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Impacts of rural worker migration on ambient air quality and health in China: From the perspective of upgrading residential energy consumption

Huizhong Shen[a,b], Yilin Chen[a,b], Armistead G. Russell[b], Yongtao Hu[b], Guofeng Shen[c], Haofei Yu[d], Lucas R.F. Henneman[b], Muye Ru[a], Ye Huang[a,e], Qirui Zhong[a], Yuanchen Chen[a,f], Yufei Li[g], Yufei Zou[h], Eddy Y. Zeng[i], Ruifang Fan[j], Shu Tao[a,*]

[a] Laboratory for Earth Surface Processes, College of Urban and Environmental Sciences, Peking University, Beijing 100871, China
[b] School of Civil & Environmental Engineering, Georgia Institute of Technology, Atlanta, GA 30332, USA
[c] Oak Ridge Institute for Science and Education (ORISE) postdoctoral fellow at U.S. Environmental Protection Agency, RTP, NC 27709, United States
[d] Department of Civil, Environmental and Construction Engineering, University of Central Florida, Orlando, FL 32816, USA
[e] Laboratoire des Sciences du Climat et de l'Environnement/Institut Pierre Simon Laplace, Commissariat à l'Énergie Atomique et aux Énergies Alternatives–CNRS–Université de Versailles Saint-Quentin, Université Paris-Saclay, F-91191 Gif-sur-Yvette, France
[f] College of Environment, Zhejiang University of Technology, Hangzhou, Zhejiang 310014, China
[g] School of Public Policy, Georgia Institute of Technology, Atlanta, GA 30332, USA
[h] Earth and Atmospheric Sciences, Georgia Institute of Technology, Atlanta, GA 30332, USA
[i] School of Environment, Jinan University, Guangzhou, Guangdong 510632, China
[j] Key Laboratory of Ecology and Environmental Science in Guangdong Higher Education, School of Life Science, South China Normal University, Guangzhou, Guangdong 510631, China

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ABSTRACT

In China, rural migrant workers (RMWs) are employed in urban workplaces but receive minimal resources and welfare. Their residential energy use mix (REM) and pollutant emission profiles are different from those of traditional urban (URs) and rural residents (RRs). Their migration towards urban areas plays an important role in shaping the magnitudes and spatial patterns of pollutant emissions, ambient $PM_{2.5}$ (fine particulate matter with a diameter smaller than 2.5 μm) concentrations, and associated health impacts in both urban and rural areas. Here we evaluate the impacts of RMW migration on REM pollutant emissions, ambient $PM_{2.5}$, and subsequent premature deaths across China. At the national scale, RMW migration benefits ambient air quality because RMWs tend to transition to a cleaner REM upon arrival at urban areas—though not as clean as urban residents'. In 2010, RMW migration led to a decrease of 1.5 μg/m^3 in ambient $PM_{2.5}$ exposure concentrations (C_{ex}) averaged across China and a subsequent decrease of 12,200 (5700 to 16,300, as 90% confidence interval) in premature deaths from exposure to ambient $PM_{2.5}$. Despite the overall health benefit, large-scale cross-province migration increased megacities' $PM_{2.5}$ levels by as much as 10 μg/m^3 due to massive RMW inflows. Model simulations show that upgrading within-city RMWs' REMs can effectively offset the RMW-induced $PM_{2.5}$ increase in megacities, and that policies that properly navigate migration directions may have potential for balancing the economic growth against ambient air quality deterioration. Our study indicates the urgency of considering air pollution impacts into migration-related policy formation in the context of rapid urbanization in China.

1. Introduction

Rapid urbanization in China has attracted hundreds of millions of people into cities during the last three decades (National Bureau of Statistics of the People's Republic of China, 2014; Yang et al., 2013). Among these newly-settled migrants, growing numbers of rural migrant workers (RMWs) have received much attention from the government and the public (Ru et al., 2015; Shi, 2008; Wong et al., 2007; Zhao, 1999). RMWs are individuals registered with rural identity (in the Chinese household registration system) and employed in urban

Abbreviations: RMWs, rural migrant workers; REM, residential energy use mix; $PM_{2.5}$, fine particulate matter with a diameter smaller than 2.5 μm; C_{ex}, ambient $PM_{2.5}$ exposure concentration calculated as ambient population-weighted $PM_{2.5}$ concentration; URs, urban residents; RRs, rural residents; EF, emission factor

* Corresponding author.
E-mail address: taos@pku.edu.cn (S. Tao).

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workplaces (Wong et al., 2007; Zhang and Shunfeng, 2003). Statistics from the 6th National Census indicated that the number of RMWs in China was 138 million in 2010, or 40% of the urban working population (National Bureau of Statistics of the People's Republic of China, 2016). 58% of the RMWs are within-province who work in the provinces where they are registered; the remaining 42% are trans-province RMWs who migrate from one province to another. The preferred destinations of trans-province RMWs are large (population between 1 million and 10 million) and mega- (population greater than 10 million) cities with more job opportunities and relatively high levels of income. In five Chinese megacities (Beijing, Tianjin, Shanghai, Guangzhou, and Shenzhen), one-third of their urban population on average is RMWs. The percentage is the highest in Shenzhen, where RMWs account for 60% of its population (National Bureau of Statistics of the People's Republic of China, 2016).

RMWs have contributed substantially to recent economic growth in China (Shi, 2008). However, they remain marginalized in urban society (Meng and Zhang, 2001; Shi, 2008; Wong et al., 2007). Most RMWs live in privately built shanties in urban villages (Feng et al., 2002; Ru et al., 2015). The registration system and their minimal wages block their access to social welfare and urban energy infrastructures (Chan and Zhang, 1999; Chan, 2010; Ru et al., 2015). Differences in living conditions between RMWs and registered urban residents (URs) who hold an urban identity lead to disparities in residential energy use mix (REM) (the amount and structure of direct energy consumed for daily residential cooking, heating, lighting, and household appliance operation) (Shen et al., 2017). Residential emissions have been found to be one of the major sources of various air pollutants (Huang et al., 2014; Lei et al., 2011; Ohara et al., 2007; Shen et al., 2013; Wang et al., 2012). Ru et al., 2015 reported that RMWs in Beijing tend to use more coal for cooking and heating than registered URs because natural gas and centralized heating systems are less accessible (Ru et al., 2015). Consequently, per-capita pollutant emissions of RMWs are different from those of URs, though their contributions to air pollution have been poorly understood. The inflow of RMWs into urban regions also leads to large-scale relocation pollutant emissions (Shen et al., 2017), thereby reshaping the geographic distributions of both pollutants emissions and population exposure. Elucidating the impacts of RMWs' migration on ambient air quality in urban and rural areas is critical for addressing public health concerns from air pollution exposure in the context of rapid urbanization in China (Gong et al., 2012).

In our previous study, we investigated the impact of overall population migration on ambient air quality in China with a focus on residential and transportation sectors (Shen et al., 2017). All migrants, comprised of RMWs and newly-registered URs, were included. Our results suggested that migration is favorable for reducing ambient national $PM_{2.5}$ concentrations, as migrants generally shift to cleaner REMs after settling in cities (Shen et al., 2017). Here, by incorporating more detailed data, we further examine the impacts of RMW migration alone. Given the low vehicle ownership among RMW population, we assume little impacts of RMW migration on the transportation source and focus solely on residential emission sources. We use a chemical transport model (Grell et al., 2005) and the Integrated Risk Function (Burnett et al., 2014) to evaluate the impact of RMW migration on ambient $PM_{2.5}$ concentrations and quantitatively assess the cross-province health impacts due to RMW migration.

