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Estimating household air pollution exposures and health impacts from space heating in rural China

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ABSTRACT

Exposure to and the related burden of diseases caused by pollution from solid fuel cooking, known as household air pollution (HAP), has been incorporated in the assessment of the Global Burden of Diseases (GBD) project. In contrast, HAP from space heating using solid fuels, prevalent in countries at middle or high altitudes, is less studied and missing from the GBD assessment. China is an ideal example to estimate the bias of exposure and burden of diseases assessment when space heating is neglected, considering its remarkably changing demands for heating from the north to the south and a large solid-fuel-dependent rural population. In this study, based on a meta-analysis of 27 field measurement studies in rural China, we derive the indoor $PM_{2.5}$ (fine particulate matter with an aerodynamic diameter smaller than 2.5 μm) concentration for both the heating and non-heating seasons. Combining this dataset with time-activity patterns and percentage of households using solid fuels, we assess the population-weighted annual mean exposure to $PM_{2.5}$ (PWE) and the health impacts associated with HAP in mainland rural China by county for the year 2010. We find that ignoring heating impacts leads to an underestimation in PWE estimates by 38 μg/m^3 for the nationwide rural population (16 to 40 as interquartile range) with substantial negative bias in northern provinces. Correspondingly, premature deaths and disability-adjusted life years will be underestimated by approximately 30×10^3 and 60×10^4 in 2010, respectively. Our study poses the need for incorporating heating effects into HAP risk assessments in China as well as globally.

1. Introduction

Increased evidence suggests that $PM_{2.5}$ (fine particulate matter with an aerodynamic diameter smaller than 2.5 μm) exposure from household solid-fuel use is associated with an increase in the risk of cardiovascular and pulmonary diseases in rural China (Zhang and Smith, 2007). Household air pollution (HAP) from solid cooking fuel use is ranked by the Global Burden of Disease project as the second most important environmental risk factor for premature deaths in China, leading to 605 thousand premature deaths in 2016 (IHME, 2016). Population-weighted annual mean exposure to $PM_{2.5}$ (PWE, μg/m^3) in this risk assessment is estimated on the basis of cooking fuel types (Forouzanfar et al., 2016), whereas impacts of heating are not considered. This means an undifferentiated exposure level during heating and non-heating seasons.

Unlike other major countries relying on solid fuels, including India and sub-Saharan African countries where heating needs are rare, a substantial amount of solid fuels are combusted for heating in rural China, especially the northern parts (Duan et al., 2014). According to a nationwide residential energy consumption survey, space heating accounts for almost 50% of total residential energy use in China (Wei et al., 2016). For rural households without access to district heating, burning solid fuels in heating stoves or “kangs” in winter remains to be the most common home-heating practices, which are known for high emissions of various pollutants and smoke backflow even for the improved types (Chen et al., 2016a; Zhuang et al., 2009). Recent studies

Abbreviations: HAP, household air pollution; GBD, Global Burden of Diseases; $PM_{2.5}$, fine particulate matter with an aerodynamic diameter smaller than 2.5 μm; PWE, population-weighted annual mean exposure to $PM_{2.5}$; DALYs, disability-adjusted life years; HD, number of heating days; ALRI, acute lower respiratory infection; LC, lung cancer; COPD, chronic obstructive pulmonary disease; IHD, ischemic heart disease; PAFs, population attributable fractions; IER models, integrated exposure-response models; RR, relative risk; LPG, Liquefied Petroleum Gas; SNG, synthetic natural gas; WHO, World Health Organization

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have already revealed significant contributions of heating to ambient air pollution in winter, especially in northern China (Archer-Nicholls et al., 2016; Liu et al., 2016).

Evidently, space heating in winter will also contribute to indoor air pollution with more solid fuel combustion and worse ventilation (MEP, 2013). In addition, the amount of time people spend indoors, especially during the heating season, makes exposure to indoor pollutants an even more important concern. While several field measurement studies have reported increased indoor $PM_{2.5}$ concentrations and personal exposure levels in rural China in winter (Alnes et al., 2014; Baumgartner et al., 2011; Wu et al., 2015; Zhong et al., 2012), the heating contribution has not been incorporated, so far, into regional or nationwide assessment of indoor air pollution exposure and health impacts. In this study, we re-assess the PWE to HAP and quantify the bias if heating impacts were neglected, by differentiating indoor $PM_{2.5}$ concentration in heating and non-heating seasons. The exposures were calculated with the inclusion of time-activity patterns and an updated indoor $PM_{2.5}$ database with a close examination of the heating impact. Associated health impacts were also estimated and bias resulting from neglecting space heating was quantified.

2. Method

This study adopted the time-activity pattern method to estimate PWE from space heating in rural China at the provincial level from 1980 to 2012 based on an updated indoor $PM_{2.5}$ database. Premature deaths and disability-adjusted life years (DALYs) were calculated as metrics to assess the burden of diseases from HAP for the years of 1990, 1995, 2000, 2005 and 2010. Detailed methods are described below, and a flowchart of the assessment is attached (Fig. 1). Specifically, county-level PWE and burden of diseases were estimated for the year 2010 to characterize the spatial pattern.

