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Internal migration and urbanization in China: Impacts on population exposure to household air pollution (2000–2010)

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HIGHLIGHTS

- We identify changes in household fuel use in China from 2000 to 2010.
- We estimate how the population exposure to $PM_{2.5}$ changed over the decade.
- ~60% of the total exposure reduction of about 50 μg/m^3 can be linked to migration.
- Annual mean $PM_{2.5}$ exposure of rural–urban migrants was reduced by about 215 μg/m^3.
- The annual health benefit from the energy transition is about 30 billion USD.

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ABSTRACT

Exposure to fine particles ≤2.5 μm in aerodynamic diameter ($PM_{2.5}$) from incomplete combustion of solid fuels in household stoves, denoted household air pollution (HAP), is a major contributor to ill health in China and globally. Chinese households are, however, undergoing a massive transition to cleaner household fuels. The objective of the present study is to establish the importance of internal migration when it comes to the changing household fuel use pattern and the associated exposure to $PM_{2.5}$ for the period 2000 to 2010. We also estimate health benefits of the fuel transition in terms of avoided premature deaths. Using China Census data on population, migration, and household fuel use for 2000 and 2010 we identify the size, place of residence, and main cooking fuel of sub-populations in 2000 and 2010, respectively. We combine these data with estimated exposure levels for the sub-populations and estimate changes in population exposure over the decade. We find that the population weighted exposure (PWE) for the Chinese population as a whole was reduced by 52 (36–70) g/m^3 $PM_{2.5}$ over the decade, and that about 60% of the reduction can be linked to internal migration. During the same period the migrant population, in total 261 million people, was subject to a reduced population weighted exposure (ΔPWE) of 123 (87–165) g/m^3 $PM_{2.5}$. The corresponding figure for non-migrants is 34 (23–47) g/m^3. The largest ΔPWE was estimated for rural-to-urban migrants (138 million people), 214 (154–283) g/m^3. The estimated annual health benefit associated with the reduced exposure in the total population is 31 (26–37) billion USD, corresponding to 0.4% of the Chinese GDP.

1. Introduction

Photos and news stories from today's Chinese cities often tell a story of extreme urban air pollution. According to the comparative risk assessment of the Global Burden of Disease Study 2010 (Lim et al., 2012; IHME, 2013), ambient urban particulate air pollution (fine particles ≤2.5 μm in aerodynamic diameter ($PM_{2.5}$)) causes 1.2 million premature deaths annually in the country, making it the fourth most important risk factor for premature death. One may presume that migrating from rural to urban areas in China entails an increased exposure burden for the individual migrant. In actual fact it entails an increased exposure to *urban ambient* $PM_{2.5}$ pollution. Whether it entails an increased *overall* exposure to $PM_{2.5}$ depends on the migrant's previous exposure to $PM_{2.5}$. As the majority of the rural population in China still uses traditional fuels and inefficient stoves, rural–urban migrants often come from a setting of high exposures to smoke particles ($PM_{2.5}$) from household stoves, so-called household air pollution (HAP).

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On an annual basis HAP is estimated to cause about 1 million premature deaths in China, making it the fifth most important risk factor in 2010, down from number one in 1990 (Lim et al., 2012).

The reduced role of HAP as a contributor to ill health in China is a result of the transition to cleaner fuels that is taking place in Chinese households. In the decade from 2000 to 2010, the number of households reporting to have solid fuel (firewood or coal) as their main cooking fuel fell substantially, from 900 million to 650 million. In 2010, 80% of urban and 23% of rural households reported to have clean fuels (gas or electricity) as their main cooking fuel (AMCR, 2004; NBS, 2012).

Several factors have contributed to household fuel switch in China. Income and education level have been identified as robust determinants of household energy choices. In addition, accessibility of energy resources has been identified as a key determinant (Jiang and O'Neill, 2004; O'Neill et al., 2012a; Papineau et al., 2009; Peng et al., 2010). Since access to modern fuels depends on infrastructure for their distribution, urbanization as such plays a key role in energy transition (Krey et al., 2012; Leach, 1992; O'Neill et al., 2012a, b). Moving from a rural to an urban area is likely to enhance access to cleaner household fuels. Thus, the massive migration from rural to urban areas taking place in China likely played an important role for the household energy transition happening during the last decade.

Urban–rural migration likely reduces the exposure to $PM_{2.5}$ from HAP. At the same time exposure to $PM_{2.5}$ from urban ambient sources may increase. To our knowledge, no previous study has attempted to quantify the impact of migration on the overall population exposure to $PM_{2.5}$ in China or elsewhere. Such knowledge would be important e.g., for formulating migration policies and shaping urban green growth, as reducing the overall exposure to pollutants is important for creating healthy living conditions and enhancing welfare. The objective of the current paper is to estimate how the exposure to $PM_{2.5}$ pollution in the Chinese population has changed over the period 2000 to 2010 as a result of migration on the one hand and general household fuel switch on the other hand. We also estimate health effects in terms of avoided premature deaths from the estimated changes in population exposure and the monetized value of the avoided deaths.

2. Materials and methods

2.1. Population data

We use China Census data to establish the number of internal migrants in China in 2010 and the population residing in urban and rural areas in China's 31 provinces/autonomous regions/municipalities (denoted provinces below) in 2000 and 2010 (Table 1 and Fig. 1) (ACMR, 2004, 2012; NBS, 2012). To be counted as a migrant in the China Census database a person needs to have stayed away from home, i.e. the place where he or she has the household registration, *hukou* in Chinese, for at least 6 months. There are two types of hukou in China, those born in rural areas generally get agricultural hukou while those born in cities get nonagricultural hukou. The two groups are often referred to as rural and urban hukou, and we use these terms in the following (see Meng, 2012 for a description of the household registration system in China).

In the China Census database migrants' current residence (i.e. location of immigration) is defined by the administrative type of setting, and is divided into three: City, town, or rural. We pool the two first groups into 'urban immigrants', i.e. migrants that live in urban areas. In the data for emigration (i.e. from where migrants originally came and still have their household registration) rural areas are differentiated into two, thus there are four types: City, town, village, or township. The first two groups are urban or semi-urban and refer to those with urban hukou. In the following they are pooled into 'urban emigrants', i.e. migrants that come from urban areas. The last two are rural or semi-rural and refer to those with rural hukou. In the following these are pooled into 'rural emigrants'.

The total number of migrants in China in 2010 was 261 million. Of these, 138 million, i.e. 53%, came from rural areas and settled in urban areas (Table 1). About two thirds (67%) of the migrants are intra-provincial migrants, i.e. they have not left for another province, as opposed to inter-provincial migrants, who have left for another province. Nearly half (48%) of the migrant population are women.

