Atmos. Chem. Phys. Discuss., https://doi.org/10.5194/acp-2017-751
Manuscript under review for journal Atmos. Chem. Phys.
Discussion started: 27 September 2017

Atmospheric Chemistry and Physics Discussions

Assessment and economic valuation of air pollution impacts on human health over Europe and the United States as calculated by a multi-model ensemble in the framework of AQMEII3

Ulas Im[1*], Jørgen Brandt[1], Camilla Geels[1], Kaj Mantzius Hansen[1], Jesper Heile Christensen[1], Mikael Skou Andersen[1], Efisio Solazzo[2], Ioannis Kioutsioukis[3], Ummugulsum Alyuz[4], Alessandra Balzarini[5], Rocio Baro[6], Roberto Bellasio[7], Roberto Bianconi[7], Johannes Bieser[8], Augustin Colette[9], Gabriele Curci[10,11], Aidan Farrow[12], Johannes Flemming[13], Andrea Fraser[14], Pedro Jimenez-Guerrero[6], Nutthida Kitwiroon[15], Ciao-Kai Liang[16], Guido Pirovano[5], Luca Pozzoli[4,2], Marje Prank[17,18], Rebecca Rose[14], Ranjeet Sokhi[12], Paula Tuccella[10,11], Alper Unal[4], Marta Garcia Vivanco[9,19], Jason West[16], Greg Yarwood[20], Christian Hogrefe[21], Stefano Galmarini[2]

[1] Aarhus University, Department of Environmental Science, Frederiksborgvej 399, DK-4000, Roskilde, Denmark.
[2] European Commission, Joint Research Centre (JRC), Ispra (VA), Italy.
[3] University of Patras, Department of Physics, University Campus 26504 Rio, Patras, Greece
[4] Eurasia Institute of Earth Sciences, Istanbul Technical University, Istanbul, Turkey
[5] Ricerca sul Sistema Energetico (RSE SpA), Milan, Italy
[6] University of Murcia, Department of Physics, Physics of the Earth, Campus de Espinardo, Ed. CIOyN, 30100 Murcia, Spain
[7] Enviroware srl, Concorezzo, MB, Italy
[8] Institute of Coastal Research, Chemistry Transport Modelling Group, Helmholtz-Zentrum Geesthacht, Germany
[9] INERIS, Institut National de l'Environnement Industriel et des Risques, Parc Alata, 60550 Verneuil-en-Halatte, France
[10] Dept. Physical and Chemical Sciences, University of L'Aquila, L'Aquila, Italy
[11] Center of Excellence CETEMPS, University of L'Aquila, L'Aquila, Italy
[12] Centre for Atmospheric and Instrumentation Research (CAIR), University of Hertfordshire, Hatfield, UK
[13] European Centre for Medium Range Weather Forecast (ECMWF), Reading, UK
[14] Ricardo Energy & Environment, Gemini Building, Fermi Avenue, Harwell, Oxon, OX11 0QR, UK
[15] Environmental Research Group, Kings' College London, London, UK
[16] Department of Environmental Sciences and Engineering, University of North Carolina at Chapel Hill, Chapel Hill, North Carolina, USA
[17] Finnish Meteorological Institute, Atmospheric Composition Research Unit, Helsinki, Finland
[18] Cornell University, Department of Earth and Atmospheric Sciences, Ithaca, USA
[19] CIEMAT. Avda. Complutense 40., 28040 Madrid, Spain
[20] Ramboll Environ, 773 San Marin Drive, Suite 2115, Novato, CA 94998, USA
[21] Computational Exposure Division, National Exposure Research Laboratory, Office of Research and Development, United States Environmental Protection Agency, Research Triangle Park, NC, USA.

Correspondence to: Ulas Im (ulas@envs.au.dk)

Abstract

The impact of air pollution on human health and the associated external costs in Europe and the United States (U.S.) for the year 2010 is modelled by a multi-model ensemble of regional models in the frame of the third phase of the Air Quality Modelling Evaluation International Initiative (AQMEII3). This is the first study known to use a common health assessment approach across the two continents. The modelled surface concentrations of O_3, CO, SO_2 and

$PM_{2.5}$ from each model are used as input to the Economic Valuation of Air Pollution (EVA) system to calculate the resulting health impacts and the associated external costs. Along with a base case simulation, additional runs were performed introducing 20% emission reductions both globally and regionally in Europe, North America and East Asia.

Health impacts estimated by different models can vary up to a factor of three in Europe (twelve models) and the United States (three models). In Europe, the multi-model mean number of premature deaths is calculated to be 414 000 while in the U.S., it is estimated to be 160 000, in agreement with previous global and regional studies. In order to estimate the impact of biases coming from each model, two multi-model ensembles were produced, the first attributing an equal weight to each member of the ensemble, and the second where the subset of models that produce the smallest error compared to the surface observations at each time step. The latter results in increase of health impacts by up to 30% in Europe, thus giving significantly higher mortality estimates compared to available literature. This is mostly due to a 27% increase in the domain mean $PM_{2.5}$ levels, along with a slight increase in O_3 by ~1%. Over the U.S., the mean $PM_{2.5}$ and O_3 levels decrease by 11% and 2%, respectively, when the optimal ensemble mean is used, leading to a decrease in the calculated health impacts by ~11%. These differences encourage the use of optimal-reduced multi-model ensembles over traditional all model-mean ensembles, in particular for policy applications.

Finally, the role of domestic versus foreign emission sources on the related health impacts is investigated using the 20% emission reduction scenarios applied over the source regions as defined in the frame of HTAP2. The differences are calculated based on the models that are common in the basic multi-model ensemble and the perturbation scenarios, resulting in five models in Europe and all three models in the U.S. A 20% reduction of global anthropogenic emissions avoids 54 000 and 27 500 premature deaths in Europe and the U.S., respectively. A 20% reduction of North American emissions foreign emissions avoids ~1 000 premature deaths in Europe and 25 000 premature deaths in the U.S. A 20% decrease of emissions within the European source region avoids 47 000 premature deaths in Europe. Reducing the East Asian emission by 20% avoids ~2000 premature deaths in the U.S. These results show that the domestic emissions make the largest impacts on premature death, while foreign sources make a minor contributing to adverse impacts of air pollution.

1. Introduction

According to the World Health Organization (WHO), air pollution is now the world's largest single environmental health risk (WHO, 2014). Around 7 million people died prematurely in 2012 as a result of air pollution exposure from both outdoor and indoor emission sources (WHO, 2014). WHO estimates 3.7 million premature deaths in 2012 from exposure to outdoor air pollution from urban and rural sources worldwide. According to the Global Burden of Disease (GBD) study, exposure to ambient particulate matter pollution remains among the ten leading risk factors. Air pollution is a transboundary and scale dependent challenge with global, regional, national and local sources, leading to large differences in the geographical distribution of human exposure. Short-term exposure to ozone (O_3) is associated with respiratory morbidity and mortality (e.g. Bell et al., 2004), while long-term exposure to

O_3 has been associated with premature respiratory mortality (Jerrett et al., 2009). Short-term exposure to particulate matter ($PM_{2.5}$) has been associated with increases in daily mortality rates from respiratory and cardiovascular causes (e.g. Pope and Dockery, 2006), while long-term exposure to $PM_{2.5}$ can have detrimental chronic health effects, including premature mortality due to cardiopulmonary diseases and lung cancer (Burnett et al., 2014). The Global Burden of Disease Study 2013 (GBD 2013) estimated 210 000 premature deaths per year associated with ambient O_3 and 2.9 million associated with anthropogenic $PM_{2.5}$ (Forouzanfar et al., 2015).

Changes in emissions from one region can impact air quality over others, affecting also air pollution-related health impacts due to intercontinental transport (Anenberg et al., 2014; Zhang et al., 2017). In the framework of the Task Force on Hemispheric Transport of Air Pollution (TF-HTAP), Anenberg et al. (2009) found that reduction of foreign ozone precursor emissions can contribute to more than 50% of the deaths avoided by simultaneously reducing both domestic and foreign precursor emissions. Similarly, they found that reducing emissions in North America (NA) and Europe (EU) reduced the ozone-related premature deaths more outside the source region than within (Anenberg et al., 2009). This result agrees with Duncan et al. (2008), which showed for the first time that emission reductions in NA and EU have greater impacts on ozone mortality outside the source region than within. Anenberg et al. (2014) estimates that 93–97 % of $PM_{2.5}$-related avoided deaths from reducing emissions occurs within the source region while 3–7 % occur outside the source region from concentrations transported between continents. In spite of the shorter lifetime of $PM_{2.5}$ compared to O_3, it was found to cause more deaths from intercontinental transport (Anenberg et al., 2009; 2014). In the frame of the second phase of the Task Force on Hemispheric Transport of Air Pollution (TF-HTAP2; Galmarini et al., 2017), an ensemble of global chemical transport model simulations calculated that 20% emission reductions from one region generally lead to more avoided deaths within the source region than outside (Liang et al., 2017). They calculated that source emission reductions avoided ozone- and $PM_{2.5}$-related premature deaths within the source region by 84% and 89%, respectively, indicating that most avoided mortality occurred within the source region in general.

Recently, Lelieveld et al. (2015) used a global chemistry model to calculate that outdoor air pollution led to 3.3 million premature deaths globally in 2010, while in Europe and North America, 381 000 and 68 000 premature deaths occurred, respectively. They have also calculated that these numbers are likely to roughly double in the year 2050 assuming a business-as-usual scenario. Silva et al. (2016), using the ACCMIP model ensemble, calculated that the global mortality burden of ozone is estimated to markedly increase from 382 000 deaths in 2000 to between 1.09 and 2.36 million in 2100. They also calculated that the global mortality burden of $PM_{2.5}$ is estimated to decrease from 1.70 million deaths in 2000 to between 0.95 and 1.55 million deaths in 2100. Silva et al. (2013) estimated that in 2000, 470 000 premature respiratory deaths are associated globally and annually with anthropogenic ozone, and 2.1 million deaths with anthropogenic $PM_{2.5}$-related cardiopulmonary diseases (93%) and lung cancer (7%).

In Europe, recent results show that outdoor air pollution due to O_3, CO, SO_2 and $PM_{2.5}$ causes a total number of 570 000 premature deaths in the year 2011 (Brandt et al., 2013a; 2013b). The external (or indirect) costs to society related to health impacts from air pollution are tremendous. OECD (2014) estimates that outdoor air pollution is costing its member countries USD 1.57 trillion in 2010. Among the OECD member countries, the economic valuation of air pollution in the U.S. was calculated to be ~500 billion USD and ~660 USD in Europe. In the whole of Europe, the total external costs have been estimated to approx. 800 billion Euros in year 2011 (Brandt et al., 2013a). These societal costs have great influence on the general level of welfare and especially on the distribution of welfare both within the countries as air pollution levels are vastly heterogeneous both at regional and local scales and between the countries as air pollution and the related health impacts are subject to long-range transport. Geels et al. (2015), using two regional chemistry and transport models, estimated a premature mortality of 455 000 and 320 000 in Europe (EU28 countries) for the year 2000, respectively, due to O_3, CO, SO_2 and $PM_{2.5}$. They also estimated that climate change alone leads to a small increase (15%) in the total number of O_3-related acute premature deaths in Europe towards the 2080s and relatively small changes (<5%) for $PM_{2.5}$-related mortality. They found that the combined effect of climate change and emission reductions will reduce the premature mortality due to air pollution, in agreement with the results from Schucht et al. (2015).

The U.S. Environmental Protection Agency estimated that in 2010 there were ~160 000 premature deaths in the U.S. due to air pollution (U.S. EPA, 2011). Fann et al. (2012) calculated 130,000 and 350,000 O_3- and $PM_{2.5}$ - related premature deaths in the U.S., respectively, for the year 2005. Caizzo et al. (2013) estimated 200 000 cases of premature death in the U.S. due to air pollution from combustion sources for the year 2005.

