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ORIGINAL ARTICLE

Future climate and adverse health effects caused by fine particulate matter air pollution: case study for Poland

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Abstract Ground level air pollution, especially fine particulate matter ($PM_{2.5}$), has been associated with a number of adverse health effects. The dispersion of $PM_{2.5}$ through the atmosphere depends on several mutually connected anthropogenic, geophysical and meteorological parameters, all of which are affected by climate change. This study examines how projected climate change would affect population exposure to $PM_{2.5}$ air pollution in Poland. Population exposure to $PM_{2.5}$ in Poland was estimated for three decades: the 1990s, 2040s and 2090s. Future climate conditions were projected by Regional Climate Model RegCM (Beta), forced by the general atmospheric circulation model ECHAM5. The dispersion of $PM_{2.5}$ was simulated with chemical transport model CAMx version 4.40. Population exposure estimates of $PM_{2.5}$ were 18.3, 17.2 and 17.1 μg/m^3 for the 1990s, 2040s and 2090s, respectively. $PM_{2.5}$ air pollution was estimated to cause approximately 39,800 premature deaths in the population of Poland in the year 2000. Our results indicate that in Poland, climate change may reduce the levels of exposure to anthropogenic particulate air pollution in future decades and that this reduction will reduce adverse health effects caused by the air pollution.

Keywords Air pollution · Particulate matter · Climate change · Health · Exposure · Poland

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Introduction

Fine particulate matter ($PM_{2.5}$, particulate matter with an aerodynamic diameter of less than 2.5 μm) causes adverse health effects worldwide. The previous health impact assessment studies have estimated that $PM_{2.5}$ causes over 800,000 premature deaths annually, worldwide (Cohen et al. 2005), and around 350,000 premature deaths in Europe alone (Watkiss et al. 2005). The World Health Organization (WHO) estimates that, worldwide, $PM_{2.5}$ "*cause[s] about 8 % of lung cancer deaths, 5 % of cardiopulmonary deaths and about 3 % of respiratory infection deaths*" (WHO 2009). In high-income countries, ambient $PM_{2.5}$ is the most significant environmental health risk (WHO 2009).

Climate change refers to the change in the state of the climate, both due to natural variability and as a result of human activity (IPCC 2007a). The main indicators of climate change are (1) the concentrations of greenhouse gases (GHG) and aerosols, (2) land cover and (3) radiation (IPCC 2007a). As a result of climate change, the global temperature is expected to increase 1.8–4.0 °C between

1980–1999 and 2090–2099, depending on emission scenarios (IPCC 2007b). Climate change is one of the main environmental challenges for humankind in the twenty-first century.

$PM_{2.5}$ air pollution and climate change are linked on several levels. Particulate matter (as an aerosol) cools the climate (IPCC 2007a). Additionally, the same source emits both GHG and $PM_{2.5}$. This means that the mitigation of either $PM_{2.5}$ or GHG emissions will likely also affect the other (e.g. ApSimon et al. 2009). The integrated assessment models in place, such as GAINS (Greenhouse Gas and Air Pollution Interactions and Synergies, http://gains.iiasa.ac.at/gains/), assess cost-effective mitigation strategies jointly for $PM_{2.5}$ and GHG emissions.

Climate change affects $PM_{2.5}$ air pollution concentrations through changes in meteorology. The formation of secondary $PM_{2.5}$ and dispersion of $PM_{2.5}$ depends on meteorological parameters, and a change in these parameters will also change the formation and dispersion of PM. A number of studies have examined this issue (e.g. Racherla and Adams 2006; Tagaris et al. 2007, 2009). The review published by Ebi and McGregor (2008) concluded that climate change would increase ozone concentrations if the precursor gas emissions did not change in future decades. Regarding $PM_{2.5}$, they considered evidence less robust due to lack of published studies. Another review by Jacob and Winner (2009) concluded that different published studies estimate changes of ± 1 $\mu g/m^3$ in annual mean $PM_{2.5}$ concentrations due to climate change in the United States (US) and Europe, with little consensus between studies.

In this study, we examined on how climate change would affect exposure to and adverse health effects caused by $PM_{2.5}$ air pollution in Poland. We calculated $PM_{2.5}$ concentration over Poland for three decades: the 1990s, 2040s and 2090s. To account for the impact of climate change, the same population, anthropogenic air pollution emission and background health data were used for all the decades. The exposure for $PM_{2.5}$ across Poland and in different Polish provinces was predicted for each year.

Materials and methods

Climate simulations

For climate simulations, a three-dimensional hydrostatic regional climate model, RegCM3, was employed. The RegCM model was originally developed by Dickinson et al. (1989) and Giorgi and Bates (1989) and is used in a wide range of applications (Giorgi and Mearns 1999; Giorgi et al. 2006), including climate change simulations (Giorgi et al. 1992; Hirakuchi and Giorgi 1995; Diffenbaugh et al. 2005; Gao et al. 2006; Meleux et al. 2007).