2. Methods

2.1. County-to-county RMW migration data

Our previous study established a dataset containing the geographic distribution of RMW at the 1-km spatial resolution for the year of 2010 (Supplementary Fig. S1) (Shen et al., 2017). On the basis of this dataset, here we derived detailed origin and destination information from census data to characterize population movements. Data from the 6th National Census (National Bureau of Statistics of the People's Republic of China, 2016) were used in this study: the county-level census data that covers the entire population and the province-level long-table sampling survey data that covers 10% of the total population. The county-level census data provides information for each county, on the numbers of the migrants who migrated within this county (herein referred as *NM1*), migrated from other counties but within the same province (*NM2*), and migrated from other provinces (*NM3*). The long-table survey data classified all migrants into four groups: rural-to-urban migrants (i.e. RMWs in this study), urban-to-urban migrants, rural-to-rural migrants, and urban-to-rural migrants. For each group, the data provided a province-to-province population movement matrix (National Bureau of Statistics of the People's Republic of China, 2016). We integrated the information from these two datasets and derived county-to-county RMW migration matrix containing quantitative and directional information on RMW migration for the year 2010 (Supplementary Text and Supplementary Data).

2.2. RMWs' REM consumption data

Due to the lack of RMWs' REM data from official statistics, we conducted two-stage questionnaire surveys (comprised of community and intercept stages) to determine RWMs' REM. The questionnaire survey framework was summarized in Supplementary Fig. S2. The questionnaire covers information on personal and family (including year of arrival, family size, family members, and income), residence information (including current address, original address, housing type, and housing expenditure), and energy use (including energy types and expenditures for cooking and heating, possession of electricity appliances, and total expenditure on electricity). A sample of the questionnaire is listed in Supplementary Fig. S3 showing the detailed information on various energy types being investigated. These energy types were then aggregated to six major types comprised of coal, liquefied petroleum gas (LPG), natural gas, electricity, heat, and biomass based on a price-to-energy conversion following the method of a previous study (Ru et al., 2015). The community stage is to distribute questionnaires within study cities, and the intercept stage is to conduct face-to-face interviews to validate the representativeness of the survey (see Supplementary text and Supplementary Fig. S2 for survey framework and sample selection criteria). Previous studies found a decrease in life expectancy due to sustained exposure to air pollution from centralized heating in northern cities in China (Chen et al., 2013; Zhang and Smith, 2007). Thus, we collected questionnaires among RMWs in Beijing and Guangzhou, representing northern (with centralized heating) and southern (without centralized heating) China, respectively, to determine REMs for RMWs. A total of 1700 questionnaires were distributed during the community stage with 526 valid questionnaires being collected. Our intercept surveys collected 491 valid questionnaires through face-to-face interviews. Comparison between REMs derived from the two survey stages confirmed the sample representativeness (Supplementary Fig. S4). Combining the two stages, a total of 1017 questionnaires were used for data analysis. The origins of the surveyed population showed a similar pattern to that reported in the sixth National Census in both cities (Supplementary Fig. S5). For other provinces, we estimated RMWs' REMs based on the compositions of RMWs' original provinces and the origin-specific REMs obtained from the surveys. Provinces with and without district heating were adjusted separately using data collected in Beijing and Guangzhou, respectively. Detailed information on the survey conducted in Beijing can be found in a previous study (Ru et al., 2015). For registered urban (URs) and rural residents (RRs), their corresponding REMs were directly derived from the government statistics by province (Ministry of Agriculture of the People's Republic of China, 1997–2008; Energy Statistics Division of National Bureau of Statistics, 1986–2013). This study focuses on REM consumption, whereas direct and indirect emissions from other sectors such as transportation and construction sectors may be also affected by

RMWs, which warrants further investigation.

2.3. Evaluation of the RMW migration impacts on pollutant emissions, ambient $PM_{2.5}$ concentrations, and premature deaths

We used an integrated model framework, following our previous study (Shen et al., 2017), to investigate the impacts of RMWs on pollutants emissions, ambient $PM_{2.5}$ concentrations, and premature deaths in 2010. Detailed information on model configuration and the comparison of modeled $PM_{2.5}$ concentrations with observations (Supplementary Fig. S6) are provided in Supplementary materials. Based on the collected REM information and emission factors (EF, pollutant emission amount per unit fuel consumed) derived from the EF database of PKU Inventory (Wang et al., 2012; Huang et al., 2014; Huang et al., 2017), we developed residential emission inventories for various PM-related pollutants including primary PM components (organic carbon (OC), black carbon (BC), $PM_{2.5}$, particulate matter with a diameter of 10 μm or less (PM_{10}), and total suspended particulate matter (TSP)) and secondary PM precursors (sulfur dioxide (SO_2), nitrogen oxides (NO_x), ammonia (NH_3), and non-methane volatile organic compounds (NMVOCs)). Emissions for the three groups of RMWs, URs, and RRs were evaluated separately. Using the REM emission inventories together with emissions from other sources (Shen et al., 2014) as model inputs, the Weather Research and Forecasting model coupled with Chemistry (WRF/Chem) version 3.5 (Grell et al., 2005) was integrated with a postprocessing Gaussian downscaling approach (Shen et al., 2014) to generate near-surface $PM_{2.5}$ (primary + secondary) concentration maps over China at a 5 km × 5 km spatial resolution. $PM_{2.5}$ exposure concentrations (C_{ex}, calculated as the ambient population-weighted $PM_{2.5}$ concentration) were then calculated based on spatial distributions of $PM_{2.5}$ concentrations and population (Shen et al., 2017). Annual deaths from ischemic heart disease, cerebrovascular disease (stroke), chronic obstructive pulmonary disease, lung cancer, and acute lower respiratory infection attributable to long-term inhalation exposure to ambient $PM_{2.5}$ were evaluated based on modeled C_{ex}, provincial background disease burdens (Supplementary Tables S1-S5) (IHME, 2016; NHFPC, 2015), and the new Integrated Exposure-Response (IER) model (Cohen et al., 2017). The new IER functions use age-specified dose-response relationships to estimate relative risks of stroke and ischemic heart disease. Compared to the old version (Burnett et al., 2014), the new IER curves are less sensitive to $PM_{2.5}$ changes for older-age groups and/or at higher $PM_{2.5}$ levels. As a result, estimated premature deaths using the new IERs are likely 10%–20% lower than those using the old ones (Cohen et al., 2017).

We performed simulations for three scenarios (a base scenario, a non-migration scenario, and a policy intervention scenario). Each scenario covered a 1-year period. The base scenario used the established actual emission inventory to simulate $PM_{2.5}$ concentrations for the year of 2010. The non-migration scenario used a counterfactual emission inventory whereby all RMWs were relocated back to the locations (in the rural area) where their rural identities were issued on the basis of the established county-to-county migration matrix. RMWs' REMs were changed back to rural REMs accordingly. We thus identified the impacts of RMW migration on pollutant emissions, ambient $PM_{2.5}$ concentrations, and premature deaths by directly comparing the differences between these two scenarios.

A policy intervention scenario was developed that had the spatial distribution and per-capita energy amounts of RMWs unchanged from the base scenario, but shares of individual fuel types were shifted to those of URs' (Supplementary text). Comparison between the base and policy intervention scenario allowed for a quantitative assessment of the health benefits from upgrading RMWs' REMs in cities.