The health impacts of air pollution and associated external costs are estimated based on observed and/or modelled air pollutant levels. Observations have spatial limitations particularly when fine scale assessments are needed. The impacts of air pollution on health can be estimated using models, where the level of complexity can vary depending on the geographical scale (global, continental, country or city), concentration input (observations, model calculations, emissions) and the pollutants of interest that can vary from only few ($PM_{2.5}$ or O_3) to a whole set of all regulated pollutants. The health impact models normally used may differ in the geographical coverage, spatial resolutions of the air pollution model applied, complexity of described processes, the exposure-response functions (ERFs), population distributions and the baseline indices (see Anenberg et al., 2015 for a review).

Air pollution related health impacts and associated costs can be calculated using Chemical Transport Model (CTM) or with standardized source-receptor relationships characterizing the dependence of ambient concentrations on emissions. (e.g. EcoSense model: ExternE, 2005, TM5-FASST: Van Dingenen et al., 2014). Source-receptor relationships have the advantage of reducing the computing time significantly and have therefore seen extensive application in systems like GAINS (Amann et al., 2011). On the other hand, full CTM simulations have the advantage of better accounting for non-linear chemistry-transport processes in the atmosphere.

CTMs are useful tools to calculate the concentrations of health-related pollutants taking into account non-linearities in the chemistry and the complex interactions between meteorology and chemistry. However, the CTMs include different chemical and aerosol schemes that introduce differences in the representation of the atmosphere as well as differences in the emissions and boundary conditions they use (Im et al., 2015a,b). These different approaches are present also in the health impact estimates that use CTM results as basis for their calculations. Multi-model (MM) ensembles can be useful to the extent that allow us to take into consideration several model results at the same time, define the relative weight of the various members in determining the mean behavior, and produce also an uncertainty estimated based on the diversity of the results (Potempski and Galmarini, 2010; Riccio et al., 2013; Solazzo et al., 2013).

The third phase of the Air Quality Modelling Evaluation International Initiative (AQMEII3) project brought together fourteen European and North American modelling groups to simulate the air pollution levels over the two continental areas for the year 2010 (Galmarini et al., 2017). Within AQMEII3, the simulated surface concentrations of health related air pollutants from each modelling group serves as input to the Economic Valuation of Air Pollution (EVA) model (Brandt et al., 2013a; 2013b). This is the first study in our knowledge that uses a common approach across the two continents regarding health impact assessment of air pollution and in particular, their economic valuation (Andersen, 2017). The EVA model is used to calculate the impacts of health-related pollutants on human health over the two continents as well as the associated external costs. We adopt a multi-model ensemble (MM) approach, in which the outputs of the modelling systems are statistically combined assuming equal contribution from each model and used as input for the EVA model. In addition, the human health impacts (and the associated costs) of reducing anthropogenic emissions, globally and regionally have been calculated, allowing to quantify the trans-boundary benefits of emission reduction strategies. Finally, starting from the conclusions of Solazzo and Galmarini (2015), the health impacts have been calculated also using a so-called optimal ensemble of models, determined by error minimization and redundancy analyses in the framework of the AQMEII3 project. This provided us with an assessment that guaranteed for the absence of recursive or redundant results due to the lack of model independence that we can compare with the classically derived estimates based on model averaging.

2. Material and Methods

2.1. AQMEII

In the framework of the AQMEII3 project, fourteen groups participated to simulate the air pollution levels in Europe and North America for the year 2010. In the present study, we use results from the thirteen groups that provided all health-related species (Table 1: Solazzo et al., 2017a). The base-case emission inventories that are used in AQMEII for Europe and North America are extensively described in Pouliot et al. (2015). For Europe, the 2009 inventory of TNO-MACC anthropogenic emissions was used. In regions not covered by the emission inventory, such as North Africa, five modelling systems have complemented the standard inventory with the HTAPv2.2 datasets (Janssens-Maenhout et al., 2015). For the

North American domain, the 2008 National Emission Inventory was used as the basis for the 2010 emissions, providing the inputs and datasets for processing with the SMOKE emissions processing system (Mason et al., 2012). For both continents the regional scale emission inventories were embedded in the global scale inventory (Janssens-Maenhout et al., 2015) used by the global-scale HTAP2 modelling community so that to guarantee coherence and harmonization of the information used by the regional scale modelling community. The annual totals for European and North American emissions in the HTAP inventory are the same as the MACC and SMOKE emissions. However, there are differences in the temporal distribution, chemical speciation and the vertical distribution used in the models. The IFS model (Flemming et al., 2015) provided chemical boundary conditions. Galmarini et al. (2017) provides more details on the setup of the AQMEII3 and HTAP2 projects.

In addition to the base case simulations in AQMEII3, a number of emission perturbation scenarios have been simulated (Table 1). The perturbation scenarios feature a reduction of 20% in the global anthropogenic emissions (GLO) as well as the HTAP2-defined regions of Europe (EUR), North America (NAM) and East Asia (EAS), as explained in detail in Galmarini et al. (2017). To prepare these scenarios, both the regional models and the global IFS-MOZART model that provides the boundary conditions to the participating regional models have been operated with the reduced emissions. For example, for the NAM scenario, the IFS model makes a 20% reduction on the anthropogenic emissions from the HTAP-defined NAM region, which is then used by the regional models as boundary conditions, in addition to no emission perturbation for the European domain but the same perturbation for the North American domain. Therefore, the NAM scenario is used to see the impact of long-range transport of emissions from North America to Europe, while it shows the impact of the North American sources on the North American air pollutant levels.

In addition to individual health impact estimates from each model, a multi-model mean dataset (MM_m, in which all the modelling systems are averaged assuming equally weighted contributions) has been created for each grid cell and time step, hence creating a new model set of results that have the same spatial and temporal resolution of the ensemble-contributing members. In addition to this simple MM_m, an optimal MM ensemble (MM_{opt}) has been generated. MM_{opt} is created following the criteria extensively discussed and tested in the previous phases of the AQMEII activity (Riccio et al., 2012; Kioutsioukis et al., 2016; Solazzo and Galmarini, 2016), where it was shown that there are several ways to combine the ensemble members to obtain a superior model, mostly depending on the feature we wish to promote (or penalize). For instance, generating an optimal ensemble that maximizes the accuracy would require a minimization of the mean error or of the bias, while maximizing the associativity (variability) would require maximize the correlation coefficient (standard deviation). In this study, the sub-set of models whose mean minimize the mean squared error (*MSE*) is selected as optimal (MM_{opt}). The *MSE* is chosen for continuity with previous AQMEII-related works. The *MSE* is chosen in the light of its property of being composed by bias, variance and covariance types of error, thus lumping together measures of accuracy (bias), variability (variance) and associativity (covariance) (Solazzo and Galmarini, 2016). The minimum *MSE* has been calculated at the monitoring stations, where observational data

are available and then extended to the entire continental areas. This approximation might affect remote regions away from the measurements. However, considering that for the main pollutants (O_3 and $PM_{2.5}$) the network of measurements is quite dense around densely populated areas (where the inputs of the MM ensemble are used for assessing the impact of air pollutants on the health of the population), errors due to inaccurate model selection in remote regions might be regarded as negligible (Solazzo and Galmarini, 2015). It should be noted that the selection of the optimal combinations of models is affected by the model's bias that might stem from processes that are common to all members of the ensemble (e.g. emissions). Therefore, such a common bias does not cancel out when combining the models, possibly creating a biased ensemble. Current work is being devoted to identify the optimal combinations of models from which the offsetting bias is removed (Solazzo et al., 2017b).

2.2. EVA

The EVA system (Brandt et al., 2013a, b) is based on the impact-pathway chain (e.g. Friedrich and Bickel, 2001), consisting of the emissions, transport and chemical transformation of air pollutants, population exposure, health impacts and the associated external costs. The EVA system requires hourly gridded concentration input from a regional-scale CTM as well as gridded population data, exposure-response functions (ERFs) for health impacts, and economic valuations of the impacts from air pollution. A detailed description of the integrated EVA model system along with the ERFs and the economic valuations used are given in Brandt et al. (2013a).

The gridded population density data over Europe and the U.S. used in this study are presented in Fig. 1. The population data over Europe are provided on a 1km spatial resolution from Eurostat for the year 2011 (http://www.efgs.info). The U.S. population data has been provided from the U.S. Census Bureau for the year 2010. The total populations used in this study are roughly 532 and 307 million in Europe and the U.S., respectively. As the health outcomes are age-dependent, the total population data has been broken down to a set of age intervals being babies (under 9 months), children (under 15), adult (above 15), above 30, and above 65. The fractions of population in these intervals for Europe is derived from the EUROSTAT 2000 database, where the number of persons of each age at each grid cell was aggregated into the above clusters (Brandt et al., 2011), while for the U.S. they are derived from the U.S. Census Bureau for the year 2010 at 5-year intervals.

The EVA system can be used to assess the number of various health outcomes including different morbidity outcomes as well as short-term (acute) and long-term (chronic) mortality, related to exposure of O_3, CO and SO_2 (short-term) and $PM_{2.5}$ (long-term). Furthermore, impact on infant mortality in response to exposure of $PM_{2.5}$ is calculated. EVA calculates and uses the annual mean concentrations of CO, SO_2 and $PM_{2.5}$, while for O_3, it uses the SOMO35 metric that is defined as the yearly sum of the daily maximum of 8-hour running average over 35 ppb, to calculate the acute effects of O_3. The morbidity outcomes include chronic bronchitis, restricted activity days, congestive heart failure, lung cancer, respiratory and cerebrovascular hospital admissions, asthmatic children (<15 years) and adults (>15 years), which includes bronchodilator use, cough, and lower respiratory symptoms. The total

number of premature death is calculated as the sum of the acute deaths due to O_3 and the chronic YOLL (years of life lost) by $PM_{2.5}$ divided by 10.6 as recommended by the CAFÉ (Clean Air for Europe Programme) report (Watkiss et al., 2005). The ERFs used, along with their references, in both continents as well as the economic valuations for each health outcome in Europe and the U.S., respectively, are presented in Table 2. For a full description of the ERFs, see Andersen et al. (2004). The economic valuations are provided by Brandt et al. (2013a); see also EEA (2013).

Table 2 lists the specific valuation estimates applied in the modelling of the economic valuation of mortality and morbidity effects. A principal value of EUR 1.5 million was applied for preventing an acute death, following expert panel advice (EC 2001). For the valuation of a life year, the results from a survey relating specifically to air pollution risk reductions were applied (Alberini et al., 2006), implying a value of EUR 57.500 per year of life lost (YOLL). With the more conservative metric of estimating lost life years, rather than 'full' statistical lives, there is no adjustment for age. Most of the excess mortality is due to chronic exposure to air pollution over many years and the life year metric is based on the number of lost life years in a statistical cohort. Following the guidelines of the Organisation for Economic Co-operation and Development (OECD, 2006), the predicted acute deaths, mainly from O_3, are valuated here with the adjusted value for preventing a fatality (VSL, Value of a Statistical Life). The life tables are obtained from European data and are applied to the U.S. as the average life expectancy in the U.S. is similar to that in Europe, and close to the OECD average. The willingness to pay for reductions in risk obviously differs across income levels. However, in the case of air pollution costs, adjustment according to per capita income differences among different states is not regarded as appropriate, because long-range transport implies that emissions from one state will affect numerous other states and their citizens. The valuations are thus adjusted with regional purchasing power parities (PPP) of EU27 and USA. The unit values have been indexed to 2013 prices as indicated in Table 2.

3. Results

3.1. Model Evaluation

Observed and simulated hourly surface O_3, CO, SO_2 and daily $PM_{2.5}$, which are species used in the EVA model to calculate the health impacts, over Europe and North America for the entire 2010 were compared in order to evaluate each model's performance. For a more thorough evaluation of models and species, see Solazzo et al. (2017a). The results of this comparison are presented in Table 3 for EU and NA, along with the multi-model mean and median values. The monthly time series plots of observed and simulated health-related pollutants are also presented in Figs. 2 and 3. The results show that over Europe, the temporal variability of all gaseous pollutants is well captured by all models with correlation coefficients (r) higher than 0.70 in general. The normalized mean biases (*NMB*) in simulated O_3 levels are generally below 10% with few exceptions up to -35%. CO levels are underestimated by up to 45%, while the majority of the models underestimated SO_2 levels by up to 68%, while some models overestimated SO_2 by up to 49%. $PM_{2.5}$ levels are underestimated by 19% to 63%.