The dynamical core of the RegCM3 is based on the hydrostatic version of the NCAR-PSU Mesoscale Model version 5 (MM5) (Grell et al. 1994). In this study, the Grell convective scheme (Grell 1993) with the Fritsch and Chappell (1980) closure assumption was adopted. The RegCM3-beta version (Torma et al. 2011), designed for high-resolution simulations (10 km $\times$ 10 km), was implemented for the modelling domain, centred around Poland (52.00°N, 19.30°E) on a grid with 120 $\times$ 109 points and a resolution of 10 km (Reizer and Juda-Rezler 2010). The map projection choice was Lambert Conformal Conic while the domain's vertical profile contained 18 layers of varying thickness, extending up to 50 hPa (study area is presented in Fig. 1).

To assess meteorological boundary conditions in relation to the regional climate model, we used a double nesting approach, based on the EU-FP6 project ENSEMBLES (http://ensembles-eu.metoffice.com/) simulations. The ECHAM5 global climate model (Roeckner et al. 2003), forced under the SRES A1B IPCC greenhouse gas emission scenario (Nakicenovic et al. 2000) with a 300-km resolution, drove a Europe-wide version of RegCM3 with a 25-km resolution, which in turn drove the RegCM3 10 km version applied in this study. The ECHAM5-RegCM3 European simulations were performed at Abdus Salam International Centre for Theoretical Physics (ICTP). The meteorological output fields of RegCM3 were used to drive offline the chemical transport model (CTM) CAMx.

The results of climate simulation are given in Reizer and Juda-Rezler (2010) and Huszar et al. (2011). They calculated warming during the summer (2 m temperature, T), which ranged from 0.4 to 1 °C in the 2040s, and from 2 to 3 °C in the 2090s, in comparison with the 1990s. The summer shortwave radiation levels showed slight decrease (2–10 W/m^2) in the 2040s and a slight increase in the same kind in the 2090s. Possible changes in horizontal and vertical mixing were accounted for by the ventilation coefficient, which is defined as the product of horizontal wind speed at the surface and planetary boundary layer height (PBLH), according to Rao et al. (2003). In the 2040s, the ventilation coefficient increased during winter in the southern area of Poland (the area of highest industrial emission) by up to 400 m^2/s while in autumn, there was no change in the ventilation coefficient for this region and a moderate decrease (up to 200 m^2/s) in the rest of the country. In the 2090s, a decrease in up to 400 m^2/s was observed for both seasons (Huszar et al. 2011).

Emissions

In order to study the impact of climate on air quality alone, the anthropogenic primary PM from the year 2000 and its precursors [CO, NO_x, SO_2, NH_3, non-methane volatile

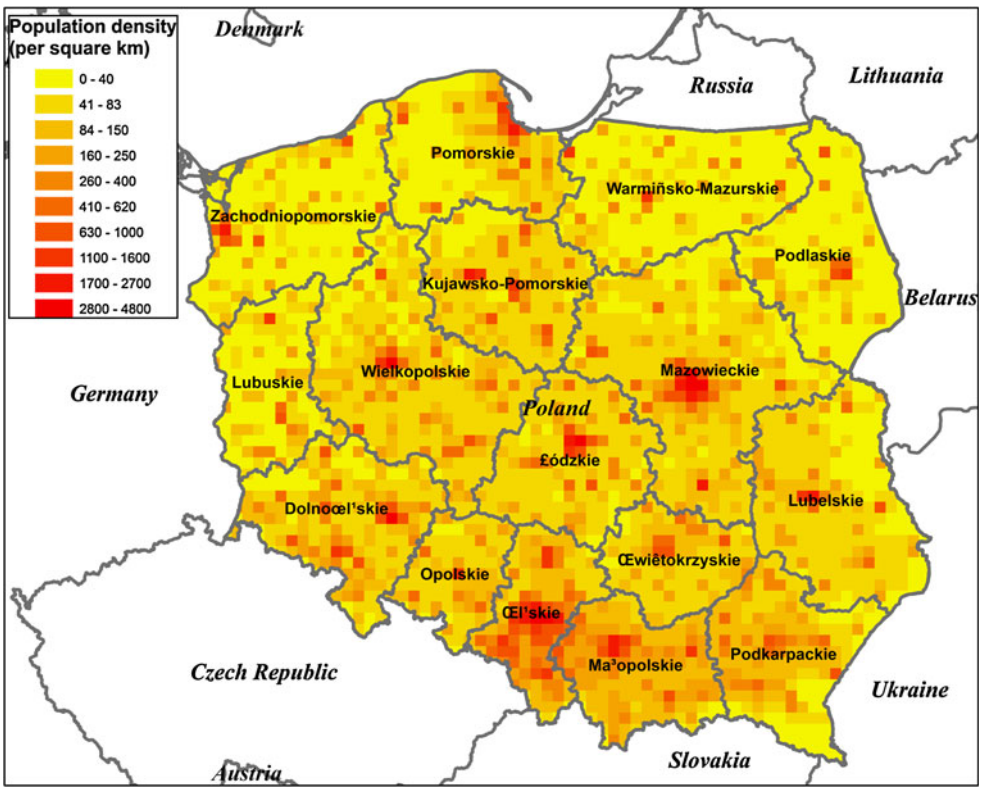

Fig. 1 The study area, location of the provinces and population density for Poland

organic compounds (NMVOC) emissions] were used in all simulations. Anthropogenic emissions for Poland were calculated with the emission model EMIL (EMIssion modeL) (Trapp et al. 2010). EMIL follows a modular setup of the SMOKE emission model, which is a part of USEPA Models-3 Community Modelling and Analysis System (CMAS) (Byun and Schere 2006).

