Drivers of GHG emissions from dietary transition patterns in China: Supply vs. demand options

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Abstract: Diets have been changing drastically in China recent decades and this change has contributed considerably to greenhouse gas (GHG) emissions. In determining effective mitigation strategies for future emissions, it is necessary to know how emissions related to diet vary over time in overall magnitude and due to compositional changes driven by socioeconomic dynamics. This study evaluates the change in dietary GHG emissions in China during the 1997-2011 period by linking environmentally extended input-output tables with individual daily food intake data, and further decomposes the contribution to GHG emission changes of socioeconomic driving forces. The results show that GHG emissions related to national diet have been decreasing from 1180 Mt CO_2e to 640 Mt CO_2e (a 54% decline), largely due to technical innovation that has reduced the emissions per calorie of food (135% of the total reduction). The change in dietary patterns has had mixed effects, with a decline in calorie intake reducing emissions by 21% while increases in animal-sourced food consumption have raised emissions by 25%. Our findings stress the importance of technical progress in the historical change in dietary GHG emissions and suggest a focus on behavior change for future research and policymaking, which has the potential to promote dietary changes towards less animal product consumption. Our findings highlight the importance of both technological and demand-side behavioral options in reducing the impact of diets on GHG emissions.

Key words: diet, GHG emissions, driving force, socioeconomic transformation

1. Introduction

Human diets contribute significantly to greenhouse gas(GHG) emissions in China. To feed a population of more than 1.3 billion, the country produced substantial GHG emissions that amounted to 1,600 Mt CO_2-eq emissions in 2010 (H. Li, Wu, Wang, & Qi, 2015). GHG emissions associated with diets are predicted to reach 2,500 Mt CO_2-eq in 2050 (World Bank, 2010), equivalent to the total emissions of India in 2013 (GHG Platform India, 2017). Such a trend requires urgent policy intervention if China wants to address GHG emissions from the national food system to make sure they align with its GHG emission reduction target commitments for the Paris Agreement (Guan, Liu, Geng, Lindner, & Hubacek, 2012; Xu & Lan, 2016). In order to develop effective strategies, it is thus imperative for decision-makers to identify the key underlying driving forces of GHG emissions change over time.

The effect of socioeconomic transformation on diets and their related GHG emissions has been explored in multiple studies. Similar to other developing countries, China has been experiencing a nutritional transition characterized by reduced intake of carbohydrates and increased consumption of animal-sourced foods (Barry M Popkin, Linda S Adair, & Shu Wen Ng, 2012; Zhai et al., 2009). The consumption of meat has increased by more than 30% during the past two decades, while the consumption of cereals has decreased by 30% over the same period (J. Liu & Savenije, 2008; F. Wang, Cai, & Zhang, 2020). On the one hand, this transition contributes to increased emissions from food consumption (Tilman & Clark, 2014). On the other hand, calorie intake by individuals is declining largely because people are requiring less energy due to less physically active types of work (Du, Lu, Zhai, & Popkin, 2002; Barry M Popkin et al., 2012), which may help reduce the emissions per capita. The calorie intake for a typical Chinese individual has been decreasing from 2490 kcal*day^{-1} in 1982 to 2170 kcal*day^{-1} in 2012 (National Health and Family Planning Commission of China, 2013). This transition is further facilitated by rapid urbanization which makes restaurants and animal-sourced foods more readily available and lessens the physical activity required for an increasingly sedentary lifestyle (Monda, Gordon-Larsen, Stevens, & Popkin, 2007; Ng, Norton, & Popkin, 2009). Previous research has shown higher GHG emissions per capita for food consumption in urban households than in their rural counterparts (L.-C. Liu, Wu, Wang, & Wei, 2011; Z. Wang & Yang, 2014). Finally, population growth with aging trends, and technical innovations in agricultural production could also play important roles in the changing GHG emissions resulting from food consumption.

Recent research has rarely explored how dietary GHG emissions change over time nor the contribution of key socioeconomic drivers. Most recent studies have focused on cross-sectional quantification that links habitual food consumption with emission factors or conducted comparisons associated with counterfactual dietary scenarios (Green et al., 2015; Heller & Keoleian, 2015; Reynolds, Piantadosi, Buckley, Weinstein, & Boland, 2015; Song, Li, Fullana-i-Palmer, Williamson, & Wang, 2017; Vetter et al., 2017; Westhoek et al., 2014). Through statistical models at the individual or household level, a few works have identified socioeconomic factors that may affect diet-related GHG emissions, including personal income (Song, Li, Semakula, & Zhang, 2015; F. Wang et al., 2020), educational background (Song et al., 2015), etc. While these contribution to the literature identify the demographic characteristics that have an effect on individual dietary patterns and consequential environmental impacts, it is still unclear how much each driving force contributes, and what the key factors affecting diet-related GHG emissions are. Understanding these relationships benefits decision-makers by enabling policy design focusing on the most cost-effective measures in approaching future emission reductions and the sustainability of the food system.

This study investigates the temporal changes in GHG emissions resulting from diets in China and quantifies the contribution of each factor that drives the change. We linked individual-level nutritional survey data collected during 1997-2011 from the China Health and Nutrition Survey (CHNS) with high-resolution emissions factors from environmentally extended input-output (EEIO) tables to quantify the per-capita dietary GHG emissions by age groups in both urban and rural areas, and extrapolated the emissions for the whole country by employing detailed national statistics on population structure. Next, we conducted a logarithmic mean Divisia index (LMDI) decomposition to detect the contribution of multiple driving forces, including technical innovation, dietary structural transition, change in calorie intake, urbanization, aging, and population growth, and the changes in such contributions across the years. Based on these results, we discuss possible policy interventions from both the production and consumption sides, and their potential cost and benefits. Given the ongoing socioeconomic transformation, human nutritional requirements, and environmental change faced by the developing world, our findings not only address the challenges faced by China from the impacts on the environment associated with the dietary transition, but does also provide a more general framework applicable to study other countries on a similar track of socioeconomic transformation.

