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Revealing the Hidden Health Costs Embodied in Chinese Exports

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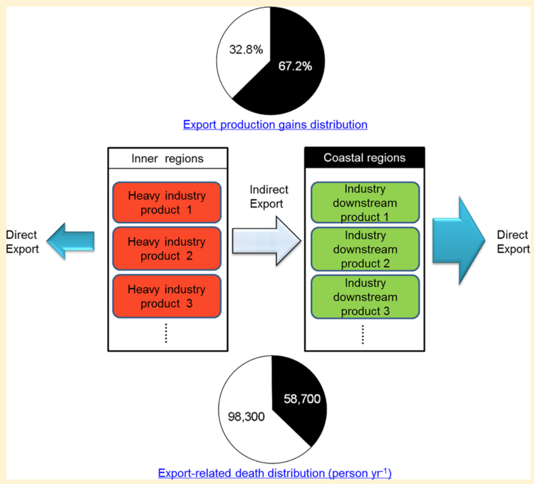
S *Supporting Information*

ABSTRACT: China emits a considerable amount of air pollutants when producing goods for export. Previous efforts have emphasized the magnitude of export-related emissions; however, their health consequences on the Chinese population have not been quantified. Here, we present an interdisciplinary study to estimate the health impact of export-related air pollution. The results show that export-related emissions elevated the annual mean population weighted $PM_{2.5}$ by 8.3 $\mu g/m^3$ (15% of the total) in 2007, causing 157,000 deaths and accounting for 12% of the total mortality attributable to $PM_{2.5}$-related air pollution. Compared to the eastern coastal provinces, the inner regions experience much larger export-related health losses relative to their economic production gains, owing to huge inter-regional disparities in export structures and technology levels. A shift away from emission-intensive production structure and export patterns, especially in inner regions, could significantly help improve national exports while alleviating the inter-regional cost-benefit inequality. Our results provide the first quantification of health consequences from air pollution related to Chinese exports. The proposed policy recommendations, based on health burden, economic production gains, and emission analysis, would be helpful to develop more sustainable and effective national and regional export strategies.



1. INTRODUCTION

Over the last 20 years, China's gross domestic product (GDP) has grown at an average rate of 10% annually, resulting in a dramatic increase in development status. However, this economic development has coincided with a surge of air pollutant emissions. From 1990 to 2005, the emissions of SO_2 and NO_x increased over 2-fold.[1,2] In 2013, China suffered extreme levels of air pollution. Heavy smog blanketed northern and southern China in January and December, affecting more than 600 million people[3] with China's air pollution being a frequent subject of international media attention. The primary component of this smog, fine particulate matter ($PM_{2.5}$), which includes sulfates, nitrate, black carbon, organics, and trace metals, is closely associated with impacts on cardiovascular and respiratory morbidity and mortality.[4−6] According to the Global Burden of Disease (GBD) 2010, 1.2 million annual deaths in China are attributable to exposure to $PM_{2.5}$ in outdoor air, representing 37% of the total global annual mortality induced by air pollution.[7]

After China's entry into the World Trade Organization (WTO), Chinese exports have grown rapidly, at an annual rate of 20% over the past ten years.[8] This significant expansion has provided a robust driving force for China's economic growth, but increasingly, it also heightened tensions between environmental and economic development. Previous studies have

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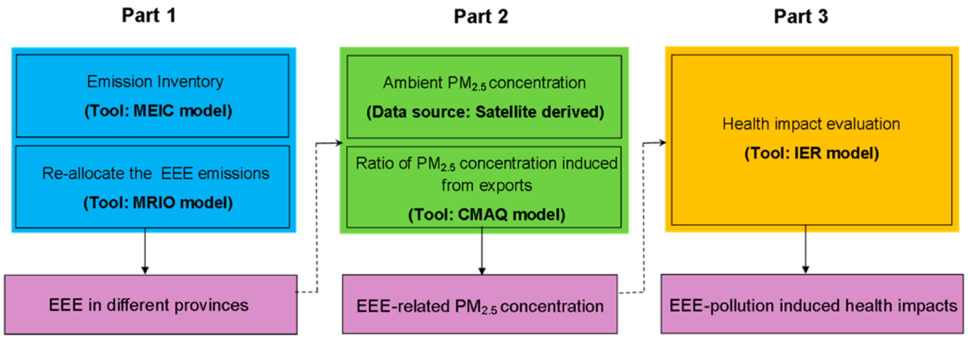