Over North America, the hourly O_3 variation is well captured by all models (Table 3), with DK1 having slightly lower *r* coefficient compared to the other models and largest *NMB* (Fig. 3a). The hourly variation of CO and SO_2 levels are simulated with relatively lower *r* values (Figs. 3b, c), with SO_2 levels having the highest underestimations. The $PM_{2.5}$ levels are underestimated by ~15% except for the DE1 model, having a large underestimation of 63% (Table 3). As DE1 and US3 use the same SMOKE emissions and CTM, the large difference in $PM_{2.5}$ concentrations can be partly due to the differences in horizontal and vertical resolutions in the model setups, as can also be seen in the differences in the CO concentrations. There are also differences in the aerosol modules and components that each model simulates. For example, DE1 uses an older version of the secondary organic aerosol (SOA) module, producing ~3 μgm^{-3} less SOA, which can explain ~20% of the bias over North America.

3.2. Health outcomes and their economic valuation in Europe

The different health outcomes calculated by each model in Europe as well as their multi model mean and median are presented in Table 4. The health impact estimates vary significantly between different models. The different estimates obtained are found to vary up to a factor of three. Among the different health outcomes, the multi model ensemble simulated the number of congestive heart failure cases to be between 19 000 to 41 000 (mean of all individual models 31 000 ± 6 500). The number of lung cancer cases due to air pollution are calculated to be between 30 000 to 78 000 (mean of all individual models 55 000 ± 14 000). Finally, the total (acute + chronic) number of premature death due to air pollution is calculated to be 230 000 to 570 000 (mean of all individual models 414 000 ± 100 000). We have also produced a multi-model mean data (MM_m) and fed it to the EVA model and obtained the number of premature death cases in Europe as 410 000 (difference from the mean of all individual model estimates is smaller than 1%). The number of premature death cases in Europe calculated by the multi model ensemble due to exposure to O_3 is 12 000 ± 6 500, while the cases due to exposure to $PM_{2.5}$ is calculated to be 390 000 ± 100 000 [180 000 – 550 000]. The O_3-related mortality well agrees with Liang et al. (2017) that used the multi-model mean of the HTAP2 global model ensemble, which calculated an O_3-realted mortality of 12 800 [600 - 28 100]. The multi-model mean $PM_{2.5}$-related mortality in the present study is much higher than that from the HTAP2 study (195 500 [4 400 – 454 800]). This difference can be attributed to the number of mortality cases as calculated by the individual models, where the HTAP2 ensemble calculates a much lower minimum while the higher ends from the two ensembles well agree.

The differences between the health outcomes calculated by the HTAP2 and AQMEII ensembles arise firstly from the differences in the concentrations fields due to the differences in models, in particular spatial resolutions as well as the gas and aerosols treatments in different models, but also the differences in calculating the health impacts from these concentrations fields. EVA calculates the acute premature death due to O_3 by using the SOMO35 metric. On the other hand, in HTAP2 O_3-related premature death is calculated by using the 6-month seasonal average of daily 1-h maximum O_3 concentrations. Both groups use the annual mean $PM_{2.5}$ to calculate the $PM_{2.5}$-related premature death. In addition to O_3

and $PM_{2.5}$, EVA also takes into account the health impacts from CO and SO_2, which is missing in the HTAP2 calculations.

Among all models, DE1 model calculated the lowest health impacts for most health outcomes, which can be attributed to the largest underestimation of $PM_{2.5}$ levels (*NMB*=-63%: Table 3) due to lower spatial resolution of the model that dilutes the pollution in the urban areas, where most of the population lives. The number of premature deaths calculated by this study is in agreement with previous studies for Europe using the EVA system (Brandt et al., 2013a; Geels et al., 2015). Recently, EEA (2015) estimated that air pollution is responsible for more than 430 000 premature deaths in Europe, which is in good agreement with the present study.

Fig. 4a. presents the geographical distribution of the number of premature death in Europe in 2010. The figure shows that the numbers of cases are strongly correlated to the population density, with the largest numbers seen in the Benelux and the Po Valley regions that are characterized as the pollution hot spots in Europe as well as in megacities such as London, Paris, Berlin and Athens.

The economic valuation of the air pollution-associated health impacts calculated by the different models along with their mean and median are presented in Table 5. A total cost of 196 to 451 billion Euros (MM mean cost of 300 ± 70 billion Euros) was estimated over Europe (EU28). Results show that 5% [1% - 11%] of the total costs is due to exposure to O_3, while 89% [80% - 96%] is due to exposure to $PM_{2.5}$. Brandt et al. (2013a) calculated a total external cost of 678 billion Euros for the year 2011 for Europe, larger than the estimates of this study, which can be explained by the differences in the simulation year and the emissions used in the models as well as the countries included in the two studies (the previous study includes e.g. Russia).

3.3. Health outcomes and their economic valuation in the U.S.

The different health outcomes calculated by each model for the U.S. as well as their mean and median are presented in Table 4. The variability among the models (~3) is similar to that in Europe. The number of congestive heart failure cases in the U.S. is calculated to be 13 000, while the lung cancer cases due to air pollution are calculated to be 22 000. Finally, the number of premature deaths due to air pollution is calculated to be 165 000 ± 75 000, where 25 000 ± 6 000 cases are calculated due to exposure to O_3 and 140 000 ± 72 000 cases due to exposure to $PM_{2.5}$. The MM_m dataset leads to a number of premature death of 149 000 that is 6% smaller than the mean of individual models. The O_3- and $PM_{2.5}$ mortality cases as calculated by the AQMEII and HTAP2 model ensembles reasonably agree. Liang et al. (2017) calculated an O_3-related mortality of 14 70 [900 – 30 400] and a $PM_{2\text{-}5}$-related mortality of 78 600 [4 500 – 162 600]. Among all models, DE1 model calculated the lowest health impacts for most health outcomes, which can be attributed to the largest underestimation of $PM_{2.5}$ levels (*NMB*=-63%: Table 4). The findings of this study are in agreement with the U.S. EPA findings of premature death cases in the U.S., as well as the Fann et al. (2012) study. The premature death cases in North America are mostly

concentrated over the New York area, as well as in hot spots over Chicago, Detroit, Houston Los Angeles and San Francisco (Fig. 4b). The figure shows that the number of cases is following the pattern of the population density by construction.

The economic valuation of the air pollution-associated health impacts calculated by the different models in the U.S. are shown in Table 5. As seen in the table, a total cost of ~145 billion Euros is calculated. Results show that ~22% of the total costs is due to exposure to O_3 while ~78% is due to exposure to $PM_{2.5}$. The major health impacts in terms of their external costs are slightly different in North America compared to Europe.

3.4. Health impacts and their economic valuation through optimal reduced ensemble subset

The effect of pollution concentrations (EVA input) on health impacts (EVA output) is investigated in order to estimate the contribution of each air pollutant in the EVA system to health impacts over different concentration levels. The technical details are provided in the supplement.

Results show that for the particular input (gridded air pollutant concentrations from individual model)-output (each health outcome) configuration, the $PM_{2.5}$ drives the variability of the health impact indices (HII) and that at least 81% of the variation of the health indices are explained by sole variations in the pollutants (i.e. without interactions: Table S1). Table S1 also shows that the most important contribution to the HIIs is from $PM_{2.5}$, followed by CO and O_3 (with much smaller influence though). The effect on the HII of perturbing $PM_{2.5}$ by a fixed fraction of its standard deviation if roughly double compared to CO and O_3.

We have run the EVA system over an all-models mean (MM_m) dataset and an optimal reduced ensemble dataset (MM_{opt}) calculated for each of the pollutants in the two domains in order to see how and whether an optimal reduced ensemble changes the assessment of the health impacts compared to an all- models ensemble mean. Table 6 shows some sensible error reduction, although the temporal and spatial averages mask the effective improvement in accuracy from MM_m to MM_{opt}. In Europe, the optimal reduced ensemble decreases the RMSE by up to 24%, while in NA, the error reduction is much larger (4% to up to 147%). On a seasonal basis, MM_{opt} reduces *RMSE* in $PM_{2.5}$ over Europe by 23% in winter while smaller decreases are achieved in other seasons (~10%). Regarding O_3, improvement is 16%-22%, with the largest improvement in spring. In NA, the improvement in winter *RMSE* in $PM_{2.5}$ is smallest (~2%) while larger improvements are achieved in other seasons (~7% - ~9%). For O_3, the largest *RMSE* reduction in NA is achieved for the summer period by 14%.

The analysis of the aggregated health indices data for Europe (Table S1) shows that EVA indices rely principally on the $PM_{2.5}$ levels and then the CO and O_3 values. Therefore, the relative improvement of the indices with the optimal ensemble should be proportional to the relative improvement in $PM_{2.5}$, CO and O_3. The proportionality rate for each pollutant is given in Table S1, assuming all pollutants are varied (from MM_m to MM_{opt}) away from their mean by the same fraction of their variance. As seen in the Table 7, from MM_m to MM_{opt}, the health indices increase by up to 30% in Europe. This increase is due to a 27% increase in the domain mean $PM_{2.5}$ levels when the optimal reduced ensemble is used, as well a slight

459 increase in O_3 by ~1%. The number of premature deaths in Europe increase from 410 000 to
460 524 000 (28%), resulting in a much higher estimate compared to previous mortality studies.
461 On the contrary, in the U.S., the mean $PM_{2.5}$ and O_3 levels decrease from 2.94 μg m^{-3} to 2.62
462 μgm^{-3} (~11%) and 18.7 ppb to 18.4 ppb (~2%), respectively. In response, the health indices
463 decrease by ~11% (Table 7). The number of premature death cases in NA decrease from
464 149 000 to 133 000.

465 3.5. Impact of anthropogenic emissions on the health impacts and their economic valuation

466 The impacts of emission perturbations on the different health outcomes over Europe and the
467 U.S. as calculated by the individual models are presented in Tables S2-S4. Table 8 shows the
468 impacts of the different emission perturbations on the premature death cases in Europe and
469 the U.S as calculated by a subset of models that simulated the base case and all three
470 perturbation scenarios MM_c. Results show that in Europe, the 20% reduction in the global
471 anthropogenic emissions leads to ~17% domain-mean reduction in all the health outcomes,
472 with a geographical variability as seen in Fig. 4c. The figure shows that the larger changes in
473 mortality is calculated in the central and northern parts of Europe (15-20% decreases), while
474 the changes are smaller in the Mediterranean region (5-10%), highlighting the non-linearity
475 of the response to emission reductions. However, it should be noted that global models or
476 coarse-resolution regional models (as in this study) cannot capture the urban features and
477 pollution levels and thus, non-linearities should be addressed further using fine spatial
478 resolutions or urban models. The models vary slightly simulating the response to the 20%
479 reduction in global emissions, estimating decreases of ~11% to 20%. The number of
480 premature deaths decreased on average by ~50 000, ranging from -39 000 (DK1) to -103 000
481 (IT1). This number is in good agreement with the ~45 000 premature death calculated by the
482 HTAP2 global models (Liang et al., 2017). The MM_c ensemble calculated a 15% and 17%
483 decrease in the O_3- and $PM_{2.5}$-related premature death cases, respectively, in response to the
484 GLO scenario. This decrease in the global anthropogenic emissions leads to an estimated
485 decrease of 56 ± 18 billion Euros in associated costs in Europe (Table 8).