The detailed data on migration pattern per province is available in the so-called Long-form database, covering approximately 10% of the total Chinese population (NBS, 2012). The total number of migrants per province is given in the Short-form database which covers 100% (NBS, 2012). For all of China in total and for the eight provinces hosting the largest number of migrants we extract the home province of the migrant population, and whether migrants come from and settled down in urban or rural areas from the Long-form database (example shown in Fig. S1). We divide the migrants' home and host province into northern and southern, defined by whether the main area is located North or South of the Yangtze river (allocation given in Fig. 1). For each province the data are scaled up to a 100% sample by applying the ratio of migrants in the 10% database to the number of migrants in the 100% database.

2.2. Estimating population weighted exposure in 2000 and 2010

We estimate the population weighted exposure to $PM_{2.5}$ (PWE) in the total Chinese population (including sub-groups according to location) and the migrant population (including sub-groups according to location of origin and destination) for 2000 to 2010. The change in PWE (ΔPWE) from 2000 to 2010 for total and migrant populations is calculated as the difference between the PWE of the given population group in 2010 versus 2000.

In the following 'migrants' refers to those who were defined as migrants in 2010 according to the definition given above. 'Non-migrants' denote those who were not migrants in 2010, i.e. those who in 2010 were living in their home settlement according to the Census data. Corresponding figures for the eight largest host provinces were also calculated (Table 2). PWE in the given year (2000 or 2010), for a population group P, is calculated as:

$$PWE_P = \frac{1}{P}\sum_{i,j}\left(P_{i,j} \cdot PWE_{i,j}\right) \qquad (1)$$

where i refers to location of P (any combination of urban or rural, North or South) and j refers to household fuel categories of P_i (clean, coal, or

Table 1
Total population in 2000 and 2010 and number of migrants in China in 2010 (million). NBS (2012) and ACMR (2004).

Total population	Total	Rural	Urban
Year 2000	1241	758	483
Year 2010	1333	659	674
Migrant population in 2010	**Total**	**Migrated from rural areas**	**Migrated from urban areas**
Total	261	164 (63%)	97 (37%)
Current residence is urban	227	138 (60%)	90 (40%)
Current residence is rural	34	27 (80%)	7 (20%)

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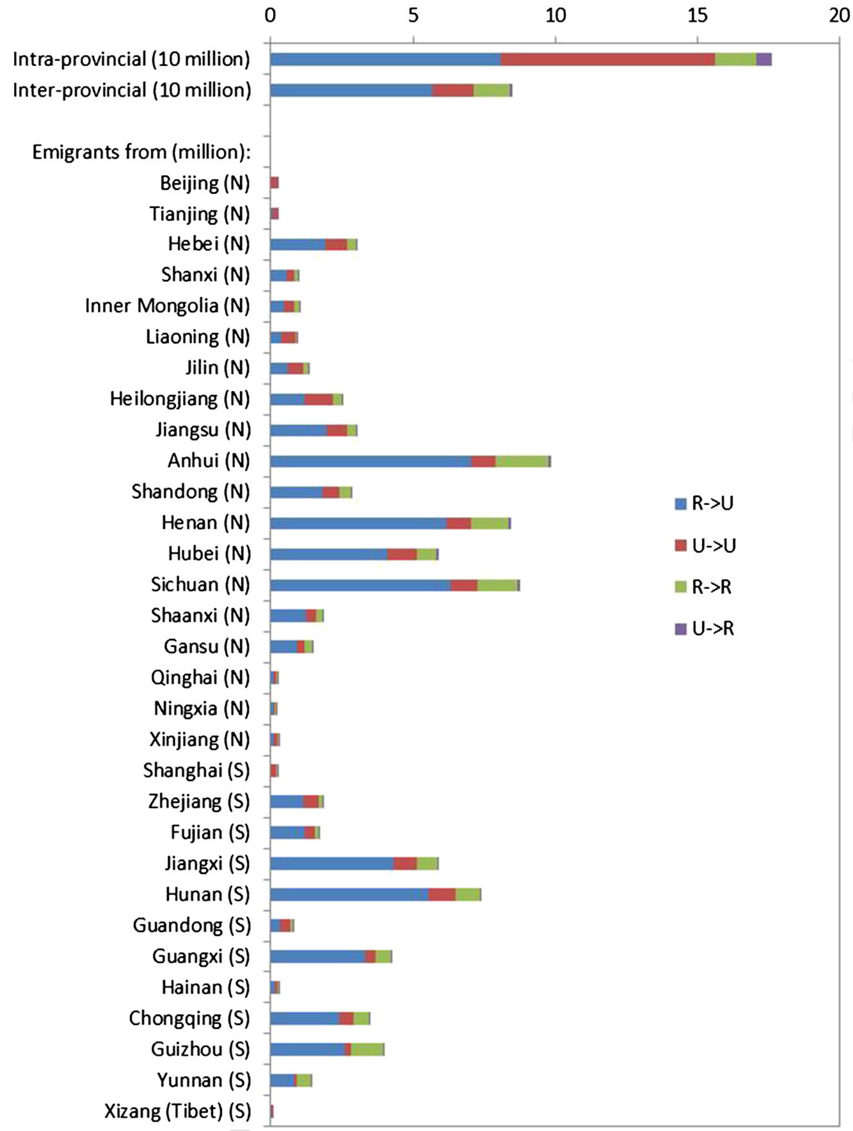

Fig. 1. Total migrant population in China in 2010 and their rural/urban origin and destination. Intra-provincial migrants have migrated within their home province, inter-provincial migrants have migrated out of their home province (both given in 10 million). Geographical location used in the allocation of provinces shown in parenthesis. R → U: rural–urban migrants; U → U: urban–urban migrants; R → R: rural–rural migrants; U → R: urban–rural migrants.

biomass). $PM_{2.5}$ data for estimating $PWE_{i,j}$ are described below. Note that to calculate the PWE of migrants in 2010, *i* refers to the location of immigration (i.e. the place where the migrants live in 2010), whereas to calculate the PWE in 2000 of current migrants, *i* refers to the place where these migrants have their household registration, i.e. place of emigration. We thus assume that the 2010 migrant population had not yet migrated in 2000. According to the Census data, less than a quarter (24%) of the migrants had left their home province more than 6 years ago. Adding a trend line to the data on how many years current migrants had been away from home in 2010, we estimate that 10–20% of current migrants had left home already in 2000. We choose to disregard this in the following (see the Discussion section below).

There is likely to be a certain fraction of children < 10 y of age in the 2010 migrant population. Data for this fraction is, however, not available. We assume that 10% of the migrant population in 2010 was < 10 y of age, i.e. that 90% of the cohort of 261 million migrants were born in 2000. In a study among the migrant population in Shanghai, 12–13% of the migrants were children < 10 y of age (Liu et al., 2010). We believe that the fraction in Shanghai is somewhat larger than in the migrant population in total.