2.1. Indoor $PM_{2.5}$ concentration

An indoor particulate matter level dataset was compiled using air pollution databases published by United Nations Environment Programme and World Health Organization (WHO), as well as updated data from field measurements published between 2009 and 2017. There were 501 publications from 2009 to 2017 that were identified from the Web of Science database relevant to indoor air pollution in China. Nine out of the 501 publications contain field measurements in rural areas and report indoor PM concentrations for households with a dominant fuel type. Measurements taken for dung cake, peat, and biomass pellet were excluded because these fuel types were not recorded in published energy databases (IEA, 2010; Wang et al., 2013), and their total consumption was assumed to be < 5% of rural residential energy consumption in China. In total, 27 studies were included in this updated dataset, covering 18 out of 33 provinces and municipalities in mainland China from both high and low space heating demanding regions. All the 27 studies were reviewed in depth and sampling details including sampled province, season, household fuel type (coal, crop residue, coal and clean energy), and indoor microenvironment (kitchen, living room, bedroom or not specified) were collected from the literature and statistically analyzed. For studies only reporting sampling periods without indicating if space heating was adopted, periods with monthly averaged

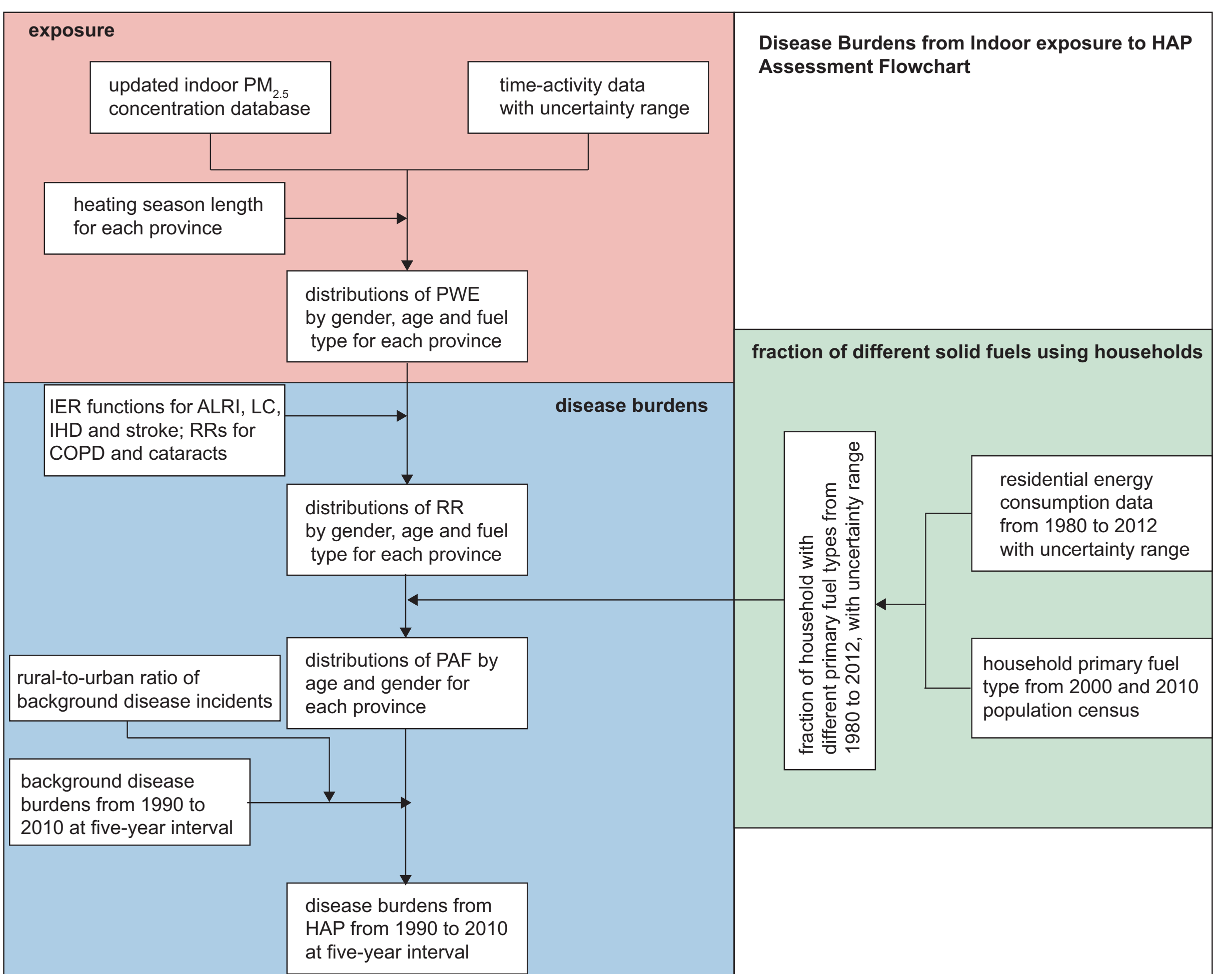

Fig. 1. Flowchart of health burdens from indoor exposure to HAP assessment.

local temperature below 5 °C were viewed as the heating season (MOHURD, 2003). All studies reviewed and included in this dataset are listed in Table S1. For studies only reporting the average concentration of several measurements, a resample was carried out according to the reported mean, standard deviation (or intervals) and sample size. Means and standard deviations of log-transformed $PM_{2.5}$ concentrations were derived to assess health impacts and uncertainties (Table S2). Measurements taken at different years were grouped together because the temporal trend of indoor $PM_{2.5}$ concentration was found to be insignificant ($p > 0.05$). This does not, however, incorporate the extra exposure in the near-household but outdoor environments from solid fuels used for space heating, for which we have no systematic measurements.