It should be noted that our health risk assessment focuses on migration-induced changes in premature deaths of the overall population instead of RMWs themselves. We also calculated the migration-induced changes in premature deaths of RMWs, which revealed that the nation-wide net change in RMW premature deaths is relatively small (as we estimated, an annual increase of 560 in premature deaths) because RMWs are mainly young people who are not sensitive to air pollution exposure (Supplementary text and Supplementary Table S6).

2.4. Evaluation of cross-province health impacts of RMW migration

On the basis of the non-migration scenario which assumed no migration, we performed a set of 30 simulations. For each simulation, we enabled RMW migration from only one specific province. The 30 simulations correspond to the 30 provinces in the mainland China (excluding Hong Kong, Macau, and Xizang due to their negligible inflows and outflows of RMWs). This set of simulations enabled us to attribute the impacts of RMW migration from one province to all individual provinces. For example, the impact of Sichuan-to-Guangzhou RMW migration on ambient $PM_{2.5}$ concentrations in Guangzhou province was evaluated by comparing the differences in estimated $PM_{2.5}$ concentrations in Guangzhou between the non-migration simulation (assuming no migration) and the simulation with migration from Sichuan province enabled.

2.5. Uncertainty analysis

Uncertainties in emissions, C_{ex}, and premature deaths were addressed whenever possible. EFs were assumed to be lognormally distributed. Their means and standard deviations were directly derived from the database of PKU Inventory (Wang et al., 2012; Huang et al., 2014; Huang et al., 2017). As adopted by previous studies, officially-reported activity rates of REMs (for URs and RRs) were assumed to be uniformly distributed with variation internals being 20% of the means for biofuels (wood, crop residue, and biogas), 5% for electricity, and 10% for all other fuel types (Shen et al., 2013; Bond et al., 2007; Ciais et al., 2010). For RMWs, we enlarged the uncertainty ranges by 50% (30% for biofuels, 7.5% for electricity, and 15% for all other fuel types) to introduce the uncertainty associated with sample representativeness. Uncertainties in emission estimate were then characterized based on Monte Carlo simulation which ran 10,000 times for each fuel type and each pollutant with each time randomly drawing an EF and an activity rate from their given distributions.

Uncertainties in C_{ex} were calculated based on a series of sensitivity simulations which determined sensitivities of ambient $PM_{2.5}$ exposure concentrations to changing emissions of individual compounds including both primary PM compounds and secondary PM precursors (Supplementary Table S7) and transferred uncertainties in emissions to those in modeled C_{ex}. Uncertainties in premature deaths were characterized using Monte Carlo simulation which ran for 10,000 times for each disease by randomly drawing one C_{ex} from its estimated distribution, one set of IER function's coefficients from the 1000 sets of function coefficients provided (Burnett et al., 2014) and one background disease burden from reported uncertainty for each disease (IHME, 2016). Medians and 90% confidence intervals were used to represent the uncertainties in this study.

3. Results

3.1. RMW-induced change in ambient $PM_{2.5}$ exposure concentration

Comparison of the simulated $PM_{2.5}$ fields before and after RMW migration suggest that the RMW migration reduces ambient $PM_{2.5}$ (primary + secondary) concentrations across almost all of China, though leads to local increases in some cities (Fig. 1). The national population exposure concentration (C_{ex}, calculated as the ambient population-weighted $PM_{2.5}$ concentration) decreases from 59.8 μg/m^3 to 58.3 μg/m^3 due to migration. The reduction in $PM_{2.5}$ exposure leads to an annual decrease in premature deaths of 12,200 (90% confidence interval 5700–16,300). C_{ex} levels in rural and urban areas decrease by

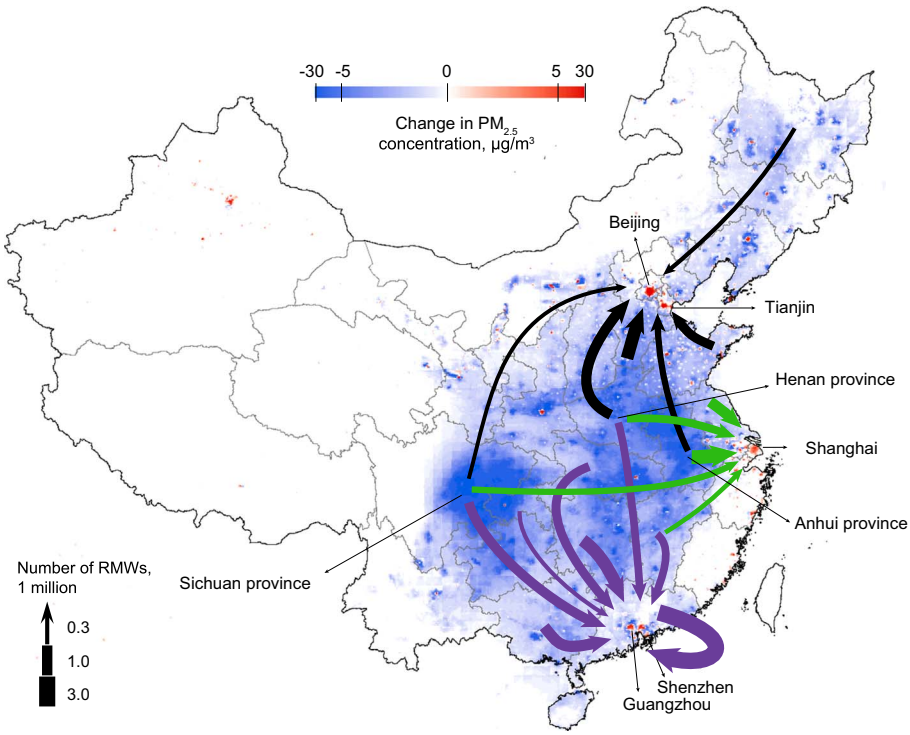

Fig. 1. Impacts of RMW migration on ambient $PM_{2.5}$ concentrations in mainland China in 2010. The migration-induced reductions in $PM_{2.5}$ concentrations were observed in the vast majority of the rural area, in contrast to the substantially increased concentrations in three metropolitan areas (containing five Chinese megacities). Migration to these three metropolitan areas is illustrated using arrows. The widths of the arrows are proportional to the numbers of RMWs. Only migrations with larger than 30,000 RMWs are shown. Due to the China Western Development policy (Lai, 2002), Urumqi, the capital city of Xinjiang province, also showed increased $PM_{2.5}$ concentrations caused by the RMW inflow.

2.2 μg/m^3 and 0.7 μg/m^3, respectively. 54% of the total population see a C_{ex} reduction of more than 1 μg/m^3. Despite the overall reduction, C_{ex} changes differently among provinces. The largest C_{ex} reduction occurs in Sichuan (−7.2 μg/m^3), whereas Beijing and Tianjin show substantial C_{ex} increase (+10.7 μg/m^3 and +9.0 μg/m^3, respectively).