Table 2
Migrant population residing in the eight largest host provinces (million), percentage of migrants coming from outside the province (inter-provincial), and percentage of the total population that are migrants.

Province	Migrant population	% inter-provincial	% of population that are migrants
Guangdong	36.8	58%	35%
Zhejiang	19.9	59%	37%
Jiangsu	18.2	40%	23%
Shandong	13.7	15%	14%
Shanghai	12.7	71%	55%
Sichuan	11.7	10%	15%
Fujian	11.1	39%	30%
Beijing	10.5	67%	54%

2.3. $PM_{2.5}$ exposure for sub-populations

To estimate the population weighted exposure (PWE) to $PM_{2.5}$ for population groups in China we use the estimates in Mestl et al. (2007a). Mestl et al. (2007a) compiled data from rural and urban settings in China on concentration levels of particulate pollution indoors in households depending on different fuels and in indoor environments away from home. Ambient air concentrations in urban areas were estimated based on measurements in 2002, whereas ambient air concentrations in rural areas were based on measurements and model results from a regional chemical tracer model using emission data for 2000. In addition, data on time–activity pattern were compiled. The annual exposure to PM_{10} for different sub-populations (in terms of age, sex, household fuel, and geographic location based on climate zone and urban or rural classification) was estimated as the sum of the exposure in the various microenvironments. The exposure in various microenvironments was estimated from the proportion of time spent in the given microenvironment multiplied by the PM_{10} concentration. A fuel mix may potentially have occurred among the variety of households included in the original studies that Mestl et al. (2007a) built on. The authors assume that the households are representative for households in the given region and urban/rural setting and for the common mix of fuel in actual use. For northern provinces (i.e. north of the Yangtze River, the 'heating zone') the estimated annual exposure was based on separate estimates for the winter and summer seasons, as the pollution level both indoors and outdoors is higher in winter due to the need for heating. Fuel use data as well as demographic data were taken from China Census 2000 and were available at a county level. Combining the estimated annual average exposure for all sub-populations with population data from China Census 2000, the estimated PWE for 12 sub-populations categorized by main cooking fuel and geographic location was derived (see Mestl et al., 2007a for further details). In summary, the 12 fuel/location exposure categories reflect the total annual PWE of these sub-populations, both to indoor and ambient PM_{10} pollution. We use a $PM_{2.5}/PM_{10}$ conversion rate of 0.5 (uncertainty range 0.4–0.6) (see the Discussion section below regarding the choice of conversion ratio). In the following we apply the estimates as shown in Table 3 both for 2000 and 2010. It may be that PWE for the sub-populations has changed over the decade in question. As the primary objective of the current paper is to estimate how the exposure to $PM_{2.5}$ pollution has changed as a result of the household fuel switch in general and of migration in particular, we suggest that the approach is justified, as we avoid introducing a number of other variables for which data are limited. For instance, there is little data showing a decline in HAP given the various cooking fuels used, and the change in outdoor $PM_{2.5}$ pollution is not clear for all regions (see below). Sensitivity analyses are, however, included below to investigate the impact on results of changing the assumption about constant PWE levels for the twelve exposure categories.

2.4. Household cooking fuel use

To allocate the sub-populations of interest into one of the exposure categories in Table 3 we use data on main cooking fuel in 2000 and 2010 per province from China Census 2000 and 2010 (ACMR, 2004; NBS, 2012). The fuel categories are gas, electricity, coal, firewood, and other (the numbers add up to 100% as they refer to the main cooking fuel only). We follow Mestl et al. (2007a) and pool gas and electricity and denote this 'clean'. The category 'other' (about 1% of the total) is pooled with firewood and denoted 'biomass'. The fuel distribution for the two years is given in Table 4.

Table 3
Population weighted exposure (PWE) to $PM_{2.5}$ ($\mu g/m^3$) for urban and rural populations in North and South China according to cooking fuel use classification (S.D.). Based on Mestl et al. (2007a).

	Urban		Rural	
	North	South	North	South
Clean	142 (18)	84 (18)	82 (7)	55 (7)
Coal	174 (18)	137 (22)	206 (15)	286 (28)
Biomass	440 (77)	485 (132)	433 (52)	496 (84)

Table 4
Main cooking fuel in Chinese households 2000 and 2010.

		Clean	Coal	Biomass	
Population (million)					
Year 2000	Total	340	335	566	1241
	Urban	267	134	82	483
	Rural	73	201	484	758
Year 2010	Total	686	190	456	1333
	Urban	536	74	63	674
	Rural	150	116	393	659
%					
Year 2000	Total	27%	27%	46%	100%
	Urban	55%	28%	17%	100%
	Rural	10%	27%	64%	100%
Year 2010	Total	52%	14%	34%	100%
	Urban	80%	11%	9%	100%
	Rural	23%	18%	60%	100%

Household cooking fuel use among migrants is not specified in the Census data. We assume that the fuel use distribution among migrants is similar to the fuel use distribution in the total population, regarding both their current residence and their home province. Thus, for the migrant population we use the fuel profile as given in Table 4. The assumption is discussed below.

For the migrant populations in the eight top host provinces corresponding profiles are derived. Whereas we have the provincial urban–rural fuel distribution data for 2010, these data were not available for 2000 (only the share of each fuel in the province as a total was available). For the eight provinces included in this study we estimate the urban–rural allocation of each fuel type in 2000 by combining the urban–rural allocation in the given region (North or South) in 2000 (Table 1 in Mestl et al. (2007a)) with the percentage share of the three fuel types in 2000 in the province. Provincial fuel profiles for 2000 and 2010 are given in Supplementary material Tables S1 and S2.

2.5. Estimating population exposure reduction linked to migration

The estimated ΔPWE for total and migrant populations reflects two factors. One is the change in population distribution, i.e. where people live. The other is the general household fuel switch that has taken place. Since the household fuel distribution is inherently a result of both factors, we cannot disentangle their relative importance neither for the total population nor for the migrant population. An estimate of the importance of migration versus fuel switch is obtained from the fact that in any given year (here 2000 or 2010) the population exposure (PWE times population) of the total population (PE_{tot}) is the sum of PE in the migrant population ($PE_{migrants}$) and the PE in the non-migrant population ($PE_{non\text{-}migrants}$):

$$PE_{tot} = PE_{migrants} + PE_{non\text{-}migrants}. \quad (2)$$

Since we know the size of both the migrant and non-migrant populations and are able to estimate the PWE for total and migrant populations in both years, we can calculate PWE for the non-migrant population for 2000 and 2010.