2. Methodology and data

2.1 Quantifying individual food intakes

The individual dietary intakes come from the China Health and Nutrition Survey (CHNS). This survey regularly collects the daily food intake of individuals along with their socioeconomic characteristics. CHNS samples from 9 Chinese provinces[1], including the more developed east coastal areas such as Shandong and Jiangsu, northeast provinces such as Heilongjiang and Liaoning, central provinces such as Henan, Hubei, and Hunan, and the less-developed southwest areas such as Guizhou and Guangxi. Although that the survey does not adopt a nationally representative sampling framework, such a heterogeneous sample represents different geographical, socioeconomic and cultural contexts, with the same communities and villages traced across the years. As shown in Figure S1 in the supporting information, these 9 provinces display diverse patterns of food consumption, urbanization rate, and per capita disposable income among all the provincial districts of China.

In the survey, each individual is requested to record her/his food intake over three continuous days based on 24-hour recalls. Cooking oil and condiment intake are estimated by measuring the weight difference in these items between the beginning and the end of the survey period for each family, a task that is assigned to each family member based on the method developed by (Du, Mroz, Zhai, & Popkin, 2004). All the food items in the CHNS are recorded with a food code that matches it with its nutrition facts in the Chinese Food Composition Tables (CFCTs), which enables us to identify the

[1] The scope of the investigation has been expanded since 2011 with more provincial administrative districts sampled. However, we still use the sample from 9 provinces for the study period as it provides the largest coverage over time.

types of foods eaten and calculate their total caloric values. We aggregate all the items in the CFCTs into 13 food groups: cereals, oils and fats (including vegetable oils and animal fats), livestock, poultry, vegetables, tubers, eggs, nuts, seafood, legumes, fruits, dairy, and others. The data collection started in 1989, and 9 waves of the survey have been conducted so far. In total, observations from approximately 7,200 households amounting to over 30,000 individuals have been recorded. Our analysis concentrates on the later waves including the years 1997, 2000, 2004, 2006, 2009, and 2011 since information on the nutrition content of the food intake is not available for earlier years.

2.2 Evaluating dietary GHG emissions

We use input-output (IO) analysis to calculate the consumption-based GHG emissions of the diets. Our aim here is to include both the direct and indirect environmental impacts of each consumed good or service per monetary unit, which can be captured by an environmentally extended input-output analysis. We start with a standard IO model of the interdependent sectors of the Chinese economy:

$$x = (I - A)^{-1} y \quad (1)$$

where x is a vector of the total sectoral economic output, A denotes the direct input-output coefficients matrix, I is the an identity matrix with same dimensions as A, and y denotes the vector of final sectoral consumption. In this way, the GHG emissions of the total output in each sector (E_{total}) can be calculated as:

$$E_{total} = f\,(I - A)^{-1} y \quad (2)$$

where f is a vector of the direct GHG emissions from the products of each sector. Accordingly, the elements of $f(I-A)^{-1}$ indicate the total GHG emissions from one monetary unit of the final product from each sector.

The input-output table used here was retrieved from the EXIOBASE database (Stadler et al., 2018). The dataset has been adopted in several studies concerning the environmental impact of food consumption (Behrens et al., 2017) and food waste (Usubiaga, Butnar, & Schepelmann, 2018). It includes global multi-regional IO tables for 1995-2011, which covers the study period of this research and thus can be matched with each wave of our longitudinal food intake data to capture the production-side changes in emissions over time. The EXIOBASE IO tables are specified in terms of 200 products, of which approximately 20 are food-related, which allows for the differentiation of the the emissions from each food category. The dataset also includes an environmental account of the main GHG emissions covering CO_2, CH_4, and N_2O. We convert all the accounts to CO_2 equivalents using 100-year global warming potential for greenhouse gases reported by the United Nations Framework Convention on Climate Change (Intergovernmental Panel on Climate Change, 2007). Emissions from land use and land cover change are excluded. Since the food products consumed in China are produced in various countries where the GHG emissions per monetary unit of product differ, we calculate the average coefficient on each product weighted by the proportion imported from each country. Although IO tables for China are also available from other data sources such as the Chinese Environmentally Extended Input-Output Database (Liang et al., 2017), they either have a lower resolution of sectors, cover fewer years, or lack environmental satellite accounts for non-CO_2 GHGs. Therefore, to cover more accurately with more detailed data the important diet transition period, we adopt the EXIOBASE dataset for our analyses.

We link the CHNS data with the emission factors from the EXIOBASE tables by food groups. Using the EXIOBASE table for the corresponding years, each food item from the CHNS dataset is associated with an economic sector as the final product from that sector(s) (The concordance table is included in the supplementary data). Since the food intake data from CHNS are measured in quantities while the emission factors from EXIOBASE are quantified in euros, we need to convert the two into the same units using food prices. We thus calculate the producer food prices of 2011 with data on production quantity and value from FAOSTAT (FAOSTAT, 2020b) to match with the basic price IO table. The prices of previous years, because they are not directly available in FAOSTAT for some food products, are extrapolated using the price indices from the same data source. To validate this extrapolation, we compare the extrapolated data with the prices of some major agricultural products at the market fairs obtained from the China Yearbook of Agricultural Price Survey. As shown in Figure S2, the prices from the two sources are strongly linearly correlated with a coefficient of 0.97, despite the fact that most producer prices from FAOSTAT are smaller than the prices in the yearbook which contains additional costs from transportation, storage, etc. Finally, the GHG emissions from the daily food intake of each individual are calculated by multiplying the factors from EXIOBASE, the extrapolated price, and the quantity of intakes from CHNS. For the food items consumed away from home, CHNS also records the type and amount of intake. We thus calculate the emissions from these food items following the same method so that the same food item results in the same emissions whether it is consumed at home or in a restaurant.

We proportionally extrapolate per capita emissions to the national level with population statistics. As a microlevel dataset, CHNS investigates 9 provinces. However, the survey does take into consideration socioeconomic heterogeneity when selecting areas for investigation and stratifies counties by income level in the sampling procedure. In this way, we assume that the CHNS reflects the dietary patterns of individuals from different age cohorts living in both urban and rural areas for the whole country and the changes in those patterns throughout the years. We calculated the per capita food intake for 9 age groups (0-10, 10-19, …, 70-79, 80+) in 2 areas of residence (urban, rural). With the proportion of the population in each cohort taken from the *China Statistical Yearbook* (National Bureau of Statistics of the People's Republic of China, 2012), we are able to calculate the total dietary GHG emissions for the whole country.

