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China's changing diet and its impacts on greenhouse gas emissions: an index decomposition analysis*

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With increasing awareness of agriculture's contribution to global greenhouse gases (GHGs) and China's position as the world's top GHG emitter, there is heightened attention to the embodied emissions in China's food consumption. China's diet has shifted to include more fruit, vegetables, meat and dairy. Not surprisingly, GHG emissions from food consumption have also increased substantially. This analysis links China's food consumption with the emissions of food production industries in China and its trade partners to determine the effects of dietary change on GHGs since 1989. We utilise high-resolution food production and emissions data to perform a logarithmic mean Divisia index decomposition to attribute changes in GHG emissions to the scale, supply structure, demand structure and efficiency effects resulting from Chinese dietary changes over a 20-year period. This study finds that while countries supplying food to China contribute little to China's food-related GHGs, demands for meat and dairy play a much larger role, driving up emissions. The overall scale of increased consumption of all food further propels growth in GHG emissions. Results indicate, however, that while food consumption in China more than doubles between 1989 and 2009 improvements in technological efficiency limit the rate of increase.

Key words: China, dietary change, embodied emissions in trade, greenhouse gas emissions, index decomposition analysis, logarithmic mean Divisia index.

1. Introduction

An increasing volume of the literature is drawing attention to the sizable share of greenhouse gas (GHG) emissions attributable to agriculture and food (Herrero *et al*. 2011; O'Mara 2011; Cederberg *et al*. 2013). Increasing attention is being paid to methods and opportunities for reducing GHGs associated with food (Steinfeld *et al*. 2006; O'Mara 2011). Most of these studies examined food production and agriculture in Western nations, such as

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the European Union, United States and Australia, with studies outside these countries, such as in India, being relatively fewer (Herrero *et al.* 2011; O'Mara 2011; Weiss and Leip 2012; Cederberg *et al.* 2013). As the world's largest producer of GHGs and with increasing influence in the Pacific Rim, there is great interest in GHG production in China, in general (Su and Ang 2013; Hawkins *et al.* 2015), but only a few studies have looked specifically at GHG production in China associated with the agricultural sector and food production industries (Wang 2010; Lin *et al.* 2014). In parallel, while the food and beverage industry has been commonly identified in decomposition analyses of energy use and GHG emissions (Ang and Zhang 2000; Ma and Stern 2008; Zhao *et al.* 2010), its use in determining the drivers for food-related GHG emissions has been limited (Hawkins *et al.* 2015). This study fills this gap in three ways. Firstly, we utilise and combine disaggregated crop and livestock sector data with a series of emission factors for production methods specific to China and its trade partners in 1989, 1999 and 2009 to provide the GHGs attributable to the production of different crop and livestock products between 1989 and 2009. Secondly, we use a logarithmic mean Divisia index (LMDI) to decompose the factors that affect the GHG generation associated with the increases and proportional changes in China's food production. Finally, we address concerns over the impacts of China's increasing consumption of meat and dairy on the GHG emissions of its Pacific Rim and other trade partners and the world as a whole.

A great amount of the literature has been dedicated to China's increasing food consumption, the changing structure of China's diet and the efficiency improvements in China's agricultural and food sectors (Huang *et al.* 1999; Rosegrant *et al.* 2001). Some of this literature has addressed the contribution of China's food production to GHG emissions, though these are largely expressed in either general terms (Steinfeld *et al.* 2006) or looking very specifically at the carbon intensities of Chinese agricultural production (Wang 2010; Lin *et al.* 2014). Review of the literature reveals how this paper can use index decomposition analysis (IDA) to show how China's changing diet affects GHG emissions by bridging the gap between the issues of increasing food consumption, the changing structure of the Chinese diet and reductions in the carbon intensity of food products.

China's increasing food consumption, particularly of animal products and fresh fruits and vegetables, has been in the spotlight for academics, governments, nongovernmental organisations and the media alike (Zhou *et al.* 2012). While the impacts of China's increasing demand and production of crops and livestock on agricultural lands are commonly identified in the literature (Naylor *et al.* 2005; Fu *et al.* 2012; Zhou *et al.* 2012), the contribution of food consumption to GHG emissions is more rarely recognised (Steinfeld *et al.* 2006; Deng *et al.* 2012; Xu *et al.* 2013) and has not been appraised in the level of detail provided in the present study.

In addition to the issue of increased food consumption, changes to the structural composition of the Chinese diet are crucial to discussions of

GHGs. China's diet in the 1990s transformed from that of a low-income, developing nation to one of a country transitioning to higher incomes, with pork, poultry, eggs and oilcrops taking on greater roles in Chinese consumption patterns (Guo *et al.* 2000; Popkin 2001). Liu *et al.* (2009) explicitly looked at different livestock products, finding that as Chinese income increases, the level of meat consumption increases and so does the composition of meat consumed, shifting from fatty meats like pork to leaner meats like beef. These changes have major ramifications for GHGs with livestock requiring more land inputs than crops which, along with enteric GHG generation, results in substantially greater carbon intensities in production of animal sourced food (Wang 2010; Lin *et al.* 2014). Other than the identification of the carbon intensity of different livestock products, however, the literature largely lacks studies connecting the structural changes in food consumption with changes in GHG emissions.