The relative importance of migration (M) regarding the change in total population exposure to HAP in China over the ten year period 2000–2010 is given by:

$$M = \Delta PE_{migrants} / \Delta PE_{tot}. \quad (3)$$

2.6. Health benefits of changes in population exposure

To provide estimates of the health effects from changes in population exposure to $PM_{2.5}$, we use exposure–response functions for risk of premature mortality and long-term $PM_{2.5}$ exposure from cohort studies in the US (Pope et al., 2011). Obviously, including morbidity end-points as well will increase health effect estimates. The exposure–response relationships established for cardiopulmonary end-points and lung cancer in Pope et al. (2011) are non-linear, i.e. they flatten at higher exposure levels. We use these functions to calculate the number of cases attributable to the given PWE level (denoted attributable cases, AC) in, respectively, 2000 and 2010 and calculate the avoided cases attributable to ΔPWE over the ten year period as the difference between AC in 2000 and AC in 2010, i.e. ΔAC. The non-linear form of the exposure–response functions implies that the mortality risk reduction from a reduction in exposure depends on the level of exposure (higher exposure levels apply to the flatter part of the curve). We follow Anenberg et al. (2010) and others to calculate AC:

Since

$$\mathrm{RR} = p/p_0 \tag{4}$$

where RR is the relative risk, p is the annual mortality rate in a polluted environment, p_0 is the annual mortality rate in a counterfactual clean environment, and

$$\mathrm{AC} = (p - p_0) \cdot P \tag{5}$$

where AC is the attributable cases, i.e. the fraction of the mortality burden attributable to the risk factor ($PM_{2.5}$ exposure), and P is the size of the exposed population, we get

$$\mathrm{AC} = [(\mathrm{RR}-1)/\mathrm{RR}] \cdot p \cdot P. \tag{6}$$

To calculate RR for cardiopulmonary diseases (CPD) and lung cancer we used the power function described in Pope et al. (2011):

$$\mathrm{RR} = 1 + \alpha(\mathrm{DD})^{\beta} \tag{7}$$

where DD is the daily dose (in mg), i.e. inhaled dose of $PM_{2.5}$, calculated as $PM_{2.5}$ in g/m^3 multiplied with 18/1000 (see details in Pope et al. (2011)). In the exposure–response function for CPD α is 0.2685 [95% CI: 0.2110, 0.3606] and β is 0.2730 [95% CI: 0.0.2146, 0.3664]. In the exposure–response function for lung cancer α is 0.3195 [95% CI: 0.2865, 0.3473] and β is 0.7433 [95% CI: 0.6666, 0.8080]. The 95% CI for the parameters was estimated from the 95% CIs of the individual studies used to derive the power function in Pope et al. (2011), excluding studies where DD was above 66 mg and, for the lung cancer studies, the data for a DD of 18 mg (seemingly an outlier). RR estimates from the power function and the individual studies are shown in Fig. S2, for CPD and lung cancer, respectively.

AC per capita is calculated for 2000 and 2010 by excluding P in Eq. (6), taking into consideration the PWE values as estimated above. Subtracting AC per capita for 2010 from AC per capita for 2000, we arrive at ΔAC per capita for the given population group. In the above equations, p refers to the mortality rate for the health end-point in question, i.e. CPD and lung cancer. The mortality rates for each health end-point are taken from IHME (2013) (see also Lim et al., 2012). We use a mortality rate for CPD of 0.0031, being the sum of mortality rates for cardiovascular diseases (CVD) (0.00234) and chronic respiratory diseases (0.00076). Mortality rate for lung cancer is 0.00038. We do not have data on mortality rates for the sub-populations and apply the China average mortality rates for 2010 for all population groups.

A study by Cao et al. (2011) is to our knowledge the only cohort study of the association between long-term exposure to PM (in terms of total suspended particulates, TSP) and premature mortality (due to cardiovascular deaths) in China. The study obtains significantly larger impact coefficients than Pope et al. (2011), especially at high air pollution levels. We use the exposure–response function from this study, 0.09% [95% CI: 0.03%, 0.15%] increased risk of CVD mortality per g/m^3 TSP in middle aged men and women, and Eq. (6) to provide an upper estimate of the health effect of ΔPWE. To apply the function in Cao et al. (2011) we assume a $PM_{2.5}$/TSP conversion ratio of 0.33 (0.54 for PM_{10}/TSP and 0.61 for $PM_{2.5}$/PM_{10}) from Ho and Nielsen (2007) (Pope and Dockery (2013) assume a similar ratio (0.3) to compare risk estimates from China and the US). p in Eq. (6) now refers to the mortality rate for CVD from IHME (2013).

To calculate the economic costs of avoided premature deaths, we follow Vennemo et al. (2009) and assume that the Value of Statistical Life (VSL) in China is 100 (50–150) times the GDP/cap. Using 2010 GDP/cap in 2010 prices and an average exchange rate of 6.24, we arrive at a VSL of approximately 480,000 USD.

In summary, the annual health benefit (H) associated with reduced exposure for a given population or sub-population is calculated as:

$$\mathrm{H} = \Delta\mathrm{AC/cap} \cdot P \cdot \mathrm{VSL} \tag{8}$$

where P is the number of people in the population (or sub-population) in 2010.

In addition to the baseline calculation, we provide estimates of the impact on PWE and health damage in the Chinese population given two counterfactual cases. The first is an assumed 100% urbanization in 2010, and the second is an assumed 100% uptake of clean fuels in 2010. Regarding the first experiment ('100% urbanization') we estimate PWE in 2010 by allocating the total population in northern China in 2010 into the three fuel categories using the urban fuel distribution for urban North in Table 3. The total population in southern China is allocated into the three fuel categories by using the urban fuel distribution for urban South. Regarding the second experiment ('100% clean scenario') we apply the PWE values for clean fuel in Table 3 for the total populations in the four zones, i.e. North (urban and rural) and South (urban and rural). We reduce the resulting PWE values by 15%. This is based on the study by Chafe et al. (submitted for publication) indicating that 15% of ambient $PM_{2.5}$ concentrations in China is due to emissions from household cooking stoves. Since outdoor pollution dominates the exposure of clean fuel users in Mestl et al. (2007b), we assume that the PWE is shifted down by the same percentage as the outdoor $PM_{2.5}$. In both experiments, the estimated ΔPWE value refers to the difference between our baseline PWE in 2010 versus the counterfactual PWE in 2010 resulting from assuming 100% urbanization or 100% clean fuel, respectively.

2.7. Uncertainty and sensitivity analyses

Uncertainty intervals below are calculated by simultaneously applying PWE values plus/minus 1 standard deviation (SD) in Table 3, the upper/lower $PM_{2.5}$/PM_{10} ratio, and the exposure–response coefficients plus/minus 1 SD.