Figure 1. Methodology framework to assess health losses from export production. The framework consists of three parts: estimates of emission embodied in exports (EEE) by province (part 1), calculation of EEE induced $PM_{2.5}$ concentration (part 2), and EEE-related health impact evaluation (part 3). The MEIC, MRIO, CMAQ, and IER models represent the Multiresolution Emission Inventory for China, Multiregional Input−Output Model, Community Multiscale Air Quality Model, and Integrated Exposure−Response Model, respectively.

shown that from 2004 to 2008, more than 30% of CO_2 emissions in China resulted from export production.[9−13] Guan et al.[14] investigated the emission drivers of primary $PM_{2.5}$ and found that, from 1997 to 2010, exports were the only final demand entity that continuously drove emission growth. Streets et al.[15] reported that 37% of SO_2, 28% of NO_x, and 8% of VOC emissions in the Pearl River Delta were linked to export related activities, responsible for 27%, 24%, and 7% of regional SO_2, NO_x, and VOC concentrations, respectively. Moreover, a recent study by Lin et al. investigated the pollution embodied in China's exported and imported goods between 2000 and 2009; they found Chinese exports to be responsible for a maximum of 34% and 23% of China's sulfate and black carbon concentrations in 2006, respectively, along with large impacts on nitrogen oxides, carbon monoxide, ozone, and primary organic carbon.[13] Given these studies demonstrating important effects of exports on Chinese air pollution and the large impacts of air pollution exposure on health in China, we sought to conduct the first quantification of the health consequences of the export-induced $PM_{2.5}$ pollution across China. While some previous studies have explored the health impacts of exports in America[16] and Asia,[17] to date, there has not been a specific focus on China, the world's largest exporter.

Further, since Chinese exports also exhibit extreme regional disparities,[18] arising from the imbalance of economic development and inequality of resources distribution, it is important to assess spatial variability in consequent air pollution and its health burden. For example, eastern coastal developed regions hold over 60% of the value of Chinese exports, but heavy industries are primarily located in less-developed regions such as central, northern, and western China. Moreover, recent studies have discovered that emission intensive production has been and continues to be shifted from China's eastern region to its central and western regions.[19−21] For instance, Zhao et al.[22] estimated that, in 2007, 22 Gg of primary $PM_{2.5}$, 45.8 Gg of SO_2, and 29.8 Gg of NO_x were generated in the central regions of China when producing intermediate products for coastal area's export. While these shifts may have resulted in reduced population exposures to air pollution, they also suggest the potential for disparities in economic benefits and health damages.

We therefore developed an integrated assessment approach to estimate the health impacts of export-relevant $PM_{2.5}$ pollution over China. Through analysis of the export structure and economic interconnection among different sectors and regions, the mortality distribution among the 30 provinces were discussed. Finally, we developed policy recommendations to sustain national and regional export strategies while minimizing pollution-related health impacts.

2. METHODOLOGY AND DATA

The methodology is built upon a cross-cutting interdisciplinary framework. Figure 1 depicts our methodology framework in which four models are linked in an integrated assessment: (1) A Multiregional Input−Output Model (MRIO), describing the interconnection among different economic sectors and regions, is used to calculate export-related emissions from different provinces. (2) We then apply the Community Multiscale Air Quality Model (CMAQ) to derive spatially varying ratios of ambient $PM_{2.5}$ concentrations induced by exports from two scenario runs. (3) These ratios are then multiplied by satellite-derived surface $PM_{2.5}$ concentrations to estimate $PM_{2.5}$ exposure levels caused by exports. (4) Finally, we utilize the integrated exposure−response model (IER) for concentration−response (C−R) functions of $PM_{2.5}$ to examine the health impacts of export-induced $PM_{2.5}$ pollution. This approach is described in more detail in the following sections.

2.1. Emission Embodied in Export (EEE) Calculation. The EEE calculation method for different provinces has been documented in our previous publication[22] (also provided in Supporting Information). In brief, we used the MRIO model, which links the intermediates (also used for export commodities) to the original production site, to conduct an environmentally extended input−output analysis, through which provincial EEE levels can be estimated. The MRIO model used in our study was developed by Liu et al.,[23] which consists of 30 provinces and 30 sectors in 2007 (excluding Tibet, Taiwan, Hong Kong, and Macao due to lack of data). This model has been previously used for assessing CO_2 and air pollutant emissions embodied in Chinese domestic and international trade flows for 2007.[22,24]