486 As seen in Table 8, a 20% reduction of anthropogenic emissions in the EUR region, as
487 defined in HTAP2, avoids 47 000 premature death, while a 20% reduction of the
488 anthropogenic emissions in the NAM region leads to a much smaller decrease of premature
489 deaths (~1 000). These improvements in the number of premature deaths are in agreement
490 with a recent HTAP2 global study that calculated reductions of ~34 000 and ~1 000 for the
491 EUR and NAM scenarios, respectively (Liang et al., 2017). Both the global and regional
492 models agree that the largest impacts of reducing emissions with respect to premature deaths
493 come from emission within the source region, while foreign sources contribute much less to
494 improvements in avoiding adverse impacts of air pollution. The decreases in health impacts
495 in EUR and NAM scenarios corresponds to decreases in the associated costs by -47 ± 16
496 billion Euros and -1.4 ± 0.4 billion Euros, respectively. This is consistent with results in
497 Brandt et al. (2012), where a contribution of ~1% to $PM_{2.5}$ concentrations in Europe is
498 originating from the NAM region.

The 20% reduction in global anthropogenic emissions leads to 18% reduction in the health outcomes (Table 8) in the U.S., with a geographical variability in the response. Fig. 4d shows that the largest decreases in mortality is calculated for the western coast of the U.S. (~20%) and slightly lower response in the central and eastern parts of the U.S. (15-20%). The number of premature death cases, as calculated by the mean of all individual models decreases from ~160 000 ± 70 000 to ~130 000 ± 60 000, avoiding 24 ± 10 billion Euros (Table 8) in external costs, also in agreement with the ensemble of HTAP2 global models (~23 000) The O_3-related premature death cases decreased by 42% while the $PM_{2.5}$-relared cases decreased by 18%. A 20% reduction of the North American emissions avoids ~25 000 ± 12 000 premature deaths (-16%), suggesting that ~80% of avoided premature deaths are achieved by reductions within the source region while 20% (~5 000 premature deaths) is from foreign sources. This number is also in good agreement with Liang et al. (2017) that estimated a reduction of premature deaths of ~20 000 due to O_3 and $PM_{2.5}$ in the United States due to an emission reduction of 20% within the region itself, using the ensemble mean of the HTAP2 global models. The corresponding benefit is calculated to be 21 ± 9 billion Euros in the NAM scenario. According to results from the EAS scenario, among these 5 000 avoided cases that are attributed to the foreign emission sources, 1 900 ± 2 000 premature deaths can be avoided by a 20% reduction of the East Asian emissions, avoiding 2.5 ± 3 billion Euros. Our number of avoided premature deaths due to the EAS scenario is much higher than 580 avoided premature deaths calculated by Liang et al. (2017).

Conclusions

The impact of air pollution on human health and their economic valuation for the society across Europe and the United States is modelled by a multi-model ensemble of regional models from the AQMEII3 project. All regional models used boundary conditions from the C-IFS model, and emissions from either the MACC inventory in Europe or the EPA inventory for the North America, or the global inventory from HTAP. Sensitivity analysis on the dependence of models on different sets of boundary conditions has not been conducted so far but large deviations from the current results in terms of health impacts are not expected. The modelled surface concentrations by each individual model are used as input to the EVA system to calculate the resulting health impacts and the associated external costs from O_3, CO, SO_2 and $PM_{2.5}$. Along with a base case simulation for the year 2010, some groups performed additional simulations, introducing 20% emission reductions both globally and regionally in Europe, North America and East Asia.

The base case simulation of each model is evaluated with available surface observations in Europe and North America. Results show large variability among models, especially for $PM_{2.5}$, where models underestimate by ~20% - ~60%, introducing a large uncertainty in the health impact estimates as $PM_{2.5}$ is the main driver for health impacts. The differences in the models are largely due to differences in the spatial and vertical resolutions, meteorological inputs, natural emissions, as well as missing or underestimated SOA mass, which is critical for the $PM_{2.5}$ mass.

The variability of health impacts among the models can be up to a factor of three in Europe (twelve models) and the U.S. (three models), among the different health impacts. The multi-model mean total number of premature death is calculated to be 414 000 in Europe and 160 000 in the U.S., where $PM_{2.5}$ contributes by more than 90%. These numbers agree well with previous global and regional studies for premature deaths due to air pollution. In order to reduce the uncertainty coming from each model, an optimal ensemble set is produced, that is, the subset of models that produce the smallest error compared to the surface observations at each time step. The optimum ensemble results in an increase of health impacts by up to 30% in Europe and a decrease by ~11% in the United States. These differences clearly demonstrate the importance of the use of optimal-reduced multi-model ensembles over traditional all model-mean ensembles, both in terms of scientific results, but also in policy applications.

Finally, the role of domestic versus foreign emission sources on the related health impacts is investigated using the emission perturbation scenarios. A global reduction of anthropogenic emissions by 20% decreases the health impacts by 17%, while the reduction of foreign emissions decreases the health impacts by less than 1%. The decrease of emissions within the source region decreases the health impacts by 16%. These results show that the largest impacts of reducing emissions with respect to the premature death come from emissions within the source region, while foreign sources contributing to much less improvements in avoiding adverse impacts of air pollution.

Outlook

Currently health assessments of airborne particles are carried out under the assumption that all fine fraction particles affect health to a similar degree independent of origin, age and chemical composition of the particles. A 2013 report from WHO concludes that the cardiovascular effects of ambient $PM_{2.5}$ are greatly influenced, if not dominated, by their transition metal contents (WHO, 2013). It is known that trace metals and traffic markers are highly associated with daily mortality (Lippmann, 2014). Even low concentrations of trace metals can be influential on health related responses.

Regarding ambient concentrations of PM and the exposure-response functions (ERFs), there is a rich set of studies providing information on total PM mass. However, only few studies focus on individual particulate species, mainly black carbon and carbonaceous particles. In addition to PM, studies on human populations have not been able to isolate potential effects of NO_2, because of its complex link to PM and O_3. The WHO REVIHAAP review from 2013 concludes that health assessments based on $PM_{2.5}$ ERFs will be most inclusive (WHO, 2013). In addition, the ERFs are based on urban background measurements, introducing uncertainties regarding non-urban areas or high pollution areas as e.g. street canyons. Current state-of-the-art health impact estimates, in particular on regional to global scales, assume a correlation with exposure to outdoor air pollution, while in reality, exposure is dynamic and depends on the behavior of the individual. In addition, differences in age groups, gender, ethnicity and behavior should be considered in the future studies. There are also uncertainties originating from the representations of the aerosols in the atmospheric models used in the

calculation of pollutant concentrations as well as the emissions. Further developments in the aerosol modules, such as the representation of organic aerosols and windblown and suspended dust, are need in order to achieve mass closure of PM to get robust estimates of health impacts.

Due to above reasons, there is a large knowledge gap regarding the health impacts of particles. There are a number of ongoing projects trying to identify the health impacts from individual particle components and produce individual ERFs for these components. NordicWelfAir project (http://projects.au.dk/nordicwelfair/) aims to investigate the potential causal impact of individual chemical air pollutants as well as mixtures of air pollutants on health outcomes. In pursuing this aim, the project uses the unique Nordic population-based registers allowing linkage between historical residential address, air pollutants over decades and later health outcomes. By linking the exposure to health outcomes, new exposure-response relationships can be determined of health effects for different population groups (e.g. age, education, ethnicity, gender, lifestyle, and working life vs. retirement conditions) related to air pollution for the individual chemical air pollutants. In addition, the high resolution simulations conducted will enable us to have a better understanding of non-linearities between the emissions, health impacts, and their economic valuation.

ACKNOWLEDGEMENTS

We gratefully acknowledge the contribution of various groups to the third air Quality Model Evaluation international Initiative (AQMEII) activity. Joint Research Center Ispra/Institute for Environment and Sustainability provided its ENSEMBLE system for model output harmonization and analyses and evaluation. Although this work has been reviewed and approved for publication by the US Environmental Protection Agency, it does not necessarily reflect the views and policies of the agency. Aarhus University gratefully acknowledges the NordicWelfAir project funded by the NordForsk's Nordic Programme on Health and Welfare (grant agreement no. 75007), the REEEM project funded by the H2020-LCE Research and Innovation Action (grant agreement no.: 691739), and the Danish Centre for Environment and Energy (AU-DCE). University of L'Aquila thanks the EuroMediterranean Center for Climate Research (CMCC) for providing the computational resources. RSE contribution to this work has been financed by the research fund for the Italian Electrical System under the contract agreement between RSE S.p.A. and the Ministry of Economic Development – General Directorate for Nuclear Energy, Renewable Energy and Energy Efficiency in compliance with the decree of 8 March 2006.

REFERENCES

Abbey, D.E., Lebowitz, M.D., Mills, P.K., Petersen, F.F., Beeson, W.L. Burchette, R.J, 1995. Long-term ambient concentrations of particulates and oxidants and development of chronic disease in a cohort of non-smoking California residents. Inhalation Toxicology 7, 19-34.

618 Alberini, A., Hunt, A. and Markandya, A., 2006, Willingness to pay to reduce mortality risks:
619 Evidence from a three-country contingent valuation study. Environmental and Resource
620 Economics, 33, 251–264.

621 Amann, M., Bertok, I., Borken-Kleefeld, J., Cofala, J., Heyes, C., Höglund-Isaksson, L.,
622 Klimont, Z., Nguyen, B., Posch, M., Rafaj, P., Sandler, R., 2011. Cost-effective control of air
623 quality and greenhouse gases in Europe: modeling and policy applications. Environmental
624 Modelling & Software, 26 (12), 1489–1501.

625 Andersen, M.S., 2017. Co-benefits of climate mitigation: Counting statistical lives or life-
626 years? Ecological Indicators, 79, 11-18.

627 Andersen, M.S., Frohn, L.M., Jensen, S.S., Nielsen, J.S., Sørensen, P.B., Hertel, O., Brandt,
628 J., Christensen, J.H., 2014. Sundhedseffekter af luftforurening – beregningspriser, Faglig
629 rapport fra DMU, nr. 507
630 (http://www.dmu.dk/1_viden/2_Publikationer/3_fagrapporter/rapporter/FR507.PDF).

631 Anderson, H.R., Ponce de Leon, A., Bland, J.M., Bower, J.S., Strachan, D.P., 1996. Air
632 Pollution and daily mortality in London: 1987-92. British Medical Journal, 312, 665-669.

633 Anenberg, S. C., A. Belova, J. Brandt, N. Fann, S. Greco, S. Guttikunda, M.-E. Heroux, F.
634 Hurley, M. Krzyzanowski, S. Medina, B. Miller, K. Pandey, J. Roos, R. Van Dingenen, 2015.
635 Survey of ambient air pollution health risk assessment tools. *Risk Analysis*. DOI:
636 10.1111/risa.12540.

637 Anenberg, S.C., West, J.J., Yu, H., Chin, M., Schulz, M., Bergmann, D., Bey, I., Bian, H.,
638 Diehl, T., Fiore, A., Hess, P., Marmer, E., Montanaro, V., Park, R., Shindell, D., Takemura,
639 T., Dentener, F., 2014. Impacts of intercontinental transport of anthropogenic fine particulate
640 matter on human mortality. Air Quality, Atmosphere & Health, 7, (3), 369-379,
641 doi:10.1007/s11869-014-0248-9.

642 Anenberg, S.C., West, J.J., Fiore, A.M., Jaffe, D.A., Prather, M.J., Bregmann, D., Cuvelier,
643 K., Dentener, F.J., Duncan, B.N., Gauss, M., Hess, P., Jonson, J.E., Lupu, A., MacKenzie,
644 I.A., Marmer, E., Park, R.J., Sanderson, M.G., Schultz, M., Shindell, D.T., Szopa, S.,
645 Vivanco, M.G., Wild, O., Zeng, G., 2009. Intercontinental impacts of ozone air pollution on
646 human mortality. Environ Science and Technology, 43,6482–6487.

647 Bell, M.L., McDermott, A., Zeger, S.L., Samet, J.M., Dominici, F., 2004. Ozone and short-
648 term mortality in 95 US urban communities, 1987–2000. Journal of. American Medical
649 Association, 292, 2372–2378.