The dietary GHG emissions in China are calculated as follows:

$$E_t = \sum_k \sum_j \sum_i em_{it} \cdot price_{it} \cdot c_{ijkt} \cdot P_{jkt} = \sum_k \sum_j \sum_i e_{it} \cdot c_{ijkt} \cdot P_{jkt} \quad (3)$$

where E_t denotes the total dietary GHG emissions in China in period t. e_{it} denote the emissions generated by producing one gram of food in food group i, which can be calculated using em_i, the total emissions from producing one monetary unit of food in food group i that comes from the elements $f(I-A)^{-1}$ for food-producing sectors in equation (2), and that food group's price, $price_{it}$. c_{ijkt} denotes the per capita daily intake of food group i in grams for individuals from age group k living in area

j, where j denotes the division of urban or rural area. P_{jkt} denotes the number of people in the cohort of age group k living in area j in period t.

2.3 Exploring the driving forces

We explore how the socioeconomic transition has restructured dietary GHG emissions in China with an additive LMDI method. Developed by Ang (Beng W Ang, Zhang, & Choi, 1998), this method is able to quantify the contribution of each driving force without residuals (B. W. Ang, 2015). In this study, we focus on how technical progress, dietary transition, and population change affect dietary GHG emissions over time. Rearranging the equation in the previous section gives

$$
\begin{aligned}
E_t &= \sum_k \sum_j \sum_i e_{it} \cdot c_{ijkt} \cdot P_{jkt} \\
&= \sum_l \sum_k \sum_j \sum_i \cdot \frac{e_{it}}{cal_{it}} \cdot \frac{c_{ijkt} \cdot cal_{it} \cdot ntr_{ilt}}{NTR_{jklt}} \cdot \frac{NTR_{jklt}}{N_{jkt}} \cdot N_{jkt} \cdot \frac{P_{jkt}}{P_{jt}} \cdot \frac{P_{jt}}{P} \cdot P \qquad (4) \\
&= \sum_l \sum_k \sum_j \sum_i ec_{it} \cdot NC_{lijkt} \cdot D_{ljkt} \cdot N_{jkt} \cdot G_{jkt} \cdot U_{jt} \cdot P
\end{aligned}
$$

where cal_i denotes calories per gram for food group i in period t. ntr_{ilt} denotes percentage of calories from nutrient l for food group i, where l indicates a macronutrient carbohydrate, fat, or protein. NTR_{jklt} is the average total calorie intake from macronutrient l for individuals from age group k living in area j. N_{jkt} is the per capita daily total intake of calories for the same individuals in the age*living area cohort. P_{jt} and P denote the number of people living in area j (either urban or rural) and the national population, respectively. Other terms are the same as in equation (3). Therefore, $ec_{it}=e_{it} / cal_{it}$ represents the technical progress in terms of per-calorie total GHG emissions; $NC_{lijkt}=c_{ijkt} \cdot cal_{it} \cdot ntr_{ilt} / NTR_{jklt}$ is the proportion of each macronutrient in each food groups, namely the percentage of carbohydrate/fat/protein provided by each food group. In this sense, $\sum_i ntr_{ilt} = 1$. $D_{ijkt} = NTR_{jklt} / N_{jkt}$ indicates the structure of energy intake in terms of the percentages of carbohydrates, fats, and proteins in the calorie supply; $G_{jkt} = P_{jkt} / P_{jt}$ captures the change in the age structure; $U_{jt} = P_{jt} / P_t$ gives the ratio of the urban and rural populations to the whole population and thus shows the level of urbanization. We are interested in how the factors in the final form of equation (3) contribute to the change in total dietary GHG emissions, both in directions and magnitudes.

The change in E_t between two periods can be expressed as:

$$E_t - E_0 = \Delta E_{tech} + \Delta E_{ntrcom} + \Delta E_{enstr} + \Delta E_{intake} + \Delta E_{aging} + \Delta E_{urban} + \Delta E_{pop} \quad (5)$$

In equation (4), E^T and E^0 refer to the emissions during periods t and 0, respectively. The contribution of each driving force can be calculated with

$$\Delta E_{tech} = \sum_k \sum_j \sum_i \frac{E_{ijkt} - E_{ijk0}}{\ln E_{ijkt} - \ln E_{ijk0}} \ln \frac{ec_{it}}{ec_{i0}} \quad (6)$$

$$\Delta E_{ntrcom} = \sum_k \sum_j \sum_i \frac{E_{ijkt} - E_{ijk0}}{\ln E_{ijkt} - \ln E_{ijk0}} \ln \frac{NC_{ijklt}}{NC_{ijkl0}} \quad (7)$$

$$\Delta E_{enstr} = \sum_k \sum_j \sum_i \frac{E_{ijkt} - E_{ijk0}}{\ln E_{ijkt} - \ln E_{ijk0}} \ln \frac{D_{ijkt}}{D_{ijk0}} \quad (8)$$

$$\Delta E_{intake} = \sum_k \sum_j \sum_i \frac{E_{ijkt} - E_{ijk0}}{\ln E_{ijkt} - \ln E_{ijk0}} \ln \frac{N_{jkt}}{N_{jk0}} \quad (9)$$

$$\Delta E_{aging} = \sum_k \sum_j \sum_i \frac{E_{ijkt} - E_{ijk0}}{\ln E_{ijkt} - \ln E_{ijk0}} \ln \frac{G_{jkt}}{G_{jk0}} \quad (10)$$

$$\Delta E_{urban} = \sum_k \sum_j \sum_i \frac{E_{ijkt} - E_{ijk0}}{\ln E_{ijkt} - \ln E_{ijk0}} \ln \frac{U_{jt}}{U_{jt}} \quad (11)$$

$$\Delta E_{pop} = \sum_k \sum_j \sum_i \frac{E_{ijkt} - E_{ijk0}}{\ln E_{ijkt} - \ln E_{ijk0}} \ln \frac{P_t}{P_0} \quad (12)$$

3. Results

3.1 Dietary transition in China

The Chinese have been reducing their calorie intake while shifting to a higher intake of non-starchy food, especially meat and oil and fat. The age-cohort-weighted average calorie intake, although increasing slightly from 2180 kcal/day in 1997 to 2238.9 in 2000, started to decline from 2000 until reaching 1960 kcal/day in 2011. Statistics from other national statistics show similar results: the averaged calorie intake of Chinese decreased from 2330 kcal/day in 1992 to 2250.5 kcal/day in 2002, and then to 2170 kcal/day in 2012 (National Health and Family Planning Commission of China, 2013). Concomitantly, starchy food has been slowly replaced by animal products. Cereals accounted for 64.3% of daily calorie intake in 1997 but 48.2% in 2011. This decline is steeper than the decline in total calorie intake, indicating that Chinese people are replacing cereals with other foods while eating fewer calories in total. The replacement mainly comes from livestock products, and oils and fats with increases from 8.2% to 11.5% and 13.8% to 20.6%, respectively. Other foods, including poultry products, seafood, dairy products, eggs, legumes, and fruits, while accounting for no more than 4% of the calorie intake each, all exhibit a slight increase in consumption as well. The intake levels of tubers, vegetables, and other foods fluctuate within a narrow range over time and show no obvious trends.