As China's constraints on land and water with regard to the production of livestock are of great concern, the literature is heavy with studies on improvements in the technical efficiency of livestock production. These studies typically focus on the topics of feed conversion ratios (Huang *et al.* 1999), increased yield and especially productivity (Rae and Hertel 2000; Nin *et al.* 2004; Rae *et al.* 2006). Lin *et al.* (2014), however, not only calculate the carbon intensity of crop and livestock products using a hybrid economic input–output and life cycle assessment (EIO-LCA) but find the carbon intensities over time for the years 1979, 1989, 1999 and 2009. Lin *et al.* (2014) calculate the carbon intensities using both direct and indirect GHG sources, including fertiliser and pesticide production, fertiliser application, enteric fermentation, manure management and direct energy use as well as indirect sources such as facilities, veterinary medicine and feed. By combining implied emission factors derived from FAO data with Lin *et al.*'s carbon emission factors (CEF) as a benchmark (Hawkins *et al.* 2016), this study is the first to account for direct and indirect GHG sources over time and in conjunction with structural change and increased consumption levels to determine how China's changing preferences for different crop and livestock products affect GHG emissions.

A wide variety of studies have examined the embodied emissions in trade with regard to China (Xu and Dietzenbacher 2014; Malik and Lan 2016). When food and agriculture are mentioned in these studies; however, they are typically mentioned in passing or in an appendix as just one of many sectors included in the analysis (Guo *et al.* 2012a,b; Dong *et al.* 2013; Vause *et al.* 2013). There are no published studies currently that disaggregate the food and agriculture sectors into specific food types to determine the embodied emissions of food coming from other countries. While Hendrie *et al.* (2014) look at different categories of food in their analysis of changes in CO_2 emissions associated with a change in the diet of Australians, consideration of trade is limited between Australia and 'Rest-of-World'. This study is the first to disaggregate embodied emissions not only by types of food products but also by the country producing and exporting the food.

Many Chinese studies using structural decomposition analysis (SDA) or IDA to analyse energy and GHG emissions include, typically briefly, discussion of the food and beverage or agricultural sector. These are most often taken within the greater context of the Chinese economy as a whole, presented in comparison with the manufacturing, electricity production, mining and other sectors (Ma and Stern 2008; Ma 2010; Minx *et al.* 2011; Wang *et al.* 2011; Su and Ang 2012). This study is the first to use IDA to directly link GHG emissions with specific changes in China's diet and consumption levels over time.

2. Material and methods

To understand the drivers behind the changes in GHGs associated with China's changing diet, we analyse food-related carbon dioxide equivalent (CO_2-e) emissions for China's food consumption between 1989 and 2009. These data are developed using crop and livestock production, import and export quantities for China and its trade partners in conjunction with CEFs for individual food categories, determined for China and its trade partners, derived from FAO data. With the data ranging from 1989 to 2009, we perform an IDA on CO_2-e levels associated with Chinese food consumption. We expand on the conventional 3-factor IDA which typically disentangles the effects of *structure*, *scale* and *efficiency*. In this discussion of China's diet, *structure* describes the mix of different livestock products within the overall diet, *scale* presents the level of consumption of livestock products and *efficiency* represents the carbon intensity of the livestock products. In this study, however, we further split the analysis of *structure* into *demand structure*, which represents the changes in the proportions of food eaten in China relative to the overall food consumed, and *supply structure*, which represents the proportion of different food supplied to China by its own production and trade partners.

2.1 Data

China's food consumption quantities are calculated by taking Chinese domestic food production quantities, subtracting China's food production that is exported to other nations and adding in quantities of food imported to China from other nations. Food production, export and import data are all taken from FAOSTAT for the years 1989, 1999 and 2009 (FAOSTAT 2015). FAO food data are aggregated from 138 categories of food crop products to nine crop categories including maize, rice, wheat, sugar, fruit, oilcrops, pulses, roots and vegetables. The FAO 12 categories of livestock food products are aggregated to six livestock categories including milk, beef, sheep, pork, poultry and eggs. Data from 1989, 1999 and 2009 as well as the aggregation of food products are selected to correspond with the Chinese benchmark data by Lin *et al.* (2014).

FAO data are collected and collated from national statistical agencies or other international organisations through forms and questionnaires. As with other international organisations such as the International Monetary Fund or the Organisation for Economic Co-operation and Development, the quality of the FAO data is largely dependent on the quality of data collected by individual countries and organisations. China's National Bureau of Statistics and other statistical agencies are not established in a political vacuum and are often pressured by other agencies to provide statistics that conform to agency goals or policies (Guan *et al*. 2012). Under the FAO quality assurance framework, however, statistical outputs are assessed and validated on a regular basis, errors are measured and documented, methods for preventing and reducing errors are in place and implemented, and the revision policy is made publicly available (FAO 2014).

As shown in numerous studies, the CEFs for agricultural products vary widely from nation to nation, from region to region or even from farm to farm based on different production practices (Dyer *et al*. 2010; Herrero *et al*. 2011; O'Mara 2011; Hawkins *et al*. 2016). Along with geographic heterogeneity, it is also important to consider how CEFs change over time as production processes become more or less CO_2-efficient or -intensive (Cederberg *et al*. 2013; Lin *et al*. 2014). Lin *et al*. (2014) attribute technological improvements for reducing the CEFs of certain food categories, but also point out examples such as shifts from the use of manure application to synthetic fertilisers as increasing the CEFs of other food categories over time.

To account for China's food imports as well as its domestic production in determining the GHGs associated with its changing diet, we require a set of CEFs for our 15 categories of food products that cover China and the nations exporting food to China. Unfortunately, a review of the literature finds that while CEFs for certain food or in particular nations have been studied to a greater or lesser degree, the assumptions upon which the CEFs are made vary widely, raising questions as to their comparability between studies (Flysjo *et al*. 2011; Herrero *et al*. 2011; Desjardins *et al*. 2012; MacLeod *et al*. 2013; Hawkins *et al*. 2016).