3.2. Factors affecting C_{ex}

The associations between RMW migration and C_{ex} reduction is due to RMWs shifting to cleaner REM after migrating into urban areas, which leads to reductions in regional pollutant emissions (Supplementary Figs. S7 and S8 and Supplementary Table S8). Taking primary $PM_{2.5}$ for example, most RMWs switch from biomass fuels that are associated with high $PM_{2.5}$ emissions, to increasing use of gas, electricity, and centralized heating (in northern China) (Fig. 2). Hence, the average per-capita $PM_{2.5}$ emission of RMWs decreases from 7.7 kg/person to 1.8 kg/person after migration. This change yields a total reduction of 830 Gg per year in national primary $PM_{2.5}$ emissions, leading to a considerable decrease in average C_{ex}. In addition to the emission reduction, C_{ex} is further reduced by the change in source proximity to people—residential emissions from biomass fuel burning are often close to residents, while emissions from electricity and centralized heating typically take place away from residential areas. Spatially, RMWs migration leads to a decrease in regional emissions but an increase in urban local emissions. The change in urban C_{ex} is a combined result of both local and regional emission changes (Supplementary Fig. S9). Our simulation shows that the impacts of regional emission decrease are more pronounced than local emission increase across a vast range of the urban area, resulting in an overall decrease in urban C_{ex}. However, large and mega cities (such as Beijing and Tianjin) that attract large numbers of RWMs and are less surrounded by rural areas show significant increases in C_{ex}. The changing patterns of emissions and C_{ex} are illustrated and discussed in detail in the following sections.

Our questionnaires revealed that despite living in urban areas, RMWs use a REM much different from those of URs—RMWs rely on a larger fraction of higher-emitting fuels (e.g. biomass and coal) (Fig. 2). This difference is possibly caused by inequality of incomes between RMWs and URs. For example, most RMWs in northern China cannot afford to live in an apartment with centralized heating or covered by natural gas pipeline networks. In this case, RMWs often turn to more coal (occupying 49% of the RMW per-capita residential energy consumption on average in northern China) for heating and cooking to compensate for the reduced availability of cleaner fuels. Similarly, in southern China, RMWs still use biomass fuels (14%). As a result, the national average per-capita emissions of $PM_{2.5}$, carbon monoxide, non-methane volatile organic compounds, and ammonia for RMWs are 8%, 11%, 12%, and 54% higher than those for URs, although the latter have higher per-capita energy consumption (11.8 GJ/person for RMWs vs 13.8 GJ/person for URs). The disparity in REMs is particularly important for air quality management in mega cities that attract more RMWs than elsewhere.

The change of emissions varies geographically. For instance, primary $PM_{2.5}$ emissions in rural areas show a reduction of 1070 Gg due to RMW outflow, while an emission increase of 240 Gg is observed in urban areas because of the inflow. Nine western inland provinces, including Sichuan, Anhui, Guizhou, Henan, Hubei, Hunan, Jiangxi, Nei Mongol, and Chongqing have a 510 Gg $PM_{2.5}$ emission reduction, whereas the emission changes in six eastern provinces, including Beijing, Tianjin, Shanghai, Zhejiang, Fujian, and Guangdong, are close to zero on average.

3.3. RMW-induced emission changes by county

It is suspected that the disproportionate distributions of emission changes linked with the dynamic of migration are driven mainly by the unbalanced economic developments among regions. To investigate this assumption, we plotted the emission changes in urban and rural areas against the per-capita gross domestic product (GDP_{cap}) by county (in China, an administrative county typically contains both urban and rural areas) (Fig. 3A and B). The increases in urban $PM_{2.5}$ emissions in individual counties range from 0 to 6.3 Gg/year. Counties with higher GDP_{cap} levels attract more RMWs and thus experience higher urban emission increases. The highest emission increase occurs in Beijing (6.3 Gg/year), followed by Tianjin (3.8 Gg/year), and the emission increases are concentrated in a few most-developed counties. Of all 2373 counties, the 5% with the highest emission increases account for more

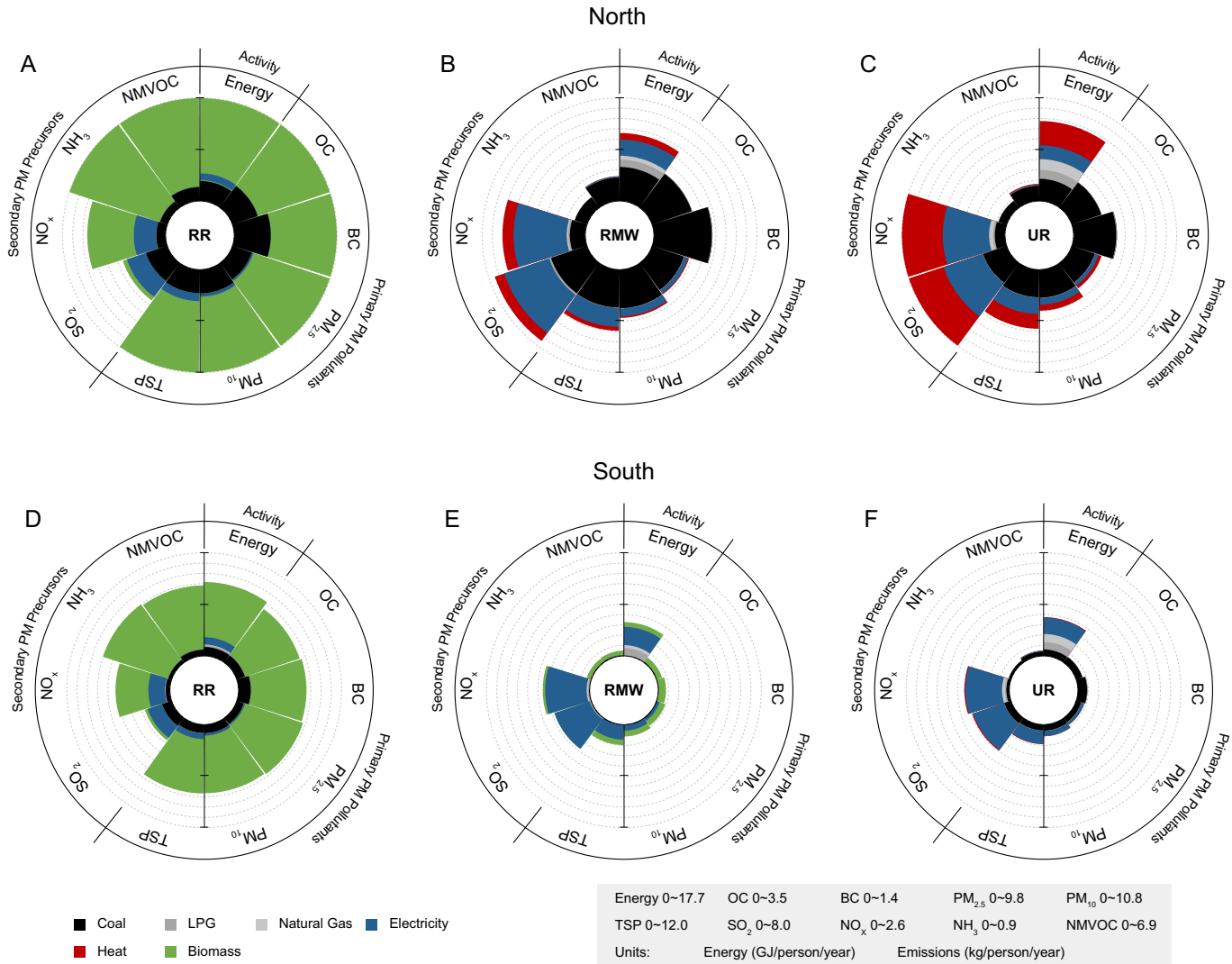

Fig. 2. Per-capita consumption of REMs and per-capita emissions of primary PM pollutants and secondary PM precursors by fuel type and population group. (*A–C*) Regional average per-capita REM consumption and pollutant emissions of RRs (*A*), RMWs (*B*), and URs (*C*) in northern China in 2010. (*D–F*) Regional average REM consumption and per-capita emissions of RRs (*D*), RMWs (*E*), and URs (*F*) in southern China in 2010. Northern China refers to the 15 provinces located north of the Qinling-Mountains-Huaihe-River Line where centralized heating systems are available in urban areas. Southern China refers to the 19 provinces located south of the Qinling-Mountains-Huaihe-River Line where centralized heating systems are not available. The per-capita emissions are illustrated using wind rose plots. Each plot represents the emission profile of one specific population group, whereas each direction is associated with one specific pollutant. Contributions of different fuel types are marked with different colors (as shown in the legend). For different pollutants, the scales along the radial direction are different; the whole ranges of the emission (or consumption) values from the smallest circle to the largest dashed circle are provided in the legend. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