Area, mobile and point sources are calculated by different modules and merged into a single output file. For area (municipal, agricultural), mobile and small industrial sources (plants with a thermal input of less than or equal to 50 MWth), the model generates gridded 2D emissions. This is based on detailed emission source inventory that was composed for the reference year of 2000. It used a 1 km × 1 km resolution, meteorological data and terrain characteristics as well as data on population density, sector-specific activity, fuel demands and characteristics, and sector-dependent emission factors for Polish sources. For large combustion plants (LCPs) with a thermal input greater than 50 MWth, a 3D emission database was created. It contains data for 220 stacks and consists of the following stack data: name of the LCP, stack number, geographical coordinates, installed technological units, stack height and diameter, temperature of exhaust gases, velocity of exhaust gases and total emission amount. The vertical distribution is calculated in CAMx via Plume in Grid (PiG) algorithms using stack and meteorological data. This leads to a more realistic spatial and temporal disaggregation of emissions from LCPs.

For other countries belonging to the model domain, the anthropogenic emission database is based on the European Monitoring and Evaluation Program (EMEP) emission inventory (Vestreng et al. 2007). It is available in gridded form on a 50 km × 50 km EMEP grid system that is based on a polar-stereographic projection. The inventory contains the total annual emission of PM precursors and primary PM, reported as $PM_{2.5}$, and coarse particles (2.5–10 μm). The emissions are distributed over 11 SNAP97 source sectors (Selected Nomenclature for sources of Air Pollution). SNAP97 is a standard defined by the CORINAIR guidebooks, which ensure that emissions reported by different nations are comparable (EEA 2007). These annual total emissions were further disaggregated by time and space. The temporal distribution was determined via sectoral emission profiles from the inventory by Winiwarter and Zueger (1996). For every sector, different distributions accounting for the month, the day of the week and the hour of the day were applied. For the neighbouring countries of Austria, Czech Republic, Hungary and Slovakia the EMEP data were scaled down to a spatial resolution of 5 km × 5 km. An emission inventory by Winiwarter and Zueger (1996) for these countries was used as the basis for the spatial disaggregation of emission within the 50 km × 50 km EMEP grid cells. The resulting emission

data were converted into the Lambertian projection. There was no vertical distribution of emission; the whole emission flux was assumed to be released into the surface layer as the area source. Finally, the chemical speciation for NO_x, NMVOC and $PM_{2.5}$ was done, for compatibility with the CBM-IV mechanism of Comprehensive Air quality Model with extensions (CAMx) (see Krüger et al. 2008).

Biogenic emissions were calculated as functions of meteorological parameters and land use. Thus, the biogenic emissions varied with changes in climate. Although changes in land use may be expected in the future, we did not include such changes in our study and kept the land-use data constant. The data from the terrain pre-processor of the regional mesoscale model MM5 (http://www.mmm.ucar.edu/mm5/) were used. They distinguish 25 land-use categories and are based on global data from the United States Geological Survey (USGS). Biogenic emissions of isoprene and monoterpenes were calculated as functions of 2 m temperature, while global radiation and land-use categories were calculated with the procedure proposed by Guenther et al. (1993). The procedure used for biogenic emission calculations is reported in Katragkou et al. (2010).

Air quality modelling

The air quality model simulations were carried out with version 4.4 of the CAMx (http://www.camx.com). CAMx is the state-of-the-art three-dimensional chemical transport model that has undergone continuous refinement by the science community over the past decade. It is one of the leading CTMs used for policy and research applications related to ozone and aerosols (Morris et al. 2005; Tesche et al. 2006). CAMx has been applied widely for a variety of purposes (e.g. Morris et al. 2005; Tesche et al. 2006; Andreani-Aksoyoglu et al. 2008; Lonati et al. 2010; Fountoukis et al. 2011). The model simulates the following major chemical and physical processes that are thought to be important for understanding atmospheric trace gas transformations and distributions: (1) emission injections, (2) horizontal and vertical advection, (3) horizontal and vertical diffusion, (4) dry and wet depositions, (5) gas-phase and aqueous-phase chemical reactions and (6) aerosol dynamics, thermodynamics and chemistry. CAMx uses a mass conservative and consistent transport numerics in parallel processing. It allows for integrated assessments of gaseous and particulate air pollution over many scales ranging from suburban to continental. The chemistry mechanism invoked is Carbon Bond v. 4 (CB-IV) (Gery et al. 1989), including 117 reactions—11 of which are photolytic—and up to 67 species (37 state gases, up to 18 state particulates and 12 radicals). For aerosol processes, the ISORROPIA thermodynamic equilibrium model (Nenes et al. 1998) is used.