Consequently, this study utilises a set of implied CEFs derived from FAOSTAT production, yield and GHG data for the crop and livestock categories used in this study. As described by Hawkins *et al*. (2016), the FAO provides implied emission factors for livestock products in kg GHG/head. These emission factors are multiplied by the inverse of the production yield to generate a coefficient of kg GHG/kg of animal product. As the FAO provides no implied emission factors for crop products, we derive our own from the FAO data. The FAO calculates crop GHGs as guided by the IPCC (2006) not by crop type but by agricultural activity such as manure application, energy use and cultivation of organic soils. These activities are multiplied by an emission factor and summed to give an overall quantity of agricultural GHGs. We derive an implied emission factor by dividing the agricultural GHGs by the quantity of crop products to get a set of generic baseline emission factors for crops. These are then modified for different food types based on cultivation activities, the gases emitted and annual yields to get specific and individual CEFs for each of the nine crop categories. This process is repeated for the crop and livestock categories for all of China's trade partners and for the years 1989, 1999 and 2009.

As described for the process by Hawkins *et al.* (2016), the CEFs found by Lin *et al.* (2014) for food production in China are used as the benchmark to calibrate the CEFs derived from the FAO data as it is the most comprehensive in scope of the food products studied, the integration of direct and indirect GHG emissions, inclusion of time series data, and aggregates food categories in a manner similar to that of the FAO. Lin *et al.*'s study includes the direct emissions sources used by the FAO, such as fertiliser application, direct energy use, enteric fermentation, and manure management but also includes indirect sources such as emissions embodied in agricultural production inputs such as fertiliser and pesticide as well as animal housing, veterinary medicine, and, importantly, animal feed. Accounting for these indirect sources as well as direct sources, Lin *et al.* (2014) provide a more complete picture of food production emissions than illustrated by the FAO figures. With this in mind, the CEFs for China based on FAO data and Lin *et al.*'s CEFs are set as a ratio representing the difference between accounting for direct emissions only and the sum of direct and indirect emissions. This ratio is then used to calibrate the FAO CEFs for the food categories of other nations to produce CEFs based on FAO yield and emissions data but also including a factor for indirect emissions. Uncertainty associated with the CEFs used in this study is addressed by Hawkins *et al.* (2016). Sensitivity analysis of the CEFs is hampered by a lack of comparable CEFs for food products available for China and its trade partners. For more detail, readers are referred to Hawkins *et al.* (2016).

FAO production, export and import quantities are used in conjunction with the CEFs to determine the GHGs for each of the food categories (FAOSTAT 2015). Chinese food export quantities are deducted from production quantities for each of the 15 food categories for 1989, 1999 and 2009 to determine the country's consumption of domestically produced food. These quantities are multiplied by the appropriate CEF for that product and year, resulting in the total CO_2-e emissions produced by that food category for the specified year for China. For each nation exporting food to China, the quantities of the food types are multiplied by the appropriate CEFs to give the CO_2-e emissions by category and year. With the inclusion of both exports and imports in addition to domestic production, we are able to look deeper into the drivers of how changes in China's overall food consumption have changed GHG emissions over time.

2.2 Methodology

With the calculation of GHG emissions for different years, the changes over time can be analysed to determine their drivers. In this study, the share of a product's CO_2-e with respect to the total change in CO_2-e is ascribed to four factors:

- Demand structure (ΔV_{strd}) – the change in CO_2-e associated with the proportion of a food product relative to all food products resulting from shifting demands in China's diet;
- Supply structure (ΔV_{strs}) – the change in CO_2-e associated with the proportion of a food product a single nation (including China) supplies to China relative to all food products supplied for China's consumption;
- Scale (ΔV_{sc}) – the change in CO_2-e associated with the quantity of food products; and
- Efficiency (ΔV_{ef}) – the change in CO_2-e produced per unit of food product.

This analysis uses IDA to determine the contribution of each of these factors to the overall change in CO_2-e emissions produced with respect to China's food consumption (ΔV_{tot}). Variables with superscripts 0 and T represent that variable at the baseline (in this case, 1989) and time T (in this case, 2009).

$$\Delta V_{\text{tot}} = V^T - V^0 = \Delta V_{\text{strd}} + \Delta V_{\text{strs}} + \Delta V_{\text{sc}} + \Delta V_{\text{ef}}$$

ΔV_{tot} is the total difference in CO_2-e emissions between 1989 and 2009 and is also the sum of the demand structure effect, supply structure effect, the scale effect and the efficiency effect. This study draws on the LMDI-I methodology described by Ang and Zhang (2000), Ang (2004), Ma and Stern (2008), Su and Ang (2012), and Ma (2014) to decompose the factors that are balancing China's increasing food production. The CO_2-e emissions are decomposed in the Stata 14 data analysis software package using the following equation:

$$V = \sum_i \sum_j V_{ij} = \sum_i \sum_j Q \frac{Q_i}{Q} \frac{Q_{ij}}{Q_i} \frac{V_{ij}}{Q_{ij}} = \sum_i \sum_j QDSI,$$

where V, total CO_2-e produced (for all food types) (tonnes); Q, overall production level (for all food types) (tonnes); V_{ij}, CO_2-e produced by food type i in country j (tonnes); Q_i Production level of food type i (tonnes); Q_{ij}, Production level of food type i in country j (tonnes); D, Demand share of food type i from country j; S, Supply share of food type i from country j; I, Emission intensity of food type i in country j; L, Logarithmic mean weight function of relative changes.