The EEE calculation was based on the Multiresolution Emission Inventory for China (MEIC).[1] MEIC is a unit/technology-based, bottom-up air pollutant emission inventory developed by Tsinghua University, which covers 10 pollutants and greenhouse gases (SO_2, NO_x, CO, NMVOC, NH_3, CO_2, $PM_{2.5}$, PM_{10}, BC, and OC) from more than 700 emission sources and production categories. Methodologies and data on which the MEIC is established have been described in our previous studies.[25−28]

2.2. EEE Related $PM_{2.5}$ Concentration Simulation. The EEE related $PM_{2.5}$ concentration was determined by integrating

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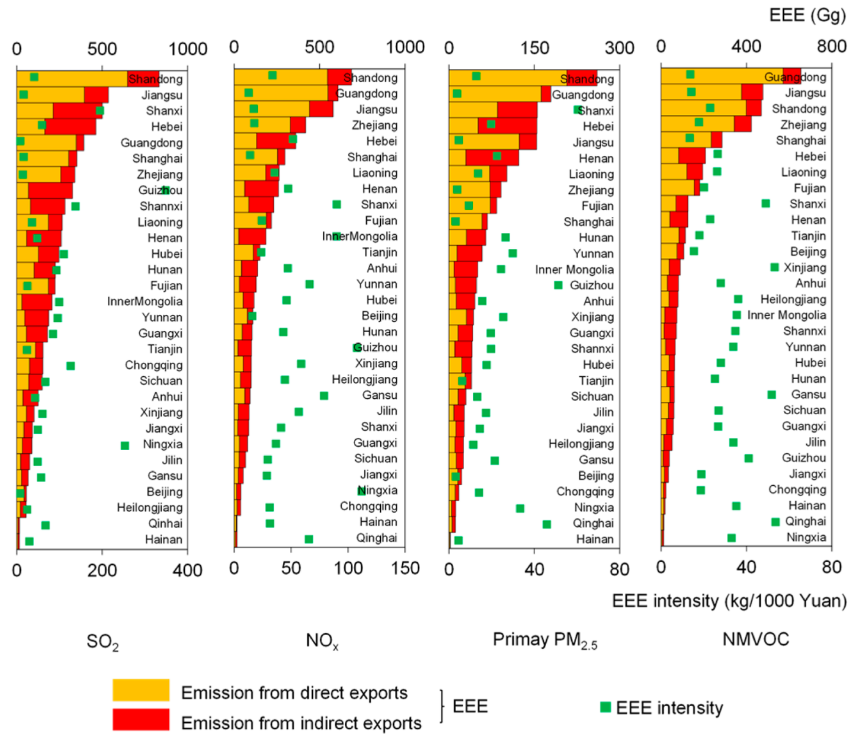

Figure 2. Emission embodied in exports (EEE, upper axis for bars) and their intensities (EEE/production gains, bottom axis for dots) in different provinces for 2007. The EEE consists of two parts: direct and indirect export emissions. Direct export emissions result from the production of goods and services shipped directly out of the country, and the indirect export-related emissions arise from goods and services production that is used to support the direct export in the same and/or different regions.

the CMAQ simulation with satellite-based $PM_{2.5}$ concentrations at 36 km horizon resolution. We used the CMAQ model to obtain the spatially varying $PM_{2.5}$ ratios by running the model both with and without EEE contributions. The EEE related ambient $PM_{2.5}$ concentrations were then calculated by multiplying these ratios to the satellite-retrieved $PM_{2.5}$ concentration. The procedure for calculating EEE-induced $PM_{2.5}$ concentrations can be found in Supporting Information.

Satellite-based $PM_{2.5}$ concentrations were estimated from our latest research,[29] which provides a detailed presentation of the satellite $PM_{2.5}$ concentration retrieval. In brief, aerosol optical depth (AOD) derived from satellite and conversion factors between AOD and $PM_{2.5}$, simulated by the chemical transport model, were utilized to calculate ground-level $PM_{2.5}$ concentrations. AOD retrievals were generated by MODIS[30] and MISR[31] instruments onboard the Terra satellite and were then combined after the AOD ground measurement filtration.[32,33] Temporally and spatially variable $PM_{2.5}$-AOD relationships were estimated using the nested-grid GEOS-Chem model v9-01-02 over China at a resolution of $0.5° \times 0.667°$ (http://geos-chem.org) and the MEIC model. The satellite-based surface $PM_{2.5}$ concentrations are developed at a resolution of 0.1° lat × 0.1° long and then regridded to 36 km to match the horizon resolution of CMAQ model. We took the average $PM_{2.5}$ concentration for three years (2006−2008) to represent the annual mean concentration for 2007 to reduce errors due to insufficient samples in satellite data. The satellite-derived $PM_{2.5}$ concentrations used in our study show good spatial agreement with ground-level measurements ($r = 0.85$, slope = 1.17), which is critical for accurate health impact estimation.