We carry out a sensitivity analysis to test the impact of altering the assumption that PWE values for the 12 fuel/location categories in Table 3 have not changed from 2000 to 2010. Since the publication of Mestl et al. (2007a), studies continue to report high levels of indoor air pollution in homes where solid fuels are used for cooking, see e.g. Wang et al. (2010), Aunan et al. (2013) and Alnes et al. (submitted for publication) for measurements in Guizhou and comparison of findings with other studies. To our knowledge, no comprehensive comparison of the situation in 2000 versus 2010 when it comes to HAP exposure given the various stoves and fuels in use has, however, been carried out in China. In the first sensitivity test we thus assume that indoor concentrations in homes using the three types of fuels have not changed during the decade 2000–2010. We do, however, change the assumptions about outdoor concentrations. Regarding urban ambient PM pollution, PM_{10} concentrations declined steadily in 86 key cities during 2001–2011. Annual average PM_{10} in the most polluted cities, located primarily in the North, was reduced by 47% over the decade, whereas the level in medium polluted cities was reduced by 18%. PM_{10} in the least polluted cities, primarily in the South, increased by 24% during

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the decade (Cheng et al., 2013a). We allocate the city data for 2001 and 2011 used in Cheng et al. (2013a) into North and South in the same way as was done for the provinces (Fig. 1) and find that PM_{10} in cities located in northern China on average decreased by 28% over the decade. PM_{10} in cities located in southern China increased on average 8%. For simplicity, we assume that $PM_{2.5}$ is changed in the same way as PM_{10}. In reality, a rising contribution of secondary fine particles to the PM load in China, due to substantive increases in important precursor gases, is indicated in urban areas as well as on a regional scale (Cheng et al., 2013b; Lin et al., 2010; Wang et al., 2011; Zhao et al., 2013). A quantification of a possible increase in the annual average $PM_{2.5}/PM_{10}$ ratio in urban ambient air was, however, not available. Regarding rural ambient $PM_{2.5}$ pollution, we use the provincial level estimates of changed mean outdoor $PM_{2.5}$ concentration in the period 2000–2010 from the GAINS-China model (IIASA, 2013; UNEP and ECLIPSE estimates) in the sensitivity analysis. The increase was 21% in the North and 15% in the South.

In the sensitivity calculation we use data from Mestl et al. (2007b) to separate the fraction of PWE in the 12 fuel/location categories that is due to outdoor $PM_{2.5}$ (Table S3). The part of PWE due to HAP is denoted PWE_{HAP}. The part of PWE due to outdoor $PM_{2.5}$ (PWE_{out}) is altered as described above for urban and rural areas in North and South, respectively. We add the resulting PWE_{out} values to the PWE_{HAP} values (Table S4). Table S5 shows the resulting PWE to outdoor $PM_{2.5}$ in various regions given the baseline and the sensitivity analysis.

In another sensitivity test we reduce PWE of all biomass users in 2010 by 25%. This is to accommodate a possibility that HAP associated with biomass fuels in 2010 may have become lower than it was in 2000 as a result of a growing private infrastructure for marketing improved biomass stoves resulting from the National Improved Stove Program (NISP) that was implemented in China during the 1980s and 1990s (Spautz et al., 2006). Also, whereas the program has ended, improved stoves still is an element in other programs targeting the rural poor (Sinton et al., 2004). In the sensitivity test, the PWE in the rural South becomes 372 (248–522) g/m^3, close to a recent estimate of exposure to indoor and outdoor $PM_{2.5}$ among rural biomass users in Guizhou (Aunan et al., 2013). We also test the sensitivity of changing the assumed percentage share of migrants in 2010 that was not yet born in 2000 to 5% instead of 10%, i.e. we assume that 5% of the migrants are children <10 y of age.

3. Results

As shown in Fig. 2 there has been a marked increase in the number of people using clean household fuels from 2000 to 2010, in total 346 million. The increase is particularly large in the urban North. Simultaneously, there has been a large reduction in people using coal and biomass as their primary fuel, 145 and 110 million, respectively. The discrepancy between these figures is the population growth. Particularly in north China, fewer people use coal for cooking. Whereas the absolute number of rural people using biomass has been substantially reduced, the share in rural areas that depends on this fuel has not changed much. As shown in Table 4 the share was 64% in 2000 and 60% in 2010. This implies that the rural poor only to a very limited extent has taken part in the household energy transition that has occurred during the decade, unless they have become migrants. Note that while there were 91 million fewer rural biomass users over the period, there were in 2010 according to our calculation 78 million rural–urban migrants that had used biomass as their main cooking fuel in their home province.

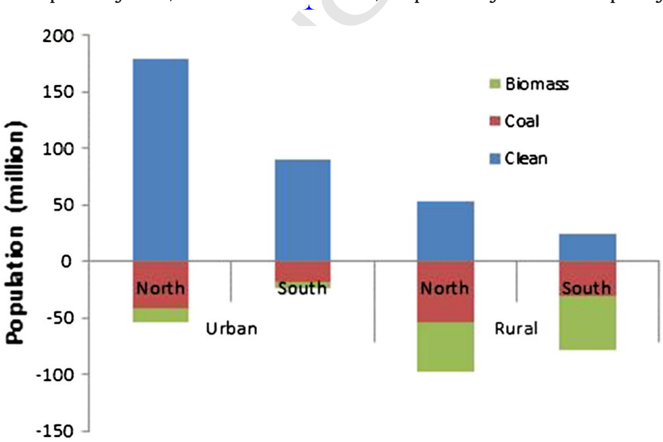

Fig. 2. Changes in the population depending on various household cooking fuels in China from 2000 to 2010.

We find large reductions in $PM_{2.5}$ exposure in the Chinese population over the decade 2000–2010. The estimated PWE in 2000 and 2010 and ΔPWE over the period are given in Table 5 for the Chinese population as a whole and for various sub-populations. We estimate a reduced PWE of 52 (36–70) g/m^3 $PM_{2.5}$ for the total population. The reduction is substantially larger for migrants (123 (87–165) g/m^3) compared to non-migrants (34 (23–47) g/m^3), and particularly large for rural–urban migrants, whose estimated ΔPWE is 214 (154–283) g/m^3. Among the eight largest host provinces, migrants to Zhejiang had the largest reduction in PWE, an estimated 199 (141–266) g/m^3. According to our estimates, migrants to Shanghai had the lowest PWE in 2010, 94 (59–137) g/m^3. PWE results for a larger selection of sub-populations are given in Supplementary material Table S6. Note that ΔPWE for the total population is larger than ΔPWE for both urban and rural populations. This is due to the urbanization taking place during the decade, i.e. the rural population has declined.

