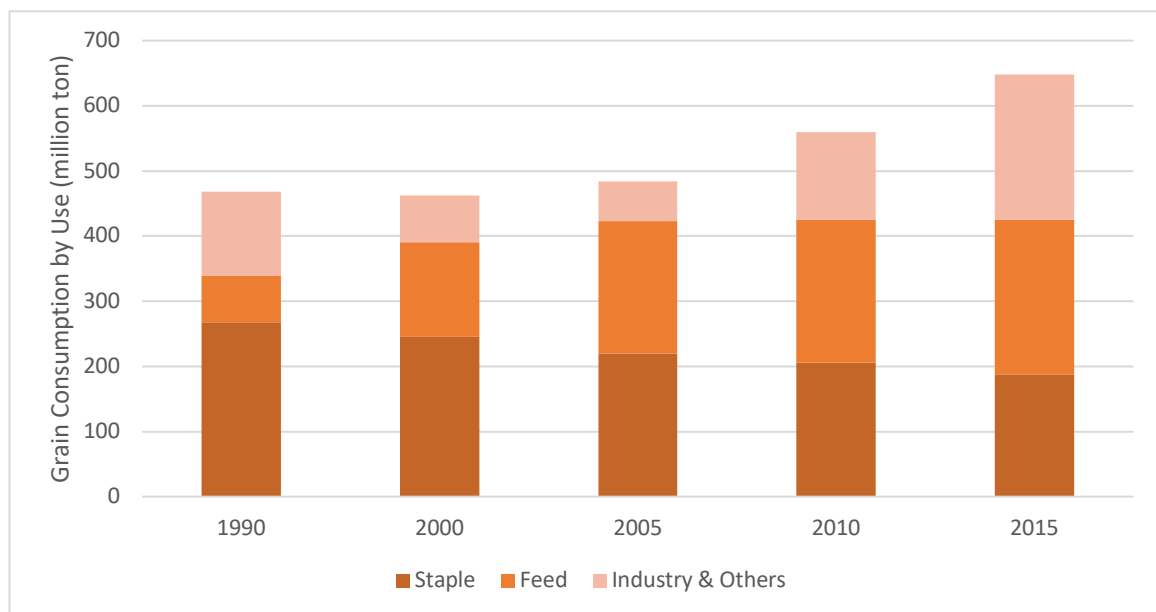
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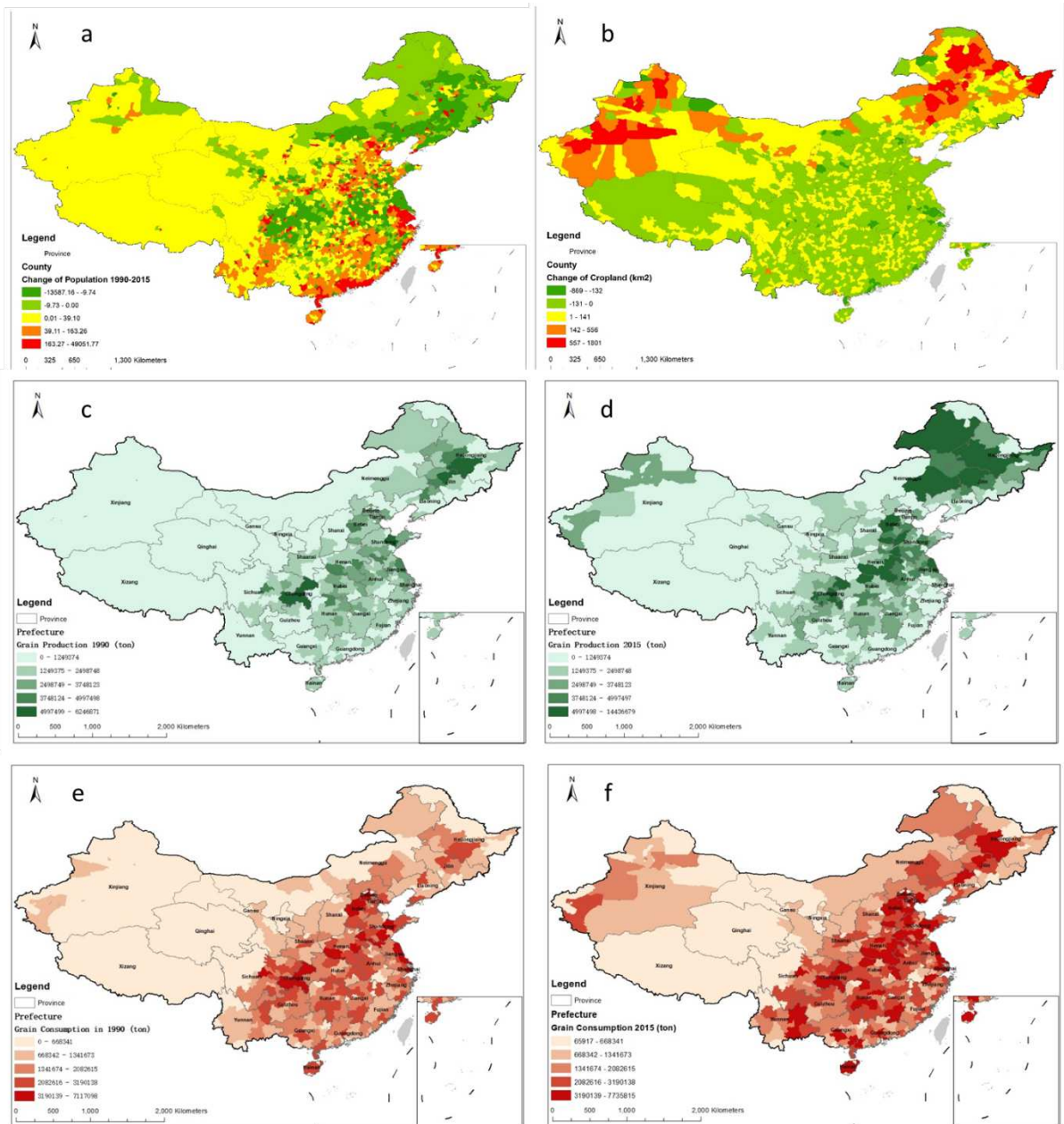
Extended Data