2.3. Health Impact Estimation. Ambient and EEE-related $PM_{2.5}$ concentrations estimated as described above were applied to integrated exposure response functions to estimate mortality impacts (eq 1). The IER functions developed by Burnett et al.,[34] incorporating data from cohort studies of ambient air pollution, and second-hand and active tobacco smoke, are used to describe the concentration−response relationship throughout the full distribution of ambient $PM_{2.5}$ concentrations, including the high levels present in China.[34] This approach is used to estimate air pollution health impacts at high levels, in the absence of epidemiologic studies of the effects of long-term exposure to $PM_{2.5}$. Given its applicability to a wide range of $PM_{2.5}$ concentrations, the GBD project employed these functions to estimate the global mortality due to ambient particulate and household air pollution in 2010.[7] Following the GBD approach, mortality was estimated for four leading causes of deaths: ischemic heart disease (IHD), chronic obstructive pulmonary disease (COPD), stroke, and lung cancer (LC). These four specific diseases share the same health impact function (as shown in eq 1), while the parameters vary with different diseases (Table S1, Supporting Information).[34]

The relative risk (RR) for mortality estimation was calculated as,

$$RR(C) = \begin{cases} 1 + \alpha(1 - e^{-\gamma(C-C_0)^{\delta}}), & \text{if} C > C_0 \\ 1, & \text{else} \end{cases} \quad (1)$$

where C is the annual $PM_{2.5}$ concentration obtained in the previous sector; C_0 is the counterfactual concentration; α, γ, and δ are parameters used to describe the shape of the concentration−response curve as presented in Table S1, Supporting Information.

The RR was then converted to the attributable fraction (AF), which is defined as,

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$$AF = \frac{RR - 1}{RR} \quad (2)$$

$$E = AF \times B \times P \quad (3)$$

Health outcomes attributable to $PM_{2.5}$ were then estimated using eq 3,[35] in which B is the national-level incidence of a given health effect in 2007[36,37] and P is the size of exposed population taken from the LandScan global population database for 2006 at 1 km resolution.[36] It should be noted that, because provincial baseline incidence data were limited, we applied national-level incidence information to all provinces.

The nonlinear shape of the IER function for certain specific health impacts (most notably IHD and stroke) renders the contribution of one source highly dependent on the absolute value (C). That is, a larger change in risk would be observed at lower concentrations.[38] Since we are not able to know where the EEE-related pollution occurs along the IER nonlinear curve, we reasonably assume that the concentrations associated with EEE pollution contribute equally throughout the distribution of concentrations and accordingly average the RRs which can be mathematically represented as[38]

$$\mathrm{ave\,RR}(x_j) = \frac{1}{N}\sum_{i=0}^{N-1}\frac{\mathrm{RR}(i\eta + \Delta_j)}{\mathrm{RR}(i\eta)} \quad (4)$$

$$\eta = \frac{(x_j - \Delta_j)}{N - 1} \quad (5)$$

where x_j is the absolute $PM_{2.5}$ level to be evaluated; N is the sampling number; i is the integer from 0 to $N - 1$; η is the increment at which concentrations of $PM_{2.5}$ are to be evaluated; Δ_j is the $PM_{2.5}$ concentration associated with EEE pollution. In this study, we set N as 1000.

Additionally, to serve as a comparison, the simpler direct proportion of burden approach was also employed to determine the health effects of EEE-relevant pollution. Further details about this method are presented in Supporting Information.

3. RESULTS

3.1. EEE and EEE Intensity. The EEE consists of two parts: direct and indirect export emissions. Direct export emissions result from the production of goods and services shipped directly out of the country, and the indirect export-related emissions arise from goods and services production that is used to support the direct export in the same and/or different regions. In 2007, China emitted 2.0, 4.3, 5.7, and 7.0 Tg of primary $PM_{2.5}$, NMVOC, NO_x, and SO_2,[22] respectively, when producing goods and services that were ultimately consumed outside of the country. Although direct export emissions contribute more than 50% of the EEE, emissions from indirect exports also reached a noticeable share, that is, 45.0%, 38.5%, 40.0%, and 31.4% of total export-related SO_2, NO_x, $PM_{2.5}$ and NMVOC emissions, respectively. Further details on Chinese direct and indirect exports are provided in the Supporting Information.