2.2. Population-weighted $PM_{2.5}$ exposure assessment

Daily exposure to $PM_{2.5}$ from HAP was estimated for residents who choose coal, crop residue, wood, and clean energy as their primary fuel source, respectively, using corresponding indoor $PM_{2.5}$ concentrations and time-activity patterns. $EXP_{j,f,h}$ is the daily exposure of subpopulation j using fuel f as the primary fuel source for heating or non-heating season calculated using the equation below:

$$EXP_{j,f,h} = \sum_k t_{j,k,h} \cdot cf_{f,k,h} \tag{1}$$

where $t_{j,kdays,h}$ is the proportion of time a subpopulation j spent in microenvironment k in a heating or non-heating season (h); $c_{f,k,h}$ is the area concentration of $PM_{2.5}$ in microenvironment k in a heating or non-heating season (h) in a household using fuel type f. The microenvironments in rural China were grouped into three categories, including kitchen, living room, and bedroom. To identify time-activity patterns for different subpopulations, rural residents in each province were divided into eight subpopulations based on gender and age, i.e., under 5 years old, between 5 and 15 years old, between 15 and 65 years old and over 65 years old for both males and females. The time spent outdoors and total time spent indoors were taken from the Exposure Factors Handbook for the Chinese population for different age groups, respectively (MEP, 2013, 2016a, 2016b). The ratios of time spent in different rooms (kitchen, living room, and bedroom and the other unspecified indoor microenvironments) were obtained directly from data compiled in a previous study (Mestl et al., 2007). When the calculated time spent in the kitchen for children and teenagers exceeds that of adult women using the ratios adopted, we assume that the amount of time spent in the kitchen for them is equal to the adult women and the remaining time is spent in the bedroom or the other unspecified indoor microenvironments. The detailed time-activity pattern used for each subpopulation in this study is provided in Table S3.

Annual average PWE is the average of PWE for heating and non-heating seasons weighted by the number of heating days (HD, defined as days with daily average temperature below 5 °C). Provincial HD from 1980 to 2012 and county-level HD for 2010 was calculated based on 2-m temperature from ERA-Interim reanalysis (Dee et al., 2011). The overall PWE for rural residents was calculated as the population-weighted average of EXPs for individual population groups.

$$PWE_{p,y,h,f} = \frac{1}{P_{p,y}} \sum_j (EXP_{j,f,h} \cdot P_{p,y,j}) \tag{2}$$

where $P_{p,y,j}$ is the size of subpopulation j in the province (or county) p and year y, derived from population censuses and the statistical yearbook (NBS, 1981–2013, 2011).

The fractions of households using different types of energy were derived from the population census which involved the major fuel type surveys, the China Energy Statistical Yearbook, and the China Rural Energy Statistical Yearbook (MOA, 1997–2008; NBS, 1986–2013, 2001, 2011). In addition, we downscaled the household fractions to a county level for the year 2010 using a series of multivariant regression models based on socioeconomic and physical indices. Detailed approaches to addressing temporal trends of provincial-level household fractions and spatial downscaling can be found in *SI Text*.

Limited by data availability, the time-activity pattern adopted for different subpopulations, especially children and teenagers, was subject to large uncertainty. In addition, potential factors influencing indoor $PM_{2.5}$ concentration including stove types, different heating practices (heating with cooking stoves, separate heating stove, or kang) were left unexplored because of limited number of field measurements (Hu et al., 2014; Stove Summit, 2017). Better characterization of PWE, as well as influencing factors, demands more field measurements with detailed information on the sampling households and more population surveys on indoor time-activity patterns in the future.

2.3. Evaluation of PWE estimates against measured personal exposure levels

To evaluate the PWE estimates from our study, we compared the estimates for each subpopulation to the personal exposure measurement results in solid fuel-using households in rural China during heating and non-heating seasons, respectively. Those reporting results from the same field measurement were identified as one study. In total, 10 field measurement studies were identified from the literature review with most studies focusing on adult female in the households. The $PM_{2.5}$ exposure level and corresponding information including fuel type, sampled subpopulation, province, heating condition (heating, non-heating, and both heating and non-heating) were listed in Table S4. By plotting the measured personal exposure level in each study against the PWE estimate for the corresponding subpopulation group, we found that most of the data pairs fall around the 1:1 ratio line within the 50%–200% range (Fig. S1). The exposure level in the heating season is consistently around twice as high as that in the non-heating season for both PWE estimates and measured personal exposure levels. Therefore, our estimated PWE based on indoor $PM_{2.5}$ concentrations and the time-activity patterns can well approximate the personal exposure levels in rural China where direct measurements are limited.

2.4. Health impact assessment

To assess burden of diseases from HAP in rural China, this study considered premature deaths and DALYs for acute lower respiratory infection (ALRI) of children under five years old, lung cancer (LC), stroke, chronic obstructive pulmonary disease (COPD) and ischemic heart disease (IHD) of adults over 25 years old and DALYs for cataracts of female over 25 years old. These metrics are consistent with those considered by GBD 2015 (Forouzanfar et al., 2016).

There are, unfortunately, various estimates of population-level health impacts from $PM_{2.5}$ exposures available over different years from the Global Burden of Diseases (GBD) project (Institute for Health Metrics and Evaluation, IHME) and WHO, which will undoubtedly change further in the future. We thus do not attempt to determine the total burden of household fuel use from the additional exposure due to the inclusion of space heating. We do apply our data to the published version of the integrated exposure-response functions (IERs) used in the last full revision of the GBD (Burnett et al., 2014) and newly published IERs (Cohen et al., 2017). The central estimates of premature deaths and DALYs derived by applying the two sets of IERs were reported as a range. The IER models describe the relative risks of certain diseases as a function of PWE (Cohen et al., 2017) and have been widely used to assess the health impacts from household air pollution at global and regional levels including China (Smith et al., 2014). IER only concerns the $PM_{2.5}$ exposure regardless of its source specification and thus should be suitable for both cooking and heating, two major sources of household air pollution, which are both associated with elevated $PM_{2.5}$ exposure.