In this study, CAMx was off-line coupled with RegCM3-beta using an interface, which reads the basic meteorological parameters from RegCM3 and converts them into a format accepted by CAMx. These parameters include fractional land-use distribution, including 11 land-use categories, three-dimensional gridded fields of height/pressure, horizontal wind components, temperature, water vapour concentrations, cloud parameters, vertical diffusivity and two-dimensional surface temperature (ENVIRON 2006). The same horizontal modelling grids with Lambert Conformal Conic map projection were used for RegCM3 and CAMx models in order to avoid interpolations between grids. Vertically, 18 σ-levels were used in RegCM3 while CAMx was run over the lowermost 12 levels of varying thickness. Layer 1 was 36 m deep and the uppermost layer was 1.2 km thick extending to about 6.5 km. The lateral chemical boundary conditions were provided by the 50 km × 50 km resolution CAMx simulations performed at University of Natural Resources and Life Sciences in Vienna (Krüger et al. 2008) under the EU-FP6 project CECILIA (http://www.cecilia-eu.org/). Top boundary conditions for all gaseous species were kept constant, with no seasonal variation and annual variability; ozone was set to 40 ppb, NO_2 to 1 ppb and CO to 200 ppb.

The results of RegCM3-beta/CAMx simulations were evaluated for the calendar year 2000 using surface observations from 17 monitoring sites from the EMEP (http://www.emep.int) and AirBase (http://acm.eionet.europa.eu/databases/airbase) databases. For winter, annual and daily mean PM_{10} concentrations and fractional bias (FB) model results amounted to −2.7, −39.8 and −26.0 %, fractional error (FE) amounted to 38.1, 41.9 and 62.3 % and index of agreement (IA) amounted to 0.81, 0.61 and 0.57, respectively (Juda-Rezler et al. 2012). These results show that our modelling system fulfils the skill criteria of FB $\leq$ ±60 % and FE $\leq$ +75 % given by Boylan and Russell (2006) as well as the benchmark for IA $>$ 0.6 proposed by Gilliam et al. (2006), thus justifying the use of proposed modelling system for future time projections.

Population data and exposure calculations

The exposure of the population to $PM_{2.5}$ air pollution was calculated by combining population location data with concentration data. Population data were based on the Central Statistical Office of Poland population density estimates from the year 2004. Population densities across Poland are illustrated in Fig. 1 and presented in Table 1. The same 10-km spatial resolution was used in both $PM_{2.5}$ concentration and population data.

Two different exposure estimates were calculated:

- Average concentration;
- Population-weighted concentration.

Population-weighted concentration took into account the correlation between $PM_{2.5}$ concentrations and the location of population and was calculated with the following equation:

$$E = \Sigma_i(C_i \times \mathrm{Pop}_i)/\Sigma_i(\mathrm{Pop}_i)$$

In this equation, E is exposure (Population-weighted concentration of $PM_{2.5}$) (unit: $\mu g/m^3$) in the given area, C is $PM_{2.5}$ concentration (unit: $\mu g/m^3$) and Pop is the number of people in grid cell i.

All the exposure calculations assume that outdoor concentration of $PM_{2.5}$ at the home addresses represents the population exposure. A similar assumption has been used in a number of previous studies assessing the scope of adverse health effects caused by $PM_{2.5}$ (e.g. Greco et al. 2007; Tainio et al. 2010). Exposure estimates were calculated for the population of Poland and for each of its provinces. For the latter, province boundaries from the year 1999 were used (Fig. 1). For the purpose of this study, the population of each 10×10 km grid belonged to the province in which the central point of the grid was located. The spatial calculations were done with ESRI ArcMap version 9.3. Calculated exposure data were combined with R version 2.9.2. The locations of provinces are illustrated in Fig. 1 and the population estimates for different provinces in Table 1.

Premature mortality due to $PM_{2.5}$

The magnitude of the adverse health effects caused by air pollution in Poland was illustrated by calculating premature mortality due to $PM_{2.5}$. Premature mortality was chosen because it has been widely used in previous impact calculations (e.g. Ebi and McGregor 2008; Tagaris et al. 2009; Tainio et al. 2010). The premature mortality rate was estimated by year and by Polish province. The premature mortality rate was calculated with following equation:

$$M = \mathrm{Mb} \times E \times (\mathrm{ER}/100)$$

In the equation, M is premature mortality due to $PM_{2.5}$ air pollution, Mb is background non-accidental mortality, E is exposure level and ER is exposure–response function (% change in the background mortality due to 1 $\mu g/m^3$ $PM_{2.5}$ exposure).