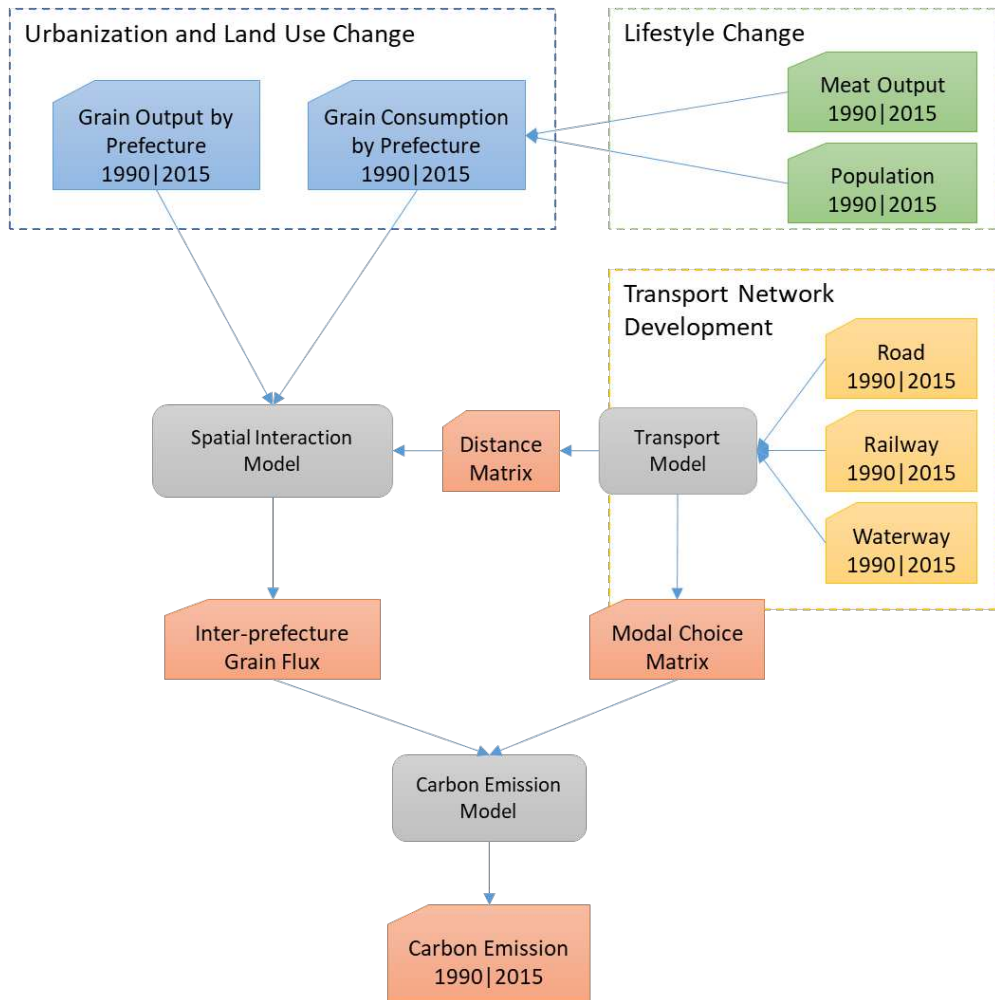
Extended Data Fig.1 Changes of Mean Centres of Cropland and Population at National Level, Chinese mainland, 1990-2015