Regarding PE (PWE times population), the reduction in the migrant population over the decade 2000–2010 is larger than the corresponding reduced PE in non-migrants (Fig. 3). Using Eq. (3) we calculate that 58% (57%–59%) of the reduced exposure from changing household fuel pattern over the decade, ΔPE_{tot}, can be linked to migration, while the remaining fraction is due to a genuine fuel switch in the non-migrant population. Fuel switch among non-migrants may have happened as a result of several factors. An estimated 11% of the non-migrant population in 2010 had been reclassified from rural to urban in administrative terms (not necessarily with respect to their hukou) during the ten year period, implying a likely upgrading of housing for this group. In addition, land may in actual fact have been urbanized (and housing upgraded) while not formally reclassified. This development is driven by a number of policies promoting urbanization or targeting rural development, as outlined in China's recent five year plans. For instance, policies targeting an efficient use of rural land have been shown to entail centralization of rural housing, again entailing upgrade of energy services for rural populations (Huang et al., 2013). We suggest that structural policies leading to formal and informal urbanization may have had a larger impact on household fuel switch in China during 2000–2010 than policies specifically targeting household fuel use as such, as, e.g., banning of household coal in cities and subsidy programs for biogas digesters, policies which may not always be effectively implemented at the local level (Gan and Yu, 2008; Ma, 2011; Zhang and Smith, 2007).

The health benefits associated with the calculated ΔPWE values for various population groups are shown in Table 5. For the total population we estimate that about 64,000 (53,000–78,000) premature deaths due to cardiopulmonary diseases and lung cancer are avoided annually as a consequence of the changes in $PM_{2.5}$ exposure in China (71% (68%–76%) of avoided cases are CPD deaths). This translates to 31 (26–37) billion USD, which is approximately 0.4% of China's GDP in 2010. Applying the exposure–response function from Cao et al. (2011) we arrive at substantially higher health benefits, an estimated 409,000 (195,000–704,000) avoided CVD deaths, worth 197 (94–338) billion USD.

In addition to estimates for the various sub-populations addressed by this study, estimates of the impact of the two counterfactual cases, 100% urbanization and 100% clean fuels, are included in Table 5. In the urbanization experiment the resulting PWE in 2010 is estimated at

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Table 5
Population weighted exposure (PWE) in 2000 and 2010, ΔPWE over the same period (g/m^3 $PM_{2.5}$), population size in 2010, and health benefits associated with ΔPWE in terms of annual avoided cases and monetized cases. See text for definition of migrants.

	PWE 2000	PWE 2010	ΔPWE	Pop 2010 (million)	Annual avoided cases of CPD and lung cancer (1000)	Annual health benefit (billion USD)	Annual health benefit (USD/person)
All China	291 (201–398)	240 (165–328)	52 (36–70)	1333	64 (53–78)	31 (26–37)	23 (19–28)
All migrants	297 (204–407)	174 (117–242)	123 (87–165)	261	34 (29–41)	16 (14–20)	62 (53–75)
All non-migrants	290 (200–396)	256 (177–349)	34 (23–47)	1072	33 (27–41)	16 (13–20)	15 (12–18)
Migrants from rural to urban	361 (251–490)	147 (97–207)	214 (154–283)	137	30 (26–35)	14 (12–17)	104 (91–124)
Migrants to Guangdong	309 (209–429)	117 (74–171)	192 (135–258)	36.8	8.5 (7.4–10.0)	4.1 (3.5–4.8)	110 (96–131)
Migrants to Zhejiang	317 (216–436)	118 (75–171)	199 (141–266)	19.9	4.7 (4.1–5.6)	2.2 (2.0–2.7)	113 (98–134)
Migrants to Jiangsu	315 (220–426)	160 (112–217)	155 (108–209)	18.2	3.0 (2.5–3.7)	1.4 (1.2–1.8)	79 (65–97)
Migrants to Shandong	308 (216–417)	203 (140–276)	106 (76–141)	13.7	1.9 (1.6–2.3)	0.9 (0.8–1.1)	66 (56–79)
Migrants to Shanghai	256 (177–348)	94 (59–137)	161 (117–211)	12.7	2.9 (2.6–3.5)	1.4 (1.2–1.7)	111 (98–131)
Migrants to Sichuan	320 (224–434)	199 (138–270)	122 (86–163)	11.7	1.4 (1.1–1.7)	0.7 (0.6–0.8)	56 (47–68)
Migrants to Fujian	309 (210–427)	120 (75–174)	189 (134–253)	11.1	2.5 (2.2–3.0)	1.2 (1.0–1.4)	108 (95–128)
Migrants to Beijing	251 (176–339)	150 (104–202)	102 (72–136)	10.5	1.3 (1.1–1.6)	0.6 (0.5–0.8)	59 (49–74)
100% urbanization	240 (165–328)[a]	156 (104–217)	84 (61–111)[b]	1333	137 (119–164)	66 (57–79)	50 (43–59)
100% clean fuels	240 (165–328)[a]	82 (57–111)	158 (108–217)[b]	1333	328 (262–418)	158 (126–201)	118 (95–151)

[a] PWE in 2010, not 2000 (see text).
[b] PWE 2010 in baseline calculation minus PWE 2010 in the counterfactual case.

156 (104–217) g/m^3. In the clean fuel experiment the resulting PWE for the total population is estimated at 82 (57–111) g/m^3. The estimated benefit of 100% clean fuels, 158 (126–201) billion USD, corresponds to 2.2% of GDP. The lion's share, 77%, is due to exposure reductions in the rural population. In a previous study in Chongqing we also found that fuel switch in rural areas brought the largest health benefits (Wang et al., 2008).

On a per capita basis, the health benefit follows the same patterns as ΔPWE. Migrants reap the largest benefits, and especially if migrating from areas where solid fuels are commonly used into areas where clean fuels are commonly used.

In the sensitivity test where we altered the assumption about outdoor $PM_{2.5}$ levels, the PWE values for 2010 for all China, the migrant and the non-migrant population became only slightly lower, 236 (162–323) g/m^3, 161 (108–225) g/m^3, and 254 (175–346) g/m^3, respectively. The estimated fraction M in Eq. (3) becomes slightly higher than the baseline estimate, 59% (58%–61%). The ΔPWE values for 2010 for rural to urban migrants increased from 214 to 227 g/m^3. In the sensitivity test where we reduced PWE values for biomass users with 25% (the NISP adoption scenario), the PWE values for 2010 for all of China, the migrant and the non-migrant population became markedly lower, 201 (139–275) g/m^3, 156 (105–216) g/m^3, and 212 (147–289) g/m^3, respectively. M in Eq. (3) becomes 31% (30%–32%). In the sensitivity test where we assume that only 5% of the migrants are children <10 y of age, PWE values are not changed, but the fraction M in Eq. (3) increases to 67% (66%–68%).