Premature deaths and DALYs attributable to HAP were calculated by multiplying background premature deaths and DALYs for all risk

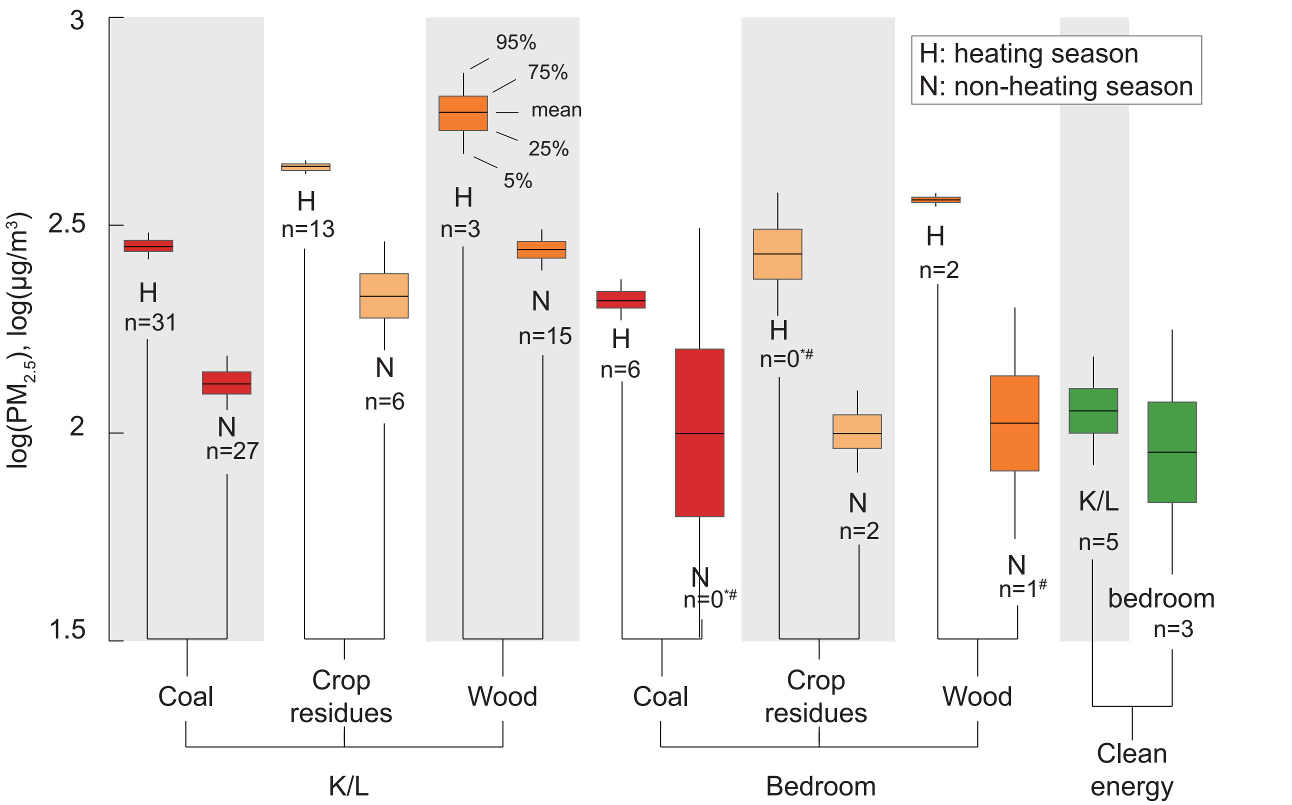

Fig. 2. Boxplot of modeled indoor $PM_{2.5}$ concentrations (log value) for the kitchen/living rooms (K/L) and bedrooms during both the heating (H) and non-heating (N) seasons for households using different fuel types. "*" means there is no measurement reported for these categories. The means were assumed to be proportional to the concentration in K/L for the same fuel type, and the $PM_{2.5}$ concentration ratio between K/L and B were assumed to be equal in both heating and non-heating seasons. "#" means standard deviations were adjusted from the corresponding K/L category with measurement number (assuming n equals 2).

factors with corresponding population attributable fractions (PAFs) of HAP, for the six causes, respectively. Provincial background premature deaths and DALYs from 1990 to 2010 at five-year intervals for the rural population were derived by multiplying reported numbers from GBD (Zhou et al., 2016) with the rural-to-urban ratio (NHFPC, 2015).

PAFs were calculated at the provincial level from 1980 to 2012 using the equations below (Lim et al., 2012) for each subpopulation with the province, gender, age and fuel type specific exposure levels. RR denotes relative risk faced by subpopulations.

$$\mathrm{PAF} = \frac{\mathrm{RR} - 1}{\mathrm{RR}} \tag{3}$$

For ALRI, LC, IHD, and stroke, two sets of RRs were calculated based on the two sets of integrated exposure-response (IER) models for $PM_{2.5}$ exposure developed by Burnett et al. (2014) and Cohen et al. (2017) and PWEs. RR values for COPD (2.00 and 2.07 for male and female, respectively) and cataracts (2.56 for female) used in this study were derived from the results of a meta-analysis because the COPD IER model developed for ambient $PM_{2.5}$ exposure did not align with indoor air pollution; and the cataract IER model was not available (Smith et al., 2014).