650 Brandt, J., J. D. Silver, L. M. Frohn, C. Geels, A. Gross, A. B. Hansen, K. M. Hansen, G. B.
651 Hedegaard, C. A. Skjøth, H. Villadsen, A. Zare, and J. H. Christensen, 2012. An integrated
652 model study for Europe and North America using the Danish Eulerian Hemispheric Model
653 with focus on intercontinental transport. *Atmospheric Environment*, Volume 53, June 2012,
654 pp. 156-176, doi:10.1016/j.atmosenv.2012.01.011

655 Brandt, J., Silver, J. D., Christensen, J. H., Andersen, M. S., Bønløkke, J. H., Sigsgaard, T.,
656 Geels, C., Gross, A., Hansen, A. B., Hansen, K. M., Hedegaard, G. B., Kaas, E., and Frohn,
657 L. M.: Contribution from the ten major emission sectors in Europe and Denmark to the
658 health-cost externalities of air pollution using the EVA model system – an integrated
659 modelling approach. Atmospheric Chemistry and Physics, 13, 7725–7746, doi:10.5194/acp-
660 13-7725-2013, 2013a.

661 Brandt, J., Silver, J. D., Christensen, J. H., Andersen, M. S., Bønløkke, J. H., Sigsgaard, T.,
662 Geels, C., Gross, A., Hansen, A. B., Hansen, K. M., Hedegaard, G. B., Kaas, E., and Frohn,
663 L. M.: Assessment of past, present and future health-cost externalities of air pollution in
664 Europe and the contribution from international ship traffic using the EVA model system.
665 Atmospheric Chemistry and Physics, 13, 7747-7764, doi:10.5194/acp-13-7747-2013, 2013b.

666 Brandt, J., Silver, J.D., Frohn, L.M., Christensen, J.H., Andersen, M.S., Bønløkke, J.H.,
667 Sigsgaard, T., Geels, C., Gross, A., Hansen, A.B., Hansen, K.M., Hedegaard, G.B., Kaas, E.,
668 2011. Assessment of Health-Cost Externalities of Air Pollution at the National Level using
669 the EVA Model System, CEEH Scientific Report No 3, pp. 96.
670 (www.ceeh.dk/CEEH_Reports/Report_3)

671 Burnett, R. T., Arden Pope, C., Ezzati, M., Olives, C., Lim, S. S., Mehta, S., Shin, H. H.,
672 Singh, G., Hubbell, B., Brauer, M., Ross Anderson, H., Smith, K. R., Balmes, J. R., Bruce, N.
673 G., Kan, H., Laden, F., Prüss-Ustün, A., Turner, M. C., Gapstur, S. M., Diver, W. R., Cohen,
674 A., 2014. An integrated risk function for estimating the global burden of disease attributable
675 to ambient fine particulate matter exposure. Environmental Health Perspectives, 122, 397–
676 403, doi:10.1289/ehp.1307049.

677 Caiazzo, F., Ashok, A., Waitz, I., Yim, S.H.L., Barrett, S.R.H., 2013. Air pollution and early
678 deaths in the United States. Part I: Quantifying the impact of major sectors in 2005.
679 Atmospheric Environment, 79,198–208, doi:10.1016/j.atmosenv.2013.05.081.

680 Dab, W., Medina, S., Quénel, P., le Moullec, Y., le Tertre, A., Thelot, B., Monteil, C.,
681 Lameloise, P., Pirard, P., Momas, I., Ferry, R., Festy, B., 1996. Short Term Respiratory
682 Health Effects of Ambient Air Pollution: Results of the APHEA Project in Paris. Journal of
683 Epidemiology and Community Health, 50 (suppl 1), S42-S46.

684 Duncan, B.N., West, J.J., Yoshida, Y., Fiore, A.M., Ziemke, J.R., 2008. The influence of
685 European pollution on the air quality in the Near East and northern Africa. Atmospheric
686 Chemistry and Physics, 8, 2267–2283.

687 Dusseldorp, A., Kruize, H., Brunekreef, B., Hofschreuder, P., de Meer, G., Oudvorst, A.B.
688 van, 1995. Associations of PM10 and Airborne Iron with Respiratory Health of Adults Living
689 Near a Steel Factory. American Journal of Critical Care Medicine, 152, 1932-1939.

690 European Commission (EC) 2001, Recommended interim values for the value of preventing
691 a fatality in DGEnvironment Cost Benefit analysis, Bruxelles:
692 (http://ec.europa.eu/environment/enveco/others/pdf/recommended_interim_values.pdf),
693 accessed 28 Sept 2016.

694 EEA, 2015. Air quality in Europe, Technical report 5/2015, Copenhagen: European
695 Environment Agency.

696 EEA, 2013. Road user charges for heavy goods vehicles: Tables with external costs of air
697 pollution, Technical report 1/2013, Copenhagen: European Environment Agency.

698 ExternE, 2005. Externalities of Energy Methodology 2005 update, European Commission,
699 Directorate-General for Research Sustainable Energy Systems. Brussels, www.externe.info.

700 Fann, N., Lamson, A.D., Anenberg, S.C., Wesson, K., Risley, D., Hubbell, B., 2012.
701 Estimating the national public health Burden associated with exposure to ambient PM2.5 and
702 ozone. Risk Anal. 32, 81-95.

703 Flemming, J., Huijnen, V., Arteta, J., Bechtold, P., Beljaars, A., Blechschmidt, A.-M.,
704 Diamantakis, M., Engelen, R. J., Gaudel, A., Inness, A., Jones, L., Josse, B., Katragkou, E.,
705 Marecal, V., Peuch, V.-H., Richter, A., Schultz, M. G., Stein, O., and Tsikerdekis, A., 2015.
706 Tropospheric chemistry in the Integrated Forecasting System of ECMWF. Geoscientific
707 Model Development, 8, 975-1003, doi:10.5194/gmd-8-975-2015.

708 Forouzanfar, M. H., Alexander, L., Bachman, V. F., et al.: Global, regional, and national
709 comparative risk assessment of 79 behavioral, environmental and occupational, and
710 metabolic risks or clusters of risks in 188 countries, 1990–2013: a systematic analysis for the
711 Global Burden of Disease Study 2013, The Lancet, 386, 2287–2323, doi:10.1016/S0140-
712 6736(15)00128-2, 2015.

713 Friedrich, R. and Bickel, P., 2001, Environmental External Costs of Transport, München:
714 Springer.

715 Galmarini, S., Koffi, B., Solazzo, E., Keating, T., Hogrefe, C., Schulz, M., Benedictow, A.,
716 Griesfeller, J. J., Janssens-Maenhout, G., Carmichael, G., Fu, J., and Dentener, F., 2017.
717 Technical note: Coordination and harmonization of the multi-scale, multi-model activities
718 HTAP2, AQMEII3, and MICS-Asia3: simulations, emission inventories, boundary
719 conditions, and model output formats. Atmospheric Chemistry and Physics, 17, 1543-1555,
720 doi:10.5194/acp-17-1543-2017.

721 Geels, C., C. Andersson, O. Hänninen, A. S. Lansø, P. Schwarze and J. Brandt, 2015. Future
722 Premature Mortality due to Air Pollution in Europe – Sensitivity to Changes in Climate,
723 Anthropogenic Emissions, Population and Building stock. International Journal of
724 Environmental Research Public Health, 12, 2837-2869.

725 Global Health Data Exchange (GHDx): Global Burden of Disease Study 2010 – Ambient Air
726 Pollution Risk Model 1990–2010.

727 Im, U., Geels, C., Hansen, K.M., Christensen, J.H., Brandt, J., Solazzo, E., Alyuz, U.,
728 Balzarini, A., Baro, R., Bellasio, R., Bianconi, R., Bieser, J., Colette, A., Curci, G., Farrow,
729 A., Flemming, J., Fraser, A., Jimenez-Guerrero, P., Kitwiroon, N., Pirovano, G., Pozolli, L.,
730 Prank, M., Rose, R., Sokhi, R., Tuccella, P., Unal, A., Vivanco, M.G., Yardwood, G.,

731 Hogrefe, C., Galmarini, S., 2017. Impacts of emission perturbations on multi-model
732 simulations of major air pollutants over Europe and North America in frame of AQMEII3, In
733 Preparation for Atmospheric Chemistry and Physics.

734 Im, U., Bianconi, R., Solazzo, E., Kioutsioukis, I., Badia, A.,Balzarini, A., Baro, R., Bellasio,
735 R., Brunner, D., Chemel, C.,Curci, G., Denier van der Gon, H., Flemming, J., Forkel,
736 R.,Giordano, L., Jimenez-Guerrero, P., Hirtl, M., Hodzic, A., Honzak,L., Jorba, O., Knote,
737 C., Makar, P. A., Manders-Groot, A.,Neal, L., Pérez, J. L., Pirovano, G., Pouliot, G., San
738 Jose, R., Savage,N., Schroder,W., Sokhi, R. S., Syrakov, D., Torian, A., Tuccella,P., Wang,
739 K., Werhahn, J., Wolke, R., Zabkar, R., Zhang,Y., Zhang, J., Hogrefe, C., and Galmarini, S.:
740 Evaluation of operational online coupled regional air quality models over Europe and North
741 America in the context of AQMEII phase 2, Part II: particulate matter, Atmos. Environ., 115,
742 421–441, 2015a.

743 Im, U., Bianconi, R., Solazzo, E., Kioutsioukis, I., Badia, A.,Balzarini, A., Baro, R., Bellasio,
744 R., Brunner, D., Chemel, C.,Curci, G., Flemming, J., Forkel, R., Giordano, L., Jimenez-
745 Guerrero, P., Hirtl, M., Hodzic, A., Honzak, L., Jorba, O., Knote,C., Kuenen, J. J. P., Makar,
746 P. A., Manders-Groot, A., Neal, L.,Pérez, J. L., Pirovano, G., Pouliot, G., San Jose, R.,
747 Savage, N.,Schroder, W., Sokhi, R. S., Syrakov, D., Torian, A., Tuccella, P.,Werhahn,
748 J.,Wolke, R., Yahya, K., Zabkar, R., Zhang, Y., Zhang,J., Hogrefe, C., and Galmarini, S.:
749 Evaluation of operational online-coupled regional air quality models over Europe and
750 NorthAmerica in the context of AQMEII phase 2, Part I: ozone, Atmos. Environ., 115, 404–
751 420, 2015b.

752 Janssens-Maenhout, G., Crippa, M., Guizzardi, D., Dentener, F., Muntean, M., Pouliot, G.,
753 Keating, T., Zhang, Q., Kurokawa, J., Wankmüller, R., Denier van der Gon, H., Kuenen, J. J.
754 P., Klimont, Z., Frost, G., Darras, S., Koffi, B., and Li, M., 2015. HTAP_v2.2: a mosaic of
755 regional and global emission grid maps for 2008 and 2010 to study hemispheric transport of
756 air pollution. Atmospheric Chemistry and Physics, 15, 11411–11432, doi:10.5194/acp-15-
757 11411-2015.

758 Jerrett, M., Burnett, R.T., Pope, C.A., Ito, K., Thurston, G., Krewski, D., Shi, Y.L., Calle, E.,
759 Thun, M., 2009. Long-term ozone exposure and mortality. New England Journal of
760 Medicine, 360, 1085–95

761 Kioutsioukis, I., Im, U., Solazzo, E., Bianconi, R., Badia, A., Balzarini, A., Baró, R.,
762 Bellasio, R., Brunner, D., Chemel, C., Curci, G., van der Gon, H. D., Flemming, J., Forkel,
763 R., Giordano, L., Jiménez-Guerrero, P., Hirtl, M., Jorba, O., Manders-Groot, A., Neal, L.,
764 Pérez, J. L., Pirovano, G., San Jose, R., Savage, N., Schroder, W., Sokhi, R. S., Syrakov, D.,
765 Tuccella, P., Werhahn, J., Wolke, R., Hogrefe, C., Galmarini, S., 2016. Insights into the
766 deterministic skill of air quality ensembles from the analysis of AQMEII data. Atmospheric
767 Chemistry and Physics, 16, 15629–15652, doi:10.5194/acp-16-15629-2016.

768 Lelieveld, J., Evans, J. S., Fnais, M., Giannadaki, D., Pozzer, A., 2015. The contribution of
769 outdoor air pollution sources to premature mortality on a global scale. Nature, 25, 367-371.