And using the additive formulae for LMDI-I:

$$\Delta V_{\mathrm{strd}} = \sum_i \sum_j L \ln\left(\frac{D_T}{D_0}\right),$$

$$\Delta V_{\mathrm{strs}} = \sum_i \sum_j L \ln\left(\frac{S_T}{S_0}\right),$$

$$\Delta V_{\mathrm{sc}} = \sum_i \sum_j L \ln\left(\frac{Q_T}{Q_0}\right),$$

$$\Delta V_{\mathrm{ef}} = \sum_i \sum_j L \ln\left(\frac{I_T}{I_0}\right),$$

$$L = \frac{V_{ij_T} - V_{ij_0}}{\ln\left(\frac{V_{ij_T}}{V_{ij_0}}\right)}.$$

3. Results

Following development of the time series, we examine the trends among different food types within the crop production and livestock categories with respect to consumption levels and production of CO-e emissions. Using the IDA of the CO_2-e emissions over the study period, we then identify the drivers of changes in the Chinese food-related GHG emissions.

3.1 Food consumption and GHG production trends

Table 1 shows the CO_2-e emissions calculated for each of the nine crop and six livestock food categories for China in 1989, 1999 and 2009. These totals include Chinese domestic production minus exports combined with imports from China's food supplying trade partners. For comparison purposes, Table 1 also includes the quantities of each food category consumed in 1989, 1999 and 2009. In 1989, crops made up 95 per cent of China's food consumption measured in this study by weight, with livestock products comprising the remaining 5 per

Table 1 Greenhouse gas emissions and consumption quantities for crop and livestock products in China between 1989, 1999 and 2009

Food consumption	Crop and livestock products consumed (millions of tonnes)														
	Milk	Beef	Fruit	Poultry	Sheep	Vegetables	Eggs	Oilcrops	Pulses	Pork	Maize	Sugar	Roots	Rice	Wheat
1989	4.7	0.9	20.8	3.5	1.0	122.2	7.5	33.4	11.9	22.0	80.3	66.6	143.2	182.5	118.2
1999	8.9	4.4	65.6	13.4	2.6	272.2	21.7	64.0	17.0	35.2	129.0	89.1	192.7	200.4	128.9
2009	38.0	6.5	118.0	17.8	3.9	477.5	27.8	124.1	29.2	49.3	168.4	124.7	178.0	196.7	125.0
$\Delta_{1989–2009}$	701%	592%	466%	408%	304%	291%	273%	271%	146%	124%	110%	87%	24%	8%	6%

Greenhouse gas emissions	Crop and livestock CO_2-e (millions of tonnes)														
	Milk	Beef	Fruit	Poultry	Sheep	Vegetables	Eggs	Oilcrops	Pulses	Pork	Maize	Sugar	Roots	Rice	Wheat
1989	8.3	45.2	33.6	8.4	42.0	47.2	17.6	17.8	2.4	143.9	26.0	8.1	17.8	357.4	32.0
1999	26.8	101.1	62.3	19.0	58.5	111.0	31.5	23.9	3.1	125.0	37.6	8.9	23.5	360.1	31.2
2009	62.8	136.8	104.3	20.5	82.8	124.7	31.9	64.8	9.7	147.5	60.1	7.7	33.8	344.2	52.8
$\Delta_{1989–2009}$	659%	202%	210%	143%	97%	164%	82%	264%	300%	2%	131%	−4%	90%	−4%	65%

cent. The amount of livestock products increases to 7 per cent in 1999 and to 9 per cent by 2009. With the volume of the literature highlighting the contributions of livestock products to GHGs (Herrero *et al.* 2011; MacLeod *et al.* 2013), it is unsurprising that CO_2-e attributed to the consumption of animal products is 33 per cent of food consumption-related emissions in 1989, 35 per cent in 1999 and 38 per cent in 2009.

Consumption increases across all food categories, with the greatest growth among livestock products. Consumption of pork increases the least, more than doubling between 1989 and 2009, and milk products grow the most with an almost sevenfold increase. Among crop products, roots and tubers, rice, and wheat, consumption remains relatively flat, growing between 6 per cent and 24 per cent during the 20-year study period. Other crop product consumption grows considerably more rapidly. Sugar, maize and pulse consumption approximately doubles, oilcrops and vegetables nearly quadruple, and fruit consumption has an almost fivefold increase.

In terms of GHG emissions, however, the increased consumption in most food categories is not matched by increases in emissions. While emissions associated with livestock products grow, the increases are all less than the growth in consumption. The increased emissions for milk products approach the growth in consumption of its products but remain less. Emissions from pork are nearly flat over the study period, increasing only 2 per cent. Among crop products, emissions associated with rice and sugar actually decrease between 1989 and 2009 by 4 per cent. Fruit, oilcrops and vegetables all saw increases in GHG emissions for the period but less than their corresponding increases in consumption. The increases in CO_2-e emissions associated with maize, wheat, pulses and roots, however, are all above the rates of growth in their corresponding consumption.

Emissions from rice comprise the largest proportion of overall emissions, making up 44 per cent of food-related CO_2-e emissions in 1989. This decreases to 27 per cent in 2009 but is still the largest single contributor to China's food-related GHGs. Emissions associated with pork are the next largest source of CO_2-e but remain less than half of the emissions from rice. Vegetable product and beef GHGs fall below the quantities associated with pork, but both grow substantially and at similar rates between 1989 and 2009. Although GHGs associated with sheep are greater than those for fruit in 1989, by 1999 and through 2009, fruit CO_2-e had grown to outpace GHGs from sheep.

These changes over the 20-year study period reflect the effects of the demand structure, supply structure, technological efficiency and scale on food consumption-related GHG emissions, as described further below.