than 50% of the total emission increase. Reductions in rural $PM_{2.5}$ emissions range from 0 to 14.6 Gg/year. Populous counties such as Chengdu (a 14.6 Gg/year reduction), Chongqing (13.0 Gg/year), and Hefei (6.2 Gg/year) show the largest rural emission decreases. Compared with urban emission increases, rural emission reductions are distributed more evenly across China. The 13% of the counties with the largest emission reductions contribute to 50% of the total rural emission reduction. Surprisingly, counties experiencing the highest emission reductions are mostly associated with median-to-high levels of economic development ($GDP_{cap} \sim 4000$ US$). The least-developed counties ($GDP_{cap} < 1000$ US$) show the smallest urban emission increase or rural emission reduction. In fact, according to census data (National Bureau of Statistics of the People's Republic of China, 2016), 45% of RMWs are from counties with GDP_{cap} between 2000 US$ and 6000 US$ which contribute to 40% of the rural emission reduction. This means median-developed regions are currently the largest source of RMWs. Despite socioeconomic factors (e.g. a surplus of labor, a rise in prices, and an awareness of broadening individual vision), there are political reasons leading to the RMW outflow from these regions. In Sichuan, for instance, the local government provides technical training courses and encourages rural people to seek jobs in coastal provinces because RMWs have increased the average income of local farmers and reduced the unemployment rate (China.org.cn, 2016). Consequently, the large RMW outflow leads to a 130 Gg $PM_{2.5}$ emission reduction in Sichuan, contributing to 14% of the total emission reduction in rural areas of all 30 provinces.

3.4. RMW-induced $PM_{2.5}$ exposure concentration changes by county

We modeled the changes in C_{ex} of ambient $PM_{2.5}$ (primary + secondary) due to RMW-induced changes in emissions of primary PM compounds and secondary PM precursors (Fig. 3C and D). C_{ex} reductions are observed in almost all rural areas due to RMW migration, with an average reduction of 2.2 μg/m^3. 56% of all counties show a C_{ex} reduction of more than 1 μg/m^3. The largest reduction occurs in Chengdu (−29.1 μg/m^3) and Hefei (−23.0 μg/m^3). Rural emission

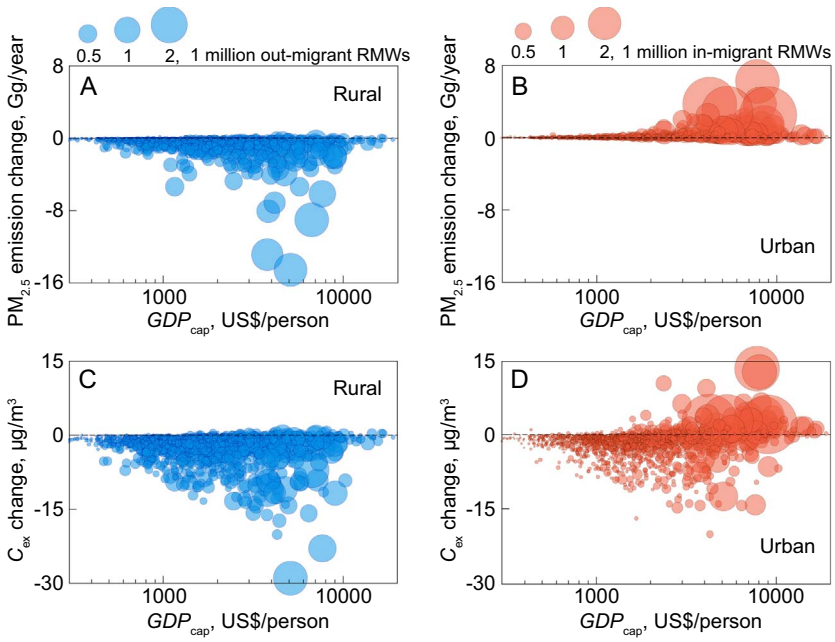

Fig. 3. The relationship between migration impacts and county-level GDP_{cap} in 2010. (*A*) Migration-induced changes in rural primary $PM_{2.5}$ emissions by county. (*B*) Migration-induced changes in urban primary $PM_{2.5}$ emissions by county. (*C*) Migration-induced changes in rural $PM_{2.5}$ exposure concentrations (primary and secondary) by county. (*D*) Migration-induced changes in urban $PM_{2.5}$ exposure concentrations (primary and secondary) by county. The bubbles represent individual counties. The areas of the bubbles are proportional to the numbers of out-migrant RMWs from rural areas in *A* and *C* and the numbers of in-migrant RMWs in urban areas in *B* and *D*. In each county, urban and rural areas share the same GDP_{cap} value.

reductions lead to decreases in urban background concentrations, which essentially offset elevated concentrations caused by increased urban emissions. Consequently, 75% of the counties show a decrease in urban C_{ex}. The urban areas in median-developed counties (with GDP_{cap} between 2000 US$ to 4000 US$) exhibit the largest decrease in C_{ex} (decreasing by 1.2 μg/m^3 on average). However, the most-developed counties (with GDP_{cap} higher than 6000 US$) show an average increase of 0.7 μg/m^3 in C_{ex}, which means the decrease in background $PM_{2.5}$ concentrations is overwhelmed by the urban emission increases in these areas. In particular, the RMW migration yields 13.6 μg/m^3 and 12.9 μg/m^3 C_{ex} increases in the metropolitan regions of Beijing and Tianjin, respectively. One reason for such substantial C_{ex} increases is that, except for massive RMW inflows, Beijing and Tianjin (in northern China) have higher within-city RMW emissions from heating when compared with other mega cities in the south. Further, since they are isolated from rural areas, the background concentrations in these two cities are less affected by rural emission decreases.

According to the model simulation, the impacts of RMW migration are mainly caused by the changes in primary PM emissions. Secondary $PM_{2.5}$ concentrations show much smaller spatial changes in response to RMW migration since emissions of secondary PM precursors from REM sources only contribute small fractions of total emissions. In addition, the transport model omits some key atmospheric chemistry processes associated with aerosol formation (Wang et al., 2016; Cheng et al., 2016; Guo et al., 2017), and thus the modeled secondary $PM_{2.5}$ concentrations tend to be biased low, which may lead to an underestimation of spatial changes in secondary $PM_{2.5}$ concentrations due to migration.