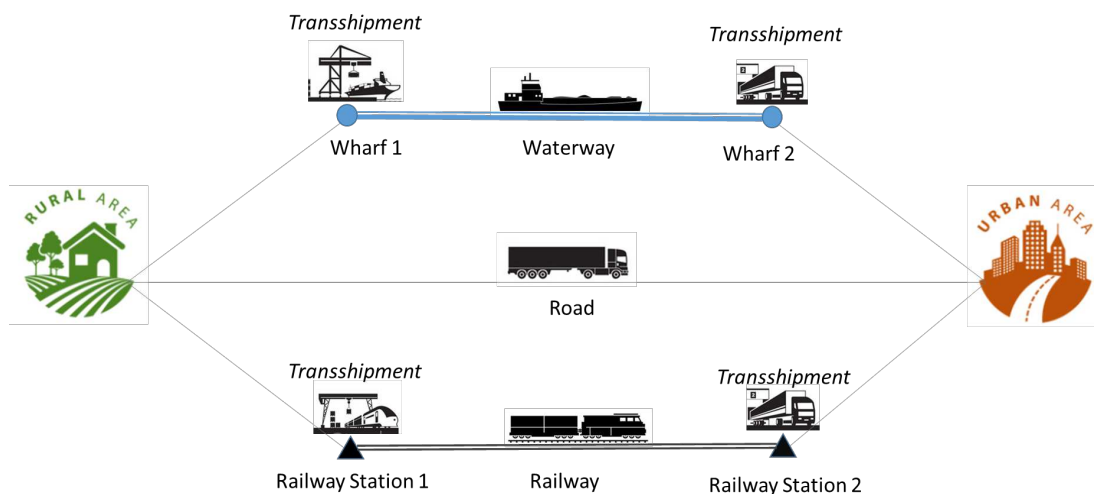
Extended Data Fig.2 Grain consumption by use 1990-2015



Extended Data Fig.3 a) Change of population by county 1990-2015; b) Change of cropland area by county 1990-2015; Grain output of each prefecture in c) 1990 and d) in 2015; Grain consumption of each prefecture in c) 1990 and d) in 2015;