To get the central estimates of PAFs for a certain group of population (e.g., solid fuel using population or the total rural population), PAFs were calculated using the equation below, where F_i denotes the fraction of the population in exposure group i. The fraction of different populations has been described in the PWE assessment section above.

$$\mathrm{PAF} = \frac{\sum_{i=1}^{n} F_i \times (RR_i - 1)}{\sum_{i=1}^{n} F_i \times (RR_i - 1) + 1} \tag{4}$$

Total premature deaths and DALYs attributable to HAP were the sums of premature deaths and DALYs for all subpopulations. The excess burden of diseases attributable to space heating is characterized by the difference in premature deaths and DALYs estimates calculated using PWE from both cooking and space heating and that from cooking only.

2.5. Uncertainty analysis

Two loops of Monte Carlo were applied to evaluate the uncertainties of exposure and health burdens. The first loop was run 1000 times for each province, gender, age, and fuel-specified subpopulations. As a result, a set of 1000 simulations of PWE, RRs, PAFs, premature deaths and DALYs was generated for each subpopulation to characterize their distributions. For the uncertainty associated with PWE, variations in $PM_{2.5}$ concentrations, time-activity patterns as well as the proportion of different fuel-using populations were considered. Log-normal distributions were applied for $PM_{2.5}$ concentrations with standard deviations calculated above. Deviations of time-activity data were directly derived from the literature (Mestl et al., 2007). The fractions of the subpopulation were assumed to be uniformly distributed with a coefficient of variation of 10%. For the health burden estimates, only uncertainties in PWE were considered.

The second loop of Monte Carlo simulation was run 10,000 times to evaluate the overall uncertainty of the metrics of interest (i.e., PWEs, PAFs and health burdens) of the total rural population. Each subpopulation was sampled using its fraction in the total population as the weight. Medians and interquartile ranges were used to represent the uncertainty in this study.

2.6. Sensitivity analysis

To illustrate the impact of non-linearity of IER functions on health burdens estimates, the sensitivity of disease burden estimates to the central PWE estimates was analyzed based on the IER functions. Corresponding premature deaths and DALYs were calculated while varying PWEs from 2.5% to 97.5% confidence levels at 5% intervals using the distribution generated in the first loop of Monte Carlo simulation and fixing the other influencing factors for the year 2010 (Smith et al., 2014).

3. Results

3.1. PWE to household air pollution due to solid-fuel use

Although not included in existing estimates of the burden from household fuels, solid-fuel use for space heating is a major factor elevating household pollution in winter in rural China, where district heating is not provided (Jin et al., 2006; Jin et al., 2005; Zhong et al., 2012). Fig. 2 compares $PM_{2.5}$ concentrations in heating and non-heating seasons in pairs for all three solid fuel types and different indoor microenvironments. $PM_{2.5}$ concentrations measured in households using clean energy, including electricity and Liquefied Petroleum Gas (LPG) were also plotted on the right. Households that utilized solid fuels in their living rooms or kitchens had $PM_{2.5}$ concentrations between 337 and 585 μg/m^3 during the heating season, which is over five times greater than those with clean energy. $PM_{2.5}$ concentrations in heating seasons are on average 50% (in kitchen/living rooms of coal-using households) to 200% (in bedrooms of wood-using households) higher than those in non-heating seasons. The differences are significant for various fuel-compartment combinations ($p < 0.05$), except for bedrooms in coal-using households due to the lack of measurement data available for this category (Table S2). Compared to coal, households using biomass show greater concentration differences between heating and non-heating seasons possibly because of higher $PM_{2.5}$ emissions in biomass-reliant heating facilities due to a relatively unstable burning condition, as documented in previous studies (Liu et al., 2008; Zhang and Smith, 2007).

Based on summarized indoor $PM_{2.5}$ concentration data, we calculated PWE using time-activity patterns[4] and derived PWE for both the total rural population and the rural population using solid fuels on national, provincial as well as county levels, on the basis of EXP levels and fractions of subpopulations (NBS, 1981–2013, 2011). We found that those provinces with large rural populations experienced the highest $PM_{2.5}$ exposure (Table S5). Fig. 3(A) shows the geographic distribution of PWE with a clear decreasing trend from north to south. A clear positive correlation exists between PWE and HD for provincial PWE estimates from 1980 to 2012 (*see Supporting information*). In comparison, the previous study neglecting space heating derived an increasing PWE trend from north to south China (Mestl et al., 2007). When major fuel type difference is the only spatial difference considered, PWE for rural residents in the south who rely more on biomass was calculated to be higher than PWE for residents in the north because biomass usually corresponds to higher indoor $PM_{2.5}$ concentrations than coal (Fig. 2). When spatial differences in fuel type and heating need are considered simultaneously, however, the increasing trend of PWE from the south to the north caused by increasing heating need overwhelms the decreasing trend caused by primary fuel type difference. Comparing PWE estimate based on indoor $PM_{2.5}$ concentration from both non-heating and heating seasons and that from non-heating season only, we find that PWE would be substantially underestimated if the heating impact is neglected, especially for the population in the north (Fig. 3(B)). PWE for rural residents would be underestimated by 20% for counties close to the boundary of district heating to 50% for counties in northeast China and Tibet with long and cold winters, which means the PWE would be underestimated by 40 to 120 μg/m^3. Overall, PWE was estimated to be 163 μg/m^3 (115–194 μg/m^3 interquartile range) for rural residents in China in 2012. In addition, PWE for the solid fuel using population was 182 μg/m^3 (160–209 μg/m^3). The estimates were comparable to direct personal exposure measurements from rural solid fuel using households (113 to 490 μg/m^3), while different direct measurements showed more variation (Baumgartner et al., 2011; Hu et al., 2014; Jiang and Bell, 2008). If the effect of elevated $PM_{2.5}$ concentration during the heating season is neglected, PWE in rural China would be 125 μg/m^3 (99–154 μg/m^3) in 2012 on average, 23% lower than the estimate with the heating impact considered. In addition to the spatial difference in PWE to HAP due to heating needs in northern China, it is important to know that we need to bring down the national PWE to HAP from an even worse level than we thought before due to the inclusion of space heating. This highlights the priority of mitigating HAP among various environmental concerns and the importance of targeting space heating for HAP mitigation.