3.2 Drivers of food consumption-related emissions

Figure 1 shows the effects of scale, efficiency, demand structure and supply structure on CO_2-e emissions due to changes in China's food consumption in

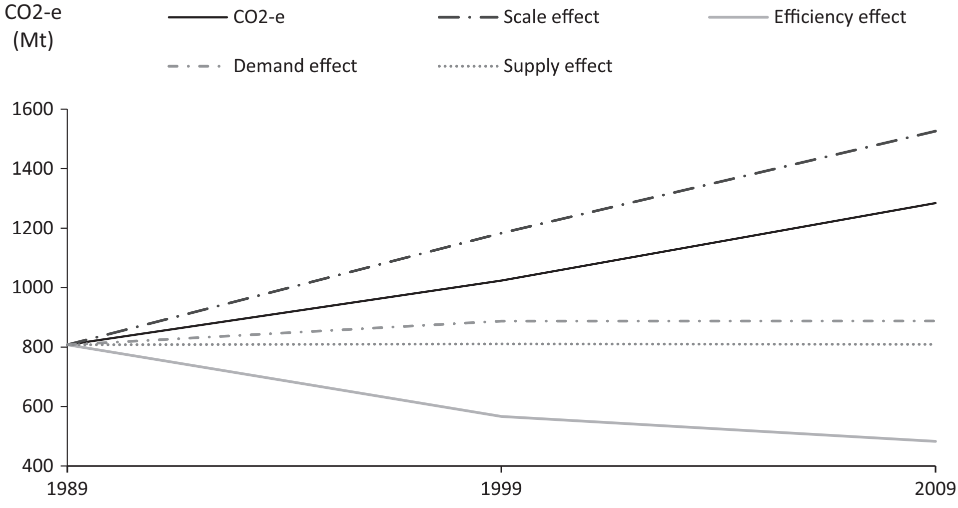

Figure 1 Magnitude of the effects of efficiency, supply structure, demand structure and scale on CO_2-e emissions resulting from changes in Chinese food consumption between 1989 and 2009.

between 1989 and 2009. The scale effect plays a dominating role in the increase in CO_2-e, responsible for a 375 Mt increase in GHG emissions between 1989 and 1999 and increasing up to 719 Mt by 2009. The effect of changing demands for products is positive but small relative to the scale effect. The demand structure effect results in an increase in 80 Mt of CO_2-e overall between 1989 and 2009, with 79 Mt of this increase occurring between 1989 and 1999. The supply structure effect plays very little role with regard to China's food consumption CO_2-e emissions, increasing CO_2-e by 3 Mt between 1989 and 1999, but then decreasing by 1.6 Mt to 2009. The efficiency effect counteracts the emissions associated with increases in production by approximately 41 per cent between 1989 and 2009, reducing GHG emissions levels 241 Mt between 1989 and 1999, and further decreasing to 325 Mt of CO_2-e by 2009.

3.2.1 Supply and demand structure effects

The combined supply structure and demand structure effects comprise a total of 82 Mt of GHGs between 1989 and 2009, approximately 17 per cent of the total growth in China's food consumption-related CO_2-e emissions in that period.

Supply structure effects are responsible for <1.4 Mt of China's GHG emissions between 1989 and 2009, approximately 0.3 per cent of food consumption CO_2-e emissions for the period studied. The composition of different nations' contributions to this effect, however, is fairly complex. As shown in Figure 2, China is responsible for 2.1 Mt of the 3 Mt supply structure effect between 1989 and 1999. The United States contributes 1.2 Mt to the supply structure effect, while Australia reduced the supply structure

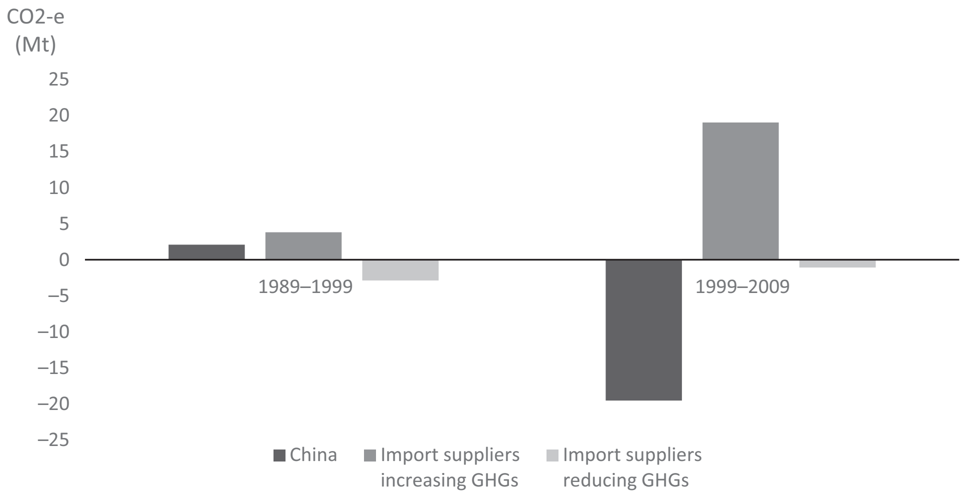

Figure 2 Contributions of China and its food import suppliers to the supply structure effect on CO_2-e emissions between 1989–1999 and 1999–2009.

effect by 1.2 Mt CO_2-e. Increases in China's supply structure-related CO_2-e in 1989 from other nations such as the United States, Canada and Brazil total 3.8 Mt of CO_2-e. These increases are counterbalanced by reductions in CO_2-e from other nations such as Australia, the Netherlands and Argentina totalling −2.9 Mt of GHGs. Between 1999 and 2009, however, the supply structure effect decreases by 1.6 Mt of GHGs. Accounting for the overall period between 1989 and 2009, we find changes in China's supply structure responsible for a 17 Mt decrease in CO_2-e. As with the 1989 to 1999 period, we find countries such as United States, Brazil and Canada among nations responsible for increasing CO_2-e emissions by almost 23 Mt over the 20-year period. Australia and the Netherlands with other countries reduce CO_2-e due to supply structure by almost 4 Mt between 1989 and 2009. Imports from countries such as the United States, Canada and Brazil contribute 4.8 per cent of the increase in China's food consumption-related GHGs for the study period. In contrast, however, China's own domestic production, along with that of countries such as Australia and the Netherlands, offset this increase by 4.5 per cent.