3.5. Health impacts of trans-province RMW migration

Migratory flows of RMWs linking C_{ex} reductions/increases between locations, also link potential health impacts between locations as well. The origin province typically obtains a health benefit due to the C_{ex} reduction, while the destination province typically sees an increase in C_{ex}, indicating a transfer of health disbenefits from origin provinces to destinations. By tracking the relocation of pollutant emissions along the routes of RMW migration, we evaluated the cross-province health impacts of trans-province RMW migration on the basis of a set of 30 simulations with each opening up the RMW outflow from one province (Methods). Here, we used premature deaths as the health indicator (Fig. 4). Among all provinces, Sichuan gains the largest health benefit from trans-province RMW migration. The outflow of RMWs from Sichuan has improved regional air quality, leading to an annual reduction of 2400 premature deaths within the province. However, the inflow of the same RMWs from Sichuan leads to increases in premature deaths (570) in destination provinces, such as Guangdong (160), Fujian (50), Zhejiang (50), Beijing (30), and Shanghai (20). A similar RMW-induced spatial transfer of health disbenefits can be found with consistent directions from the other more populous inland provinces (e.g. Anhui, Hubei, Hunan, and Jiangxi) to the most-developed eastern provinces. The spatial transfer of health disbenefits has thus considerably increased the health burdens on eastern megacities (Figs. 1 and 4).

We calculated the ratios of inflow-induced premature deaths to the number of in-migrant RMWs (R_{in}) for each province (Table 1) and found that R_{in} in Beijing (0.12×10^{-3}) and Tianjin (0.14×10^{-3}) are higher than the national average (0.09×10^{-3}). A higher R_{in} indicates more health disbenefits caused by RMW inflows. The R_{in} of 0.12×10^{-3} in Beijing suggests that an increase of one in-migrant RMWs in Beijing leads to an annual increase of 0.12×10^{-3} premature death associated with ambient $PM_{2.5}$ exposure. Together, the inflow of 4.2 million trans-province RMWs contributes to an annual increase of 500 premature deaths in the capital of China. Correlation analysis revealed a significant positive correlation between R_{in} and provincial heating degree days (HDD) ($p < 10^{-5}$), and, controlled by HDD, a significant positive partial correlation between R_{in} and the urban population density of the provinces ($p < .1$). Linear regression analysis showed that the two variables (i.e. HDD and urban population density) together explain 40% of the R_{in} variation, where higher HDDs and higher urban population densities are associated with higher R_{in} (Supplementary text). This is because higher HDDs (or lower temperature in winter) have higher per-capita REMs consumption for heating, leading to higher per-RMW emissions and consequently higher per-RMW health impacts (R_{in}), whereas a higher urban population density means more people being readily affected by per-unit emission increases and thus are associated with a higher R_{in}. Ambient air quality in Beijing and Tianjin is more sensitive to the increasing inflows of RMWs as compared to other mega cities in the south because of the greater HDDs and population densities. Therefore, from geographic and health perspectives, Beijing and Tianjin are not ideal cities for RMW in-migration but can also benefit most from lowering RMW emissions.

We then calculated the ratio of the outflow-induced premature

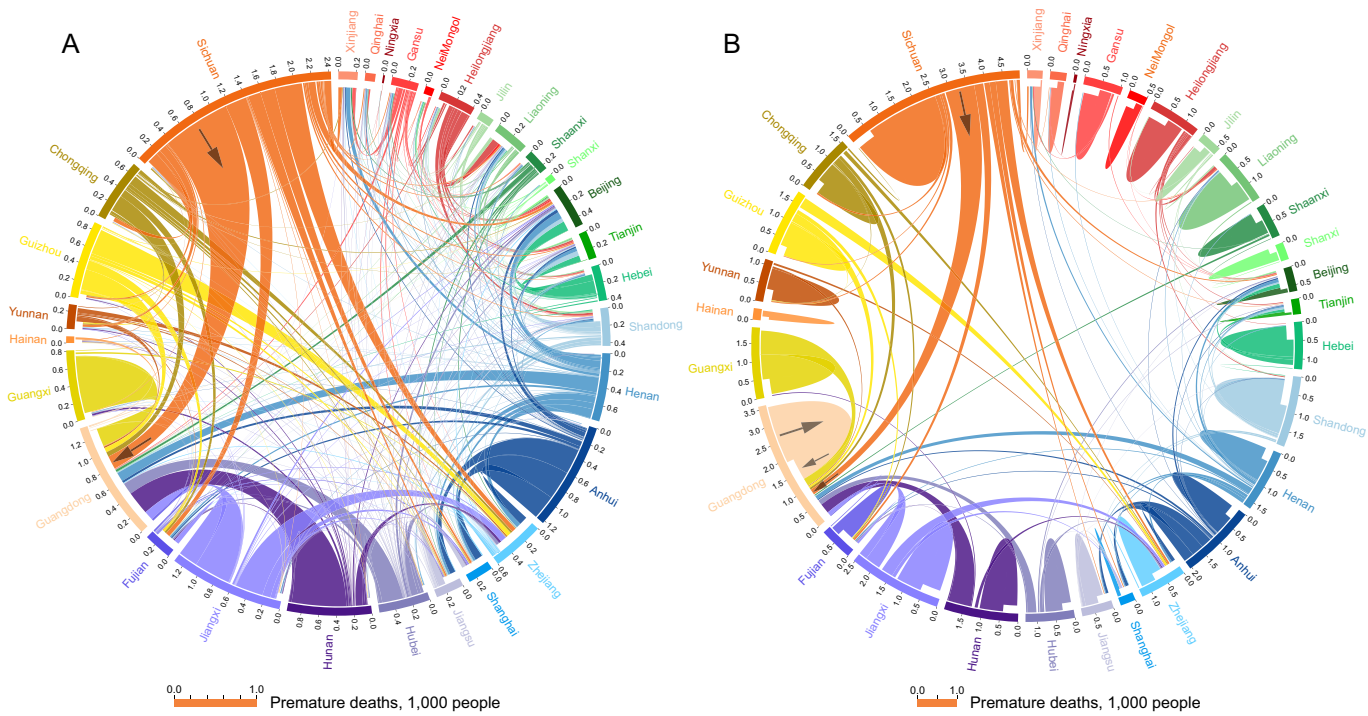

Fig. 4. Cross-province health impacts of RMW migration. RMW migration leads to decreases in $PM_{2.5}$ concentrations and premature deaths in origin provinces but corresponding increases in destination provinces. The circular plots link decreases in air pollution-related premature death in the origin provinces with increases in the destination provinces for individual province-to-province migration flows. The flow directions shown in the plots are encoded by the color of the origin province and a gap between the flow and the province segment. Tick marks show an increased number of deaths due to the inflow of RMWs (with a gap between the flow and the segment) or decreased number of deaths due to outflow (no gap) in thousands. (*A*) Changes in premature deaths caused by trans-province migration. (*B*) Changes in premature deaths caused by both trans-province and within-province migration (only numbers larger than 10 are shown). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

death changes to the number of out-migrant RMWs (R_{out}). The R_{out} in Sichuan is -0.38×10^{-3}, which means, for the actual outflows of 6.4 million RMWs, about 2400 premature deaths are avoided in the origin province. Typically, a higher positive R_{in} is often associated with a more negative R_{out} of the same province (Table 1)—the correlation coefficient r between provincial R_{in} and R_{out} is -0.61 ($p < 10^{-3}$), because they are both subject to population densities and climate conditions of the province (Supplementary text). The health impact of an entire trans-province migration flow is determined by both the origin's R_{out} and the destination's R_{in}. For example, migration of one RMW from Sichuan to Beijing will lead to a net decrease of 0.26×10^{-3} premature death (calculated as R_{in} in Beijing + R_{out} in Sichuan). In contrast, an RMW migrating from Nei Mongol to Beijing will lead to a net increase of 0.05×10^{-3} premature death (the R_{out} in Nei Mongol is -0.07×10^{-3}, a lower absolute value, because of lower population densities across the province). Our quantitative assessment on the provincial R_{out} and R_{in}, as listed in Table 1, allows for easier evaluation of RMW's trans-province health impacts and could benefit policy decisions for properly addressing the current air quality-related health issues raised by RMW migration in China.