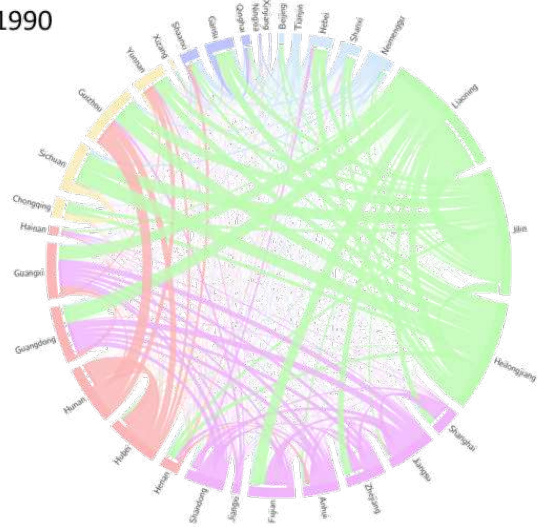


Extended Data Fig.4 Modelling Framework for Estimating Carbon Emission of Grain Transport

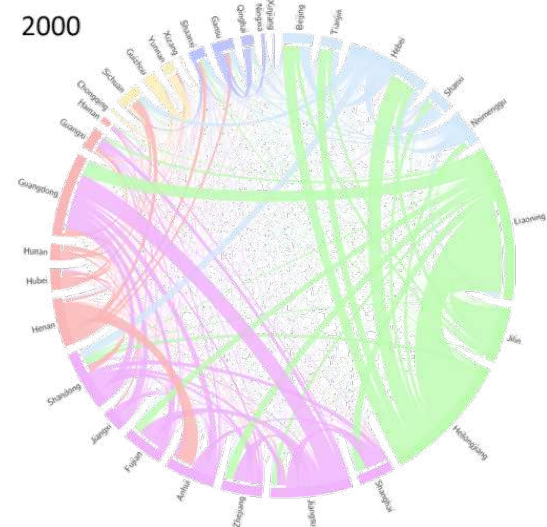


Extended Data Fig.5 Illustration of Transport Cost and Modal Choice Model

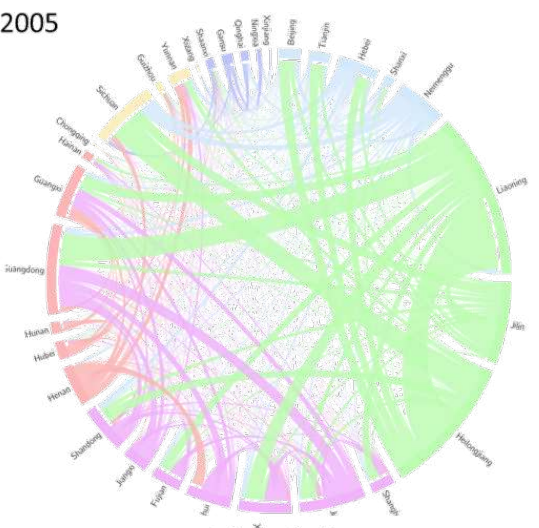
1990



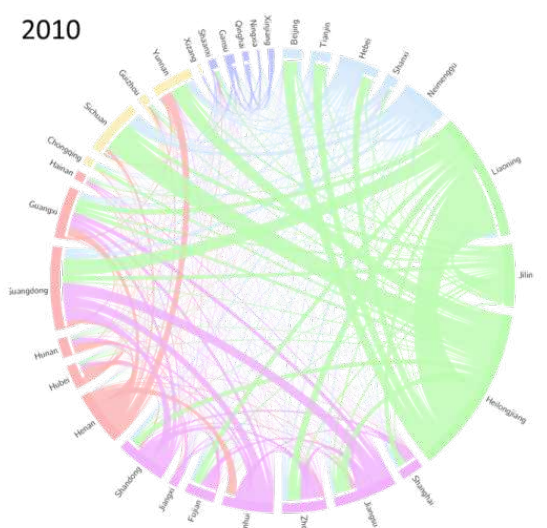
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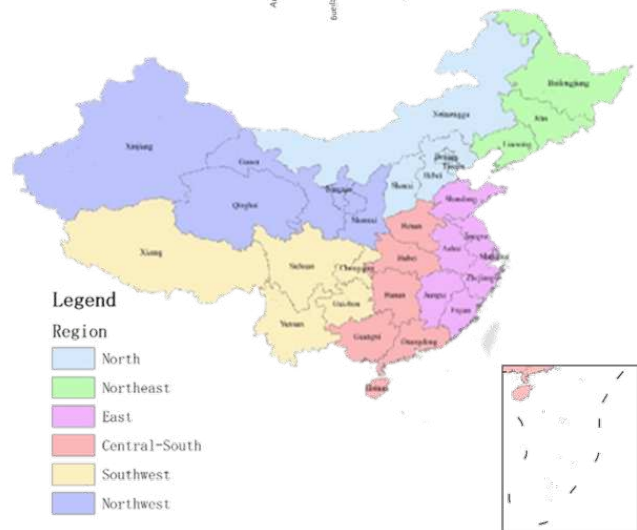
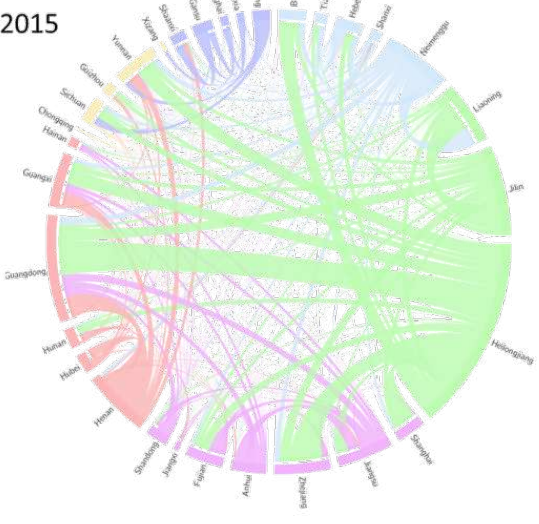
2005



2010



2015



Extended Data Fig.6 Inter-Provincial transport of grain in China, 1990-2015

Extended Data Table 1 Provincial average carbon footprints of grain transport for consumption and supply

| Province | Carbon Intensity of Grain Transport for Consumption (kgCO ₂ e/ton) | Carbon Intensity of Grain Transport for Supply (kgCO ₂ e/ton) |
|----------------------------|---|--|
| Beijing | 38.45 | 13.75 |
| Tianjin | 39.98 | 43.84 |
| Hebei | 6.27 | 7.03 |
| Shanxi | 10.28 | 9.31 |
| Neimenggu (Inner Mongolia) | 6.69 | 41.98 |