In total, approximately 0.67–0.93 million premature deaths, or 7.6%–10.6% of all deaths, and 14.0–17.7 million DALYs, or 4.2%–5.3% of total DALYs, could be attributed to HAP from cooking and heating in rural China in 2010. The overall population attributable fraction (PAF) was 28%–39%. Cooking-related exposure alone accounted for 0.64–0.91 and 13.2–17.1 million estimated deaths and DALYs. Spatial distributions of the relative difference between premature deaths and DALYs estimates with and without space heating only show a small increment of health burdens from the inclusion of space heating with such significant differences in PWE (Fig. S2). This is because the IER

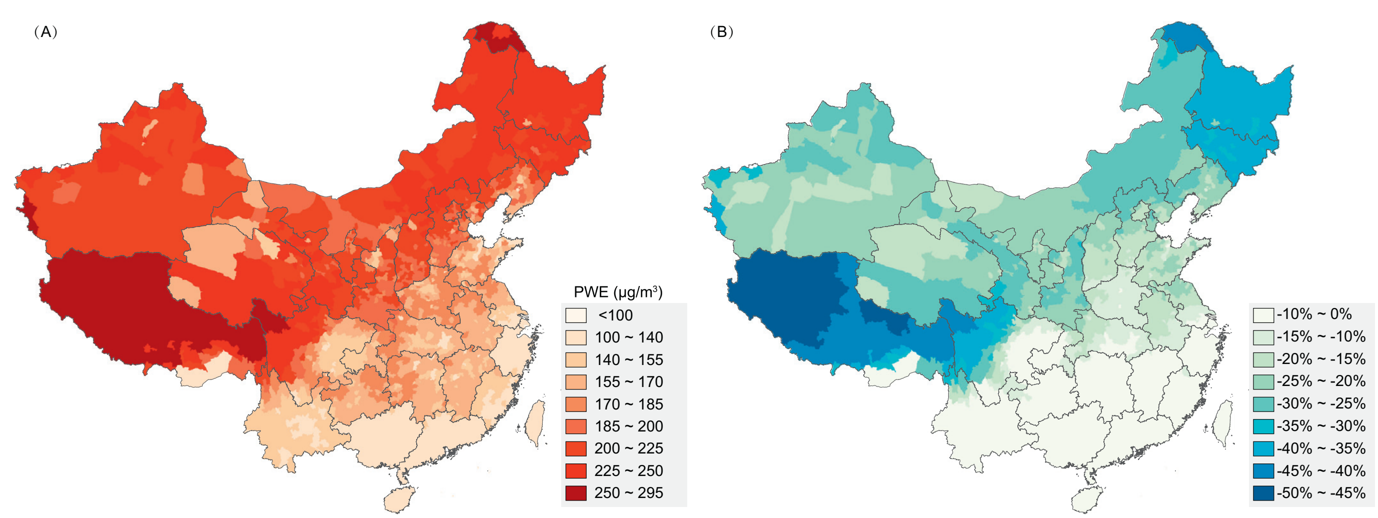

Fig. 3. Geographical distribution of PWE from the use of household solid fuels in mainland rural China in 2010 (A). Relative difference of PWE estimates in 2010 with and without considering heating impacts (B).

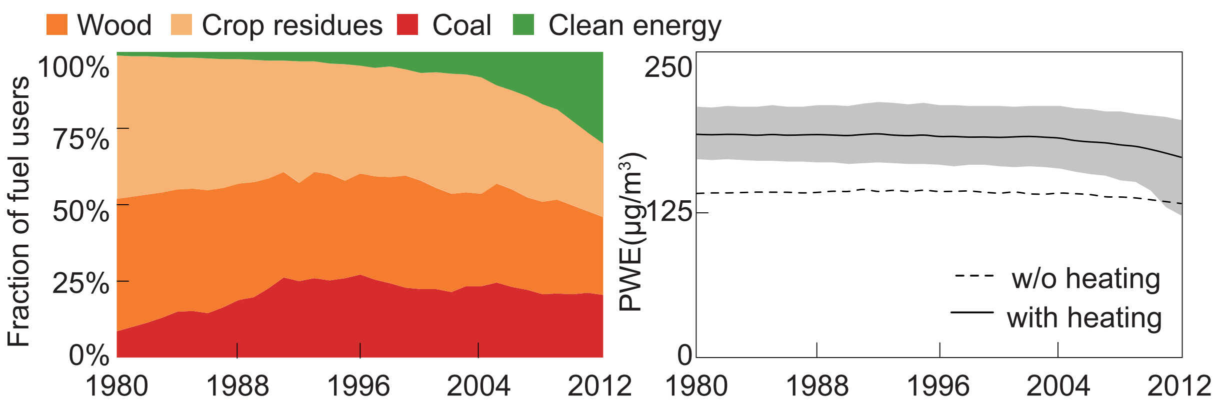

Fig. 4. Temporal trend of the fraction of different fuel users (left) and national PWE level (right) for the rural population from 1980 to 2012. The solid line and dash line represents PWE estimates with and without considering additional exposure from heating. The shaded area represents the interquartile range of PWE estimates including heating impacts.

functions are rather insensitive to exposure change at the high exposure end (Fig. S3). However, the increment is expected to grow since the PWE is expected to decrease in the future with a wide range of projects promoting clean cookstoves and clean cooking fuels (Chen et al., 2016b; Smith et al., 1993) and IER functions are more sensitive to exposure change at the lower exposure end.