Figure 2 depicts provincial EEE and their intensities (EEE/economic production gains, details about economic production gains are provided in Supporting Information) for 2007. Scaling down the emissions to the provincial level, we notice that provincial EEE correlate with local export-related production gains (Figure S1, Supporting Information). Coastal and northern regions such as Shandong, Guangdong, and Jiangsu are in the position of being both the largest beneficiaries and emitters, followed by the northeastern and central regions, which both profit and emit less. Western regions such as Qinghai get both the lowest production gains and the lowest EEE volumes. The emission share for indirect exports varies greatly across provinces, with coastal areas generally representing the lowest and western areas showing the highest contributions. Especially high shares are observed in Inner Mongolia, Guizhou, and Shannxi provinces, all above 70% for four air pollutants. More than 50% of the exports in central and western regions are indirect exports (Figures S2 and S3, Supporting Information) and, moreover, the bulk of which stem from energy and emission intensive industries such as metals and chemicals, leading to an extraordinarily high share of indirect emissions in these regions.

EEE intensity (mass/economic value) distributions, however, exhibit a different pattern from those of EEE. Central (e.g., Hubei and Anhui) and western regions (e.g. Qinghai, Ningxia, Guizhou) display the highest emission intensity whereas eastern regions show the lowest. Export structure plays a critical role here. Exports in coastal areas are primarily supported by high-value-added products and downstream goods within the industry chain such as electronic and electric equipment and machinery (Figure S1, Supporting Information). For example, roughly 30% of total exports in Guangdong, Shanghai, and Jiangsu originate from these industries. In contrast, inner areas (northern, northeastern, central, and western) produce more low-value added but high energy and emission intensive goods (Figure S1, Supporting Information). In particular, 40% of exports in Hunan and Guizhou originate from metal smelting and pressing and chemical industries. Emission-intensive export structure determines the high EEE levels in the inner areas. Moreover, in addition to the export structure, significant disparities in production and emission control technologies are another crucial reason for the huge discrepancy of EEE intensity. Technologies that support exports actually rely on the local technology level. Previous studies conducted by Liu et al.[20] have shown a huge disparity in regional technology by comparing the sectoral CO_2 emission intensities among different provinces. Additionally, end-pipe emission control technology usage (e.g. flue gas desulfurization and selective catalytic reduction) is expected to be much higher in developed eastern regions relative to developing inner regions, further exaggerating the disparities.

3.2. EEE-Related $PM_{2.5}$ Pollution. Through CMAQ simulations, we estimated the proportion of ambient $PM_{2.5}$ concentrations attributable to EEE pollution. $PM_{2.5}$ concentrations arising from EEE pollution were then calculated by multiplying these proportions by the satellite-derived surface $PM_{2.5}$ concentrations. Figure 3A illustrates the distribution of EEE-induced $PM_{2.5}$ concentration over China. At the national level, we estimate that approximately 15% of population-weighted mean ambient $PM_{2.5}$ (8 $\mu g/m^3$) arises from EEE pollution, elevating the population weighted annual mean $PM_{2.5}$ concentration from 56 to 64 $\mu g/m^3$ in 2007. More than 1.1 billion people, or 90% of China's total population, are affected by pollution attributable to EEE (Figure 3B). In particular, the fraction of people living in areas with annual average $PM_{2.5}$ concentrations over 35 $\mu g/m^3$ (the interim target established by the World Health Organization (WHO)) increased by 10% due to the role of EEE. Coastal, central, and northern areas, which are home to 0.9 billion people, are the most affected regions; the annual average $PM_{2.5}$ concentrations attributable to

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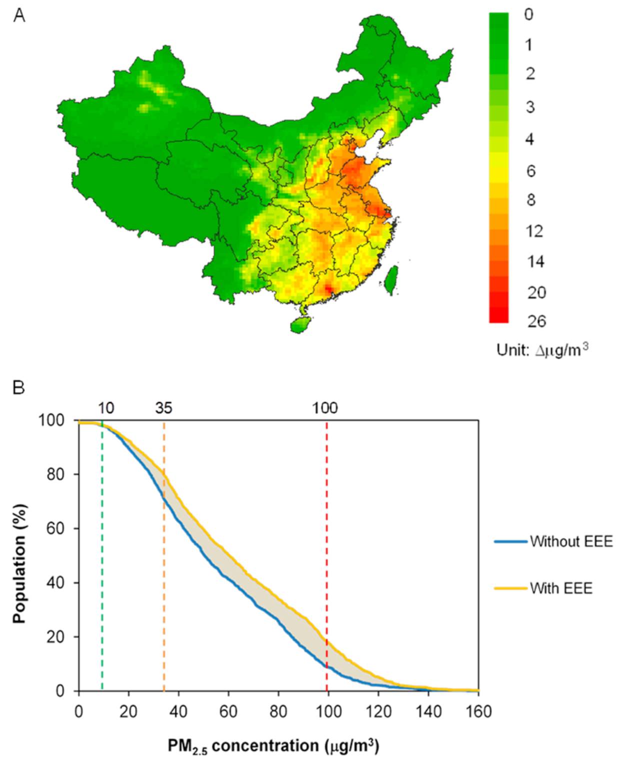