For the value of ER, we used the estimates from the European expert judgment study (Cooke et al. 2007; Tuomisto et al. 2008). In that study, six European air pollution experts estimated the exposure–response function for the $PM_{2.5}$ air pollution. The expert estimates were combined with two different methods: equal-weight (EDM) decision maker and performance-based (PDM) decision maker. The main difference between these two methods was that in EDM each expert's answer was combined, giving equal weight to

Table 1 Population and background mortality data for different provinces

Acronym	Province (Polish name)	Province (English name)	Population	Background non-accidental mortality
POL	Polska	Poland	37,055,702	338,950
DLS	Dolnośląskie	Lower Silesian	2,727,991	26,019
KPM	Kujawsko-Pomorskie	Kuyavian-Pomeranian	2,103,549	18,388
LBL	Lubelskie	Lublin	2,208,251	21,454
LBU	Lubuskie	Lubusz	973,692	8,170
LDZ	Łódzkie	Łódź	2,567,235	29,620
MLP	Małopolskie	Lesser Poland	3,448,207	26,563
MAZ	Mazowieckie	Masovian	5,337,043	47,687
OPL	Opolskie	Opole	964,950	8,751
PKR	Podkarpackie	Subcarpathian	2,161,395	16,843
PDL	Podlaskie	Podlaskie	1,265,272	10,836
POM	Pomorskie	Pomeranian	1,415,512	16,750
SLK	Śląskie	Silesian	4,365,674	41,996
SWK	Świętokrzyskie	Świętokrzyskie	1,165,895	12,459
WMZ	Warmińsko-Mazurskie	Warmian-Masurian	1,419,529	10,688
WKP	Wielkopolskie	Greater Poland	3,358,632	29,427
ZPM	Zachodniopomorskie	West Pomeranian	1,506,820	13,299

The population data are based on year 2004 population counts and background mortality to year 2000 data. For the location of provinces, see Fig. 1

each response, while in PDM the weights were based on so-called calibration questions that aimed to measure the performance of the experts. The estimated mean percentage change in mortality due to 1 μg/m^3 exposure change in $PM_{2.5}$ in Europe was 0.62 and 0.95 % for EDM and PDM, respectively (Tuomisto et al. 2008). In this study, the EDM estimate was used. The exposure–response function was assumed to be linear between exposure levels and the zero-exposure. Thus, we did not assume any threshold value for the mortality effect (WHO 2003).

The background non-accidental mortality (Mb) data were calculated for different provinces for the year 2000 (Table 1). Data were prepared by the National Institute of Public Health-National Institute of Hygiene (NIZP-PZH). The exposure and premature mortality calculations were done with the Monte-Carlo simulation program Analytica (version 4.4.).

Results

The predicted $PM_{2.5}$ concentrations across Poland in 2000 are illustrated in Fig. 2, and the change in $PM_{2.5}$ concentrations in Poland over several decades is presented in Fig. 3. The population-weighted concentration of $PM_{2.5}$ for the 1990s, 2040s and 2090s were 18.3, 17.2 and 17.1 μg/m^3, respectively (Table 2). The difference in concentrations between the first decade (1990s) and two other decades is statistically significant (*T* test, *p* value less than 0.05). On the average, the population-weighted concentration of $PM_{2.5}$ was 6 and 7 % lower in the 2040s and 2090s in comparison with the 1990s (Table 2).

The average change in $PM_{2.5}$ concentrations in different provinces is presented in Table 2. The average $PM_{2.5}$ concentrations were estimated to change from 2 to −13 % between 1990s and 2040s in different provinces. Only two provinces showed an increase in $PM_{2.5}$ concentration between the 1990s and the 2040s. Examining different regions of Poland separately revealed that the reduction in $PM_{2.5}$ concentration is highest in the eastern part of the country and lowest in the western part. In the 2090s, $PM_{2.5}$ concentrations were lower than the 1990s concentrations in all the provinces.

The $PM_{2.5}$ air pollution was estimated to cause 39,800 premature deaths in Poland in 2000 (Fig. 4). If all other parameters except climate were to stay constant, the premature deaths would be 35,800 and 34,900 per year in 2050 and 2100, respectively. Thus, lower exposure would result in a significant reduction in adverse health effects.

Discussion

In this study, we have estimated the change in exposure to $PM_{2.5}$ in Poland and the resulting premature mortality rates between current climate and projected future climate. We discovered that exposure to $PM_{2.5}$ and the associated adverse health effects could decrease in the future due to projected changes in climate. On the average, $PM_{2.5}$ concentrations were 6 % lower in the 2040s than in the 1990s. Expanding our analysis for the years 2091–2100 did not show significant change in PM concentrations. Examining the impact of climate change in different parts of Poland showed that the reduction in $PM_{2.5}$ concentration is highest in the eastern part of the country and lowest in the western part. The decrease in $PM_{2.5}$ concentration is assumed to be a result of the increase in winter precipitation (Juda-Rezler et al. 2012). In two provinces, $PM_{2.5}$ concentrations were expected to increase between the 1990s and the 2040s. $PM_{2.5}$ caused an estimated 39,800 premature deaths annually in Poland in the year 2000.

A reduction in 6 % in $PM_{2.5}$ concentrations between 1990s and 2040s may have a significant effect on calculating cost-effectiveness. In a cost -effectiveness study conducted in the US (Liao et al. 2010), the research team concluded that 4.1 billion dollars are required to offset the increase in $PM_{2.5}$ and ozone concentration between years 2000–2002 and 2049–2051. Although their study included both ozone and $PM_{2.5}$ and they estimated that the concentrations of both pollutants would increase in future, the study gives an indication of the cost of combating air pollution problems. Based on our study, the situation is different in Poland since climate change itself will presumably reduce $PM_{2.5}$ concentrations. This means that fewer preventative actions may be required to meet guidelines such as the European Union Air Quality Directive (2008/50/EC).