Again, however, this is not the full impact of exposure due to household heating, since we only account for indoor concentrations and not the near-household exposures due to the emissions. Nor does it account for the portion of general ambient air pollution due to household heating.

3.2. Temporal trends

The national PWE for the rural population was estimated to decrease by 19 μg/m^3 from 1980 to 2012 (182 to 163 μg/m^3) with the decrease of solid fuel users. Without including exposure from space heating, PWE would only decrease by 9.0 μg/m^3 (134 to 125 μg/m^3) because the exposure reduction would be smaller when solid fuel users switch to clean energy. Fig. 4 depicts the interannual change in PWE as well as the fraction of solid fuels and clean energy users for the total rural population in the past three decades. Temporal trends of residential fuel and electricity consumptions in rural China from 1980 to 2012 are shown in Fig. S4. Solid fuels dominated the rural energy use over the entire study period and accounted for over 70% of the total consumption for all years. However, their relative contribution has decreased at an exponential rate over the last three decades from 99% to 70%. Since the early 1980s, China has experienced a rapid socio-economic transition (Zhu, 2012). Consequently, residential energy profiles have shifted from being primarily solid fuel dominated to now being occupied by clean energy in the forms of LPG and electricity (Duan et al., 2014; Zhang et al., 2009), especially after 2005 when the latter began to be widely marketed (Higashi, 2009; Ngan, 2010). Further transition from solid fuels to clean energy, especially electricity, as the primary cooking fuel in rural households in recent years has also been confirmed by a nationwide follow-up survey (Chen et al., 2016b). As a result, 14 out of 19 μg/m^3 PWE reduction occurred after 2005. The pace of PWE decrease varied among different provinces. More developed coastal provinces, including Shanghai, Zhejiang, and Guangdong, saw over 20% reduction in PWE. In contrast, less-developed western provinces, including Xizang, Qinghai, and Gansu, saw < 6% decrease in PWE from 1980 to 2012, and the provincial PWE was still over 200 μg/m^3.

Fig. 5 provides corresponding health burden changes calculated based on IERs from Cohen et al. (2017) from 1990 to 2010 at five-year intervals. To distinguish the impact of the change in background disease rate and switching from solid fuels to clean energy, the burden of diseases avoided by switching to clean energy was also calculated and pictured as hollow stacks. The burden of diseases avoided was defined as the difference between actual burden of diseases and the counterfactual loss if the fraction of solid fuel using households remained at 1990 level. Change in death and DALY rates are also depicted. In total, 0.30 million premature deaths and 15 million DALYs attributable to HAP were avoided over the last two decades. Among them, the decrease of the fraction of solid fuel users contributed to 0.11 and 2.0 million avoided premature deaths and DALYs from HAP in rural China in 2010, respectively. The premature death and DALYs in 2015, would be 0.78 and 16 million, respectively, if the fraction of households relying on solid fuel in rural China stays the same as that of 1990, 30% higher than the deaths and DALYs loss estimates from household air pollution in 2016 from GBD estimates (IHME, 2016). DALY rates from solid fuel use decreased steadily for rural residents from 35 × 10^2 to 21 × 10^2 per 100,000, mainly because switching from solid fuels to clean energy effectively reduce the DALYs from ALRI for children under five years old. In contrast, death rates stayed constantly from 1990 to 2005 when the fraction of solid fuel using households only decreased by 8.2% during this period. The death rate dropped from 112 to 100 per 100,000 from 2005 to 2010, when the transition from solid fuel to clean energy accelerated by increased accessibility to and affordability of clean energy. In addition, a spatial imbalance in terms of health burden change exists between more developed east coastal area and other parts of China (right panel). The decreasing rate of burden of diseases attributable to HAP is slower in central and northeastern China, indicating the needs for more targeted policies to accelerate household clean energy adoption for those areas.

4. Discussion

Comparing the indoor $PM_{2.5}$ concentration reported for heating and non-heating seasons in rural China, this study revealed that the $PM_{2.5}$ level in the heating season is significantly higher than that in the non-heating season for almost all fuel and indoor microenvironment categories, which is consistent with the tendency found by personal exposure measurements (Baumgartner et al., 2011; Ni et al., 2016; Zhong et al., 2012). We estimated indoor PWE from HAP to be 163 μg/m^3 (115–194 μg/m^3). Corresponding burden of diseases to be 0.67 (0.58–0.75) million premature deaths and 14.0 (12.1–15.7) million DALYs in 2010, approximately 23% and 8.0% higher, respectively, than the estimates without considering additional exposure from heating. Although the total health burden estimates are comparable to the GBD15 estimates (IHME, 2016), partly due to the nonlinearity of the IER functions, we found an obvious spatial variation in health burdens attributable to HAP. For instance, corresponding PAFs in 10 out of 16 northern and western provinces, where district heating is provided in