Diets have been transitioning along a similar trend for individuals from different groups but with different levels. We present the change in calorie intake and its composition by food group in Figure 1. On average, urban residents show lower calorie intake and a steeper decline in calorie intake over the study period, from 2270 kcal/day to 1890 kcal/day. In contrast, their rural counterparts first increased from 2131.2 kcal/day to 2259.3 kcal/day in 2000 and started to decrease to 2026.2 kcal/day in 2011. These observations are also consistent with the data from the national nutritional survey, which showed the calorie intake changing from 2394.6 kcal/day in 1992 to 2134 kcal/day in 2002 and 2052.6 kcal/day in 2012 in the urban area, and from 2294 kcal/day to 2295.5 kcal/day and 2286.4 kcal/day in the rural area (National Health and Family Planning Commission of China, 2013). Concerning the composition of calorie intake, urban residents derive a larger percentage of calories from non-starchy food in general, but their rural counterparts are catching up rapidly. For urban residents, the calories from cereal were reduced from 55.0% in 1997 to 43.7% in 2011, while these values were 69.0% and 52.5% for rural residents. In the meantime, calories from fats and oils increased from 16.0% to 20.9% and 12.6% to 20.4% for urban and rural residents, respectively. Calories from livestock also increased in rural areas from 6.5% to 10.3% while ranging from 11.6% to 13.9% in urban areas without any evident temporal trend. A similar comparison applies to other food groups. The difference across age groups is less apparent. Figure 1 shows that adults and adolescents exhibited a steeper decline in calorie intake compared to other age groups . However, the changes in the macronutrient composition of calorie intake are similar for each age cohort.

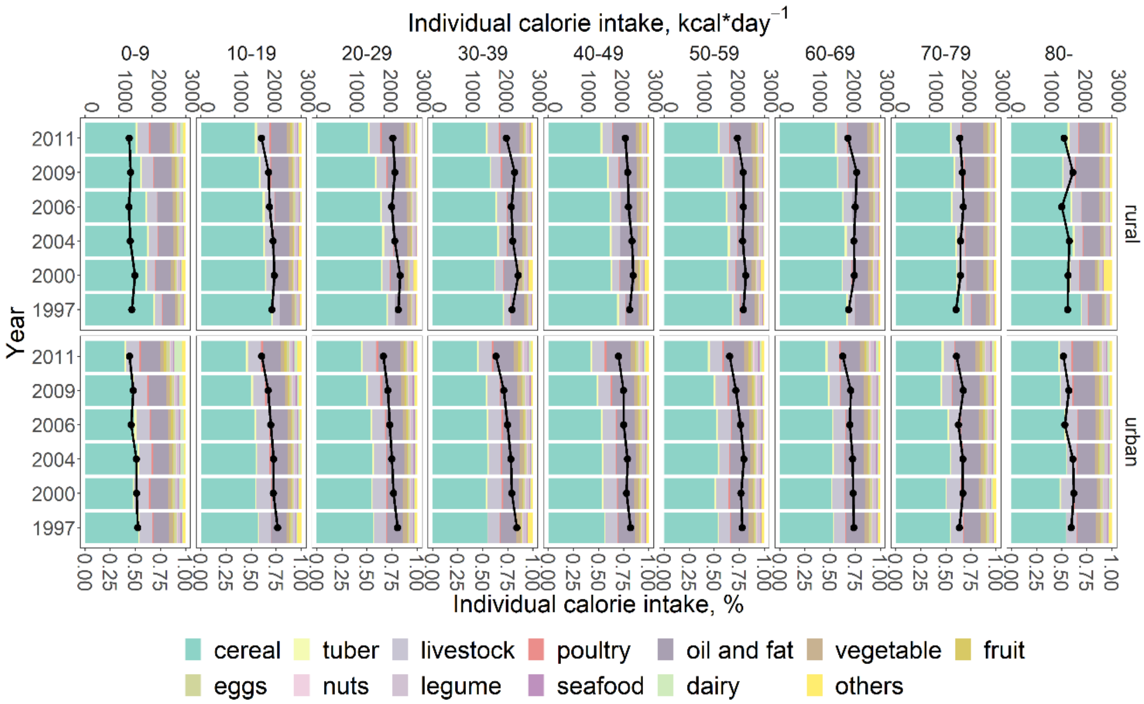

*Figure 1 Individual calorie intake and its composition for different age*living area cohorts. The points and lines show the trend of total calorie intake per capita (scaling shown on the top); the stacked bars show the composition of the calorie intake (scaling shown at the bottom).*

3.2 Dietary GHG emissions and its composition

Dietary-related GHG emissions have fallen over time, with the decline mainly coming from reduced emissions related to meat, cereals, and vegetables despite the fact that the meat intake has been increasing. Emissions at the individual level by residence area and age group are shown in Figure 2. The age-cohort-weighted average dietary GHG emissions decreased from 2.61 kg CO_2e/(capita*day)$^{-1}$ in 1997 to 1.32 kg CO_2e/(capita*day) in 2011. This value resembles the results of a previous study (Song et al., 2015), which estimates a 5-95% interval of 2.12 to 3.87 kg CO_2e/(capita*day) and a mean of 2.96 kg CO_2e/day during the period of 2004-2009. Our results are also smaller than those for developed countries, e.g. 2.94-5.93kgCO_2e/day in the UK (Scarborough et al., 2014), and 4.17kgCO_2e/day in France (Vieux, Darmon, Touazi, & Soler, 2012), partially due to lower consumption of meat and dairy products. The largest decrease occurred during the 1997-2000 period, after which the emissions bounced back during 2000-2004 and then continued to decrease. Livestock, cereal, and vegetable accounted for the majority of the dietary emissions (more than 60% in total), and contributed the largest reduction. The emissions per capita from livestock products changed from 1.75 kg CO_2e*day^{-1} to 0.57 kg CO_2e*day^{-1}),cereal (from 0.34 kg CO_2e*day-1 to 0.29 kg CO_2e*day^{-1}), and oils and fats (from 0.23 kg CO_2e*day^{-1} to 0.03 kg CO_2e*day^{-1}). The emissions from poultry products per capita experienced a slight increase from 0.025 kg CO_2e*day^{-1} to 0.84 kg CO_2e*day^{-1}. A decrease in emissions is also observed for other food groups, with a reduction of no more than 0.15 kg CO_2e*day^{-1} each. This downward trend is mainly due to technical advances despite the increased intake of some food groups as will be explained in a later section.