Demand structure accounts for over 80 Mt of the change in China's food consumption-related CO_2-e emissions in the 20-year study period. As shown in Figure 3, the demand structure effects associated with beef, vegetables, fruit, oilcrops and all other categories of livestock products are responsible for increases in GHGs between 1989 and 2009. With the exception of milk, the demand structure effects of livestock products are greater between 1989 and 1999 than in the period between 1999 and 2009. The demand structure effects for beef are the greatest among categories between 1989 and 1999, increasing CO_2-e emissions by 78 Mt, but in 1999 to 2009, contributing a little over 10 Mt. Milk, however, with an almost fivefold increase in

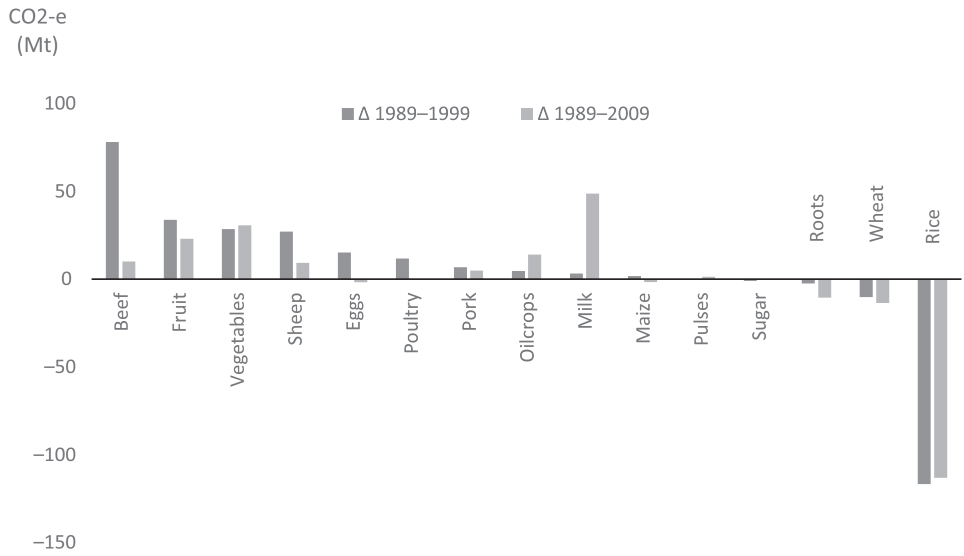

Figure 3 Contributions of demand structure effects by different food types on CO_2-e emissions between 1989–1999 and 1999–2009.

consumption between 1999 and 2009 is responsible for the strongest demand structure effects during this same period, 52 Mt of GHGs. Demand structure effects from cereals and grains, consumption of which has declined in China, remain relatively flat. The demand structure effect associated with relative decreases in consumption of sugar, roots and tubers, and especially rice is reflected in decreases in CO_2-e emissions during the study period. The demand structure effect for rice decreases GHGs by almost 230 Mt.

3.2.2 Scale and efficiency effects

As opposed to the supply and demand structure effects which contribute 82 Mt of CO_2-e to China's food consumption-related GHGs, scale effects increase CO_2-e levels by 719 Mt. This is, in turn, offset through efficiency effects reducing CO_2-e by 325 Mt.

As shown in Figure 4, scale effects for all food products are positive. Although the decreasing relative demand for rice results in a strong decreasing demand structure effect, this is countered by substantial overall increases rice consumption resulting in a scale effect of 257 Mt of CO_2-e for the study period. The overall increased consumption of pork contributes 97 Mt of GHGs and the scale effects of vegetables and beef add similar amounts of roughly 65 Mt over the 20-year study period. The remaining categories are each responsible for <50 Mt of GHGs, with increased consumption of pulses adding only 2.8 Mt.

In contrast, the efficiency effect offset increases by 325 Mt, reducing GHGs by 241 Mt between 1989 and 1999 and a further, but slowing, 84 Mt reduction in GHGs between 1999 and 2009. As illustrated in Figure 5, the