3.6. Health impacts of within-province RMW migration and net RMW migration

In addition to trans-province migration mentioned above, the within-province migration also plays an important role in terms of health impacts. The ratio of within-province-migration-induced premature death relative to the number of within-province RMWs (R_{inner}) can be simply calculated as (R_{in} + R_{out}) of the same province. 77% of the Chinese provinces exhibit negative R_{inner} values, suggesting a net decrease in premature deaths due to within-province migration. Correlation analysis revealed a significant positive correlation between R_{inner} and HDD ($p < .01$) and a negative correlation between R_{inner} and total population density ($p < .02$). Such associations indicate that, in those provinces with warmer climates and higher population density, within-province RMW migration tends to cause larger decreases in overall premature deaths. In many provinces, within-province RMW migration reduces C_{ex} in both urban and rural areas because increased urban emissions tend to be countered by substantial rural emission decrease and atmospheric transport. Sichuan gains the largest health benefit—a decrease of 1500 premature deaths from the within-province RMW migration, followed by Guangdong with a decrease of 1000 premature deaths.

Unlike trans-province migration which consistently increases health burdens on destination provinces, within-province migration is beneficial to ambient air quality in most provinces and to a certain extent, depending on the fractions of the rural population in the provinces, compensates for the health disbenefits induced by trans-province migration. For example, among the 25 million RMWs in Guangdong province, one-third are within-province RMWs, which compensates for ~80% of the increased premature deaths caused by trans-province migration. Thus, although Guangdong contains the largest RMW population among provinces, migration only leads to a net increase of 250 premature deaths, which, in terms of per-RMW premature deaths, is substantially less than Beijing who has 4.7 million RMWs but an overall increase of 430 premature deaths (Table 1 and Fig. 4). Further assessments based on Table 1 show that if all RMWs in China were subject to within-province migration, the net health benefits nationwide would remain roughly the same (a decrease of 10,600 premature deaths) as those caused by the actual migration, but the adverse health impacts in the most-developed provinces would be entirely diminished (the migration-induced changes in premature deaths in the actual and the assumed cases are +430 and −60, respectively, in Beijing; +210 and −80 in Shanghai; +250 and −1020 in Guangdong).

Table 1
Numbers of in-migrant, out-migrant, and within-province RMWs and the associated net health impacts on the local population.

Province	RMW_{in} ($\times 10^3$)	RMW_{out} ($\times 10^3$)	RMW_{inner} ($\times 10^3$)	R_{in} ($\times 10^{-3}$)	R_{out} ($\times 10^{-3}$)	R_{inner} ($\times 10^{-3}$)	Deaths ($\times 1$)
Anhui	330	6980	3065	0.10	−0.17	−0.08	−1400
Beijing	4224	33	505	0.12	−0.22	−0.11	430
Fujian	3320	1198	4305	0.08	−0.06	0.02	270
Gansu	225	971	1339	0.31	−0.25	0.06	−90
Guangdong	17,035	350	8262	0.08	−0.19	−0.12	250
Guangxi	464	3223	2971	0.15	−0.23	−0.08	−920
Guizhou	429	2537	2032	0.12	−0.33	−0.21	−1200
Hainan	356	143	644	0.11	−0.25	−0.13	−80
Hebei	702	2219	3251	0.12	−0.15	−0.03	−320
Henan	308	6274	4864	0.06	−0.12	−0.06	−1010
Heilongjiang	234	1278	1663	0.31	−0.27	0.04	−200
Hubei	520	4071	3529	0.07	−0.14	−0.06	−730
Hunan	373	5424	3796	0.10	−0.17	−0.07	−1150
Jilin	204	657	1007	0.21	−0.20	0.01	−70
Jiangxi	294	4255	2415	0.21	−0.29	−0.08	−1380
Jiangsu	4557	1964	4720	0.04	−0.08	−0.04	−150
Liaoning	1041	400	2001	0.25	−0.29	−0.04	70
Nei Mongol	835	500	2744	0.08	−0.07	0.01	60
Ningxia	229	100	449	0.07	−0.07	0.00	7
Qinghai	218	84	386	0.37	−0.47	−0.09	7
Sichuan	494	6353	5368	0.10	−0.38	−0.28	−3870
Shaanxi	540	1264	2517	0.12	−0.15	−0.03	−220
Shandong	1180	2021	5531	0.12	−0.14	−0.02	−250
Shanghai	6285	19	546	0.05	−0.18	−0.14	210
Shanxi	498	596	2653	0.06	−0.09	−0.03	−120
Tianjin	2140	56	265	0.14	−0.24	−0.10	270
Xinjiang	1061	96	780	0.20	−0.05	0.15	330
Xizang	98	14	48	0	0	0.00	0
Yunnan	781	802	2451	0.12	−0.24	−0.12	−400
Zhejiang	7998	1192	4409	0.06	−0.09	−0.03	240
Chongqing	535	2431	2152	0.14	−0.26	−0.12	−810
China	57,509	57,509	80,669	0.09	−0.20	−0.12	−12,200

Note:
Columns 2–4 show numbers of in-migrant (RMW_{in}), out-migrant (RMW_{out}), and within-province (RMW_{inner}) RMWs in individual provinces. Unit: 1000 person.
Columns 5–7 show the R values described in the text.
Column 8 shows the overall changes in provincial premature deaths caused by RMW migration. Unit: person.
Hongkong, Macau, and Taiwan are not listed here due to the lack of migration information.

3.7. Health benefits from improving RMWs' REMs

Although RMW migration has largely reduced ambient $PM_{2.5}$ concentrations in China due to an upgrade of REMs, marked disparities in REMs still exist between RMWs and URs. The relatively "dirtier" REMs among RMWs suggest a potential to further enlarge the migration-induced beneficial effects on air quality across China and can have a particularly large benefit in cities where net disbenefits are found if the RMWs' REMs could be modified to match those of their UR counterparts. To quantitatively assess the beneficial impacts of upgrading RMWs' REMs on urban $PM_{2.5}$ concentrations, we analyzed a policy intervention scenario which assumed the total amounts of RMWs' per-capita energy consumption remained unchanged but the energy structures being shifted to those of URs' (Method, Supplementary text). On the basis of this scenario, we estimate that this energy upgrade for RMWs nationwide leads to a decrease of 119 (69 to 214) Gg in $PM_{2.5}$ emissions or 0.8% (0.5% to 1.5%) of the national total $PM_{2.5}$ emissions. However, urban emission reductions in individual cities could reach up to as high as 10.7% (Shenzhen) (Supplementary Table S9) depending on the magnitude of RMW inflows. Spatially, the emission reduction concentrates within 1.4% of the country's territory where half of the Chinese population reside. Such a strong geographic overlap between emission reductions and population distributions tends to amplify the subsequent impacts on C_{ex} and health. Consequently, the national annual average C_{ex} decreases by 1.5% from 58.2 μg/m^3 to 57.3 μg/m^3, and the urban C_{ex} decreases 2.2% from 63.7 μg/m^3 to 62.3 μg/m^3 with mega cities showing the largest decreases (Fig. 5A). For instance, C_{ex} levels decrease 7.8 μg/m^3 (7.7%) and 6.3 μg/m^3 (6.6%) in Beijing and Tianjin, respectively (Supplementary Table S9). Corresponding to the C_{ex} decrease, 5390 (3500 to 8280) premature deaths associated with air pollution exposure can be avoided annually across China. Cities with higher GDP_{cap} levels and more urban populations tend to benefit more from REM upgrading (Fig. 5B). Five mega cities (Beijing, Tianjin, Shanghai, Guangdong, and Shenzhen) account for 16% of the total reduction in premature deaths (Fig. 5B and Supplementary Table S9). Beijing benefits most among all cities, with 270 premature deaths being avoided annually. We, therefore, suggest that improving RMWs' REMs can be an effective approach to reducing ambient $PM_{2.5}$ in cities, especially those located in most-developed regions attracting large RMW inflows. Still, development strategies regarding RMWs' infrastructure construction should further consider the limited space and resources in Chinese cities and thus calls for a comprehensive cost-effective assessment before fully implemented.