| Liaoning | 22.79 | 41.13 |
| Jilin | 6.21 | 37.57 |
| Heilongjiang | 5.21 | 42.87 |
| Shanghai | 34.40 | 4.57 |
| Jiangsu | 16.25 | 11.09 |
| Zhejiang | 30.24 | 2.37 |
| Anhui | 5.81 | 13.07 |
| Fujian | 35.59 | 3.33 |
| Jiangxi | 8.65 | 5.43 |
| Shandong | 8.87 | 11.16 |
| Henan | 2.54 | 11.32 |
| Hubei | 8.24 | 7.07 |
| Hunan | 5.92 | 1.47 |
| Guangdong | 45.39 | 2.06 |
| Guangxi | 31.89 | 3.44 |
| Hainan | 44.77 | 4.14 |
| Chongqing | 1.64 | 0.92 |
| Sichuan | 14.02 | 3.32 |
| Guizhou | 15.78 | 1.18 |
| Yunnan | 34.26 | 5.48 |
| Xizang (Tibet) | 77.18 | 24.80 |
| Shaanxi | 14.44 | 3.94 |
| Gansu | 11.05 | 13.22 |
| Qinghai | 57.60 | 31.04 |
| Ningxia | 6.90 | 19.98 |
| Xinjiang | 8.51 | 31.67 |

Extended Data Table 2 GHG Conversion Factors by Transport Mode

| Transport Mode | Carbon Emission Conversion Factor (kgCO ₂ e/ton-km) |
|----------------|--|
| Road (HGV) | 0.11364 |
| Railway | 0.02601 |
| Waterway | 0.01315 |

Supplementary Files

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