Individuals from different areas show similar patterns in emissions over the years but these patterns differ in extent. Urban residents are responsible for higher emissions because they eat more animal products, particularly meat, as a proportion of their daily diets. At the same time, their emissions also decrease more rapidly due to a larger reduction in total meat consumption. In urban areas, the individual dietary emissions decreased from 3.67 kg CO_2e*day^{-1} to 1.50 kg CO_2e*day^{-1} on average with emissions from livestock reducing from 2.70 kg CO_2e*day^{-1} to 0.71 kg CO_2e*day^{-1}, while in rural areas, the values are 2.11 kg CO_2e*day^{-1} and 1.12 kg CO_2e*day^{-1} with emissions from livestock reducing from 1.30 kg CO_2e*day^{-1} to 0.45 kg CO_2e*day^{-1}.

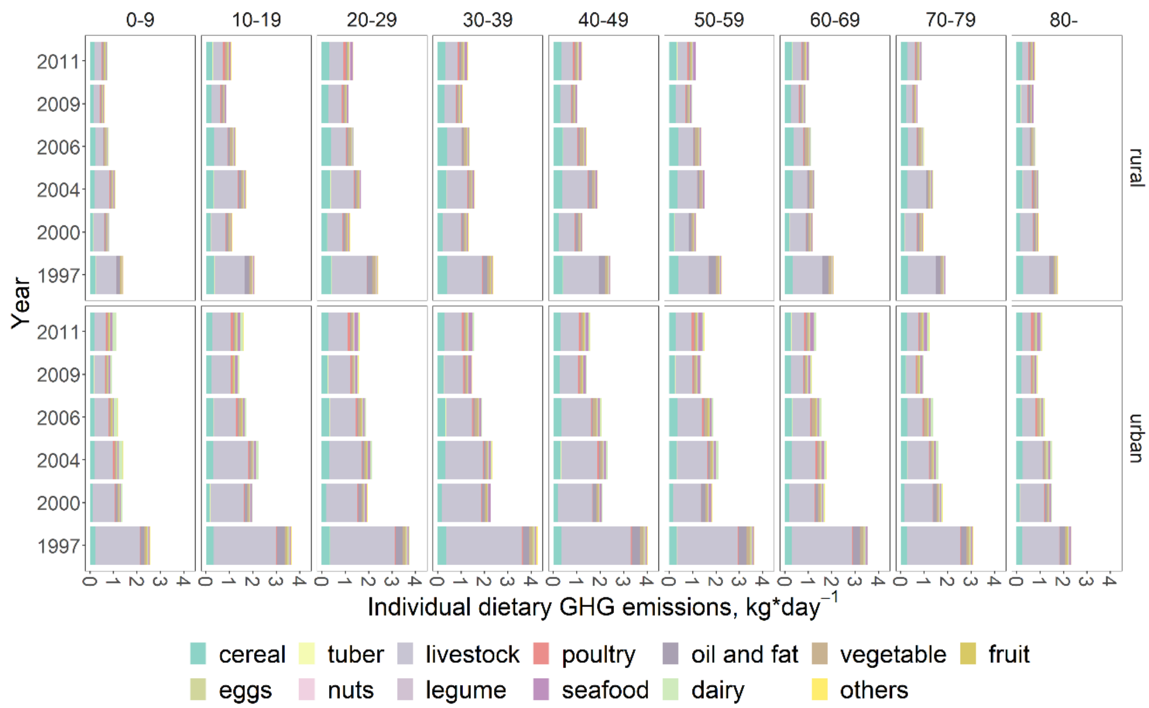

Figure 2 Per capita dietary GHG emissions by age and residence.

At the national level, dietary GHG emissions have been reduced as well. As shown in Figure 3, the total annual emissions for the whole nation declined from 1178.8 Mt CO_2e in 1997 to 556.2 Mt CO_2e in 2009, though they slightly increased to 641.9 Mt CO_2e in 2011. This range is slightly lower than that in the evaluation of Li *et.al.*, which was 1,308 to 1,618 Mt CO2-eq GHGs during 1996-2010 (H. Li et al., 2015). This is possibly because the extent of Li *et.al.* includes the food chain system, while we only address the food consumed in China, excluding exports as well as food loss and waste but involving the emissions from imported products. As China has become a leading exporter of agricultural products in the world, (FAOSTAT, 2020a) and food loss and waste accounts for 10.8% to 48.2% of the total food consumption in industrialized Asia with variation across food groups (Gustavsson, Cederberg, Sonesson, & Emanuelsson, 2011), the emissions from these factors can lead to a considerable difference across studies.

Additionally, Li *et.al.*, while showing increasing GHG emissions over time, adopt the temporally invariant IPCC emission factors for estimation so that the effect of technical progress is not embodied in the historical change in emissions. The declining trend in GHG emissions is similar to that found in Wang *et.al.* which also adopt the input-output framework for evaluation but do not include non-CO_2 GHG emissions(F. Wang et al., 2020). They present a 21% reduction in CO_2 emissions from household food consumption in China during 1992-2007. Similarly, a decreasing trend is also found in the CO_2 emissions from European agricultural production during 1995-2009, mainly due to decreasing energy intensity (T. Li, Baležentis, Makutėnienė, Streimikiene, & Kriščiukaitienė, 2016). As the factors such as technical advancement contributing to emissions reductions can reduce CO_2 and other GHGs jointly, it is possible to find a higher reduction rate when both are included in the evaluation.