4. Discussion

The migration-induced emission reduction is fundamentally due to the long-term disparity between REMs in rural and urban areas (Shen et al., 2017). In the context of rapid urbanization in China, the RMW migration may accompany with a relatively fast energy transitions from rural to "near urban" REMs. Our quantitative assessment demonstrates that the RMWs-induced transitions to cleaner REMs lead to reduced ambient $PM_{2.5}$ concentrations, overall, but there are regional and more local disparities. Despite consistent RMW-induced emission increases in urban areas, we showed that regional emission reduction induced by energy transitions is sufficient to reduce ambient $PM_{2.5}$ concentrations

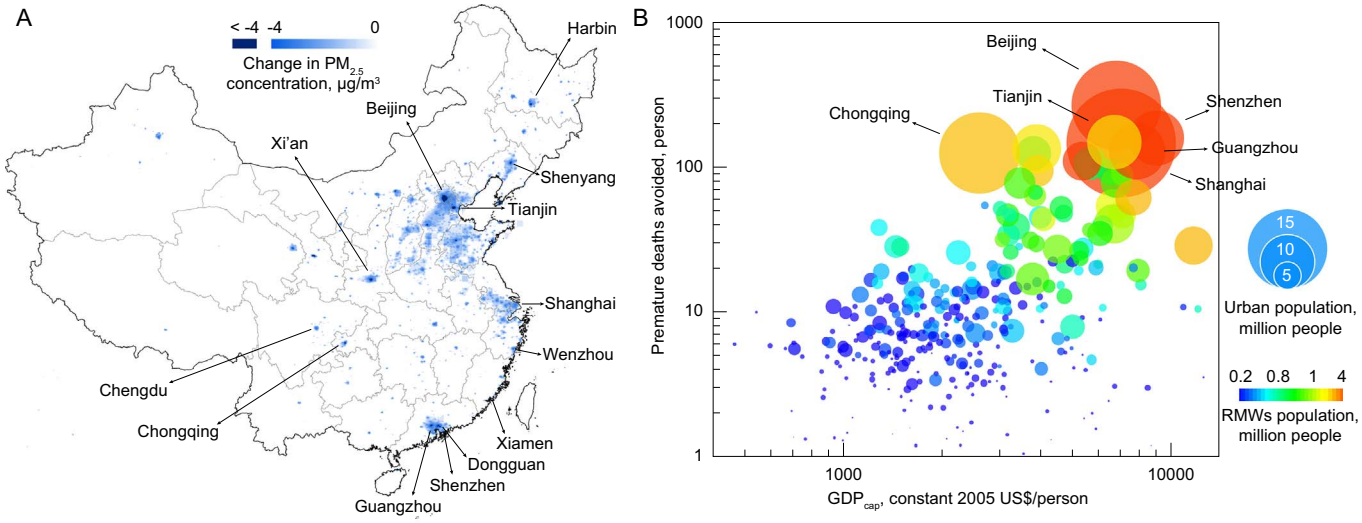

Fig. 5. Potential health benefits from upgrading RMWs' REMs in cities. (*A*) The spatial distribution of $PM_{2.5}$ concentration reduction due to upgrading RMWs' REMs across mainland China. The distribution is calculated as the differences between the policy intervention scenario and the base scenario (Methods). Cities with significant $PM_{2.5}$ reductions are marked on the map. (*B*) The relationship between city-level premature deaths avoided by upgrading RMWs' REMs and the city GDP_{cap} levels. The bubbles represent individual cities. The widths of the bubbles are proportional to urban population in the cities. The colors of the bubbles are associated with RMW population in the cities. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

both in rural (directly due to local emission reduction) and urban (through reduction in background concentrations) areas. The similar beneficial impact on health is applicable to other developing counties where large disparities in REM are observed between urban and rural residents, assuming that cleaner sources are utilized in the more economically developed urban areas (Balachandra, 2011; Barnes and Floor, 1996; Krey et al., 2012).

One of the several issues associated with RMWs in China is the Spring Festival travel rush, which is recognized as the largest annual human migration in the world (BBC NEWS, 2009; Huang et al., 2012). Every year before the Chinese New Year, a vast majority of the RMWs travel back home and reunite with rural families. Basically, this is the inverse of RMW migration. Data from the nationwide monitoring network indicate that, despite the exodus of RMWs from cities, urban air quality deteriorates during the Spring Festival (China National Environmental Monitoring Center, 2016), earning it the epithet "New Year Smog" (Mashable, 2017). Although many during-Festival activities can contribute to the "New Year Smog", including increased emissions from transportation and fireworks (Huang et al., 2012; Leng, 2017), RMW migration and their REM changes may be playing a more important role, since RMWs shift back to rural REMs, leading to a large emission increase from residential sectors.

We found that, in megacities, RMWs' REM emissions play an important role in ambient air quality deterioration and increases the cities' health burdens. This adverse impact would be mitigated if long-range trans-province migration could be partially taken place by within-province migration. For example, local strategies that plan to create more jobs and protecting RMWs' interests and rights (such as health insurance and children's education) can prevent rural labors from migrating to other cities or provinces (Seeborg et al., 2000; Wu and Zhou, 1996)—given comparable job opportunities, most RMWs are willing to work nearby their origins (China.org.cn, 2016). Similar strategies for holding within-province rural labors can not only reduce the health burdens in megacities but also maintain a balanced economic development across regions. Yet, the final formation of these strategies should consider both environmental and economic capacities at a city scale as well as the multiscalar cost-effectiveness on a regional-to-national level (Ramaswami et al., 2016).

RMW migration can lead to an increased health burden on large and mega cities, but also is a unique opportunity for intensively upgrading the REMs of hundreds of millions of people within a confined geographic region. While RMW migration transfers what used to be rural emissions to the most-developed and populous metropolitan area, the movement comes with emission reductions both on the individual and overall basis. The increased exposures in these more densely populated areas can be mitigated, in fact more than offset, by improving RMWs' REM. Energy infrastructure development for RMWs, which are often overlooked in Chinese cities, remains a critical component for air pollution mitigation. Our study also poses the need for gradually eliminating the urban-rural differences and regional disparities in REMs which is fundamental to reducing REM pollutants emissions and improving regional air quality.

5. Conclusions

We find that, from the perspective of upgrading REMs, RMW migration is beneficial to ambient air quality and population health on a national level but have regional disparities. Long-term strategies that can properly navigate RMW migration and improve RMWs' living conditions via upgrading REMs are expected to support both economic development and air quality improvement in China and allow for sustainable development in the increasingly urban future.

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Competing interests

No competing interests.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envint.2017.11.033.

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