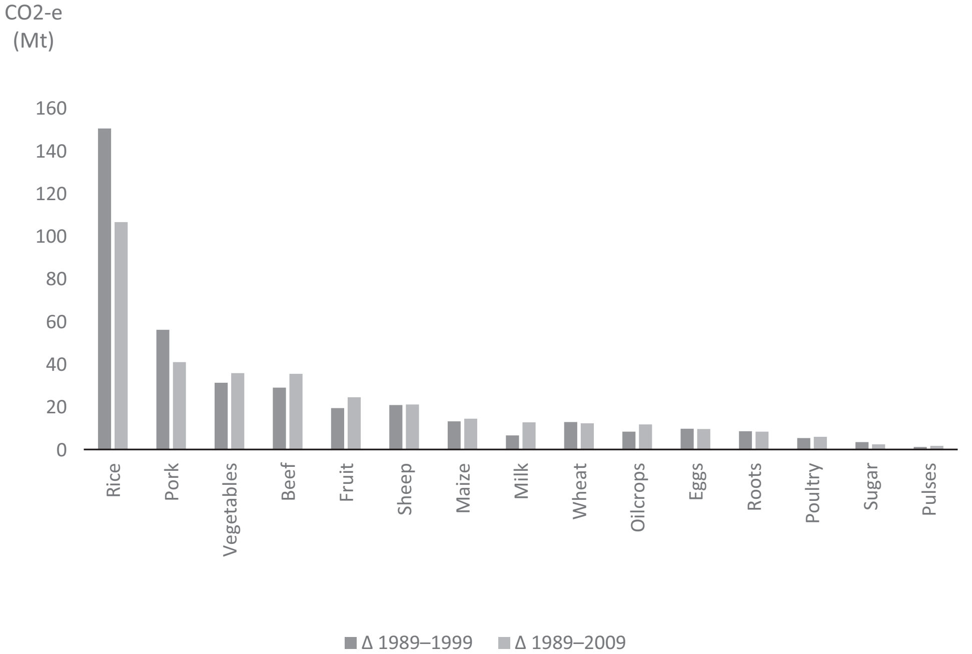

Figure 4 Contributions of scale effects by different food types on CO_2-e emissions between 1989–1999 and 1999–2009.

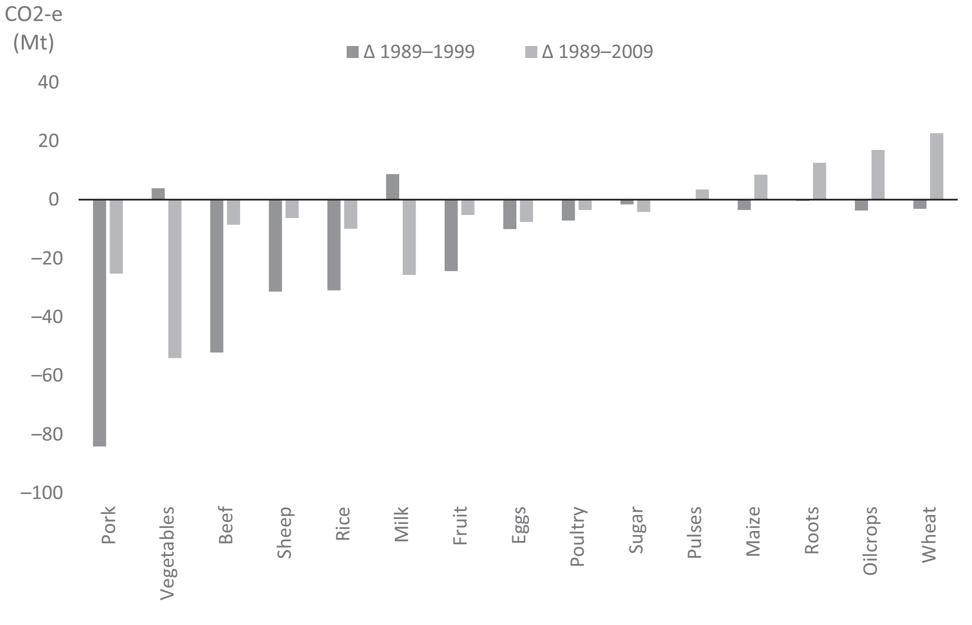

Figure 5 Contributions of efficiency effects by different food types on CO_2-e emissions between 1989–1999 and 1999–2009.

efficiency effect for all livestock products is negative overall for the 20-year study period. The efficiency effect is also negative for vegetables, rice, fruit and wheat. In the case of both vegetables and milk, efficiency effects are initially positive between 1989 and 1999, but then turn negative to decrease between 1999 and 2009, more than offsetting their increases in the earlier decade. By contrast, the efficiency effects of pulses, maize, roots and tubers, oilcrops, and sugar are all negative from 1989 to 1999, ranging from −0.2 to −3.7 Mt, before growing positive between 1999 and 2009, increasing up to 22 Mt of CO_2-e. Overall, however, in conjunction with the other effects identified in this study, the efficiency effect reduces emissions between 1989 and 2009 by 41 per cent.

4. Discussion

Comparing the food consumption quantities with their corresponding GHG emissions in Table 1, the efficiency effects for products are readily apparent. China produces almost half of the world's pork supply (Zhou *et al*. 2012). Although the OECD-FAO projects that China's pork production will slow (OECD-FAO 2013), Table 1 shows steady growth in pork production between 1989 and 2009, yet also indicates that CO_2-e emissions resulting from pork production remain nearly flat over the same period. The efficiency effect is further demonstrated through the other categories of livestock products included in this study, though not to the magnitude demonstrated with pork. While the demand structure effect for pork is small relative to other livestock products, overall increases in consumption reflected in the scale effect are responsible for 97 Mt of GHGs, but is offset by improvements in pork production methods which result in efficiency effects on the order of 109 Mt of CO_2-e in the 20-year study period. The demand structure effect for beef, which has been highlighted as a particular concern with regard to GHGs (Steinfeld *et al*. 2006), is notably the highest among all food categories but also shows the second greatest efficiency effect in limiting CO_2-e.