4. Discussion

To our knowledge there is only one study of population exposure of migrants in China, thus there is little data with which we can compare our estimated PWE values for migrants. Lejnarova (2012) measured concentration levels of $PM_{2.5}$ in indoor and outdoor environments and time–activity pattern for migrant people in urban, sub-urban, and rural districts in Shanghai. Exposure in transit and indoor micro-environments away from home, where people spent on average 2.7–7.0 h per day, was not included in the study. Shanghai is among the most economically developed Chinese provinces and its migrant population may not be representative of the average migrant in China. In the random sample of 54 households all used gas or electricity for cooking in 2011. The migrant population in urban Shanghai had smaller living space (dwellings) and lower income than the urban average. Migrants in rural areas had smaller living space but similar income as the rural average. The annual average exposure (in measured micro-environments) for migrants living in urban Shanghai was about 80 g/m^3. For migrants living in sub-urban and rural Shanghai the level was about 70 g/m^3. In our calculation Shanghai is defined as South, thus the corresponding values applied in our calculations are 84 (53–122) g/m^3 (urban) and 55 (38–74) g/m^3 (rural).

In the lack of detailed information we have assumed that the fuel use distribution and the corresponding exposure level among migrants are the same as in the total population in the given region, both regarding their current residence and their home province. We also assume that mortality rates in migrants are similar to the average in the total population. If migrants are not a representative sample of the population in the province in which they live in 2010 and were not representative of their home province population when they left, these assumptions may lead to erroneous estimates.

Representativeness may be questioned for several reasons. First, the age of migrants is likely skewed towards younger adults. Whereas data on age is not available for the migrant population in the 2010 Census data, previous studies confirm a lower average age in migrant populations (Willmore et al., 2012). According to Mestl et al. (2007a) the annual average exposure level for adults 15–64 y of age is somewhat lower than in the elderly and in small children for those using biomass, both for rural and urban areas. The differences are however not large and using separate PWE values for the age groups would probably not affect our results very much, had the detailed age group data been available. Second, it may be that migrants are not representative of the population in the area from where they left. Regarding the level of education of migrants this is somewhat higher than in the general employed population in China (ACMR, 2012), especially for intra-provincial migrants (Fig. S3). As education level and access to clean household fuels are known to correlate (Jiang and O'Neill, 2004), one could speculate that if migrants have a higher education level, their fuel profile would be skewed towards the cleaner fuels compared with the average population. We do not know, however, the education level of migrants when they left. Moreover, migrants are likely to have lower income than the average population (Wong et al., 2007) and may therefore settle down in poorer and less well developed areas. Chai and Chai (1997) and Zhu (2007) found that the living space of migrant workers often is much smaller compared to the permanent residents. This could point towards higher HAP exposure, since room volume, ventilation and housing characteristics are important factors affecting pollutant concentrations (Lejnarova, 2012). On the other hand, many publications on living conditions for migrant populations focus on the rural–urban migrants and often the more marginalized groups among them (e.g. Pai, 2013 and references therein). In the current paper we include all migrants, of which nearly 30% are urban–urban intra-provincial migrants (the percentage varies considerably across host provinces;

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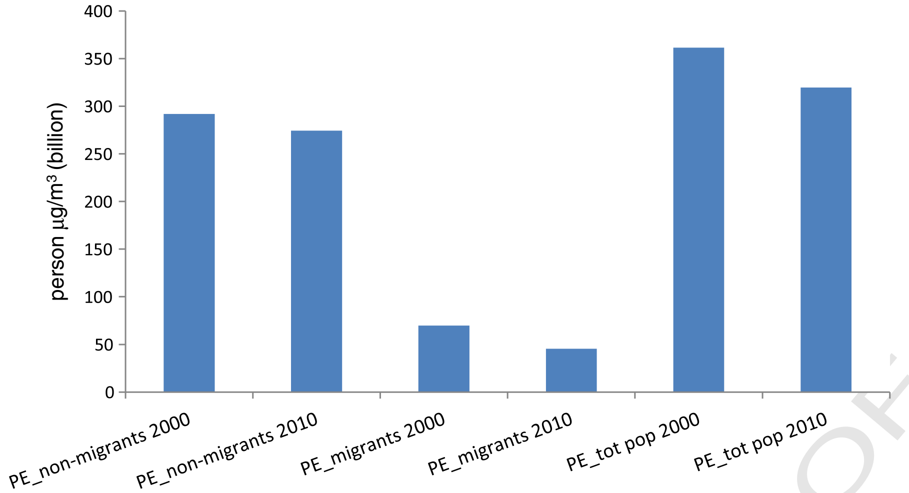

Fig. 3. Population exposure (PWE × population) for migrants, non-migrants, and the total population in 2000 and 2010 (billion person g/m^3 $PM_{2.5}$).

Table 2). Among these, quite many (13.4%) report that the reason they migrate is that their homes are being demolished or moved, which could in fact imply that they are moving into dwellings of a higher standard, including access to gas and electricity. If so, a higher share of these migrants is likely to have clean fuels as compared to the average urban population in the region where they settle down. A study based on China Census data for 2000 by Jiang (2006) shows that migrants who move because their dwellings are demolished generally improve their living condition, although some experience a worsening. Jiang (2006) compares housing facilities (including cooking fuels) among migrants settled in urban areas with those of the permanent urban population (with long-term urban hukou), those who had become permanent citizens the last 5 years (recent urban hukou), and the total urban population. The share of urban migrants using clean cooking fuels (gas or electricity) was higher than for the total urban population. The permanent urban population was worst off in terms of clean fuels and also had a higher share of slum dwellers than the urban migrant population. The share of clean fuels was highest among those who had become permanent citizens over the last 5 years. Jiang (2006) not only concludes that migration in general may help people achieve better living conditions, but also finds that the hukou status affects the living conditions of migrants, with an urban hukou vouching for better living conditions.

In summary, given the available data, we have little information to establish whether the assumption about representativeness biases our results in any particular direction. If the pattern described in Jiang (2006) for 2000 holds for 2010, the overall exposure to HAP of migrants in 2010 may be overestimated (as migrants to cities on average would have a higher share of clean fuels than urban average). On the other hand, their exposure in 2000 may also be overestimated (it may be the slightly better off who leave), hence any bias in ΔPWE for migrants, and by consequence in the ΔPWE for non-migrants, should not be very large. ΔPWE for the total population is not affected.