770 Liang, C., Silva, R.A., West, J.J., Emmons, L., Jonson, J.E., Bian, H., Pan, X., Chin, M.,
771 Henze, D., Lund, M.T., Sudo, K., Sekiya, T., Takemura, T., Flemming, J., Park, R., Lin, M.,
772 Pierce, R.B., Lenzen, A., Kucsera, T., Folberth, G., 2017. Multi-model estimates of
773 premature human mortality due to intercontinental transport of air pollution. Atmospheric
774 Chemistry and Physics, In preparation.

775 Lippmann, P., 2014. Toxicological and epidemiological studies of cardiovascular effects of
776 ambient air fine particulate matter ($PM_{2.5}$) and its chemical components: Coherence and
777 public health implications. Critical Reviews in Toxicology, 44(4), 299-347.

778 Mason, R., Zubrow, A., Eyth, A., 2007. Technical Support Document (TSD) Preparation of
779 Emissions Inventories for the Version 5.0, 2007 Emissions Modeling Platform, available at:
780 https://www.epa.gov/air-emissions-modeling/2007-version-50-technical-support-document,
781 last access: 24 May 2017.

782 OECD (2014). The Cost of Air Pollution: Health Impacts of Road Transport. OECD
783 Publishing, Paris. http://dx.doi.org/10.1787/9789264210448-en

784 OECD, 2006, Cost-benefit analysis and the environment: recent developments, Paris,
785 Organisation for Economic Co-operation and Development.

786 Ostro, B.D., 1987. Air Pollution and Morbidity Revisited: A Specification Test. Journal of
787 Environmental Economics and management, 14, 87-98.

788 Pope, C.A., Dockery, D.W., 2006. Health effects of fine particulate air pollution: lines that
789 connect. Journal of the Air and Waste Management Association, 56, 709-742.

790 Pope, C.A., Burnett, R.T., Thun, M.J., Calle, E.E., Krewski, D., Ito, K., Thurston, G.D.,
791 2002. Lung cancer, cardiopulmonary mortality and long-term exposure to fine particulate air
792 pollution. Journal of American Medical Association, 287 (9), 1132-1141.

793 Pope, C.A. Thun, M.J., Namboodiri, M.M., Dockery, D.W., Evans, J.S., Speizer, F.E., Heath
794 Jr, C.W., 1995. Particulate air pollution as a predictor of mortality in a prospective study of
795 US adults. American Journal of Respiratory and Critical Care Medicine, 151, 669-674.

796 Pope, C.A., Dockery, D.W., 1992. Acute Health Effects of PM10 Pollution on Symptomatic
797 and Asymptomatic Children. The American Review of Respiratory Disease, 145, 1123-1126.

798 Potempski, S., Galmarini, S., 2009. Est modus in rebus: analytical properties of multi-model
799 ensembles. Atmos. Chem. Phys., 9, 9471-9489, doi:10.5194/acp-9-9471-2009.

800 Pouliot, G., Denier van der Gon, H. A. C., Kuenen, J., Zhang, J., Moran, M. D., Makar, P. A.,
801 2015. Analysis of the emission inventories and model-ready emission datasets of Europe and
802 North America for phase 2 of the AQMEII project. Atmospheric Environment, 115, 345–360.

803 Riccio, A., Ciaramella, A., Giunta, G., Galmarini, S., Solazzo, E., Potempski, S., 2012. On
804 the systematic reduction of data complexity in multimodel atmospheric dispersion ensemble

modeling. Journal of Geophysical Research, 117, D05314. dx.doi.org/10.1029/2011JD016503.

Roemer, W., Hoek, G., Brunekreef, B., 1993. Effect of Ambient Winter Air Pollution on Respiratory Health of Children with Chronic Respiratory Symptoms. The American Review of Respiratory disease, 147, 118-124.

Schucht, S., Colette, A., Rao, S., Holland, M., Schopp, W., Kolp, P., Klimont, Z., Bessagnet, B., Szopa, S., Vautard, P., Brignon, J.-M., Rouil, L., 2015. Moving towards ambitious climate policies: Monetised health benefits from improved air quality could offset mitigation costs in Europe. Environmental Science & Policy, 50, 252-269.

Schwartz, J., Morris, R., 1995. Air Pollution and Hospital Admissions for Cardiovascular Disease in Detroit, Michigan. American Journal of Epidemiology, 142 (1). 23-35.

Silva, R. A., West, J. J., Lamarque, J.-F., Shindell, D. T., Collins, W. J., Dalsoren, S., Faluvegi, G., Folberth, G., Horowitz, L. W., Nagashima, T., Naik, V., Rumbold, S. T., Sudo, K., Takemura, T., Bergmann, D., Cameron-Smith, P., Cionni, I., Doherty, R. M., Eyring, V., Josse, B., MacKenzie, I. A., Plummer, D., Righi, M., Stevenson, D. S., Strode, S., Szopa, S., Zengast, G., 2016. The effect of future ambient air pollution on human premature mortality to 2100 using output from the ACCMIP model ensemble. Atmospheric Chemistry and Physics, 16, 9847-9862, doi:10.5194/acp-16-9847-2016.

Silva, R. A., West, J. J., Zhang, Y., Anenberg, S. C., Lamarque, J.-F., Shindell, D. T., Collins,W. J., Dalsoren, S., Faluvegi, G., Folberth, G., Horowitz, L. W., Nagashima, T., Naik, V., Rumbold, S., Skeie, R., Sudo, K., Takemura, T., Bergmann, D., Cameron-Smith, P., Cionni, I., Doherty, R. M., Eyring, V., Josse, B., MacKenzie, I. A., Plummer, D., Righi, M., Stevenson, D. S., Strode, S., Szopa, S., Zeng, G., 2013. Global premature mortality due to anthropogenic outdoor air pollution and the contribution of past climate change. Environmental Research Letters, 8, 034005, doi:10.1088/1748-9326/8/3/034005.

Solazzo, E., Van Dingenen, R., Riccio, A., Galmarini, S., 2017b. The role of multi-model ensembles in assessing the air quality impact on crop yields and mortality. In preparation for submission to ACP.

Solazzo, E., Bianconi, R., Hogrefe, C., Curci, G., Tuccella, P., Alyuz, U., Balzarini, A., Baró, R., Bellasio, R., Bieser, J., Brandt, J., Christensen, J. H., Colette, A., Francis, X., Fraser, A., Vivanco, M. G., Jiménez-Guerrero, P., Im, U., Manders, A., Nopmongcol, U., Kitwiroon, N., Pirovano, G., Pozzoli, L., Prank, M., Sokhi, R. S., Unal, A., Yarwood, G., Galmarini, S., 2017a. Evaluation and error apportionment of an ensemble of atmospheric chemistry transport modeling systems: multivariable temporal and spatial breakdown. Atmospheric Chemistry and Physics, 17, 3001-3054, doi:10.5194/acp-17-3001-2017.

Solazzo, E., Galmarini, S., 2016. Error Apportionment for atmospheric chemistry transport models – a new approach to model evaluation. Atmospheric Chemistry and Physics 16, 6263-6283.

Solazzo, E., Galmarini, S., 2015. A science-based use of ensembles of opportunities for assessment and scenario studies. Atmospheric Chemistry and Physics, 15, 2535-2544, doi:10.5194/acp-15-2535-2015.

Solazzo, E., Riccio, A., Kioutsioukis, I., Galmarini, S., 2013. Pauci ex tanto numero: reduce redundancy in multi-model ensemble. Atmospheric Chemistry and Physics, 13, 8315–8333.

Touloumi, G., Samoli, E., Katsuyanni, K., 1996. Daily mortality and "winter type" air pollution in Athens, Greece - a time series analysis within the APHEA project. Journal of Epidemiology and Community Health, 50 (suppl 1), S47 - S51

U.S. EPA, 2011. The Benefits and Costs of the Clean Air Act: 1990 to 2020. Final Report of U.S. Environmental Protection Agency Office of Air and Radiation, pp. 5-10.

Van Dingenen, R., Leitao, J., Dentener, F., 2014. A multi-metric global source-receptor model for integrated impact assessment of climate and air quality policy scenarios. European Geophysical Union General Assembly 2014.

Watkiss P., Pye S., Holland M., 2005. Cafe CBA: Baseline Analysis 2000 to 2020. Service Contract for Carrying out Cost-Benefit Analysis of Air Quality Related Issues, in Particular in the Clean Air for Europe (Cafe) Programme. 2005.. Available online: http://ec.europa.eu/environment/archives/cafe/activities/pdf/cba_baseline_results2000_2020.pdf, accessed on 24 May 2017.

Woodruff, T.J., Grillo, J., Schoendorf, K.C., 1997. The relationship between selected causes of postneonatal infant mortality and particulate air pollution in the United States. Environmental Health Perspectives, 105, 608-612.

Wordley, I., Walters, S., Ayres J.G., 1997. Short term variations in hospital admissions and mortality and particulate air pollution. Journal of Occupational Environmental Medicine, 54, 108-116.

World Health Organization (WHO) 2014. http://www.who.int/mediacentre/news/releases/2014/air-pollution/en/

World Health Organization (WHO), 2013. Review of evidence on health aspects of air pollution (REVIHAAP). WHO Technical Report.

Zhang, Q., Jiang, X., Tong, D., Davis, S.J., Zhao, H., Geng, G., Feng, T., Zheng, B., Lu, Z., Streets, D.G., Ni, R., Brauer, M., van Donkelaar, A., Martin, R.V., Huo, H., Liu, Z., Pan, D., Kan, H., Yan, Y., Lin, J., He, K., Guan, D., 2017. Transboundary health impacts of transported global air pollution and international trade. Nature, 543, 705-709, doi:10.1038/nature21712.

Atmos. Chem. Phys. Discuss., https://doi.org/10.5194/acp-2017-751
Manuscript under review for journal Atmos. Chem. Phys.
Discussion started: 27 September 2017

Atmospheric Chemistry and Physics Discussions

Table 1.Key features (meteorological/chemistry and transport models, emissions, horizontal and vertical grids) of the regional models participating to the AQMEII3 health impact study and the perturbation scenarios they performed.

Group Code	Model	Emissions	Horizontal Resolution	Vertical Resolution	Europe				North America			
					BASE	GLO	NAM	EUR	BASE	GLO	EAS	NAM
DE1	COSMO-CLM/CMAQ	HTAP	24 km × 24 km	30 layers, 50 hPa	×	×	×	×	×	×	×	×
DK1	WRF/DEHM	HTAP	50 km × 50 km	29 layers, 100 hPa	×	×	×	×	×	×	×	×
ES1	WRF/CHEM	MACC	23 km × 23 km	33 layers, 50 hPa	×		×					
FI1	ECMWF/SILAM	MACC	0.25° × 0.25°	12 layers, 13 km	×	×	×	×				
FRES1	ECMWF/CHIMERE	HTAP	0.25° × 0.25°	9 layers, 50 hPa	×	×	×	×				
IT1	WRF/CAMx	MACC	23 km × 23 km	33 layers, 50 hPa	×	×		×				
IT2	WRF/CHEM	MACC	23 km × 23 km	14 layers, 8 km	×	×						
NL1	LOTOS/EUROS	MACC	0.50° × 0.25°	4 layers, 3.5 km	×							
TR1	WRF/CMAQ	MACC	30 km × 30 km	24 layers, 10hPa	×	×	×					
UK1	WRF/CMAQ	MACC	15 km × 15 km	23 layers, 100 hPa	×	×	×	×				
UK2	WRF/CMAQ	HTAP	30 km × 30 km	23 layers, 100 hPa	×	×						
UK3	WRF/CMAQ	MACC	18 km × 18 km	35 layers, 16 km	×	×	×					
US3	WRF/CAMx	SMOKE	12 km × 12 km	35 layers, 50 hPa					×	×	×	×

Atmos. Chem. Phys. Discuss., https://doi.org/10.5194/acp-2017-751
Manuscript under review for journal Atmos. Chem. Phys.
Discussion started: 27 September 2017

Atmospheric Chemistry and Physics Discussions

Table 2. Exposure-response functions and economic valuations used in the EVA model.