Figure 3. Distribution of EEE-induced $PM_{2.5}$ concentrations ($\mu g/m^3$). The horizontal spatial resolution is 36 km × 36 km (A). A comparison of national cumulative distributions of $PM_{2.5}$ exposures with and without considering EEE volumes; the gray area represents excess $PM_{2.5}$ exposure due to EEE pollution. The vertical lines help guide the eye to identify the cumulative exposures at specific $PM_{2.5}$ concentrations. Ten $\mu g/m^3$ is the $PM_{2.5}$ concentration set in the WHO's Air Quality Guideline and 35 $\mu g/m^3$ is the interim target 1 (IT-1) established by the WHO (B).

EEE in these areas range from 10 to 26 $\mu g/m^3$. For some highly populated areas, such as the Jing-Jin-Ji (Beijing, Tianjin, and Hebei) region, the Yangtze River Delta, and the Pearl River Delta, export goods production-related emissions were responsible for local population weighted mean $PM_{2.5}$ concentrations of 13, 15, and 12 $\mu g/m^3$, respectively.

3.3. Health Losses. We estimate that EEE-related $PM_{2.5}$ is responsible for 157,000 deaths per year, accounting for 12% of the annual mortality resulting from $PM_{2.5}$-related air pollution in China (Table S3, Supporting Information). Specifically, 74,000 deaths (EEE related death/total death, 11.9%) were caused by stroke, 33,000 deaths (12.7%) by COPD, 35,000 deaths (12.3%) by IHD, and 15,000 deaths (12.2%) by LC. Figure 4A displays spatial distribution of mortality across China. As shown, deaths triggered by EEE-related pollution occurred throughout the country. In agreement with the $PM_{2.5}$ concentration distribution pattern and the distribution of population density, the eastern and southeastern regions suffered the most (see Figure 3A).

A comparison of regional mortality and economic production gains (Figure S1, Supporting Information) shows remarkably imbalanced distributions across China. For example, export-related production gains in Guangzhou are over 20 times higher than in Hunan while mortality is only 2.3-fold greater in Guangzhou. This issue would be more pronounced if we separate China into six regional areas. Classifications for these six regions are provided in Supporting Information. As shown in Figure 4B,D, the coastal areas suffered 37% (57,000) of all deaths and obtained 67% of export-related production gains from Chinese exports; in contrast, the central regions obtain a small proportion of overall export-related production gains, 8%, but account for nearly 30% of all deaths (47,000).

To further examine the inter-regional disparities in export-related mortality, here, we define health burden intensity as the health losses for a unit of export-related production gains. From the health perspective, the intensity of health burden quantitatively reflects the quality of regional exports; that is,