We use average mortality rates for 2010 for all population groups. In reality mortality rates for sub-populations (e.g., the migrants, the rural populations) are likely to differ from the country average, but we don't know how. Data for 2003 shows that the rural mortality rate for CVD at that time was similar to the urban average, whereas the mortality rate for pulmonary heart disease was considerably higher. The mortality rate for lung cancer was somewhat lower in rural areas (MoH, 2004). If the mortality rate for migrants is higher than the China average our approach underestimates the health benefits of exposure reductions in migrants.

The implication of the simplifying assumption made above that none of the migrants had left in 2000 is simply that the exposure reduction may have taken place over a period of more than 10 y for some of the migrants, and that the exposure reduction over this period may be slightly different than the ΔPWE for the migrants calculated here.

A possible source of error in our calculation is the assumption that PWE of the twelve population segments in Table 3 is constant over the decade 2000–2010. When we attempt to take into account the possible changes in rural and urban ambient PM levels over the decade, this does not, however, affect our estimates very much. In the sensitivity test where we reduce the 2010 PWE of biomass users, the reduction in PWE in the total population as well as the health benefit increases considerably. The benefit is less skewed towards the migrants, thus the fraction M (Eq. (3)) decreases. When we test how a lower fraction of children in the migrant population affects our results, M increases.

Another potential cause of error when it comes to the estimated levels of PWE is the assumed $PM_{2.5}/PM_{10}$ conversion ratio. We use 0.5 (0.4–0.6), which is a conservative estimate (conservative in terms of resulting in lower PWE values). Using data from Chinese cities, Ho and Nielsen (2007) suggest a $PM_{2.5}/PM_{10}$ conversion ratio of 0.61 and a PM_{10}/TSP conversion ratio of 0.54. The PM_{10} estimates in Mestl et al. (2007a) that we use in this paper were based on measurements of either total suspended particulates (TSP), PM_{10}, or PM_4 in the various indoor or ambient settings. A PM_{10}/TSP conversion ratio of 0.7 was applied (see Mestl et al., 2006). In the current paper we choose to use a $PM_{2.5}/PM_{10}$ conversion rate of 0.5 to compensate for a possible overestimation of PM_{10} in Mestl et al. (2007a). A $PM_{2.5}/PM_{10}$ conversion ratio of 0.5 (combined with a PM_{10}/TSP conversion ratio of 0.7) renders approximately the same $PM_{2.5}$ estimates as would result from converting TSP figures to $PM_{2.5}$ using the ratios suggested by Ho and Nielsen (2007). Assuming another ratio for all figures in Table 3 does affect PWE and ΔPWE for the various sub-populations (in a linear proportional way), but does not affect the fraction M in Eq. (3), and only to a minor extent affect health benefit estimates. In reality, the annual average $PM_{2.5}/PM_{10}$ ratio likely varies between the regions, fuel categories, and the various microenvironments, which would affect our estimates in a more complicated way. For instance, Wang and Hao (2012) note that the $PM_{2.5}/PM_{10}$ ratio in urban air may be as high as 0.58–0.77 in some large cities and the observed rising contribution of fine particles to the PM load in China, mentioned above, likely lead to an increasing $PM_{2.5}/PM_{10}$ ratio in ambient air.

There may be uncertainties in the China Census data on household fuel use. According to the WHO Household energy database (WHO, 2010) there was nearly no increase in the percentage of urban households in China using clean fuels from 2000 to 2006 (the figure was approximately 64% for both years). This is in stark contrast to the figures in the China Census databases used here (Table 4). In the WHO database, the percentage of rural households using clean fuels increased from 27% to 38% from 2000 to 2006, hence indicating a higher penetration rate of clean fuels than in the China Census data. Obviously, lower rates of clean fuels in urban areas and higher rates in rural areas, would imply smaller reductions in PWE for rural–urban migrants and a smaller fraction M in Eq. (3).

In this study we do not differentiate between urban and peri-urban settings. Nearly 40% of the total urban population lives in towns, while less than a quarter (24%) of the urban migrants live in towns (Table S7). As seen in Table S7, fewer people in towns have got clean fuels compared with people in the cities. Among the three types of settings, City, town, and rural, coal use is highest in towns. It may be that outdoor air pollution is worse in some towns than in cities as a consequence of combined urban and rural types of pollution and a general lower technological standard. Due to the lack of exposure estimates we have not singled out the peri-urban population as a sub-group in this paper. As migrants are less likely to live in towns than the average urban citizen, doing so would probably not have altered the finding in this paper, namely that migration leads to alleviation of $PM_{2.5}$ exposure.

Our approach to estimating $PM_{2.5}$ exposure from outdoor and indoor pollution, building on the results in Mestl et al. (2007a, b) does not include a detailed account of exposure in working environments. Typically, the jobs that rural–urban migrants take are so-called 3D — Dirty, Dangerous, and Demeaning as formulated by Meng (2012). Previous studies have reported an unhealthy work environment in enterprises employing migrant workers, e.g., in terms of dust, toxic substances and poor ventilation (Wong et al., 2007). According to data from 1997, around 20% of migrants to Beijing actually lived at their work place (Jiang, 2006). Thus, the reduction in PWE estimated for migrants in the current paper may in reality be counteracted by an increased exposure in working environment for segments of this population group, a topic warranting future research. In addition to the need for taking into account the diversity in the migrant population in terms of socio-economic conditions and determinants of exposure to health damaging air pollutants, a higher geographical resolution in exposure calculations would increase the accuracy of our estimates. Finally, whereas our findings point towards large health benefits from the ongoing urbanization in China due to reduced $PM_{2.5}$ exposure, we do not in this paper address the many important questions related to how urbanization should proceed in the country in order to protect the health and livelihoods of all migrants as well as safeguarding the local, regional, and global environment.

5. Conclusions

In China, the combination of a transition to cleaner household fuels due to expansion of energy infrastructure resulting from land urbanization, individual household choices, and a massive rural–urban migration has led to substantive reductions in the population exposure to HAP in particular and to $PM_{2.5}$ in general over the decade 2000 to 2010. We find that about 60% of the reduction in the total population exposure is linked to migration. Regarding the changes in exposure there are large differences between population groups. We estimate that rural–urban migrants have been subject to the largest exposure reduction.

The health benefits associated with the reduced exposure to $PM_{2.5}$ over the decade are substantial. Policies ensuring a continued increased access to clean household fuels have the potential to bring even larger health benefits, particularly among rural populations. This paper also supports a policy of continued urbanization, given that this results in an increased access to clean household fuels. For those migrants working and living in polluted environments, the net exposure effect of migration is not known. We suggest that further studies of possible health damaging exposures to air pollution for sub-groups of migrants are needed.

Conflict of interest Q16

The authors have no actual or potential competing financial interests.

Acknowledgments

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2014.02.073.

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