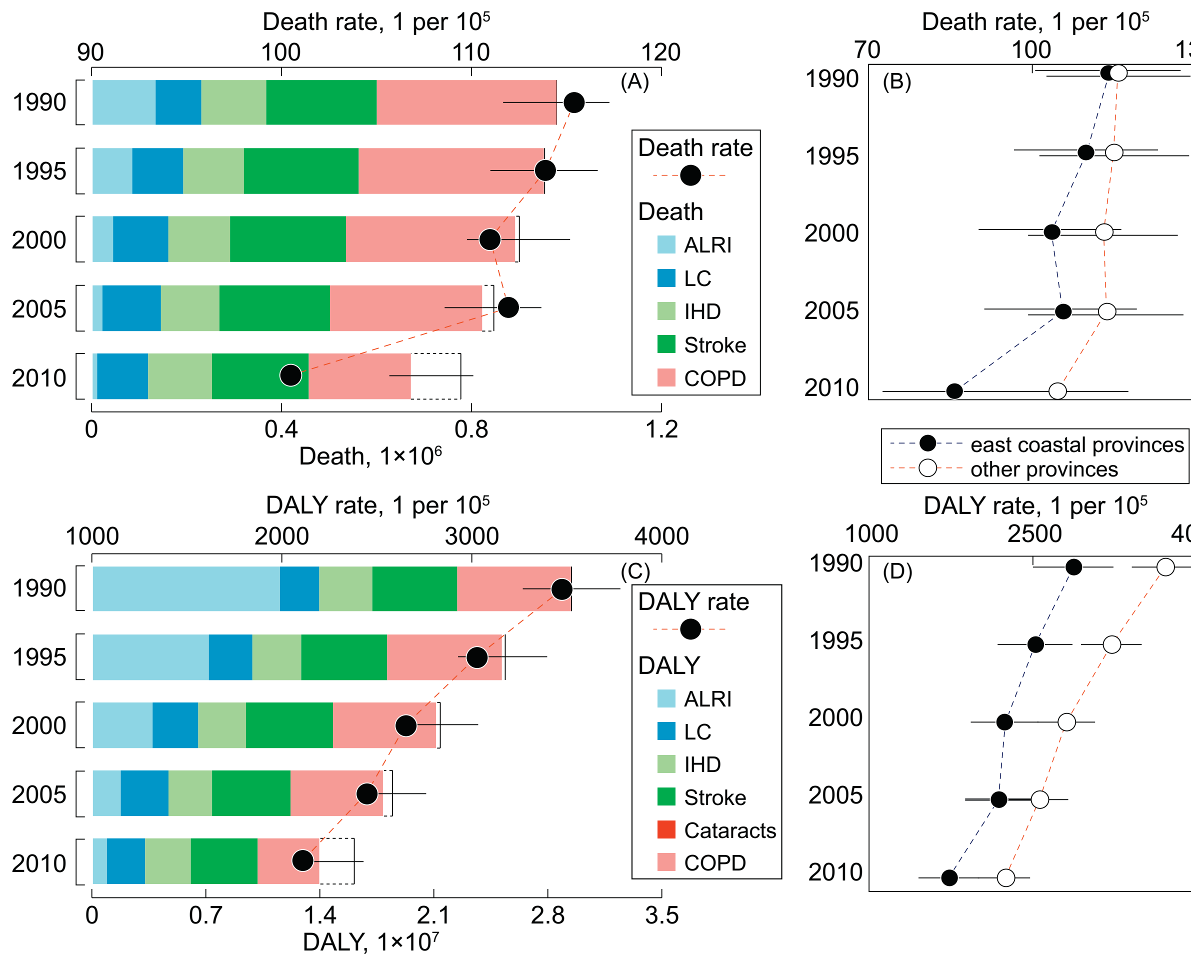

Fig. 5. The change in premature death (A) and DALYs (C) attributable to indoor exposure to HAP from solid fuel use in rural China from 1990 to 2010 and the contribution of relevant diseases. The error bars indicate the uncertainty range (interquartile range) for total death and DALYs from two-loop Monte Carlo simulation; the dashed line represents the change in death and DALYs rate.

cities were higher than the average for the total rural population in 2010. Therefore, neglecting the seasonal variation in exposure would not only lead to underestimation of total burden of diseases from HAP but also significant spatial biases. For example, PWE estimates are higher in southern provinces in a previous study focusing only on indoor PM concentration difference between biomass and coal (Mestl et al., 2007). Temporally, we found that the health burden estimates would be underestimated by approximately 7.4% from 1990 to 2010 if neglecting the heating impact. The bias persisted while the fraction of solid fuel using households decreased by 30%. In addition to China, there are many countries (such as Kazakhstan and Mongolia) where coal combustion for heating is poorly controlled and is prevalent (Kerimray et al., 2017). It is expected that health impacts of HAP in these regions have been significantly underestimated because space heating is not taken into account.

It is observed that as space heating becomes more affordable with economic development in China, the heating demands for rural residents will continue growing (World Bank, 2013). The trend is expected to continue in the future with further economic development. The Chinese government recently launched an ambitious five-year clean heating plan to convert heating with coal to natural gas by the year 2021 (Judy and Benjamin, 2017). The effectiveness of this plan would be significantly under-evaluated if impacts of solid-fuel heating are neglected in a risk assessment.

To mitigate population exposure to pollution from solid fuel cooking and heating, particularly, for provinces in western, central and northeastern China, where most rural residents live and clean energy technologies have been slow to penetrate, both short-term interventions, such as cleaner coal and improved stoves, and long-term policies to replace solid fuels with LPG, electricity, or synthetic natural gas (SNG) will be needed (Shen, 2016). Considering the substantially higher emission factors of household stoves compared to coal consumption by power plants, such interventions will also provide China with a great opportunity to meet the “Action Plan for Air Pollution Prevention and Control” (Qin et al., 2017; Sheehan et al., 2014).

Contributions

Y.C. and S. T. conceived and designed the study. Y.C. performed the analysis and prepared the initial draft of the paper. H. S., K. R. S., D. G., Y. C., G. S. contributed to results interpretation. All authors (Y.C., H.S., K. R.S., D.G., Y.C., G.S., J.L., H. C., E. Y. Z. and S.T.) participated in the writing of the manuscript. S.T. coordinated and supervised the project.

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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.2018.04.054.

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