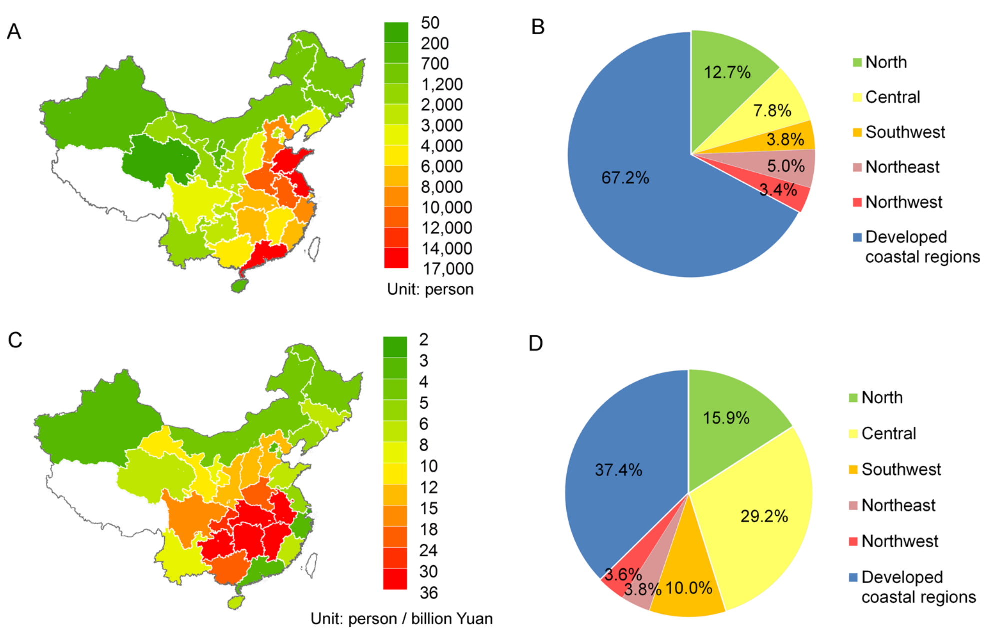

Figure 4. Distribution of EEE-induced mortality (A) and health burden intensities (C). The fraction of economic production gains (B) and deaths attributable to EEE pollution in the six integrated regions (D). Data are not available for Tibet and Taiwan. The constitution of the six integrated regions can be found in Table S4, Supporting Information.

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fewer health losses as a unit of production gains indicates higher quality exports. In coastal areas, thanks to optimized export structure (Figure S1, Supporting Information) and more advanced technologies, health burden intensities are considerably lower, reflecting relatively high quality of exports. In contrast, exports in inner areas (central, western, and northern) are primarily from energy- and emission-intensive products, and moreover, such technologies tend to be less advanced. With the large population, the health burden intensities turn into extremely high levels for some regions such as the central areas (e.g. Anhui, Hunan, and Hubei) which are home to more than 350 million (~30%) people in China (Table S4, Supporting Information).

It is worth noting that a large quantity of goods from heavy industry manufactured in central and western developing regions is not directly exported to the international market (Figure S1, Supporting Information). Instead, numerous products are exported indirectly through the domestic industry chain as essential raw materials for subsequent manufacturing.[20,22] For example, more than 75% of the goods produced in the metals smelting and pressing industry produced in Guizhou is used to feed high-value added and relatively greener commodities such as electric equipment and machinery, which are the major industries in the coastal area. In other words, considerable production gains attained in eastern developed regions through exports are at the expense of health conditions in less-developed central and western regions, rendering the mortality levels in inner areas more than their "fair share" (i.e., export-related production gains), which reveal the extreme regional inequality.

4. DISCUSSION

Recently, China has engaged in considerable efforts to combat air pollution; however, little attention has been paid to the specific role of export embodied emissions (EEE).[14] In 2007, Chinese exports contributed to 15% of the ambient $PM_{2.5}$ and 12% of the mortality attributable to $PM_{2.5}$-related air pollution. The EEE contributions were estimated to be responsible for a population weighted mean $PM_{2.5}$ concentration contribution of 8.3 $\mu g/m^3$ and 157,000 deaths annually. While we focus this paper specifically on quantification of the health impacts attributable to air pollution arising from exports, we also note that, during the same period of increasing emissions and economic development, China's health profile has also improved dramatically.[39]

Although we have used state-of-the-art methods to estimate EEE-pollution related mortality in China, there are a number of assumptions and limitations inherent to our estimates. First, uncertainty comes from the heath impact models. The IER can model health impacts over a wide range of concentrations, enabling us to capture C−R relationships at higher concentrations across China. Meanwhile, the IER models also introduce uncertainties arising from the model itself and assumptions on which the model is based. Uncertainty of the IER models has been statistically estimated by using a simulation approach described in Burnett et al.,[34] which is based on uncertainty in several components of the relative risk function. In addition, IER models are derived from epidemiologic data of cohort studies of ambient air pollution and second-hand and active tobacco smoke and follow the assumption that health effects are independent of $PM_{2.5}$ composition and exposure periods. Although there is evidence of varying toxicity for particles from different sources (e.g., Seagrave et al.[40] and Franklin et al.[41]), this equitoxicity principle has been the basis for numerous international assessments, such as the WHO Air Quality Guidelines, the International Agency for Research on Cancer Monograph, and the Global Burden of Disease. In addition, we assume that EEE contributed equally throughout the IER curve, which may introduce another uncertainty. To provide a point of comparison, we applied the direct proportion approach[38,42] to examine EEE-related health impacts. The number of deaths is estimated to be 170,000 (Table S3, Supporting Information), which is slightly higher (8%) but remains close to the value derived from the average approach. Second, uncertainty also originates from satellite based annual mean $PM_{2.5}$ concentration (11.5 $\mu g/m^3$) which has been discussed in our previous study.[29] Third, the uncertainty of EEE lies on the MEIC inventory and the economic input-out analysis. The uncertainty of MEIC has been quantified by many previous studies, and the value is estimated to be 12% for SO_2, 31% for NO_x, 68% for NWVOC, and 107% for $PM_{2.5}$ (±95% confidence intervals).[25,27] The uncertainty in input-out analysis is much smaller (<50%) (see detailed discussion in Lin et al.[13]). Fourth, simulations of chemical models like CMAQ are unavoidably affected by errors in the representation of meteorology and chemistry; however, the impact on our pollution attribution may be much reduced since we only used the modeled relative contribution of EEE-related $PM_{2.5}$ to total $PM_{2.5}$ pollution.