Our study was done with constant anthropogenic $PM_{2.5}$ and precursor gas emission, population and background health effect parameters. This does not present a realistic scenario because in the future all of these parameters will change. This modelling approach was chosen in order to separate the impact of climate change from other major factors. Therefore, these results should be viewed as more indicative of future $PM_{2.5}$ concentrations rather than as a realistic assessment of the situation in the 2040s or 2090s.

We did not specifically investigate the question of why the $PM_{2.5}$ concentrations are lower in future climate because such analyses have been performed in Juda-Rezler et al. (2012). In that study, the decrease was mainly associated with the increase in winter precipitation in future decades and small changes in mixing height were not found to significantly affect PM_{10}. Additionally, in the review by Jacob and Winner (2009), precipitation, which directly affects the main removal mechanism of PM, and wet deposition were assumed to play important role in decreasing the $PM_{2.5}$ concentrations. An increase in mixing height was also recognised to affect $PM_{2.5}$ concentration;

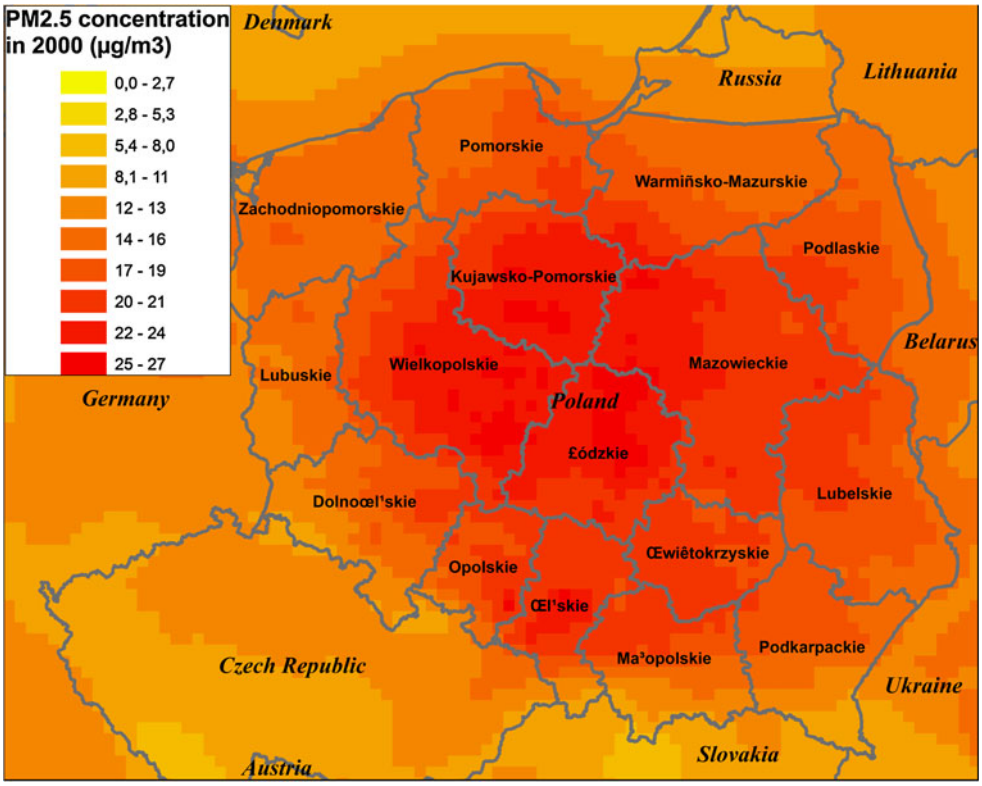

Fig. 2 $PM_{2.5}$ concentrations in Poland in 2000

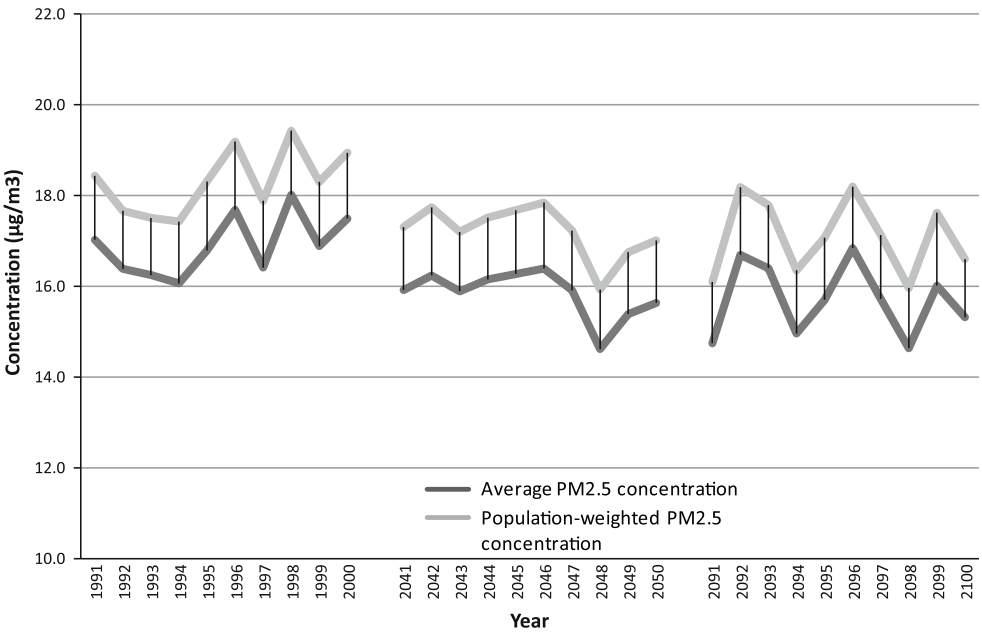

Fig. 3 Average $PM_{2.5}$ concentrations in Poland in different decades

however, the direction of change was specific to region and PM constituent (Jacob and Winner 2009). In our study, precipitation change was identified as the major reason for lower $PM_{2.5}$ concentrations in 2040s.

Climate change did not, however, have any further significant impact on $PM_{2.5}$, despite the further increase in precipitation in 2090s. The cause of this phenomenon is not easy to explain. As was concluded by Juda-Rezler et al. (2012), the main difficulty of interpreting projected PM levels has to do with the fact that the effects of