Health effects (compounds)	**Exposure-response coefficient**	**Valuation, €$_{2013}$**
	(α)	**(EU27 & NA)**
Morbidity		
Chronic Bronchitis[1], CB (PM)	8.2E-5 cases/µgm^{-3} (adults)	38,578 per case
Restricted activity days[2], RAD (PM)	=8.4E-4 days/ µgm^{-3} (adults)	98 per day
	-3.46E-5 days/ µgm^{-3} (adults)	
	-2.47E-4 days/ µgm^{-3} (adults>65)	
	-8.42E-5 days/ µgm^{-3} (adults)	
Congestive heart failure[3], CHF (PM)	3.09E-5 cases/ µgm^{-3}	10,998 per case
Congestive heart failure[3], CHF (CO)	5.64E-7 cases/ µgm^{-3}	
Lung cancer[4], LC (PM)	1.26E-5 cases/ µgm^{-3}	16,022 per case
Hospital admissions		
Respiratory[5], RHA (PM)	3.46E-6 cases/ µgm^{-3}	5,315 per case
Respiratory[5], RHA (SO_2)	2.04E-6 cases/ µgm^{-3}	
Cerebrovascular[6], CHA (PM)	8.42E-6 cases/ µgm^{-3}	6,734 per case
Asthma children (7.6 % < 16 years)		
Bronchodilator use[7], BUC (PM)	1.29E-1 cases/ µgm^{-3}	16 per case
Cough[8] – COUC (PM)	4.46E-1 days/ µgm^{-3}	30 per day
Lower respiratory symptoms[7], LRSA (PM)	1.72E-1 days/ µgm^{-3}	9 per day
Asthma adults (5.9 % > 15 years)		
Bronchodilator use[9], BUA (PM)	2.72E-1 cases/ µgm^{-3}	16 per case
Cough[9], COUA (PM)	2.8E-1 days/ µgm^{-3}	30 per day
Lower respiratory symptoms[9], LRSA (PM)	1.01E-1 days/ µgm^{-3}	9 per day
Mortality		
Acute mortality[10,11] (SO_2)	7.85E-6 cases/ µgm^{-3}	1,532,099 per case
Acute mortality[10,11] (O_3)	3.27E-6*SOMO35 cases/ µgm^{-3}	
Chronic mortality[4,12], YOLL (PM)	1.138E-3 YOLL/ µgm^{-3} (>30 years)	57,510 per YOLL
Infant mortality[13], IM (PM)	6.68E-6 cases/ µgm^{-3} (> 9 months)	2,298,148 per case

[1] Abbey et al. (1995), [2] Ostro (1987), [3] Schwartz and Morris (1995), [4] Pope et al. (2002), [5] Dab et al. (1996), [6] Wordley et al. (1997), [7] Roemer et al. (1993), [8] Pope and Dockerey (1992), [9] Dusseldorp et al. (1995), [10] Anderson (1996), [11] Touloumi (1996), [12] Pope et al. (1995), [13] Woodruff et al. (1997).

Atmos. Chem. Phys. Discuss., https://doi.org/10.5194/acp-2017-751
Manuscript under review for journal Atmos. Chem. Phys.
Discussion started: 27 September 2017

Table 3. Model evaluation over the European and North American domains (hourly for O_3, CO and SO_2 and daily means for $PM_{2.5}$). Units are % for *NMB* and *NMGE*, μg m^{-3} for all species for Europe and ppb for the gaseous species and μg m^{-3} for $PM_{2.5}$ in North America.

	O_3				CO				SO_2				$PM_{2.5}$			
Models	*r*	*NMB*	*NMGE*	*RMSE*	*r*	*NMB*	*NMGE*	*RMSE*	*r*	*NMB*	*NMGE*	*RMSE*	*r*	*NMB*	*NMGE*	*RMSE*
Europe																
DE1	0.73	9.87	4.59	13.50	0.80	-42.07	41.75	133.24	0.77	4.34	21.07	1.33	0.88	-63.08	128.10	11.95
DK1	0.88	6.71	2.59	9.99	0.74	-41.67	43.10	135.84	0.85	-47.24	56.49	1.54	0.86	-45.69	56.82	9.65
ES1	0.79	-15.16	6.59	14.21	0.59	-46.27	55.42	147.82	0.78	-68.13	182.02	2.15	0.23	-30.84	44.68	9.66
FI1	0.84	-35.87	24.00	23.58	0.85	-26.75	15.78	92.11	0.82	-20.49	17.26	1.05	0.58	-26.98	29.18	8.02
FRES1	0.78	-9.65	4.79	12.51	0.82	-39.19	34.10	123.37	0.74	-76.81	320.13	2.44	0.87	-36.16	32.25	7.88
IT1	0.90	4.20	2.45	9.60	0.82	-36.81	31.23	120.35	0.79	-29.78	28.19	1.26	0.78	-18.25	14.88	6.06
IT2	0.92	-14.26	4.46	11.76	0.77	-43.53	45.13	136.44	0.81	-54.78	87.14	1.77	0.11	-48.40	87.23	11.65
NL1	0.92	-5.06	2.01	8.30	0.69	-46.09	55.74	148.51	0.80	-51.92	83.39	1.79	0.76	-55.55	99.64	11.56
TR1	0.86	8.09	8.91	18.65	0.84	-20.11	9.40	74.24	0.43	2.28	24.06	1.40	0.60	-19.16	21.08	7.17
UK1	0.91	7.51	2.13	9.10	0.59	-41.56	44.88	138.72	0.73	-12.96	16.19	1.06	0.78	-40.32	44.67	8.97
UK2	0.83	-2.75	4.17	12.10	0.64	-42.63	45.23	138.00	0.72	20.46	17.82	1.31	0.77	-28.28	23.59	7.15
UK3	0.78	-1.01	4.04	12.01	0.80	-45.04	48.32	139.60	0.64	48.75	46.00	2.34	0.94	-43.82	42.44	8.48
MEAN	0.84	-3.95	5.89	12.94	0.75	-39.31	39.17	127.35	0.74	-23.86	74.98	1.62	0.68	-38.04	52.05	9.02
MEDIAN	0.85	-1.88	4.32	12.06	0.78	-41.87	43.99	136.14	0.77	-25.14	37.10	1.47	0.77	-38.24	43.56	8.73
North America																
DE1	0.85	5.55	11.65	4.69	0.41	-40.68	40.71	92.20	0.45	-40.20	41.05	1.35	0.74	-62.65	62.65	6.97
DK1	0.72	21.75	23.80	10.33	0.47	-7.41	18.02	47.32	0.63	-42.36	43.47	1.35	0.64	-14.08	22.21	2.86
US3	0.88	-1.53	11.18	4.51	0.44	-3.89	19.89	51.42	0.52	-12.83	23.98	0.84	0.76	17.23	23.75	3.25
MEAN	0.82	8.59	15.54	6.51	0.44	-17.33	26.21	63.65	0.53	-31.80	36.17	1.18	0.71	-19.83	36.21	4.36
MEDIAN	0.85	5.55	11.65	4.69	0.44	-7.41	19.89	51.42	0.52	-40.20	41.05	1.35	0.74	-14.08	23.75	3.25

Atmospheric Chemistry and Physics Discussions
Open Access
EGU

Atmos. Chem. Phys. Discuss., https://doi.org/10.5194/acp-2017-751
Manuscript under review for journal Atmos. Chem. Phys.
Discussion started: 27 September 2017

Atmospheric Chemistry and Physics Discussions

Table 4. Health impacts as calculated by the individual models over Europe and the United States ($\times 10^3$, except for IM). See Table 2 for the definitions of health impacts.

Models	CB	RAD	RHA	CHA	CHF	LC	BUC	BUA	COUC	COUA	LRSC	LRSA	PD	IM
Europe														
DE1	191	194 776	13	24	19	29	5 694	37 284	19 674	38 380	7 592	13 844	232	213
DK1	290	296 611	17	37	26	44	8 671	56 776	29 960	58 446	11 562	21 082	336	325
ES1	415	424 229	23	53	34	64	12 402	81 205	42 851	83 593	16 536	30 153	456	465
FI1	411	420 220	25	53	35	63	12 285	80 437	42 445	82 803	16 380	29 868	457	460
FRES1	373	381 243	22	48	32	57	11 146	72 976	38 509	75 123	14 861	27 098	419	418
IT1	507	517 996	30	65	41	78	15 144	99 153	52 322	102 070	20 191	36 818	571	568
IT2	310	317 256	18	40	27	48	9 275	60 728	32 045	62 514	12 367	22 550	345	348
NL1	264	269 418	16	34	24	40	7 876	51 571	27 213	53 088	10 502	19 150	303	295
TR1	460	470 496	29	59	40	70	13 755	90 061	47 524	92 710	18 340	33 442	538	516
UK1	343	351 026	23	44	30	53	10 262	67 192	35 456	69 169	13 683	24 950	404	516
UK2	417	425 950	28	53	35	64	12 453	81 534	43 024	83 932	16 603	30 275	488	467
UK3	342	349 974	26	44	29	52	10 231	66 991	35 350	68 961	13 642	24 875	416	383
MEAN	360	368 266	23	46	31	55	10 766	70 492	37 198	72 566	14 355	26 175	414	414
MEDIAN	358	366 135	23	46	31	55	10 704	70 084	36 982	72 146	14 272	26 024	418	439
The United States														
DE1	61	62 305	5	8	7	9	1 946	11 926	6 722	12 277	2 594	4 428	80	61
DK1	161	164 681	10	21	15	25	5 148	31 522	17 787	32 449	6 864	11 705	191	161
US3	204	209 023	13	27	18	31	6 604	40 009	22 819	41 186	8 806	14 856	224	209
MEAN	142	145 337	10	19	13	22	4 566	27 819	15 776	28 637	6 088	10 330	165	143
MEDIAN	161	164 681	10	21	15	25	5 148	31 522	17 787	32 449	6 864	11 705	191	161

Atmos. Chem. Phys. Discuss., https://doi.org/10.5194/acp-2017-751
Manuscript under review for journal Atmos. Chem. Phys.
Discussion started: 27 September 2017

Atmospheric Chemistry and Physics Discussions

Table 5. External costs (in million Euros) related to the health impacts of air pollution as calculated by the individual models over Europe and the United States.

Models	CO	SO_2	O_3	$PM_{2.5}$	TOTAL
Europe					
DE1	70	19 000	22 000	155 000	196 000
DK1	80	13 000	24 000	237 000	274 000
ES1	70	8 000	6 000	339 000	353 000
FI1	90	18 000	5 000	335 000	358 000
FRES1	90	15 000	13 000	305 000	333 000
IT1	80	17 000	21 000	413 000	451 000
IT2	70	11 000	6 000	253 000	270 000
NL1	70	12 000	18 000	215 000	245 000
TR1	110	30 000	35 000	376 000	441 000
UK1	80	28 000	25 000	280 000	333 000
UK2	80	34 000	27 000	340 000	401 000
UK3	80	47 000	25 000	279 000	351 000
MEAN	81	21 000	19 000	294 000	334 000
MEDIAN	80	17 500	21 500	292 500	342 000
The United States					
DE1	30	9 000	21 000	46 000	76 000
DK1	55	11 000	39 000	123 000	172 000
US3	60	14 000	22 000	155 000	191 000
MEAN	50	11 500	27 000	108 000	146 000
MEDIAN	55	11 000	22 000	123 000	172 000

Atmos. Chem. Phys. Discuss., https://doi.org/10.5194/acp-2017-751
Manuscript under review for journal Atmos. Chem. Phys.
Discussion started: 27 September 2017

Atmospheric Chemistry and Physics Discussions

Table 6. Annual average RMSE of the multi-model ensemble mean (MM_m) and of the optimal reduced ensemble mean (MM_{opt}) for the heath impact-related species. Units are μg m^{-3} for all species for Europe and ppb for the gaseous species and μg m^{-3} for $PM_{2.5}$ in North America.