We realize that the seriousness of EEE pollution does not necessarily suggest only restraining exports to reduce the overall pollution. Instead, we advocate developing Chinese exports in cost effective ways. In this study, we introduce an important index, health burden intensity, to quantify the quality of regional exports. On the basis of our results, ratios of health loss and export-related production gains are extremely imbalanced across China, exhibiting coastal regions with low and inner regions with high distribution patterns, with particularly high levels observed in the central and southwestern China. To improve the overall quality of Chinese exports while shrinking this regional inequality, regional exports upgrading in inner areas (e.g., central, western, and northern areas) is crucial. Meanwhile, pollution from both indirect and direct exports in inner areas should be equally and adequately addressed.

For pollution arising from direct exports in inner areas, export structural adjustment acceleration is likely the best solution. Heavy industries dominate regional exports in central and western China. Metals and chemicals, labeled as high energy- and emission-intensive goods, are staple commodities for exports. Heavy industry exports place serious health burdens on central regions. Therefore, limiting highly intensive but low value added exports becomes an urgent issue in inner regions, especially for the central areas in which nearly 30% of the total Chinese population reside. Nevertheless, such adjustments may be retarded under present pollution control policies. Almost all of the currently established environmental control targets focus on the developed eastern regions while placing loose restraints in inner areas. For example, the Action Plan for Air Pollution Prevention and Control (Action Plan), known as the toughest plan in Chinese history, set critical $PM_{2.5}$ control objectives for Jing-Jin-Ji region, Yangtze River Delta, and Pearl River Delta, to lower the $PM_{2.5}$ concentrations by 25%, 20%, and 15%, respectively; however, the goal for the rest of regions (i.e. western and central China) is only a 10% reduction.[43] Another example is the energy intensity reduction

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policy, which also set a stricter goal for eastern developed regions than for inner less developed regions.[44] Looser pollution-control targets would tempt increasing high-polluting industries to move into the inner areas, especially into the west of China. Under such circumstance, reliance on the heavy industry in inner areas might be even intensified with enormous environmental and health consequences.

In addition to structural adjustment for direct exports, it is imperative to pay more attention to the pollution from the indirect exports. China recently has devoted efforts to export optimization. However, simply upgrading direct exports will only partially limit health effects in inner areas: most of the EEE in these regions (more than 60%) result from indirect causes. Therefore, upgrading technologies throughout the domestic industry chain, and especially among upstream segments that are mostly located in central regions, would significantly improve national and regional export quality (e.g., western, northern, and central). In turn, regional inequality between the coastal and inner areas could also be narrowed. To achieve this objective, developed coastal regions should more actively help the inner regions upgrade their local technologies.[19] Export-related production gains achieved in developed eastern regions are partially based on health losses in less-developed central and western regions via shifting their emission intensive products to inner areas through the domestic supply chain.[45] Central and western regions are lagging economically and cannot afford technological innovations needed to mitigate pollution. Thus, establishing an effective collaborative framework between coastal and inner areas to technologically and financially support research and development (R&D) efforts in inner regions would help improve the overall quality of Chinese exports while shrinking regional inequality.

■ ASSOCIATED CONTENT

Supporting Information

Additional details on methodology and data; full description of economic production gains embodied in Chinese exports; fit parameters for IER models (Table S1); the Chinese EEE (Table S2); the number of deaths calculated with and without EEE pollution conditions in China (Table S3); specific information about the six integrated regions (Table S4); the provincial economic production gains from exports and their sectoral contribution (Figure S1); the share of indirect exports by sector in each province (Figure S2); the fraction of indirect exports in each province (Figure S3). This material is available free of charge via the Internet at http://pubs.acs.org.

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Notes

The authors declare no competing financial interest.

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