climate change on PM depend on PM composition and the interrelated, complex processes that PM components undergo in the atmosphere. As we simulated only $PM_{2.5}$ mass concentration, the lack of knowledge about the particulate matter species limits the interpretation. One should also consider that all presented concentrations are averaged,

Table 2 Exposure for $PM_{2.5}$ in different provinces, in three different decades

Province	Population-weighted $PM_{2.5}$ concentration			Difference (%) in comparison with 1991–2000	
	1991–2000	2041–2050	2091–2100	2041–2050	2091–2100
Poland	18.3	17.2	17.1	−6	−7
Lower Silesian	15.5	15.1	14.8	−2	−4
Kuyavian-Pomeranian	21.3	19.6	19.7	−8	−7
Lublin	17.6	16.1	15.9	−8	−10
Lubusz	14.1	14.4	14.0	2	−1
Łódź	22.1	20.6	20.5	−7	−7
Lesser Poland	17.6	16.7	16.6	−5	−6
Masovian	20.0	18.3	18.4	−8	−8
Opole	17.9	17.2	16.9	−4	−6
Subcarpathian	16.1	15.2	14.9	−6	−7
Podlaskie	16.0	14.0	13.9	−13	−13
Pomeranian	15.6	14.5	14.6	−7	−7
Silesian	20.6	19.9	19.7	−3	−4
Świętokrzyskie	18.8	17.5	17.3	−7	−8
Warmian-Masurian	15.7	14.1	14.2	−10	−9
Greater Poland	20.8	19.7	19.5	−5	−6
West Pomeranian	14.1	14.1	13.7	0	−3

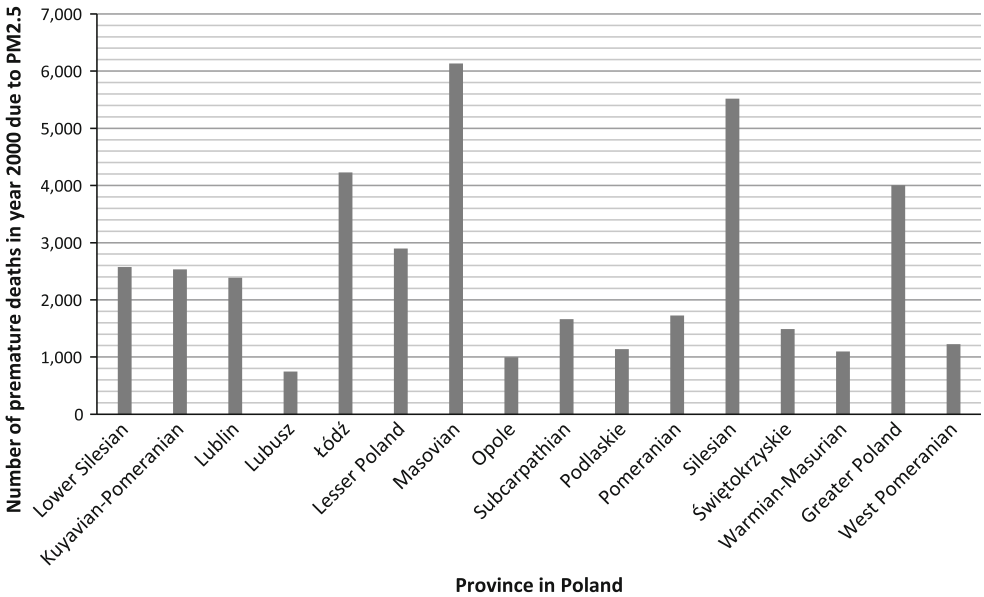

Fig. 4 Premature death estimates for year 2000 for different provinces. For the location of provinces, see Fig. 1

which substantially complicates the discussion. Although it seems that in the 2090s a precipitation-driven decrease in $PM_{2.5}$ was suppressed by other processes, these processes were probably driven by the increased T: (1) faster SO_2 oxidation to sulphate aerosol; (2) higher water vapour content related to the rise of temperatures, which also results in faster SO_2 oxidation; and (3) higher biogenic emission of hydrocarbons being precursors of SOA (Secondary Organic Aerosol). A decrease in the ventilation coefficient during winter and autumn (when the highest emissions of PM occurs) is another likely cause suppressing the aforementioned precipitation-driven decrease in $PM_{2.5}$.