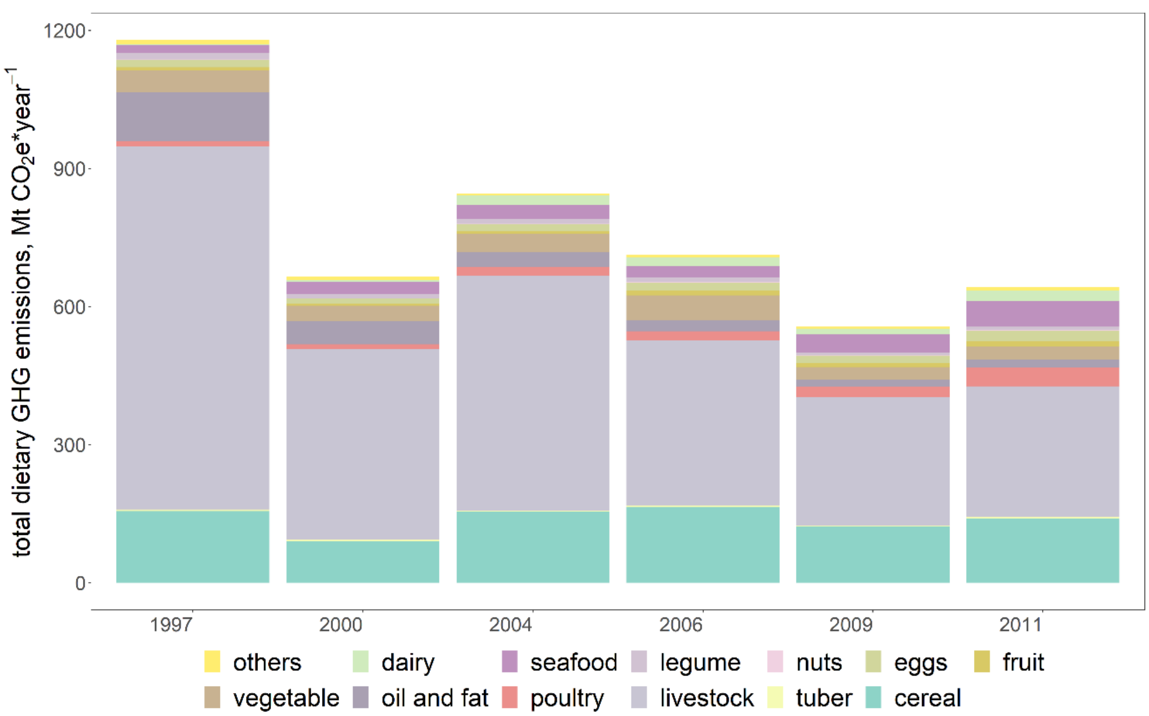

Figure 3 The composition of total dietary GHG emissions by food groups over time in China

3.3 Driving forces of the changing diet

Technical progress plays a dominant role in driving down dietary GHG emissions, while factors other than calorie intake drive emissions up. We show the decomposition of the dietary GHG emissions in China during the 1992-2012 period in Figure 4. Technical progress dominates the reduction in the total emissions over time. Its critical role is also documented in Wang *et.al.* as the consumption of all types of foods except cereal increased for Chinese households during the 1992-2007 period, while their carbon footprint dropped over the same period, indicating a reduction in CO_2 per kg embedded in Chinese food consumption (F. Wang et al., 2020). Such a reduction in carbon emission intensity has also been documented in other studies particularly for animal products such as beef and pork (B. Lin & Lei, 2015; J. Lin, Hu, Cui, Kang, & Xu, 2015). The reduction in emissions from reduced calorie intake is smaller but has been increasing critical in recent years. The structural change of diets tends to increase emissions and have a similar contribution in magnitude. The switch from foods rich in carbohydrates (e.g. cereals and tubers) to those rich in fats and proteins (e.g. oils, fats, poultry, and livestock) has led to more emissions, as the latter two are usually more emissions-intensive - fat and protein are provided primarily by cooking oils and fats, and animal and soybean products, respectively, which emit more GHGs than the starchy foods in the food chain. The nutrient composition, namely the source of carbohydrates, fats, and protein, also matters, such as switching from plant-sourced protein (soybean products) to the animal-source protein (meat). Demographic change generally leads to higher emissions primarily due to population growth. Urbanization also leads to higher emissions, attributable to the fact that more restaurants are readily accessible, which cater to the growing demand for dining out and tend to serve foods with high fat and more meat to attract customers (Byrne, Capps, & Saha, 1996; McCracken & Brandt, 1987; Barry M. Popkin, Linda S. Adair, & Shu Wen Ng, 2012). Moreover, the opportunity costs of preparing food rise due to shifts in working styles and more women entering the labor market (Wilkinson, 2004). As a result, easily and instantly available processed food items become popular, leading to easier portability, storage, and preparation at a low price due to advancements in industrial food production (Thow, 2009). As such, change tends to be more drastic for urban residents compared with their rural residents, and therefore urbanization plays a role in increasing GHG emissions. Finally, the change in population structure presents a negligible but positive contribution to the increase of dietary GHG emissions.