Among crops, the efficiency effect is also notable for sugar which increases in consumption by 87 per cent over the study period but decreases 4 per cent of associated CO_2-e emissions. Fruit and vegetables also saw consumption outpacing their GHG emissions, being in the top five categories for both scale and demand structure effects, but demonstrating strong reductions in CO_2-e emissions from the efficiency effect, particularly with vegetables becoming increasingly efficient in the period between 1999 and 2009. Rice-related emissions decrease by 4 per cent due to decreased relative demand reducing the demand structure effect and improved production efficiency counterbalancing overall scale effect increases. Many of the increases in consumption can be attributed to the removal of price controls on nonstaple food in the early 1980s (Fuller and Dong 2007), while reductions in emissions are largely a result of innovation in production techniques and the development and adoption of more efficient varieties of crops and livestock (Lin *et al*. 2014; Li *et al.* 2016). The efficiency effect is not demonstrated for roots, pulses, maize and most notably wheat, which show consumption increases of 8 per cent but increased CO_2-e emissions of 65 per cent.

The reductions in CEFs across most food categories hold back increases in CO_2-e levels from 1989 to 2009 by 59 per cent, while the overall quantity of food products consumed increases by over 100 per cent. The structural demand differences in the types of products the Chinese population consumes, however, contribute little to these totals. Similarly, the source of China's food supply, be it domestically produced or produced overseas and imported to China plays an even smaller role in China's food-related GHG emissions. The reciprocal with regard to the role that China's imported food plays on the CO_2-e emissions of its food supplying nations may not be true, however. Exports from New Zealand to China, for example, increase fourfold between 1989 and 2009. This growth represents New Zealand's exports to China growing from 3 per cent of its total agricultural exports by weight to 11 per cent of its agricultural exports and the resultant GHG emissions growing from 1.6 per cent to 10 per cent of all New Zealand's food-related GHGs. For a country where agriculture is responsible for 48 per cent of the nation's GHGs (Beukes *et al.* 2011), this is not a trivial contribution. While China's imported food represents only 4 per cent of its food consumption-related CO_2-e emissions in 2009, in absolute terms, this is 57 Mt of CO_2-e, slightly less than all of the 2010 CO_2-e emissions generated in the country of Denmark (WRI 2015).

As shown in Figures 1 and 3, China's increased consumption of beef, dairy, fresh fruits, and vegetables, and reduced consumption of rice is reflected in China's GHG emissions, representing almost 17 per cent of the food-related growth between 1989 and 2009. That said, the overall increases in food consumption are the primary driver in the massive increases in China's GHG emissions through scale effects as illustrated in Figures 1 and 4. Without the efficiency effect of technological improvements in food production shown in Figures 1 and 5, increases in the overall scale of food consumption contribute an estimated 719 Mt of CO_2-e to the atmosphere in 2009 over 1989 levels, more than the total of 2010 CO_2-e emissions for Canada. Improvements in production methods and transportation substantially reduce the carbon intensity of China's food supply and are responsible for the reduction in 325 Mt of CO_2-e in 2009, roughly the equivalent of the 2010 CO_2-e emissions for the entire country of Nigeria (WRI 2015). This pattern of increases in GHGs driven by increased consumption with minor contributions from structural changes being partially offset by efficiency improvements, while specific to China in this study, reflects larger overall trends in consumption-related emissions as demonstrated by Arto and Dietzenbacher (2014).

5. Conclusion

China is the world's foremost producer of GHG emissions, and agricultural production plays a major role. Additionally, the magnitude of China's agricultural production has expanded as its population becomes more wealthy and urbanised. With this, China's taste in food has shifted with sharply increasing demands for meat, dairy, fresh fruit and vegetables. Although demand has increased for the more GHG intensive food, the CO_2-e emissions generated by these sectors have grown at just under half the rate of production. Technological efficiencies, most notably the transition from traditional lower yield Chinese varieties to higher yield varieties, are reducing the carbon intensity of agricultural products. Technological efficiency plays a massive role in containing the growth of GHG emissions associated with food consumption, offsetting the effects of an increasing scale of production by more than 40 per cent.

It is unclear, however, how much more efficient these agricultural production systems can become. The diminishing marginal returns are becoming apparent in food products' decreasing CEFs. As demand for products continues to increase, however, the limitations of land, water and other inputs will likely result in increases in the carbon intensities of food as more energy is used in water conveyance and irrigation and increased fertiliser use to make up for the loss of arable land. While the scale and structure of the agricultural production can be influenced by economic and sociocultural tools, as wealth continues to increase, demand for products will also continue to increase. While the population, both in China and worldwide, is being encouraged to eat less meat to reduce GHG emissions, it is uncertain how socially and economically feasible this solution may be in the near term.

Our results indicate for China, however, that reduction in the overall amount consumed is nine times more effective than changing the structure of the diet for reducing CO_2-e emissions. Similarly, improvements in the technological efficiency of food production are four times more effective in reducing CO_2-e emissions than making structural changes to food consumption. This study acknowledges that energy substitutions and composition of the energy inputs into agriculture affect the GHG intensity of food products. In the absence of this, however, reductions in consumption, potentially through reductions in food waste, and encouraging the development of agricultural production technologies to reduce carbon intensities and improve efficiency appear to be the most cost-effective solution to curbing GHG emissions associated with China's food consumption. The Chinese government has already made agricultural production and food security a top priority in recent years. With increasingly limited resources, an agricultural policy approach focused on reducing waste and more efficient use of food is necessary to restrain growing CO_2-e emissions.

While the FAO data utilised in these analyses are maintained for comparability and coherence, conclusions drawn from this methodology should be made with awareness of the limitations and reliability of the data. In addition to questions of credibility of the food consumption data, this study is also limited by the assumption of uniform adoption of the IPCC guidelines as a means to adequately measure GHGs across nations. With the development of higher-resolution world input–output tables, it would be worthwhile to compare the results of this methodology against a multiregion input–output analysis of agricultural emissions to see how the results contrast.

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