Comparison to previous studies

Several studies have considered the impact of climate change on $PM_{2.5}$ concentrations in different geographical areas and with different model assumptions. For example, Racherla and Adams (2006) estimated global changes in

ozone and particulate matter concentrations between 1990s and 2050s. They concluded that annual average surface concentrations of particulate matter were increasing and decreasing in different parts of the world so that there was in practise zero change in $PM_{2.5}$ concentration on a global scale. However, they did estimate a reduction in the lifetime of $PM_{2.5}$ of 2–18 % (Racherla and Adams 2006). Racherla and Adams (2006) considered the changes in meteorological conditions between 1990s and 2050s as well as changes in natural emissions. Other main parameters were kept constant. For Poland, they estimated that the concentrations of some of the $PM_{2.5}$ components were increasing and that some were decreasing. They did not specifically investigate the reasons for these changes in Poland but seasonal-scale precipitation decrease in the region was assumed to be the likely reason for higher surface layer concentrations.

Tagaris et al. (2007) estimated $PM_{2.5}$ concentrations in the US for the year 2050 with two different models. The first model took into account changes in climate and the other model accounted for both climate change and emissions. They found out that in 2050, the annual average concentration of $PM_{2.5}$ would be approximately 1 % higher due to changes in the climate. When both meteorology and emissions reductions were taken into account, the $PM_{2.5}$ concentrations were 23 % lower in the year 2050. Tagaris et al. (2009) expanded assessment by estimating the adverse health effects caused by $PM_{2.5}$ and ozone in the US. They found out that 4,000 more premature deaths would occur in 2050 than in 2001 due to increased $PM_{2.5}$ concentrations. The same air pollution emissions data were used for both 2001 and 2050, so we may conclude that the increased mortality rate is due entirely to changes in climate conditions.

Uncertainties and limitations

We estimated $PM_{2.5}$ concentrations across Poland using a spatial resolution of approximately 10 km. The spatial variability of $PM_{2.5}$ concentrations is high due to local emissions. Using a spatial resolution of several kilometres mainly reveals the changes in background concentrations. However, people also inhale $PM_{2.5}$ air pollution next to emission sources, especially if the source has low emission height and is emitting particles near the general population (like traffic, for example). This might mean that population exposure for some of the $PM_{2.5}$ has been underestimated (Taimisto et al. 2011). However, since climate change is assumed to have the greatest effect on widely dispersed pollutants, we assume that this possible underestimation of exposure did not have any significant impact on our conclusions.

We focused on the change in climate and how that would affect $PM_{2.5}$ concentrations and adverse health effects caused by $PM_{2.5}$. Most of the previous studies mentioned in this article have estimated the change in both ozone and $PM_{2.5}$ concentrations (e.g. Racherla and Adams 2006; Tagaris et al. 2007). While the impact of climate change to $PM_{2.5}$ has been uncertain, most of the studies have estimated that ground level ozone would increase. However, when the health effects were taken into account, the increase in $PM_{2.5}$ concentration had an observed impact that was 15 times greater than ozone (Tagaris et al. 2009). In Poland, ozone limits were exceeded every year, mainly in unpolluted areas of the central and western parts of the country (IOS 2010). When compared to $PM_{2.5}$, ozone is not a significant health threat in Poland (Wagner et al. 2010), and therefore, it is unlikely that considering ozone in our calculations would change our conclusions.

Several other studies chose to vary parameters that we kept constant. Tagaris et al. (2007), for example, focused on changes in anthropogenic emissions. As has been discussed earlier, they found that the $PM_{2.5}$ concentration would decrease by 23 % in the USA from the years 2001 to 2050. Spracklen et al. (2009) estimated the change in wildfire smoke levels in USA, and they assumed that climate change would increase the area burned in the western USA by 54 %. These wildfires were then estimated to increase the summertime organic carbon concentrations by 40 % and elemental carbon concentrations by 20 % from 2000 to 2050. Biomass burning in Eastern Europe has been noticed to cause $PM_{2.5}$ concentration peaks across large areas of Europe (Niemi et al. 2009), and consequently, climate change could increase $PM_{2.5}$ episodes also in Poland.

Conclusions

Estimates indicate that $PM_{2.5}$ air pollution caused tens of thousands of premature deaths in Poland in year 2000. Climate change estimates in Poland suggest a decrease in exposure to $PM_{2.5}$ of approximately 6 % between 1990 and 2040 thus also lowering the rate of premature deaths caused by $PM_{2.5}$ by 6 %. Expansion of our analysis to the 2090s did not result in a similar reduction, indicating that the climate impact on $PM_{2.5}$ dispersion will occur in the coming few decades. These results will become extremely important in the process of planning cost-effective preventative actions in Poland.

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