	O_3		CO		SO_2		$PM_{2.5}$	
	MM_m	MM_{opt}	MM_m	MM_{opt}	MM_m	MM_{opt}	MM_m	MM_{opt}
Europe								
Winter	10.3	8.6	502.4	490.3	6.3	5.6	22.5	20.7
Spring	12.4	9.6	247.1	239.5	4.6	3.1	9.9	7.8
Summer	13.4	10.7	197.4	188.0	3.9	2.3	8.2	5.7
Autumn	10.7	8.8	314.5	305.5	4.6	3.1	11.0	8.7
Annual	11.7	9.4	315.3	305.8	4.8	3.5	12.9	10.7
North America								
Winter	10.9	10.4	356.7	328.1	5.7	5.5	8.3	8.1
Spring	12.0	11.4	288.7	270.2	5.4	5.1	7.2	6.6
Summer	15.1	13.0	258.3	238.7	5.4	5.0	9.7	8.8
Autumn	12.8	11.6	330.6	307.6	5.8	5.3	7.8	7.2
Annual	12.7	11.6	308.6	286.1	5.6	5.2	8.2	7.7

Atmos. Chem. Phys. Discuss., https://doi.org/10.5194/acp-2017-751
Manuscript under review for journal Atmos. Chem. Phys.
Discussion started: 27 September 2017

Atmospheric Chemistry and Physics Discussions

Table 7. All models-mean (MM_m) vs. optimal ensemble (MM_{opt}) calculated for different health impacts over Europe and U.S ($\times 10^3$, except for IM).

	Europe		The United States	
	MM_m	MM_{opt}	MM_m	MM_{opt}
CB	360	468	142	125
RAD	368 000	478 100	145 300	127 900
RHA	23	28	8	7
CHA	46	60	19	16
CHF	31	38	9	8
LC	55	72	22	19
BDUC	10 800	14 000	4 600	4 000
BDUA	70 500	91 500	27 800	24 500
COUC	37 200	48 300	15 800	13 900
COUA	72 600	94 200	28 600	25 200
LRSC	14 400	18 600	6 100	5 400
LRSA	26 170	34 000	10 300	9 100
PD	410	524	149	133
IM*	403	524	143	126

Atmos. Chem. Phys. Discuss., https://doi.org/10.5194/acp-2017-751
Manuscript under review for journal Atmos. Chem. Phys.
Discussion started: 27 September 2017

Atmospheric Chemistry and Physics Discussions

Table 8. Impact of the emission reduction scenarios on avoided premature death (ΔPD) and corresponding change in external cost as calculated by the multi-model mean over Europe and the United States.

Source	Receptor			
	Europe		The United States	
	ΔPD	ΔTotal Cost (billion €)	ΔPD	ΔTotal Cost (billion €)
GLO	-54 000 ± 18 000	-56 ± 18	-27 500 ± 14 000	-24 ± 10
NAM	-940 ± 1100	-1.4 ± 0.4	-25 000 ± 12 000	-21 ± 9
EUR	-47 000 ± 24 000	-47 ± 16	-	-
EAS	-	-	-1 900 ± 2 200	-2.5 ± 3

Atmos. Chem. Phys. Discuss., https://doi.org/10.5194/acp-2017-751
Manuscript under review for journal Atmos. Chem. Phys.
Discussion started: 27 September 2017

Fig.1. Population density over Europe and the United States.

Atmos. Chem. Phys. Discuss., https://doi.org/10.5194/acp-2017-751
Manuscript under review for journal Atmos. Chem. Phys.
Discussion started: 27 September 2017

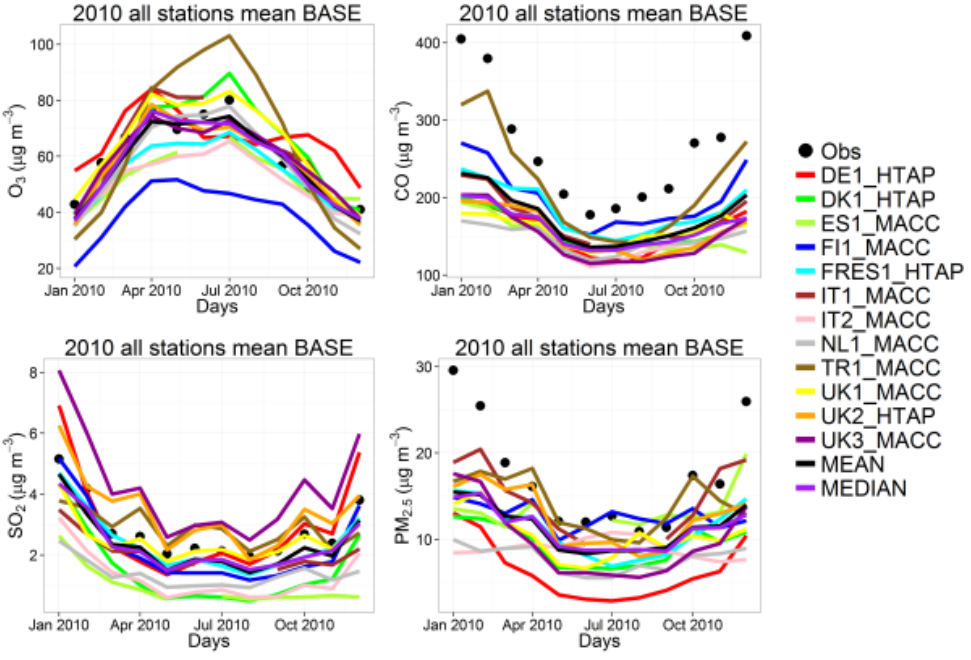

Fig. 2. Observed and simulated (base case) monthly a) O3, b) CO, c) SO2 and d) PM2.5 concentrations over Europe.

Atmos. Chem. Phys. Discuss., https://doi.org/10.5194/acp-2017-751
Manuscript under review for journal Atmos. Chem. Phys.
Discussion started: 27 September 2017

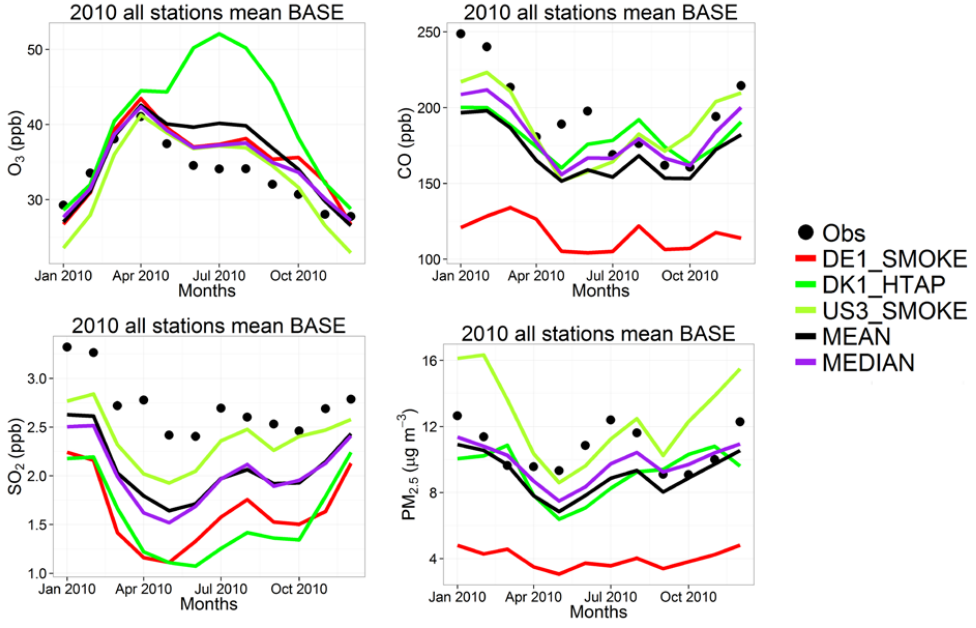

Fig. 3. Observed and simulated (base case) monthly a) O_3, b) CO, c) SO_2 and d) $PM_{2.5}$ concentrations over the U.S.

Atmos. Chem. Phys. Discuss., https://doi.org/10.5194/acp-2017-751
Manuscript under review for journal Atmos. Chem. Phys.
Discussion started: 27 September 2017

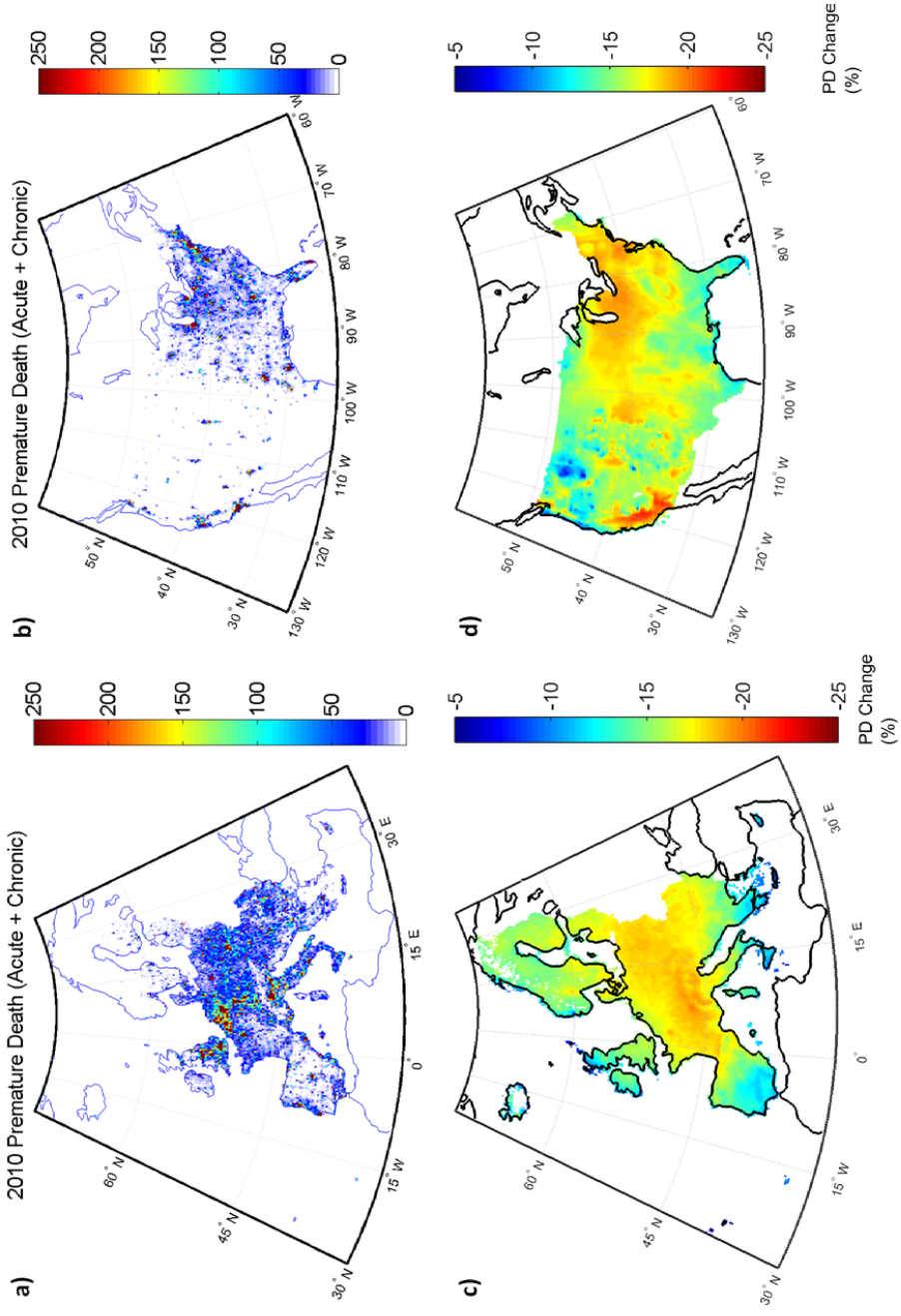

Fig. 4. Spatial distribution of the number of total premature death in Europe and the United States in 2010 as calculated by the multi-model mean ensemble.