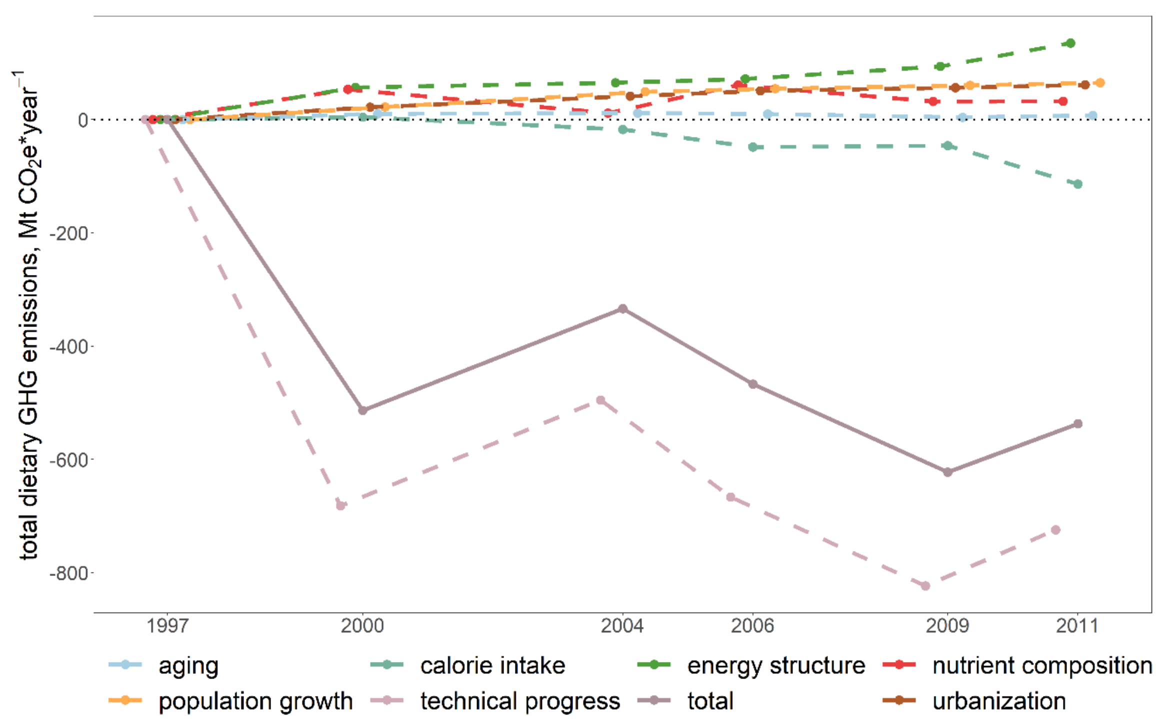

Figure 4 LMDI decomposition of the dietary GHG emissions in China

Each food group contributes to the dietary emissions change differently through both the production and consumption sides: Technical progress may differ by food production sectors and thus lead to heterogeneous emissions reduction rates; Dietary change also results in a disproportional adjustment of the amount each food group consumed in individual daily diets. To identify the key food groups for policy intervention, we further explore how each food group contributes to the change in dietary GHG emissions by adding up the components in equation (4) by food group following the method in (Zhao & Chen, 2014). The results mark meat and cereals as two major contributors to emissions reduction (Figure 5). Given that livestock consumption actually increased during the study period, this result shows that advancement in technology efficiency can compensate for emission changes in the dietary composition. The reduction started to slow in recent years, with emissions from meat even increasing slightly during the 2009-2011 period due to both smaller marginal improvement on the production side and a continuing increase in meat consumption. These two factors drove emissions in the same direction for cereals; cereal consumption went down as individuals switched to foods with more fat and protein. The contribution of other foods is primarily a result of technical progress as their intake changes only marginally. One exception is dairy products, which drive up emissions slightly, indicating that the increase in milk and yogurt (the major dairy products that consumed in China) consumption balances out the benefits from technical progress.

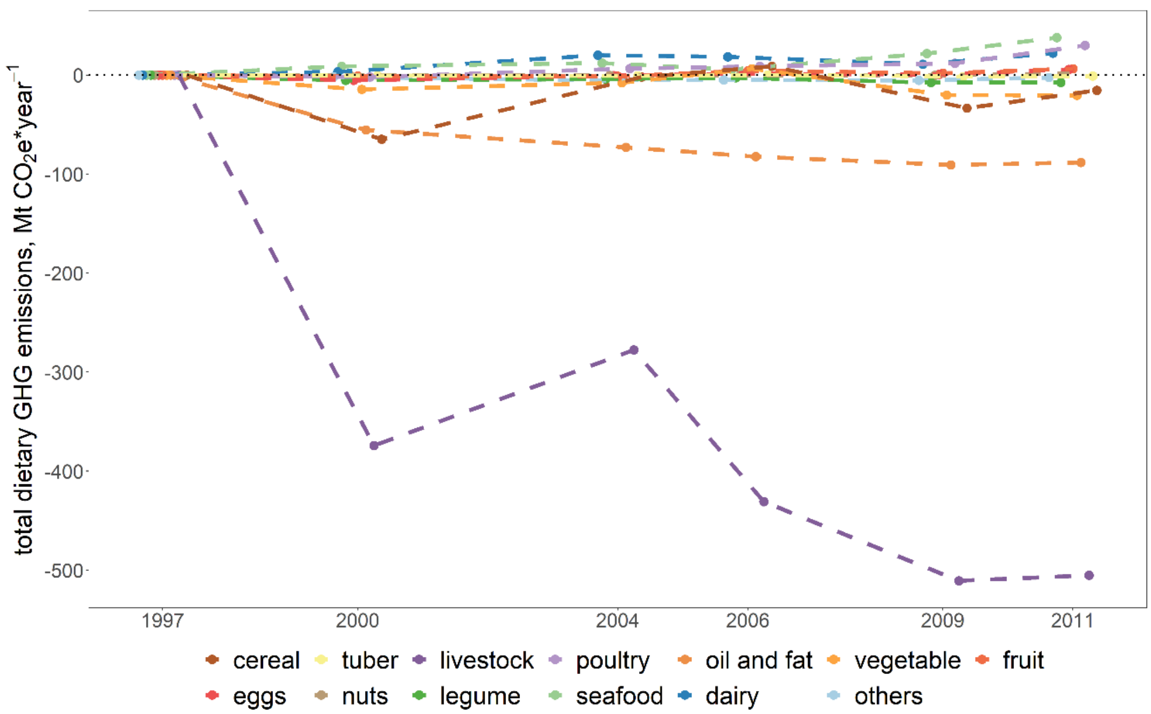

Figure 5 Contribution to dietary GHG emissions by food groups

4. Discussion

4.1 Driving forces of dietary emissions

The change in dietary composition is likely driven by interactions among concomitant factors, including lower calorie requirements due to sedentary occupations, lower food prices, greater purchasing power, and more urbanized and modernized food preparation and supply systems (Gómez & Ricketts, 2013; Kearney, 2010; Barry M Popkin, 2003; Barry M. Popkin, 2014; Barry M. Popkin et al., 2012). The decline in calorie intake is mainly related to less physically demanding jobs for a large section of the population (Ng et al., 2009; Barry M Popkin, 2001). Machines have replaced a large portion of laborers engaged in physical work, and transportation has also become more motorized. Although leisure activities are also becoming more popular, sedentary lifestyles are becoming more and more widespread which reduces individuals' energy requirements (Attard et al., 2015; Bauman, Allman‐Farinelli, Huxley, & James, 2008; Ng, Howard, Wang, Su, & Zhang, 2014). These observations are reflected in the physical activity levels of the sampled individuals. As shown in Figure S3, all age groups in both the urban and rural areas are shifting to lifestyles demanding less physical activities, particularly adults. At the same time, starchy food accounts for a larger share of the diets of people with a higher energy requirement or lower income, as such foods are the cheapest source of energy in China (Du et al., 2004). As physical activity levels (PALs) decrease and incomes increase, people can afford more expensive calories from other types of foods (particularly animal-sourced foods) and thus choose to cut down consumption on rice and flour (Du et al., 2004; Barry M Popkin & Du, 2003). Falling food prices also play a role. Because animal-sourced food has become cheaper and more affordable due to trade openness and technical advances while purchasing power has simultaneously risen, the intake levels of animal-sourced foods have increased (Barry M. Popkin et al., 2012).

The urban-rural difference in dietary structure can be attributed to disparities in lifestyle and residential environment. (Barry M. Popkin et al., 2012; Zhang, Cao, & Ramaswami, 2016). A modernized lifestyle and increased availability of restaurants and food outlets facilitated by the ongoing urbanization encourage away-from-home dining and in general the consumption of more processed food (Barry M. Popkin et al., 2012), which is usually characterized by significantly higher energy density and more animal products than that of food prepared at home (Byrne et al., 1996; McCracken & Brandt, 1987). The impact of urbanization on lifestyles in China is expected to increase in the future given the ongoing agglomeration of the population in urban area.

Increases in age are associated with increases in emissions, although quantitatively small. These increases can be explained by the different food requirement of individuals of different ages. In our study, age is represented by the percentage of the population from each age group. Fertility declines in an aging population structure, and the proportion of adults, who have a higher requirement of food, increases, which thus causes more emissions. Although elderly individuals, who require fewer calories than non-elderly adults, also account for a larger proportion of the population, the aggregated effect of population structure change is still positive. In other words, the direction of the aging effect depends on the specific shape of the population pyramid. Nevertheless, such an effect is trivial compared with other factors. Also, the study period is insufficient to separate the effect of pure aging from the effect of belonging to different generations growing up in different socio-economic environments. As more modernized and prosperous generation reaches older age, the impact is expected to be more significant. A small impact of increasing age was also found in previous studies. An evaluation of household carbon footprints in Japan, a country stepping into an aging society as well, also showed a flat trend in GHG emissions from food consumption (Shigetomi, Nansai, Kagawa, & Tohno, 2014). This latter study is also affected by a limited time span, so more research is needed to properly address this issue.

Our results show that dietary structure changes contribute more to rising GHG emissions than population growth. Other studies have shown that these findings are dependent on specific circumstances. Kastner et al. found that the effect of dietary change is slightly lower than that of population growth in determining agricultural land use in East Asia during 1963-1984, but exceeds the latter during 1984-2005; by contrast, population growth still takes the lead in less developed areas such as Africa (Kastner, Rivas, Koch, & Nonhebel, 2012). Yang and Cui also concluded that dietary change may override population growth to raise the dietary water footprint on a global scale in the future (Yang & Cui, 2014). Along with our findings, these observations reflect how economic development significantly reshapes the dietary patterns of households until their environmental impact catches up with the impact from the growing population.

Our findings add to the body of research looking at driving forces behind dietary environmental impacts. We found that technical progress has an important role in the case of China. In general, , results may differ across the types of environmental footprints studied. (Kastner et al., 2012) found that population growth balances out the effect of technology in driving up the land requirement for global food consumption. (Yang & Cui, 2014) found that technical progress has an important impact on water footprint, while Zhao and Chen concluded that economic development, population growth, and dietary change have a larger effect than changes in technology (Zhao & Chen, 2014). The conclusion reached by different researchers may also depend on the country's level of development. (Kastner et al., 2012) shows that for the agricultural land footprint, the technology has had a larger effect than population growth and dietary change over time in Europe, but the opposite is true in Asia, Africa and other areas.

4.2 Policy implications of abating dietary GHG emissions

The distinct contribution of each driving factor provides a basis to assess the effectiveness of different policies in reducing dietary environmental impacts. Given that the benefits from technical progress are large but tends to level off, it is not clear how much such progress can contribute in the future. On the other hand, as research and development costs for greener methods rises, production-side options will become increasingly less feasible. On the other hand, managing emission from the consumption-side appears to have greater potential, particularly for countries such as China with fast-growing food demand due to population growth, rapid urbanization, and increasing affluence. In particular, for developing countries like China, meat consumption is predicted to continue to increase in the future (Alexandratos & Bruinsma, 2012). In order to limit more negative effects of this trend, providing public service advertisements and dietary education that advocate healthy diets with less meat could be a “low-hanging fruit”, to help driving affordable consumer behavioral change as shown in multiple programs (Afshin et al., 2017). Such measures addressing environmental issues can also improve the nutritional quality of diets and lead to positive health outcomes (Behrens et al., 2017; Springmann, Godfray, Rayner, & Scarborough, 2016). Another strategy is to make low-carbon, healthy foods such as vegetables, fruits, or meals with reduced oil more accessible in the food supply. As an example, governments can offer financial and regulatory incentives to increase the number of healthy food retail outlets offering local produce. In particular, China is urbanizing rapidly. Since urbanization can lock people into carbon-intensive lifestyles and diets (Seto et al., 2016), it is critical to take actions immediately to create a food environment that promotes sustainable diets before the urban lifestyle has become locked into path dependence.

As the most emission-intensive food group, meat is central in terms of policy development from both the production and consumption sides. Our results show a substantial reduction in GHG emissions due to production improvements. Despite diminishing marginal environmental benefits, there is still room for further emission reduction by expanding the use of techniques such as ranching intensification and adopting best-practice animal management strategies discussed in recent studies (Herrero et al., 2016). On the other hand, consumer behavior change can make a significant difference. In China pork consumption has quadrupled since 1971, while beef consumption has increased fivefold (Westcott & Trostle, 2014). Both are predicted to continue to increase (Alexandratos & Bruinsma, 2012). Currently, the per capita meat consumption in China has exceeded dietary recommendations (Chinese Nutrition Society, 2016; Song et al., 2017), bringing about adverse impacts on both the environment and individuals' health (He, Baiocchi, Hubacek, Feng, & Yu, 2018). Studies have shown that reducing meat consumption could lead to a considerable reduction in emissions in China (Song et al., 2017). Therefore, policy tools for promoting changes in food consumption behavior can complement production-based strategies to implement emissions reductions in the Chinese food system.

Acknowledgements

Funding for this research was provided by the National Natural Science Foundation of China under Young Scholar Program Grant (71904098) and China Postdoctoral Science Foundation under Chinese Postdoc Scientific Grant (2019M650704).

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